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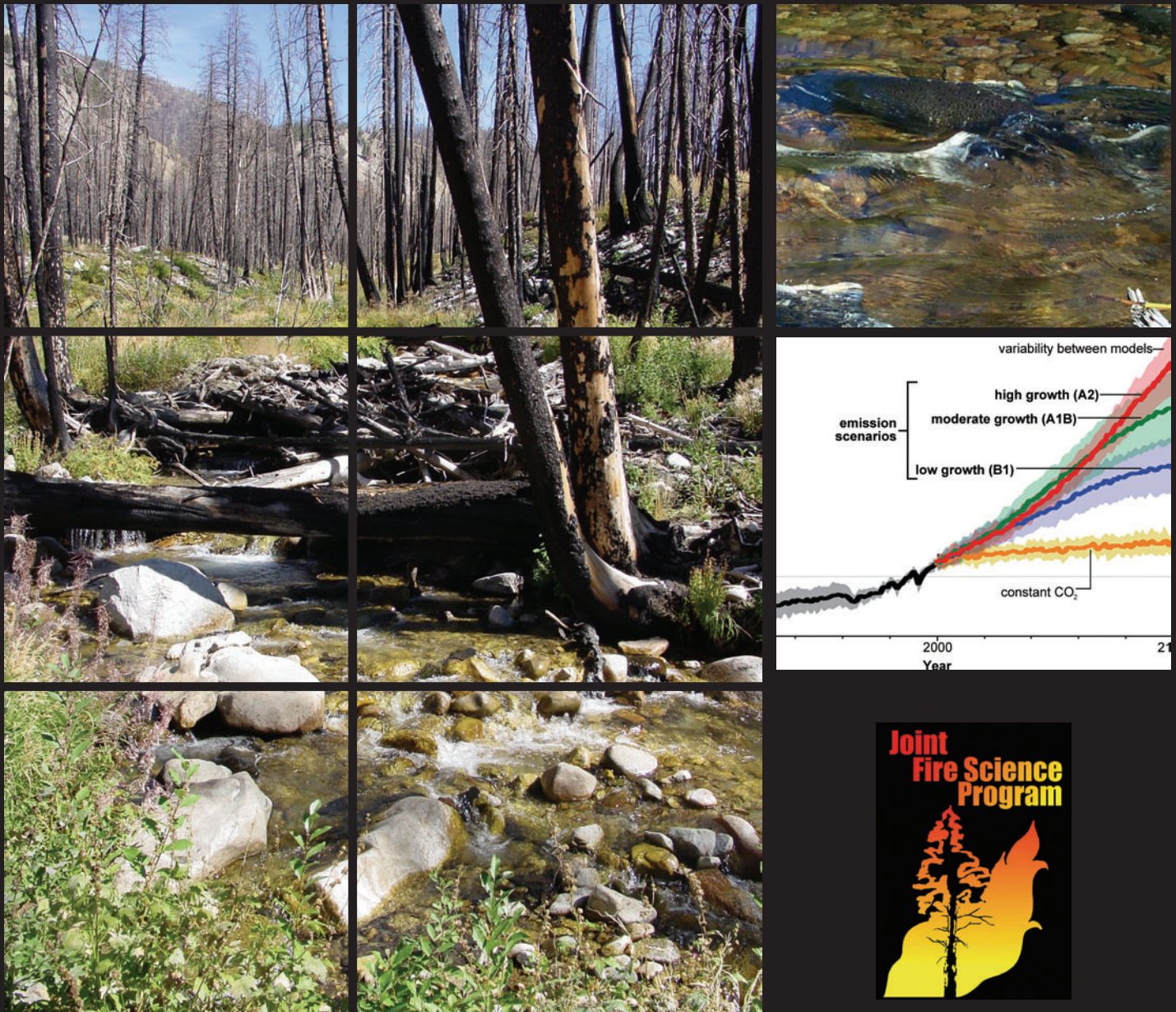
September 2012



Climate Change, Forests, Fire, Water, and Fish:

Building resilient landscapes, streams, and managers

Charles Luce, Penny Morgan, Kathleen Dwire, Daniel Isaak,
Zachary Holden, and Bruce Rieman



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ABSTRACT

Fire will play an important role in shaping forest and stream ecosystems as the climate changes. Historic observations show increased dryness accompanying more widespread fire and forest die-off. These events punctuate gradual changes to ecosystems and sometimes generate stepwise changes in ecosystems. Climate vulnerability assessments need to account for fire in their calculus. The biophysical template of forest and stream ecosystems determines much of their response to fire. This report describes the framework of how fire and climate change work together to affect forest and fish communities. Learning how to adapt will come from testing, probing, and pushing that framework and then proposing new ideas. The western U.S. defies generalizations, and much learning must necessarily be local in implication. This report serves as a scaffold for that learning. It comprises three primary chapters on physical processes, biological interactions, and management decisions, accompanied by a special section with separately authored papers addressing interactions of fish populations with wildfire. Any one of these documents could stand on its own. Taken together, they serve as a useful reference with varying levels of detail for land managers and resource specialists. Readers looking for an executive summary are directed to the sections titled "Introduction" and "Next Steps."

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Climate Change, Forests, Fire, Water, and Fish:

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Charles Luce, Penny Morgan, Kathleen Dwire, Daniel Isaak,
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with

Advanced Topics on Fish Populations and Fire
by various authors



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Introduction

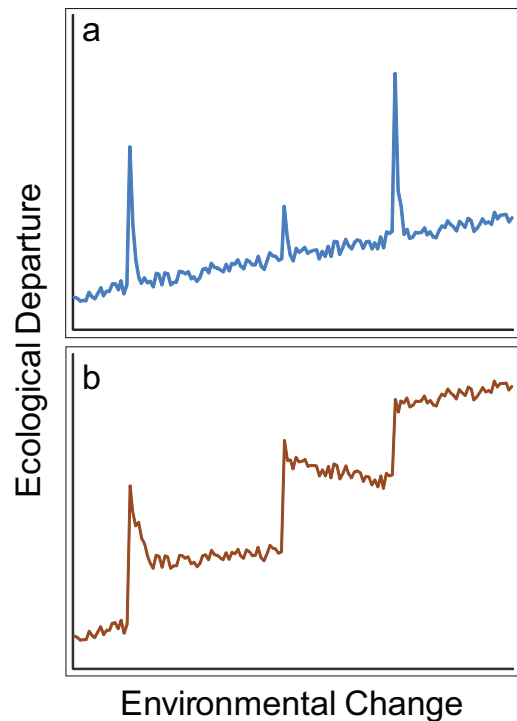
As the climate changes in the western United States, streams are warming, low flows in summer are declining, and winter floods are occurring more often in places where snowmelt is the main source of water (Stewart and others 2005; Hamlet and Lettenmaier 2007; Luce and Holden 2009; Isaak and others 2010). Some of the changes have been subtle, others more noticeable, and they are expected to shift distributions of fishes (Rieman and others 2007; Wenger and others 2011a; Wenger and others 2011b). At the same time, the terrestrial ecosystems surrounding the mountain streams of the West are changing in response to the same climatic signals. Drier years and drier summers have often led to more large fires, many of which are more severe (Dillon and others 2011). Further, fire regimes are shifting, with fires becoming more frequent in some places and less frequent in others, and potential conversion of forests to shrubs in some places (Pierce and others 2004; Breshears and others 2005; Westerling and others 2006; Morgan and others 2008; Westerling and others 2011).

Fires have long been prevalent in Western mountain landscapes. Many, but not all, ecosystems benefit from the biomass consumption, cycling of nutrients, rejuvenation of vegetation, and changing vegetation composition and structure after fires (Agee 1993). Indeed many species and ecological communities in the western United States depend on fire in some form. Some benefit from frequent fires that consume small amounts of fuel, while others, seemingly paradoxically, thrive as a result of infrequent but severe fires that consume most of the available fuel in their path. Thus, fire itself has been long recognized as a considerable influence, apart from any consideration of a changing climate, on forest and stream ecosystems (e.g., Bisson and others 2003; Shakesby and Doerr 2006; Hessburg and others 2007).

The number of large fires has increased in recent decades, and future annual area burned is likely to increase further with concurrent concerns over costs of fire management and threats to safety of people and property (NWCG [National Wildfire Coordinating Group] 2009; Spracklen and others 2009; Littell and others 2010). Although not all environments are equally prone to fire, and humans have been very effective at detecting and suppressing the majority of fires when they are small (Stephens and Ruth 2005), forest fires will continue to occur. Global, national and regional trends of increasing number of large fires in recent decades are likely to continue with implications for both terrestrial and aquatic systems.

Fire and related disturbances will be an agent of climate change in shifting forest ecosystems (Dale and others 2001; Jentsch and others 2007; Turner 2010). Tree mortality can be caused directly by climate, or it may be induced by fire that is in turn responding to climate. Sometimes, the loss of the current forest canopy can pave the way for new species and even life forms (e.g., shrubs and grasses). Thus, climate change and climate variability have both direct and indirect implications for fish, streams, and aquatic ecosystems. Fire may become a critical point in the progression of individual forest stands or streams, where ecosystems may either gradually shift in response to climate change punctuated by fire and recovery (Figure 1a), much like they always have, or where ecosystems are relatively non-responsive to climate between events that provide the catalyst to adjust to new climate conditions (Figure 1b). This new transitioning role of fire as “coup de grace” will pose new challenges for land managers who are well versed in the cyclic dynamics of forests. Of course, this simple model must be thought of in different terms in the context of changing disturbance frequency and severity as well. Providing sustained ecosystem services through seemingly unpredictable change-points may represent a primary challenge for managers.

Figure 1: Conceptual roles for disturbance in a changing climate. Disturbance could continue to operate much as it always has (a), with unique disturbance/recovery patterns, or it could become the catalyst that forces ecosystems to shift rapidly and via alternate and uncertain pathways (b) in response to climate.



While natural systems have evolved adaptations to the kinds of disturbances provided by fire, plant and animal populations may not be resilient to fires when fire regimes change, or when the landscape context of those fires changes. Large trees that survived many surface fires in the past may die in high severity fires, and regeneration of new trees may fail if fire recurs before young trees grow to fire-resistant size. Where serotinous cones have aided rapid post-fire regeneration, such regeneration will be less successful if fires recur before trees are old enough to produce abundant cones. For species relying on recolonization through dispersal from unburned refugia, very large, severe fires may present too great a barrier. Trees may not regenerate successfully following high severity fires at lower timberline if the post-fire environment is less conducive than in the past or if invasive plants pose severe competition. An awareness of how the chain of consequences from climate change interacts with natural adaptations will be critical to forming solutions that maintain valued ecosystem components and processes into the future.

Within the pantheon of adaptation and mitigation concepts and approaches, two terms, resistance and resilience, stand out as critical ideas (e.g., Holling 1973; Waide 1988; Millar and others 2007 and see Resilience text box). Resistance is the ability of an ecosystem to experience stressors but not change. For example, old ponderosa pine and Douglas-fir trees with thick bark are very resistant to surface fires. Engineers describe resilience as the ability to return to a given state despite sometimes formidable changes. Ecologists, sociologists, and psychologists share a more generalized definition of resilience as the capacity to absorb and weather change in a way that both combines and transcends the engineering concepts. Nonetheless, the engineering-oriented metaphor highlights the point that if climate changes, there seems to be a difficulty in applying either concept, as resistance must eventually be overcome, and it's difficult to “bounce back” if the driving pressures are maintained if not growing. The concept of facilitated change fills in this difficult space. For example, while thinning might be seen as a resistance step for fire in one context, it can also serve to help forests cope with a changed water balance in a more predictable manner than without treatment.

Preparation in many forms contributing to both resilience and resistance will be important, as will appropriate responses during and after major disturbances. No longer will simple protective responses to events suffice, nor even simple protective preparations. A set of strategic measures encompassing whole landscape perspectives using combinations of protective, monitoring, and corrective approaches will be necessary to manage a dynamic system suffused with uncertainty from both chance events and incomplete understanding. There will be tradeoffs between current and future risks. Management actions taken in the present will generally pose some risk, especially in the short term. The question is whether the imposed risks outweigh potential future risks. Even if they do, there are questions about scaling imposed risks, like how much at once and how much do we leave to chance in the short to medium term. While none of these questions have universal answers, there are contexts that support one approach versus others, and attentive managers teamed with researchers can learn how to describe the tradeoffs rationally.

Key Debates

With respect to forests, critical issues revolve around fire and fuels management including mechanical fuel reduction, intentional fire treatments, and natural fire treatments. Each comes with attendant risks, such as fires with unintentionally high severity or size, long duration and severity of smoke exposure from fires, potential for increase in invasive species, and impacts of roads where they are needed to facilitate management. There are costs and benefits with these actions, just as there are costs and benefits to no action. Decisions about where to prioritize work are a critical piece of the decision-making process. These decisions are made most frequently in the contexts of human habitation and threats to forests from fire, insects, and disease.

These decisions are sometimes difficult (and constrained) without considering the riparian and aquatic components of the ecosystem. Within riparian zones, most treatment options, including no action, have consequences for unique plant communities and adjacent streams. Considering aquatic communities brings in a range of other issues for water and aquatic management, some of which compete, or seem to compete, with decision space for forest management. Roads, which provide important access for silvicultural treatments and fire response, now form a threat not just to native vegetation, but also to stream communities, intensifying the tradeoffs. Reframing the decision goals to optimize both aquatic and terrestrial conditions can reveal opportunities in place of tradeoffs, particularly in previously managed areas with an existing road system (Rieman and others 2000; Rieman and others 2010).

Earlier syntheses of fire effects on fish and streams (Bisson and others 2003; Rieman and others 2003a and papers therein) provided new ideas and science that helped bridge the complexity of balancing the multiple resources. A principal idea presented in those papers was a greater reliability on natural dynamics to create resilient forest and stream ecosystems. The idea was appealing both from an economic perspective, due to reducing fuel and fire management expenses, and the perspective of persistence of key aquatic resources. The notion that while local fish populations might be severely reduced, they had life history adaptations that allowed them to persist in the long term stepped away from a static view for healthy aquatic ecosystems. The emerging view is that fire has played not just an important role in Western forests, but the streams running through them as well. This conceptualization ties well to a broader understanding of forest rejuvenation after fire and offers alternatives to artificial divisions between forest and stream ecosystems in any given place in

favor of viewing it as one jointly cycling ecosystem. Scientists and managers have warned, however, that where connectivity and fragmentation of forests and streams was changed through historical land management, the resilience that is critical to natural cycling may be compromised.

The complexities added by climate variability change the discussion about forests, fish, and fire. Where the understanding that a goal of improved resilience to fire could commonly solve both aquatic and forest issues (Rieman and others 2010), potentially with relatively low expense and public opposition, we are now more commonly faced with choices between some kind of active intervention versus prospective loss of species locally. Once, we could identify a fairly simple dichotomy between (1) places where wildfire could operate freely without dramatically changing the natural dynamics of an environment and (2) places where some restoration of forests or streams would be necessary before wildfire would be unlikely to substantially alter the natural dynamics. A concept called “historical range of variability” (e.g., Keane and others 2009) was used to describe natural dynamics in forest or stream conditions. There are likely still a very few places where the dynamics will be comparatively unchanged, but most places in the western United States will no longer have the same temporal and spatial scalings of the dynamic processes, limiting our ability to mimic cycles of the past.

We are seemingly back to similar issues that defined the debate about forests versus fish a decade ago, only with more intensity. Forests are now more urgently in need of treatment, and that need may be more geographically widespread. Aquatic ecosystems are becoming even more sensitive to either management or uncharacteristic fire. The limitation of resources for treating or responding to changes is similar, pointing again to prioritization as an important first step in reducing apparent conflicts. Human disruption of forest landscapes through timber harvest and construction of new roads has, however, decreased. Solutions are likely to be challenging, and most proposals for active intervention are likely to be controversial in the public arena (Spies and others 2010). While there is a recognition that dynamics are critical, the fact that they are no longer the same dynamics leaves questions in the minds of scientists, land and water managers, and the public as to how to proceed.

Framing Solutions

A key, though difficult, step will be articulating goals for the future (Rieman and others 2010). Communication of the goals will need to address the cross-disciplinary nature of the problem and be explicit in definitions and values that frame the goals. Climate change shifts the decision space in significant ways. It alters what the ultimate goals look like, from one where we consider the (comparatively) simple harmonic (cyclic) dynamic of forests burning and regrowing in patches to one of a moving target, sometimes gradual, sometimes rapid.

Envisioning future solutions will be facilitated through exploration of forest and stream ecosystem dynamics across landscapes containing multiple populations or patches and over long time periods containing many events and ecosystem response trajectories (White and Jentsch 2001; Jentsch and others 2003; Jentsch 2007). The nature of interventions, if any, should draw from a deep understanding of which components of ecosystem-stream dynamics are most important to their structure and function and how the spatial and temporal scales of disturbances interact through the biological system to produce the observed biodiversity. Nonlinearities in physical

and biological processes, including either threshold-like or buffering behaviors, will identify both heightened and dampened vulnerability. As shifts occur in the physical climate and novel patterns begin to emerge, then, managers may develop new coping ideas taking advantage of this understanding of ecosystem dynamics.

Climate change is shifting what is and is not possible in some areas, and goals that once included protection of some species in a given location may no longer be tenable. If the critical dichotomy 10 years ago was whether wildfire or fuel treatments were worse for aquatic ecosystems, today it could be glibly paraphrased as whether we are more interested in maintaining select species populations or maintaining ecosystem function (Rieman and others 2010). The question revolves around values, what is possible within constraints of changing climate, and how much intervention we are willing to accept, or pay for, in forests. To be effective, interventions may have to be at scales grand enough to address the issues, with consequent needs to manage the potential for unintended outcomes of our decisions.

The scientific contributions to these policy issues lie in exploring the constraints fundamentally imposed by climatic changes, the biophysical context, and what capacity we have as land and water managers to alter how ecosystems will respond. Constraint can be partially viewed as parallel to vulnerability analysis, where shifts in habitat suitability over time are explored for species or communities (e.g., Parson and others 2003; Turner and others 2003; Füssel and Klein 2006; Millar and others 2007). Habitat suitability is defined in part by disturbance regimes as well as basic climatic factors, and this particular aspect of vulnerability is key to working through issues related to fire. Our ability to alleviate ecosystem stresses and stretch possibilities around these constraints is found in adaptation. Although the concept of resilience resonates for applications related to fire, resistance and facilitation will likely also be important (Millar and others 2007), and the distinction may be blurry in application.

Although most vulnerability analyses have been done without the full knowledge of the likely interactions and indirect effects of climate change, a logical extension would include changes in expected fire regimes associated with climate shifts and the interactions of those fire regimes with species and biotic communities. The expression of disturbance regimes can in fact be a stronger determinant of species ranges than the optimal temperature and precipitation requirements of the species (Pickett and White 1985). Because fire severity and size have a significant influence on aquatic habitats, this more complex scope for vulnerability analysis to include upslope and riparian vegetation conditions and risks is a key step in building holistic plans for adaptation of forest landscapes and watersheds.

The added challenges posed by climate change are revising not only how we assess risks to aquatic resources but also how much we can rely on resilience as a primary adaptation strategy. Many fish populations, particularly in more natural settings and without pressures from invasive species, seem to be well poised for resilience to disturbances from wildfire. However, it is generally unclear which populations would continue to persist under the combined effects of stream warming and increased fire size, severity, and frequency, and for how long. There are also questions about how much time restoration practices might buy, if, for example, habitat gained with reconnection of fragmented habitats through culvert removal were eventually lost to thermal barriers and shrinkage of local patches.

Even given the recognition that rapid climate change is altering ecosystems beyond a natural capacity to adjust, there is substantial contention about the use of active

management in adaptation (Spies and others 2010). Philosophical discussions of the goals of management intervention without the context of a specific physical landscape can be interesting, but a long literature search would suggest that they might be unproductive absent a rooting in a real landscape (e.g., Cissel and others 1999; DellaSala and Frost 2001; Rhodes and Baker 2008). Discussion of future management will necessarily be filled with details, and with some uncertainty. Some details will be about tradeoffs between goals ranging from maintaining specific genetic resources (e.g., subspecies), to maintaining ecosystem functions, to maintaining general biotic assemblages (e.g., trout or forests) (Rieman and others 2010). Some details will be about how particular goals are achieved, probably reflecting a general bias toward those that are least intrusive while still somewhat effective. Other details will be about the relative risks to different resources and about acceptable levels of risk. In the translation of goals to specific objectives in the landscape, it is useful to frame alternatives as where we can maintain or restore process, and where it is necessary to impose more control (Rieman and others 2010). Recognizing and understanding uncertainty should not be a barrier to action, but rather a basis to inform, refine, and revise action through monitoring or adaptive management.

Ultimately, solutions that satisfy this high-dimensional and uncertain decision space require information and creativity. At any given location, it will be necessary to understand what climate has been doing and where it might be going, and the uncertainties inherent in the forecast. It will be important to understand physical and biological sensitivities to the changes, some layered through indirect effects and feedbacks. Learning from careful monitoring of climate shifts and their effects may be the only source of information for some processes. It will also be necessary to have ideas about potential beneficial actions and the relative value of actions from place to place.

In this monograph, we describe occurring and predicted changes to components of the climate and ecosystem to increase the understanding of how ecosystem responses to disturbance may be changing. Where we can, we delve into the intersection of fire, forests, streams, and changing climate to discuss current questions and debates and the relative contexts and conditions that might shape decision frames. Throughout, we try to provide some background to assist readers in understanding the physics and biology sufficiently to follow the rapidly evolving science and to spur creativity in local problem solving. We emphasize recent science.

We begin with a brief review of how the physical system of terrestrial and aquatic habitat, including climate, hydrology, and geomorphology, is being affected by climate change and fire, and then we proceed to discuss ecological changes to the upland and riparian vegetation and aquatic systems. Throughout, there is the opportunity to contrast the relative and combined effects of changes driven directly by the climate and those related to fire. The changes to the physical environment and upslope/upstream terrestrial ecology set the stage for a discussion of how aquatic communities will feel pressure through multiple pathways. We close by framing important components of the discussion about analyzing vulnerability and building future management options.

Resilience

Resilience is one of the most commonly discussed ecological concepts with respect to fire and climate change. Based on such broad usage, one might suppose that it is a concretely defined term. There are, however, nuances that are sometimes unclear to new readers, and different connotations may appear in a single document. Though subtle, interesting concepts, underlie the distinctions, and they are worth exploring.



The etymology of resilience is pretty simple from the Latin, re- “back” and “salire” “to jump”. Engineers tend to focus on this aspect of the word in application to material properties. Ecologists have a broader usage as “the capacity to absorb” (Walker and Salt 2006; Gunderson and others 2010). This usage is fairly similar to psychological and sociological usage. Ecologists sometimes contrast resilience with resistance to emphasize potential for recovery, and in common usage, rigidity is taken as an antonym for resilience. Yet resistant attributes can be an important component of resilience. The general upshot is that resilience is a quality describing the ability to withstand the “slings and arrows of outrageous fortune.”

Mathematics is a terse and generally unequivocal language; so it seems like a potentially fruitful place to turn for tightening the definition. Indeed, ball and cup analogies illustrating phase portraits and flow diagrams of partial differential equations are favorites for mathematically inclined ecologists. In mathematical terms, the various meanings of resilience relate to the concept of “stability.” In the literature of mathematical dynamics, there are, by one account, 57 different definitions of stability (Glendinning, 1994), providing on the one hand a vast potential insight about various themes on resilience, and, on the other, too many shades of nuance to be practical for general discussion.

Analogies for resilience are fairly common, and perhaps more helpful. Engineers like to talk about

springs, while foresters seem to prefer bristlecone pines. These capture the “rebounding” and the “tough” aspects of the concept very well. Other apt analogies might be a swarm of gnats or a rock thrown into a pond. In these cases, there is no outward appearance of bouncing or resisting, but one would be hard pressed to say whether anything happened a few minutes after the disturbance.

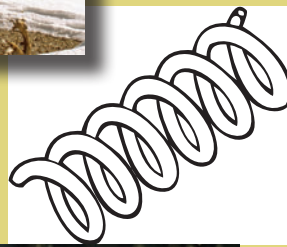
All of these analogies relate in some form to a variation on stability, and in terms of biological process, they illustrate the capacity of individuals or populations to heal after being harmed, or to avoid being harmed in the first place. This kind of resilience reflects the most common perception with respect to trees and fish after fire. This is also the conceptualization of resilience applied in

the development of forest and fishery resource management models that have been used to set “sustainable” harvest levels.

There is a need also to discuss the resilience of ecosystems more generally, e.g, how well ecosystems retain their resilience over time—a sort of resilience of resilience. This could also be framed as resilience of ecosystems to multiple interacting stressors. As the climate changes and as human developments proceed, some of the processes that provide an ability to avoid, absorb, resist, or recover from disturbance are changing. Some examples:

- Longer growing seasons at high elevations allow more fuels to grow and diseases and insects to be more effective, increasing risks for trees that historically survived through isolation;
- Fragmentations by roads has impinged on the reestablishment of fish populations;
- More frequent fires do not allow sufficient time between events for resistant species to attain sufficient size and species drawing on postfire reproduction to sufficiently mature;
- Forests or fisheries managed to maximize yields have demographics that are less able to survive major disruptions.

Although concrete definitions, synonyms, or even analogies seem elusive, the antonyms seem clear. A resilient system is not vulnerable or sensitive. Understanding the complex pathways of resilience in forest and stream ecosystems will help analyze vulnerability to future changes.



Part I: The Physical System

A. Climate

Patterns of air temperature and precipitation, the minimums and maximums, the seasonal patterns, and the correlation in timing between the two are critical elements of climate. Many biota have evolved some degree of specialization to particular temperature ranges or particular amounts of available water. Some biota trade specialization in extreme environments against open capacity for growth, while others take full advantage of mild and low variability climates. As the climate changes, the adaptations of various species and life forms will be tested. The increased number of large fires in recent decades across the western United States is explained, in part, by climate (Westerling and others 2006; Holden and others 2012). The National Wildfire Coordinating Group (2009) has predicted that the annual area burned across the United States will increase to 10-12 million acres/year (up considerably from the 10 year averages of 3.8 million acres in 1990s and 7.1 million acres 2000-2008) due to a combination of factors, including climate change. As changes in climate result in increased length of fire season (Running 2006; Westerling and others 2006) and increased tree mortality from bark beetles and drought (Breshears and others 2005), we will experience important positive feedbacks among fire, climate, and other disturbances with important implications for aquatic ecosystems.

Climate Change

Increasing concentrations of greenhouse gasses are causing the atmosphere to become warmer (see textbox on climate change mechanics). The changing heat balance of the earth is also changing atmospheric flow patterns and redistributing the wind streams that carry water vapor from oceans to land (Solomon and others 2007; Archer and Caldeira 2008; Fu and others 2010b), changing the precipitation. Temperature increases to date are already substantial compared to historical and paleoclimatic records and clearly tied to anthropogenic greenhouse gas emissions (Solomon and others 2007). Estimates of the future rate of change depends on the rate that greenhouse gases are added to the atmosphere, producing a number of estimates that depend on the economic/regulatory scenario (Figure 2, Table 1).

The changes are complex, and not all places will warm equally, nor will precipitation change in the same way or to the same degree everywhere. In the northern hemisphere, it is expected that the warming will be more pronounced in arctic and Antarctic regions (Figure 3), where more precipitation is also expected. The belt of deserts in the subtropics (25-35 degrees N latitude) will likely spread northward with expansion of the Hadley cells, a primary component of the earth's circulation. Beyond these generalities by latitude, the actual changes to any region depend on the relationship of the landmass to ocean currents. A combination of Global Circulation Models (GCMs) and historical analyses allow us to estimate what the future may bring to any given location. The implications for complex mountain terrain are poorly understood (Solomon and others 2007, Chapter 11).

Western North America is predicted to warm at rates comparable to global averages (Solomon and others 2007). For the A1B emission scenario, this is on the

Figure 2: Global temperature trends showing the 20th century rise in temperature and projections for continued global average temperature increases depending on alternative carbon dioxide emission scenarios for the future (Solomon and others 2007).

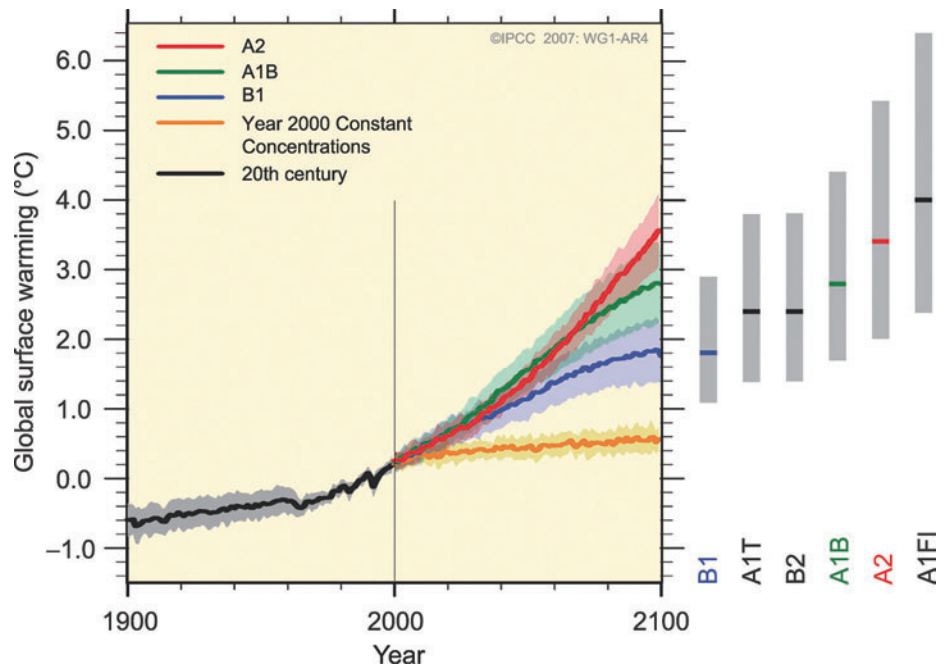


Table 1: Descriptions of the carbon emission scenarios used in IPCC reports from Special Report on Emission Scenarios (SRES: Nakićenović and Swart 2000).

Scenario	Storyline (from SRES)	Description	Cumulative Carbon Emission (Gt 1990-2100)
A1B	Rapid and successful economic development, in which regional average income per capita converge - current distinctions between “poor” and “rich” countries eventually dissolve. The scenario reflects a strong commitment to market-based solutions, high savings and commitment to education at the household level, high rates of investment and innovation in education, technology, and institutions at the national and international levels, and international mobility of people, ideas, and technology.	Initially fastest carbon emission growth rate with declining emissions starting by mid-21st century	1499
A2	Characterized by lower trade flows than A1B, relatively slow capital stock turnover, and slower technological change. Less emphasis on economic, social, and cultural interactions between regions are characteristic for this future, and economic growth is uneven and the income gap between now-industrialized and developing parts of the world does not narrow.	Accelerating carbon emission over 21st century	1862
B1	High level of environmental and social consciousness combined with a globally coherent approach to a more sustainable development.	Slowest CO ₂ emission growth, emissions declining by mid century and emissions below 2000 levels by 2100.	983

Climate Change Mechanics

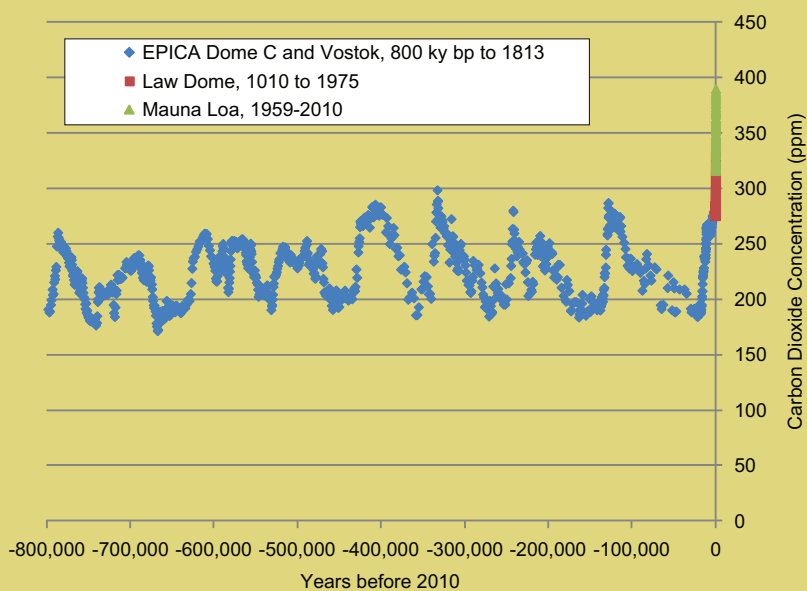
The energy balance of the earth is pretty simple. The sun shines on the earth, warming it, and the earth “shines” back into space. Because there is no other way to move heat into space, these two energy fluxes are nearly equal.

Most of the light from the sun is in the shortwave portion of the light spectrum, that is light we can see. Most of the light emitted by the earth is in the longwave portion of the spectrum, which is not visible to humans. Clouds and particles in the atmosphere do little to interrupt or reflect incoming solar radiation, but longwave radiation can be captured by greenhouse gases.

Most of the radiation emitted by the earth’s surface passes directly through the atmosphere into space. Fortunately for us, some is briefly captured by greenhouse gases in the atmosphere, which re-radiate it: half back down toward earth, and half continuing on to space. So our atmosphere acts a bit like a blanket with respect to radiant energy.

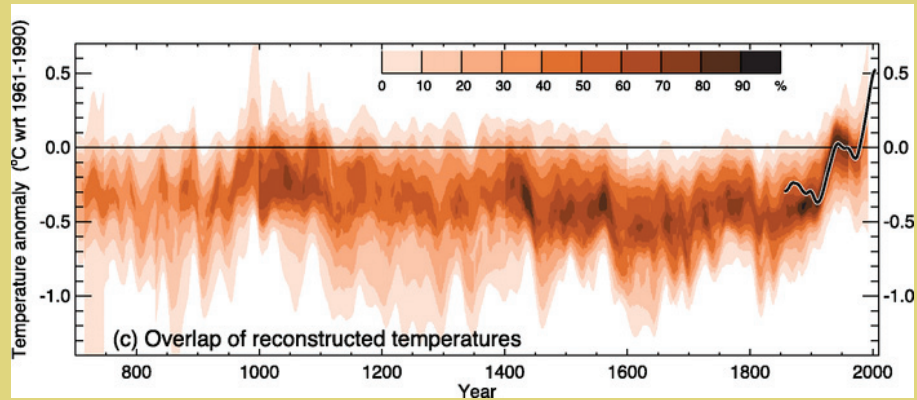
Humans have a substantial influence on the amount of CO₂ in the atmosphere, and it has been increasing over time. Although levels of CO₂ have naturally varied in the past, in part with solar cycles, levels now are well beyond any measured over the last few hundred thousand years, covering several solar cycles (Figure T2-1).

Figure T2-1: Composite of CO₂ data from 800 ky before present to 2010. Green: Mauna Loa data (Tans and Keeling 2011), Rust: Law Dome ice core data (Etheridge and others 1996), Blue: composited Vostok and Dome C ice core data (Petit and others 1999; Monnin and others 2001; Siegenthaler and others 2005; Lüthi and others 2008)



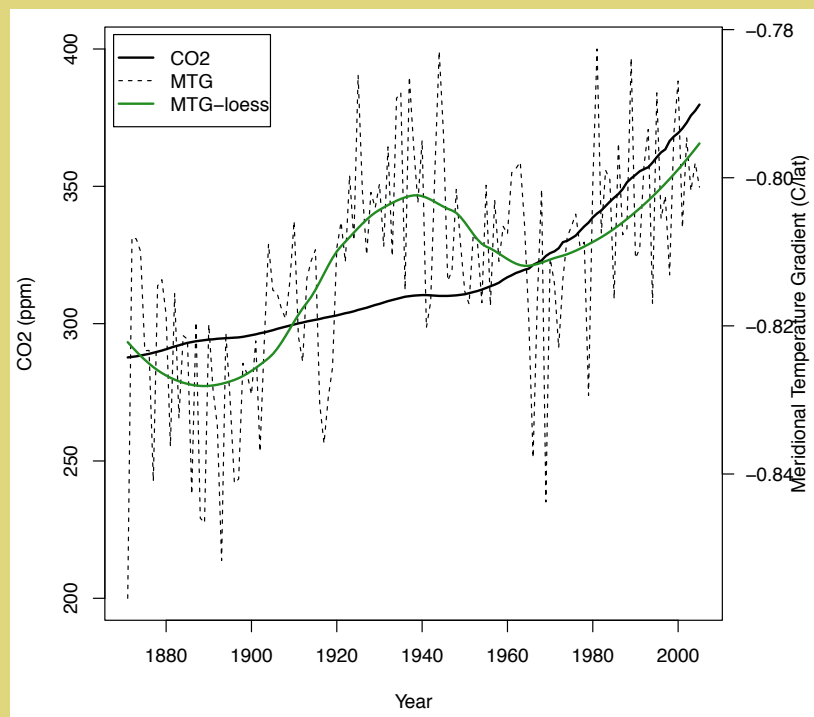
As greenhouse gas concentrations in the atmosphere increase, more of the longwave photons emitted by the earth’s surface are caught by greenhouse gas molecules, and about half of them are returned to the earth’s surface. The radiative forcing increase from CO₂ as of 2005 was about 1.5 W/m² (Solomon and others 2007), or a little less than 1 miniature tree light on every square meter. As a result, the earth is warming well beyond variations seen in proxy records we have (Mann and Jones 2003; Mann 2008; Figure T2-2).

Figure T2-2: Overlap of the published 1300-year northern hemisphere temperature reconstructions included in IPCC 4th Assessment with instrumental record shown as a black line for the last 150 yrs. Overlap is determined by the degree of agreement among the uncertainty ranges (± 1 standard error of estimate) of the 10 reconstructions (from Figure 6.10 of Solomon and others 2007).



There are differences in warming caused by increased greenhouse gases compared to increased solar radiation. The radiation blanket analogy might make it easier to explain. When you sleep in a cold room at night, it is usually your feet that get cold first, and adding a warmer blanket will warm your feet up. Contrast this to standing around a camp fire with cold feet; you usually need to actually expose your feet to the heat from the fire to warm them up. Increased solar radiation would be expected to warm places with lots of sunlight, like the tropics, more than places without much sunlight, near the poles. A CO₂ 'blanket,' though, warms the polar regions more than the tropics. What we have observed so far is that the poles have been warming more than the tropics, reducing the meridional temperature gradient (Gitelman and others 1997; Braganza and others 2004).

Figure T2-3: The meridional (equator to pole) temperature gradient over time (raw and smoothed) plotted with CO₂ concentrations. CO₂ data from Law Dome (Etheridge et al., 1996), MTG data from (Karamperidou et al., 2010). Note that the MTG axis is negative, and that the increasing trend represents a warming arctic compared to tropics.



Similarly, although you probably don't pay much attention to it, the top of a blanket in a cold room is actually still warmer than the room, because you are warming it from underneath. If you put on a thicker blanket, you, on the bottom side of the blanket, will feel warmer, while the top of the blanket gets closer to the cold room temperature. If instead of putting on a thicker blanket, you had the opportunity to let more sun shine on the blanket, both the top and the bottom of the blanket would warm. What we have observed so far is that the mean temperature of the earth's surface is warming while the stratosphere is cooling (Oort and Liu 1993; Golitsyn and others 1996; Guo and others 2008).

Figure 3: Multi-model mean surface temperature warming relative to 1980-1999 mean temperature (from Figure 10.8 in Solomon and others 2007)

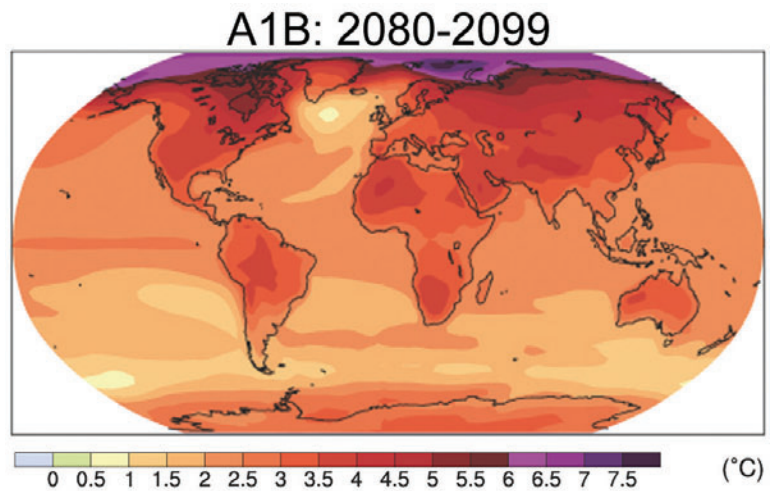
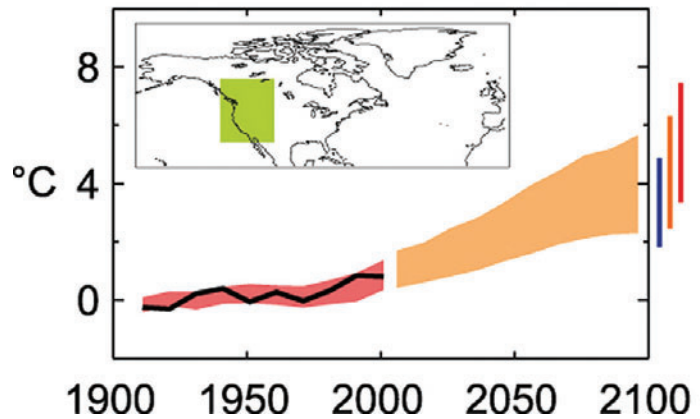


Figure 4: Temperature increases relative to 1901-1950 average temperature in western North America (from Figure 11.11 in Solomon and others 2007). Bars on right show ranges for B1 blue, A1B orange, and A2 red.



order of 2-6°C by 2100 (Figure 4). In contrast, the temperature has warmed about 0.7°C relative to a natural atmosphere over the 20th century (Solomon and others 2007). This seemingly modest amount of warming has been linked to many changes in the western United States, including plant phenology (Cayan and others 2001), snowpack reduction (Mote and others 2005; Pierce and others 2008), and earlier streamflows (Stewart and others 2005).

Projected changes in precipitation suggest strong declines in the Southwestern United States but seem less certain in the Northwestern United States (Figure 5). There is, in general, much less agreement between GCMs on precipitation than on temperature and pressure (Figure 6). Thus, even in areas where the sign of the change is not in question, there is still substantial disagreement about how large the change will be, with major implications regarding the magnitude of consequences (e.g., Barnett and Pierce 2008, 2009; Rajagopalan and others 2009).

Other predictions about future precipitation relate to variability and extreme values. One prediction is that precipitation events will be more intense when they occur (Trenberth 1993). This is derived from the slope of the saturation vapor pressure increasing with temperature, thus for a given change in temperature (e.g., from lifting over mountains), more water would be extracted from a given change in temperature. Another general prediction is increased variability in precipitation resulting from more variable storm tracks (Easterling and others 2000). Both results imply an increased flood risk from precipitation events (Easterling and

others 2000; Hamlet and Lettenmaier 2007). However, there is also recognition that increased dryness or length of dry spells could also have significant ecological consequences (Easterling and others 2000; Dale and others 2001; Westerling and others 2006; Holden and others 2012).

Variations in precipitation are influenced by sea surface temperature anomalies, such as ENSO, PDO, PNA, NAO, and AMO (e.g., Dettinger and others 1998; Cayan and others 1999; Clark and others 2001; McCabe and others 2004; Abatzoglou 2011) that influence patterns of global air pressure and therefore circulation of air masses. These phenomena operate at frequencies of one cycle every few years (ENSO range) to decadal or multi-decadal cycles (NAO and PDO ranges). Reconstructions of long-term streamflow (which relate to long-term patterns in precipitation) show significantly more variability over deep time compared to current variations (Figure 7) than do the marked shifts in temperature in recent decades (Figure T2-2). As a consequence of the strong natural variability, it is not as easy to discern the effects of anthropogenic climate change on precipitation and drought as it is to discern (and attribute) the effects of climate change on temperature (Easterling and others 2000).

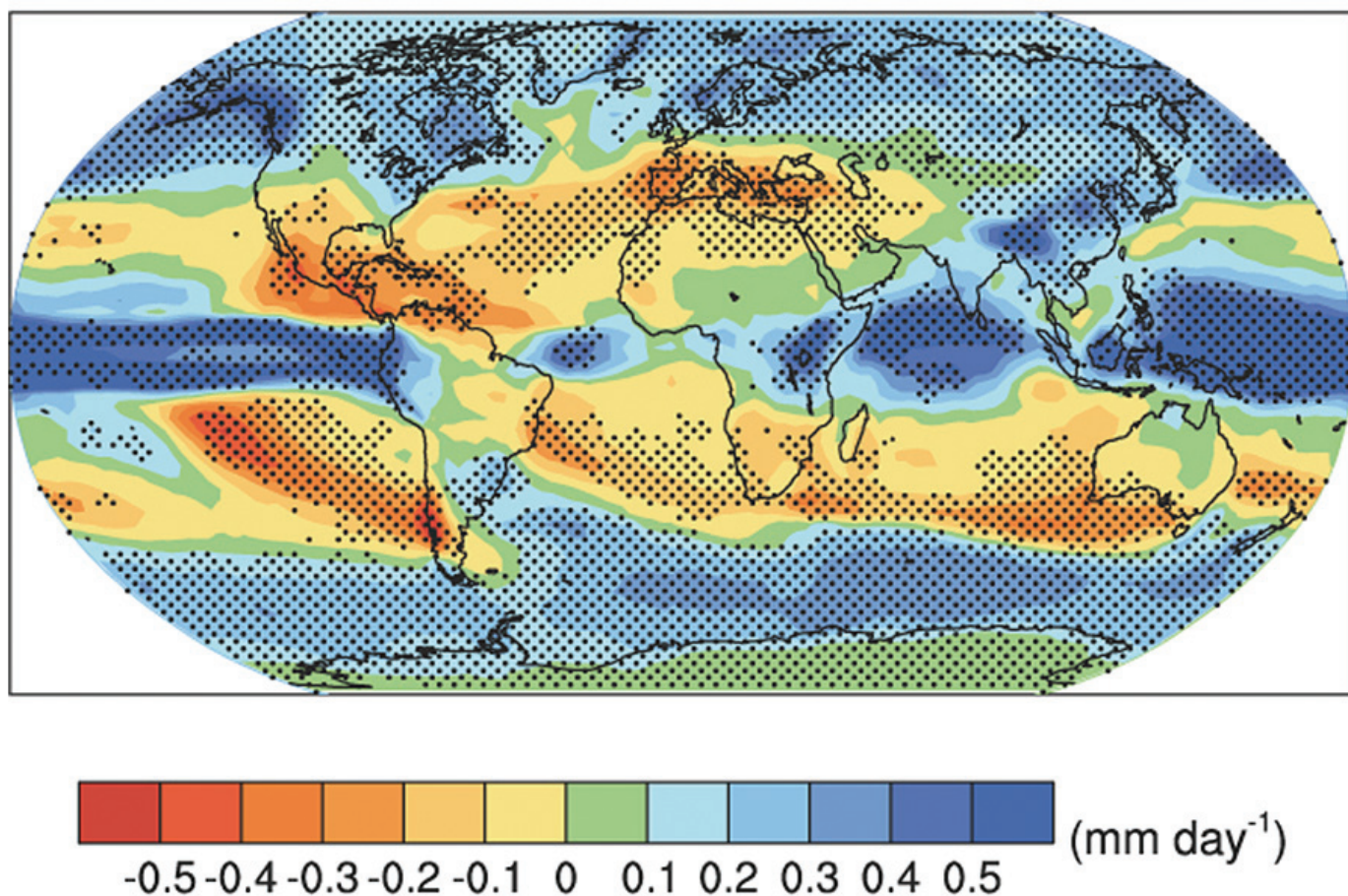


Figure 5: Precipitation anomalies (multi-model mean, A1B, 2080-2099 relative to 1980-1999). Stippling indicates areas where at least 80% of the models agree on the sign of the change.

Figure 6: Relative skill of general circulation models in predicting different climate variables over Australia (Johnson and Sharma 2009). Skill was indexed based on degree of agreement among models.

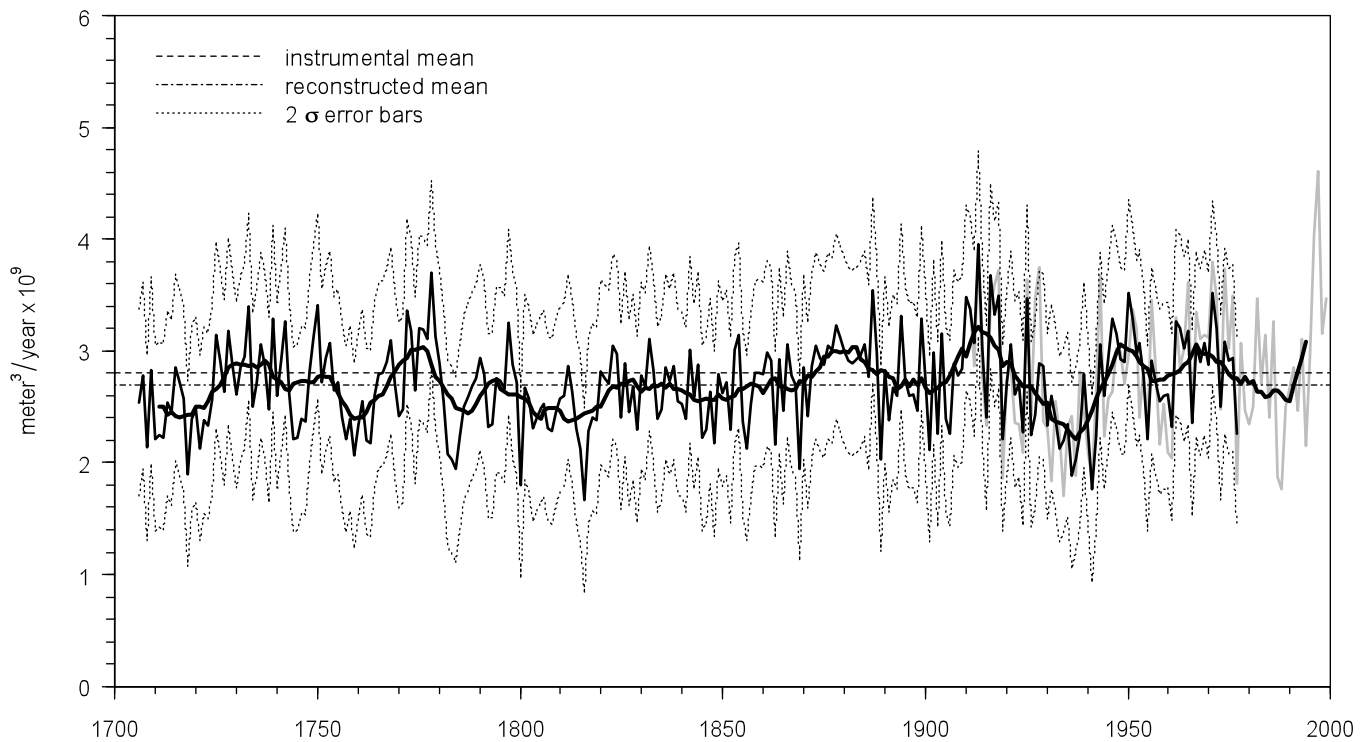
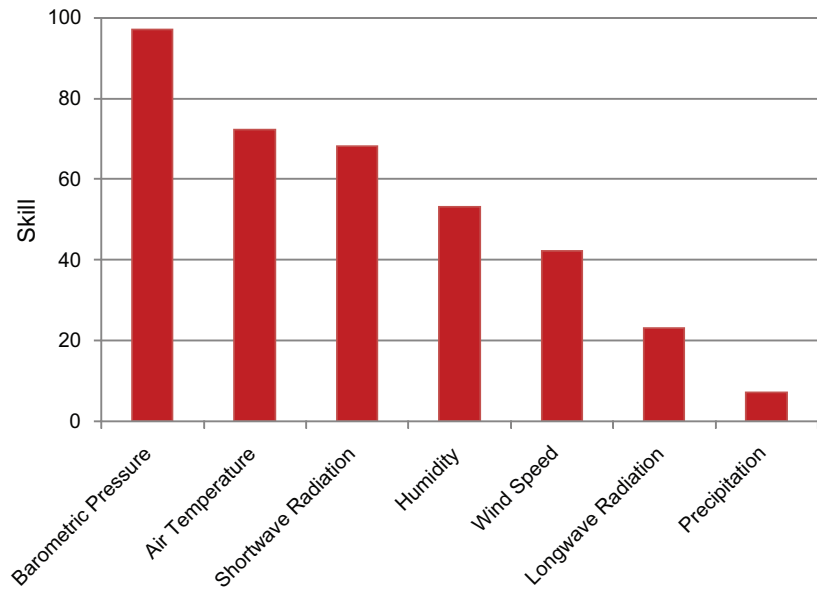


Figure 7: Streamflow reconstruction of annual flow volume of the Yellowstone River based on Tree Ring widths. Annual flows are in a solid black line bounded by dashed lines at 2-standard errors and with a 10-yr moving average smooth in a heavier black line. Reproduced from Graumlich et al. (2003) using data at ftp://ftp.ncdc.noaa.gov/pub/data/paleo/treering/reconstructions/montana/yellowstone_flow.txt.

Fire

Fire effects on climate tend to be either very local or global (note: the effects of climate on fire are discussed in sections II-B and III-A). The most important local change post-fire in forests is the loss of above-ground canopy cover, and this is seldom 100%. Vegetation cover may increase relatively quickly following less severe fires, but until it does, there may be less shading leading to increased

daytime temperature near the ground and in streams. The change in evapotranspiration rates and canopy interception of precipitation can alter soil moisture and therefore stream recharge (see Hydrology section below). While common wisdom would suggest that the forest canopy would buffer against radiative cooling of the ground in winter, the canopy still experiences radiative cooling, and the resulting cold air flows down below the canopy, where it may keep conditions cooler than the general atmospheric temperatures. Precipitation intensity increases have been noted over deforested areas in the Amazon (Chagnon and Bras 2005) related to the decreased albedo; however, circulation patterns in the tropics differ substantially from those over North America, with lower horizontal wind speeds aloft, and similar processes have not been examined in temperate latitudes. Globally, the particulates from smoke from forest fires can influence atmospheric processes (Fromm and Servranckx 2003).

B. Hydrology

The response of stream and forest ecosystems to shifts in climate will be mediated through the changes in hydrology. An overarching issue in much of the western United States is simply the availability of water. The Interior West is a dry place, and even the wetter portions of the western United States are dry in the summer. Minor changes in the water balance or timing can have more exaggerated effects on biota because of the competition for this valuable resource. The influence of water on disturbance regimes, such as insects, disease, fire, or flood, is another important linkage.

Wildfire is, itself, often an outcome of reduced water availability to forests, and it provides an important feedback to the hydrologic system, with the potential both to ameliorate and to exacerbate changes already occurring in the climate system. An important question is the relative contribution of wildfire to hydrologic changes locally and at the basin scale. A key aspect of the discussion on climate change is how much hydrologic change results from climatic change versus how much change results from land use and land cover shifts. Because there is a large legacy of research on the hydrologic effects of land use and land cover changes, this provides some leverage for understanding the potential effects of climate-induced effects. There is also the issue of cumulative effects through multiple pathways, such as the combined effects of fire and climate change together on water yield or flooding.

Climate Change

Hydrologic changes in the western United States in recent decades include both changes to timing of streamflow and the water balance. There are linkages between the two, in that changes in precipitation can cause changes to the timing of streamflow (Luce and Holden 2009). The principle changes attributable to anthropogenic warming are changes to snowpacks (Pierce and others 2008), and include reduced precipitation as snow compared to rain (SWE/P) (Knowles and others 2006), reduced snowpack on April 1st (Mote and others 2005; Regonda and others 2005), and earlier runoff timing (Stewart and others 2005). Changes to precipitation related to climate change are expected in the Southwestern United States in the coming century caused by spreading Hadley cells (Seager and al. 2007; Johanson and Fu 2009), but changes in the Northwestern United States are uncertain, leaving attribution difficult (Easterling and others 2000). Historical changes

show increases in Southwestern streamflows (Regonda and others 2005) and declines in the Northwestern United States (Luce and Holden 2009); such changes are partially consistent with general expectations for climate cycles (Dettinger and others 1998). Dry-year streamflow is better correlated with time, however, than with indices of low frequency variability (e.g., PDO), leaving questions about the relative contribution of cycles versus monotonic climate change (Luce and Holden 2009). The land use contribution to changes in streamflow are likewise a source of uncertainty as are the changes caused by increased potential evapotranspiration (Hoerling and Eischeid 2007).

Precipitation is the largest term in the terrestrial water balance, and any incoming precipitation can be partitioned into either evapotranspiration or runoff (through surface, near surface, or deeper groundwater). Most precipitation in the western United States falls in fall and winter, leaving a dry summer. Most of the precipitation in the western United States also falls in mountains. It is not surprising then that about 75% of runoff in the western United States is currently derived from precipitation that falls as snow (Service, 2004), and equally unsurprising that concerns about snowpack changes are among the most important in the western United States (Barnett and others 2005).

Snowpacks in many parts of the western United States are sensitive to variations in temperature (Mote and others 2005; Regonda and others 2005), and therefore to anthropogenic increases in greenhouse gases (Pierce and others 2008). Declines in snow water equivalent and earlier melt dates over the last half century also have a relationship to trends in precipitation and runoff, for which the connection to greenhouse gas concentrations is more uncertain within GCMs. Regardless of cause, the primary trend over the last 60 years has been for less snow in the mountains of the western United States (Barnett and others 2008). As a result, the spring freshet has become both shorter and smaller (less volume). Because these trends are partially related to temperature, which is projected to continue increasing, the expectation is that they will continue. Most of the western United States has dry summers, and the earlier and smaller spring runoff predicted for the future will hold important implications for both biota and farmers.

Warming temperatures cause less precipitation to fall as snow and more to fall as rain (Knowles and others 2006). Conceptually we expect to see higher snowline elevations for individual storms (Casola and others 2009). This means that some fall and winter storms that historically produced more snow will now produce runoff, shifting some mountain streams from snowmelt-dominated hydrographs, with peak runoff in the spring, to rain-dominated or transitional hydrographs, where the timing of flows is more related to the timing of precipitation (Stewart and others 2005). In the western United States, that means more streamflow in fall and winter and, consequently, less in the spring and summer (Figure 8). Such changes will happen soonest at mid-elevation sites, above already rain-dominated streams but below places where winter temperatures will remain cold enough for snow for some time (Regonda and others 2005; Pierce and others 2008; Nayak and others 2010).

Temperature-related shifts in timing also have implications for flood and flood effects on biota. Since fall and winter are the main precipitation seasons, a shift from snow to rain means that the likelihood of floods in late fall and winter could increase, with consequences for fall-spawning fish (Wenger and others 2011b). Floods are likely to increase in magnitude in many basins as well, both because of the increased occurrence of rain-on-snow events (Lettenmaier and Gan 1990; Hamlet and Lettenmaier 2007) in currently spring-snowmelt-dominated basins and

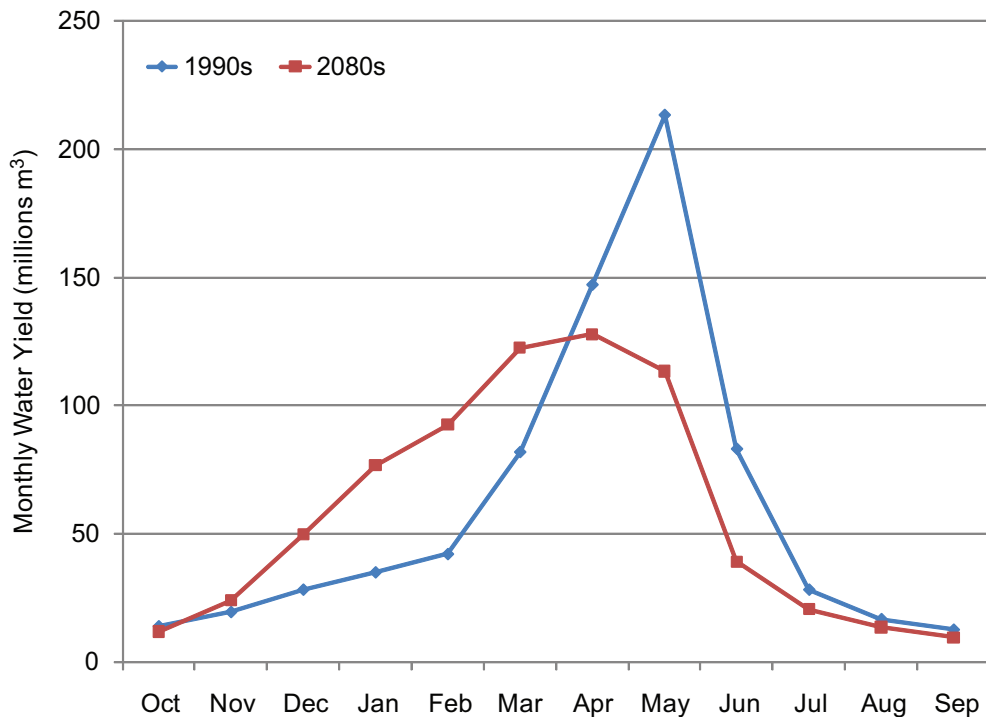


Figure 8: Average annual hydrographs for the Middle Fork Boise R. simulated using the VIC model based on historical (1990s) climate compared to the projected climate of the 2080s under an A1B scenario. Data derived from VIC runs done by University of Washington Climate Impacts Group and U.S. Forest Service Rocky Mountain Research Station (Elsner and others 2010; Wenger and others 2010).

because of increasing precipitation intensity in rain-dominated basins (Easterling and others 2000).

An important discussion is evolving in the Pacific Northwest about the roles and causes of changing precipitation in the regional hydroclimate. Earlier work suggested a lack of trend in flows in the western United States (e.g., Mote and others 2005; Regonda and others 2005) in part due to underestimating the statistical importance of increased variance over time. More recent work has identified regional trends of declining streamflows over the last half century (Moore and others 2007; Luce and Holden 2009; Clark 2010; Fu and others 2010a), particularly in the Northwestern United States and with a more pronounced decline in runoff in drier years (Figure 9). This is an example of the principle that both means and variances are shifting, increasing the likelihood of some rare events (Jentsch and others 2007).

An important question is whether the changes are precipitation or transpiration related. Some hypothesize that trends in mountain streamflow are related to precipitation (Luce and Holden 2009; Clark 2010; Fu and others 2010a), others hypothesize warmer temperatures are increasing evapotranspiration (Hoerling and Eischeid 2007), and still others suggest changes to land use and land cover (Wang and Hejazi 2011). This raises speculation, for example, that decreases in streamflow have been caused by increased forest cover due to fire suppression. Examination of the Historical Climatology Network of weather stations suggests no trend in precipitation (Mote and others 2005), supporting a stronger focus on temperature related changes to hydroclimatology, which are more easily tied to anthropogenic greenhouse gas increases. There is, however, some question as to whether the precipitation gage network represents precipitation trends in the mountains because the gages in the network are primarily at lower elevations (Mote and others 2005).

JOHNSON CR AT YELLOW PINE ID

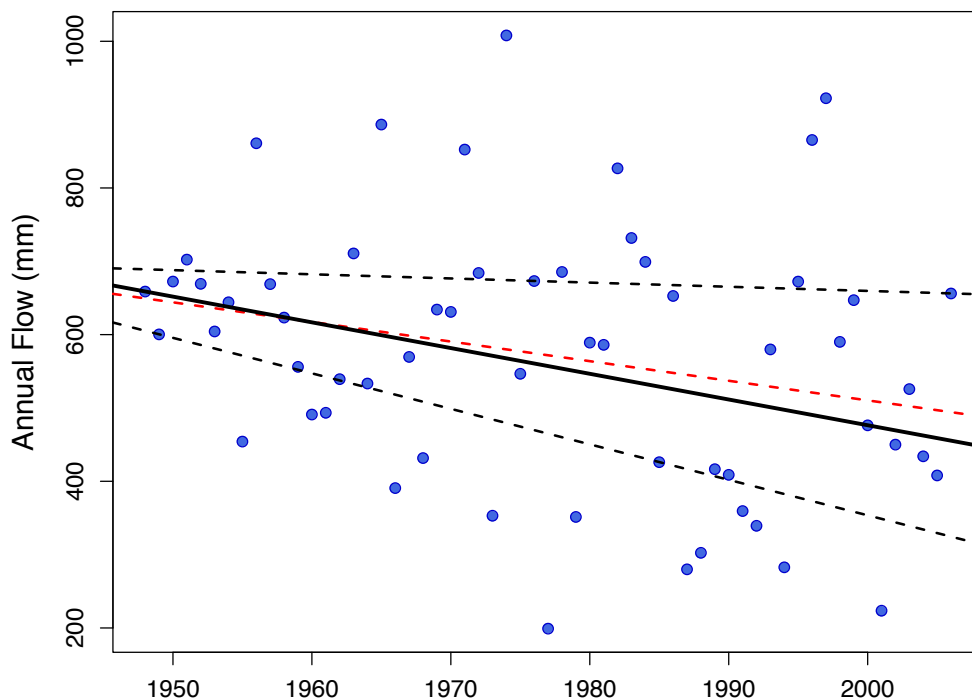


Figure 9: Trends in streamflow means and quantiles 1948-2006 for Johnson Creek at Yellow Pine, Idaho (USGS gage sta. 13313000). The dashed red line is the trend in the mean annual flow (24% decline, $P=0.049$), solid black line is the trend in the median (31% decline, $P=0.025$), the lower dashed black line is the 25th percentile flow (1 in 4-year low annual flow; 47% decline, $P=0.01$), and the top dashed black line is the 75th percentile flow (1 in 4-year high annual flow; 5% decline, $P=0.82$). This is a fairly common pattern in the Northwest, with dry years increasingly dry and wet years about as wet as they have been.

An important concept differentiating among alternative causes is that increased demand for water from either increased forest cover or warmer temperatures could not be satisfied if there is insufficient water. Essentially, this describes the distinction between potential and actual evaporation. Several have noted that changes in forest cover alter water yield primarily in wet years (Troendle and King 1987; Zhang and others 2001; Ford and others 2011). The findings of decreasing trends, particularly in the driest years and not in the wettest years, would not support the hypothesis that observed streamflow changes are caused by increased evaporative demand, leaving precipitation as the most likely driver.

This leaves the question of whether such precipitation changes are simply part of the regional climate cycles or can be attributed to increases in anthropogenic greenhouse gases. Causal linkages between decreasing streamflow and precipitation in some regions and anthropogenic climate change are not as easily identified as they are for temperature because observed variations in precipitation and streamflow are within bounds of historical and estimated paleoclimatic variation. Unfortunately, GCMs are notoriously poor at predicting precipitation (Johnson and Sharma 2009), so they are difficult to apply in formal attribution studies to discern the relative contributions of natural and anthropogenic changes on precipitation. Empirical statistical analysis supports primarily an anthropogenic contribution for the very low frequency (sometimes called *secular*) component of the trend with climate cycles playing an important role in variations over the course of a few years (Luce and Holden 2009). There is also theoretical support that the changes

are tied to anthropogenic changes; the decreasing meridional temperature gradient and ocean-land temperature contrast discussed in the earlier textbox (Gitelman and others 1997; Braganza and others 2004) would both predict decreases in precipitation for this region. Decreasing meridional temperature gradients reduce the *baroclinicity*, or storminess, at midlatitudes, which is reflected in the storm record (McCabe and others 2001). The faster warming of the land compared to the ocean means that a water vapor content that is in balance with the ocean temperature would result in reduced relative humidity over the more rapidly warming land surface, reducing orographic precipitation in mountains (Simmons and others 2010).

Even if precipitation-related changes in snowpack have not resulted from anthropogenic climate change and have a connection to natural climate cycles, they are still important influences on overall snowpack patterns observed in the last half-century and have played a dominant role in higher elevation snowpacks (Regonda and others 2005; Moore and others 2007; Luce and Holden 2009). The date that snow completely melts off of a site is earlier for shallow snowpacks than deep snowpacks, all else being equal. It is less well understood that the timing of melt for shallow snowpacks is more sensitive to the amount of accumulation than for deep snowpacks, resulting in a non-linear relationship between timing and snowpack accumulation, in turn creating a non-linear relationship between total annual streamflow and the timing of streamflow from high elevation basins (Figure 10). Concave downward relationships can yield statistically significant changes in flow timing related only to changes in total flow, without any effects from temperature (Luce and Holden 2009). While some shifts in timing are occurring because there

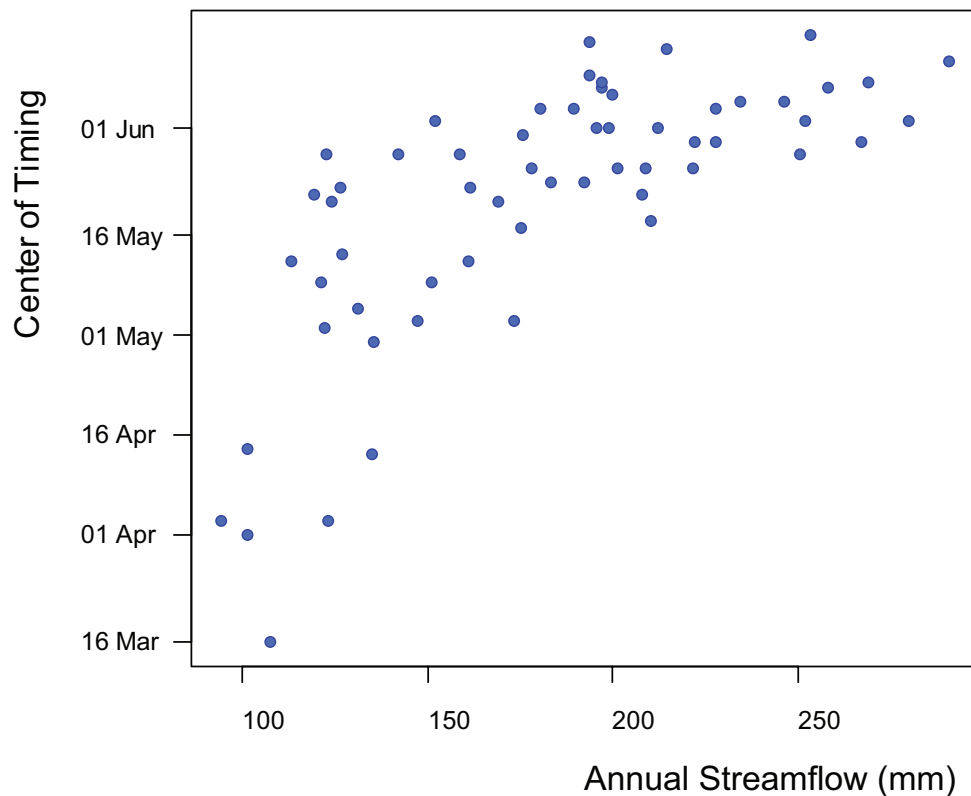


Figure 10: Relationship of snowmelt timing to total annual streamflow. Because sun angles are higher later in the spring, the last inch of snow can take weeks to melt if exposed by March but can melt in a few days if not exposed until June. The concave downward relationship has implications for causes of observed trends in streamflow timing (Luce and Holden 2009).

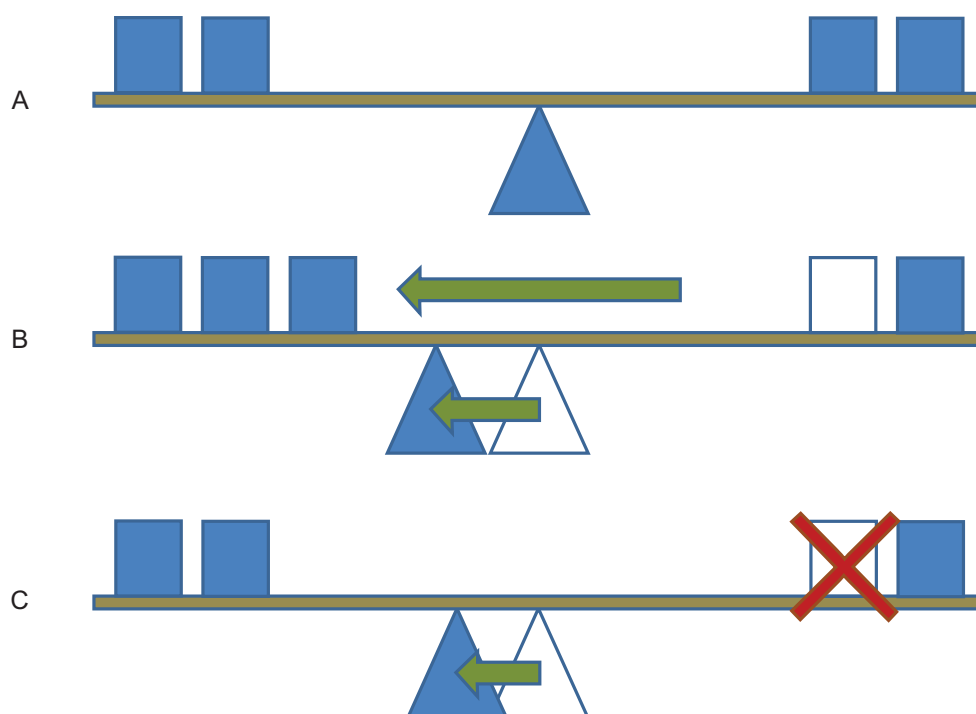


Figure 11: The flow timing seesaw. If the blue boxes represent buckets of runoff from a watershed, the center of timing for streamflow (blue triangle) is the balance point for when those buckets runoff over the year (A). If some of the flow starts coming off sooner because of earlier melt or falling as rain, the center of timing shifts earlier (B). If some of the flow in the summer is lost altogether, such as if precipitation is declining in a snowmelt dominated system, the center of timing also shifts earlier (C). Experiments that only measure the center of timing cannot distinguish between cause (B) and cause (C). Information is also needed about trends in flow or precipitation.

is more rain and less snow, others are occurring simply because there is less snow (Figure 11).

Summer streamflow provides habitat for fish rearing, carries food downstream to fish, and helps maintain cool stream temperatures. It is also related to soil moisture during the summer growing season for forests. Summer flows are mostly correlated to annual flows, and many streams are showing declines in summer flows (e.g., Luce and Holden 2009; Leppi and others 2011). Besides the evapotranspiration hypotheses already discussed, an additional mechanism suggested for declining summer flows and soil moisture is the earlier melting of the snowpack caused by warmer temperatures discussed above (e.g., Mote and others 2005; Stewart and others 2005; Westerling and others 2006; Barnett and others 2008). Because both temperature and precipitation are changing, both are contributing to the effect with different contributions in different places. Higher elevation basins, for instance, may be primarily responding to precipitation variability, while lower elevation basins in the Cascades may be responding more to temperature changes (Mote and others 2005).

With increasing variability in streamflow between years and lower low streamflows, the geologic context of streams may increase in importance. Groundwater-dominated systems such as provided by karst or recent volcanic geologies buffer short-term variations in streamflow driven by climatic variations, though at some cost through increased sensitivity to dry spells lasting several years (Lall and Mann 1995; Shun and Duffy 1999). Tague and Grant (2009) noted

an ironic exception where timing shifts in snowmelt may yield greater absolute changes in low summer streamflow in deep groundwater-fed systems primarily due to the fact that shallow groundwater systems are already nearly dry in late summer.

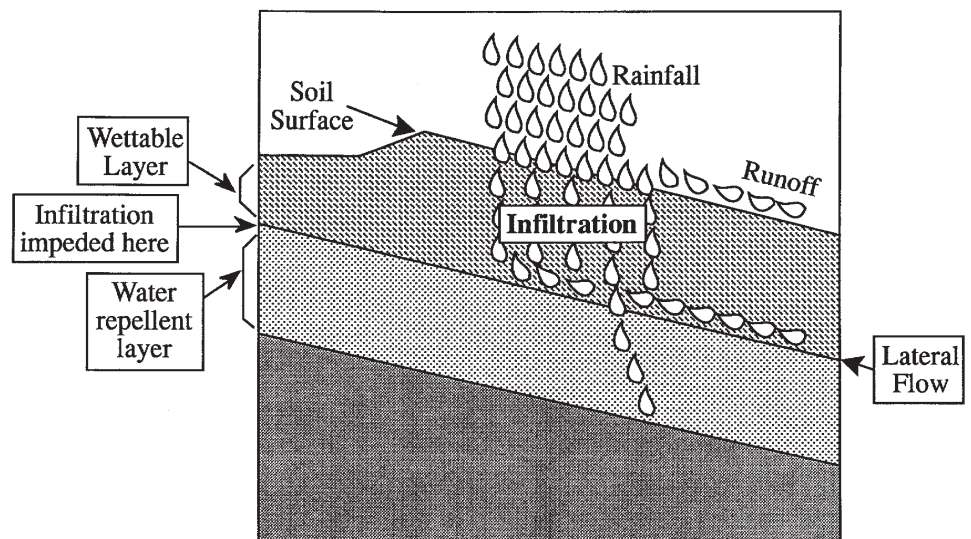
Fire

Hydrologic changes induced by fire are generally seen as somewhat more “spectacular” than the changes driven more directly by climate change. Extensive rilling, gullies, and debris flows related to post-fire runoff from water-repellent soils, for example, are sometimes dramatic after wildfire (Klock and Helvey 1976; DeBano 1981; Doerr and others 2000; Cannon and others 2001; Istanbulluoglu and others 2002; Neary and others 2003; Shakesby and Doerr 2006; Moody and Martin 2009). There are other more subtle changes, however, including changes to snowmelt, water yield, and low flows. As a result, peak flows in streams may be 200 to 450 times higher post-fire than pre-fire, though it is more frequently reported that post-fire peak flow is less than 10 times that of peak flow pre-fire (Shakesby and Doerr 2006). In some smaller basins, peak flows bulked with debris have destroyed gage sites and not been recorded (e.g., Woodsmith and others 2004)

Runoff changes after wildfire have primarily been attributed to changes in soil properties. Many have studied the formation of water repellency after fire (see, for example, Shakesby and Doerr 2006 for review). Surface sealing has been suggested as another mechanism for increased runoff after fire (Rowe 1948; Swanson 1981; Benavides-Solorio and MacDonald 2001; Meyer and Pierce 2003).

Post-fire water repellency typically occurs in a shallowly buried layer of soil and prevents infiltration of water through that layer where it occurs (DeBano 1981) (Figures 12 and 13). The layer is hypothesized to be formed when waxy substances in accumulated leaf and needle litter are volatilized by fire and recondense on cooler soils particles deeper in the soil (DeBano 1981; Doerr and others 2007). Because of the dependence on substances found in vegetative litter, it does not occur everywhere, but seems to be most commonly associated with particular vegetation communities, including (but by no means limited to) chaparral, eucalyptus, and subalpine fir, and are more likely where fires burn severely. It is also most

Figure 12: Schematic of water repellency effects on infiltration and runoff generation after fire. The water repellent layer impedes infiltration leading to saturation of the wettable layer and runoff over the surface (after DeBano 1969).



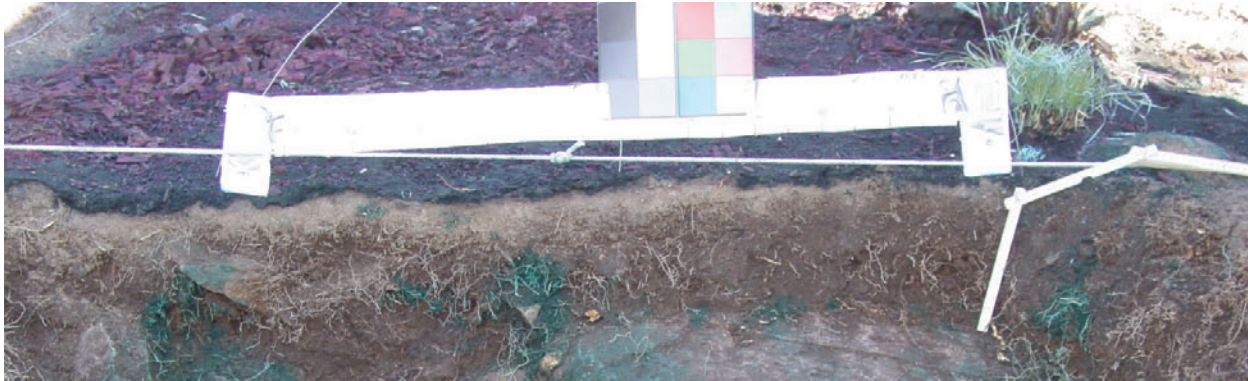


Figure 13: Photograph of wetted layer over a water repellent layer after rainfall simulation with blue dye showing substantial similarity to the schematic by DeBano (1969). The scale bar is 1-m between the legs.

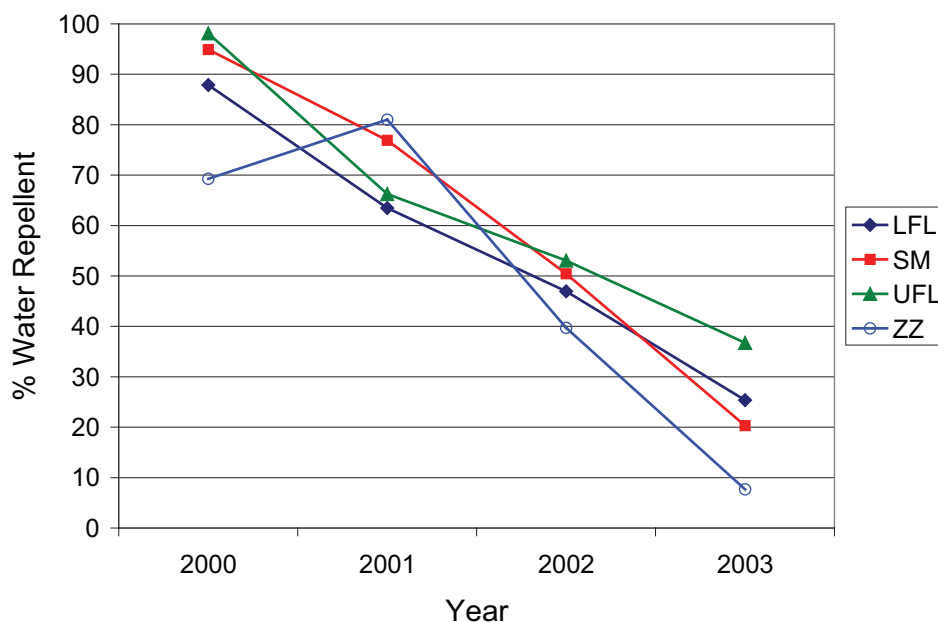
common on coarse-textured soils. Although water repellent chemicals (mostly fatty acids associated with plants) are present on soil particles prior to fire, and fires consume much of that water repellent material, water repellent areas may have as little as 1-4% of the original infiltration capacity after severe fires (DeBano 1981; Doerr and others 2000). Potentially the volatilization and recondensation of the chemicals has an annealing effect.

Water repellency is sensitive to the soil moisture state, and soils with water repellent substances generally repel water when dry (Doerr and others 2007). Water repellent soils are actually wettable, but only very slowly through vapor diffusion processes. As a consequence, soils rarely display water repellency in the wetter parts of the winter and spring, because soil moistures are maintained by frequent precipitation and melt. Dry and hot summers associated with Mediterranean climates of the western United States are ideal for bringing out water repellent behaviors.

Besides the annual disappearance and reappearance of water repellency with wetting and drying, there are longer-term patterns. The most frequently cited is a study by Dyrness (1976), showing some repellency remaining 6 years after a fire. Based on the sampling done in that study, the remaining repellency represents less than 1/8th of the area, however, which is an important consideration for broader-scale effects (Shakesby and Doerr 2006). Sampling done after fires near Boise, Idaho, and Bozeman, Montana, showed a relatively rapid decline from nearly 90% water repellency in severely burned areas to less than 50% within 3 years (Figure 14). Consideration of the hydraulic conductivity (Megahan and Clayton 1986) and precipitation intensity characteristics for the area would suggest that in excess of a 20-year precipitation event would be necessary to generate runoff after three years. This relates to the general observation that significant runoff and erosion events typically occur within 1-2 years of fire (Robichaud and Brown 1999; Shakesby and Doerr 2006).

Soil sealing is caused by the disaggregation of soil particles by raindrops, due primarily to high-energy raindrops falling on friable soils that are unprotected by vegetation or organic matter layers. Finer particles at the soil surface impede infiltration, and in extreme cases may form a crust. Much of the work on soil surface sealing has been done with agricultural and other severely disturbed soils (Mohammed and Kohl 1987; Bosch and Onstad 1988; Luce 1997), but similar behaviors have also been seen on burned soils under intense rainfall (Larsen and others 2009). Because soil particle aggregate stability is increased by organic matter (Kemper and Koch 1966), losses of organic matter through heating and increased post-fire decomposition rates may be an important contributor to the vulnerability of soils after fire.

Figure 14: Declining fractional water repellent area over time following fires near Boise, Idaho, and Bozeman, Montana. The four lines are four separate burned areas (unpublished data).



Decreases in hydraulic conductivity related to surface sealing seem to be less severe than those associated with water repellency. For example, Larsen and others (2009) measured about a 50% decline in infiltration rate by applying intense rainfall to recently burned soils in Colorado. Nonetheless, in locations where water repellency is not as prevalent, and precipitation is intense, even minor reductions in soil infiltration can have severe consequences for runoff generation.

Peak runoff rates after fire are generally tied to intense precipitation events, such as convective storms (thunderstorms). Water repellency is less prevalent in winter and spring when soils are wet, thus the timing of the most severe repellency coincides with the timing of convective storms. Soil sealing requires high raindrop energy to disperse soil aggregates, and infiltration rates after sealing are still much greater than snowmelt rates, thus soil sealing is also more important under summer storms. The consequence of the tie to convective storms is that runoff effects from post-fire events tend to be localized. In a database of 600 severe post-fire flood and related events in the western United States, the largest basin with a reported event was 122 km², and 99% of the basins were less than 25 km² (Gartner and others 2005). After the Tillamook burn in the Oregon Coast Range in 1933, Anderson and others (1976) estimated a 45% increase in the peak flow of two basins close to 400 km² in size the first year after the fire. In the Boise River, no increase in peak flow was noted at the 2,000-km² scale despite measurable changes to water yield and some dramatic events in basins up to 20 km² in extent (see textboxes on water yield increases after fire and debris flow scale in the Boise River Watershed).

Snowmelt changes after wildfire are important as well. Changes in snowmelt rate relate to the increased exposure of the snowpack to solar radiation and wind where vegetation cover post-fire is reduced. Increases in solar radiation post-fire have been linked to advances in the timing of snowmelt by 1 to 2 weeks, but not to increases in snowmelt-related peak flows in high elevation areas (Megahan and others 1995; Troendle and others 2010; also see textbox on Boise River). However at a lower elevation site, changes in the soil water balance and increases in accumulated snow combined with rapid melt during rain-on-snow increased peak flows and caused debris torrents in burned and salvage-logged basins (Klock and Helvey 1976). Turbulent transfer of heat from warm air can dramatically increase snowmelt rates and can be increased by forest harvest (Harr 1986). While the protective influence of

Water Yield Increases After Fire in the Boise River Watershed

It can generally be said that when trees are removed from the landscape, runoff increases (Zhang and others 2001; Andréassian 2004; Brown and others 2005). There are, however, questions about whether water yield increases realized in small experimental basins (typically less than 10 km²) translate into increases from large basins on the order of a few thousand km² (Troendle 1983; Troendle and others 2010). There are also questions about whether the larger fires that have been occurring in recent decades will translate to greater risks of flooding in large basins post-fire.

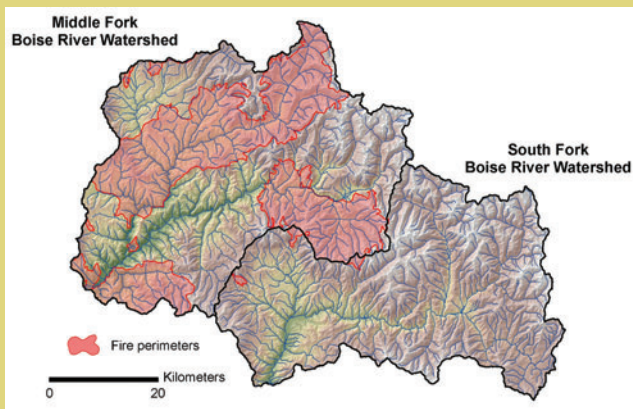


Figure T3-1: Middle and South Fork Boise watersheds with fire perimeters as of 2006.

Table T3-1: Area and elevations of the two gages.

	Area (km ²)	Gauge Elev. (m)	Mean Elev. (m)
South F.	1645	1286	2085
Middle F.	2150	993	1936

Table T3-2: Fire events in the Middle Fork Boise River basin, 1992-2003

Year	Acres in MF	% Basin
1992	30,000	6%
1994	152,000	29%
2000	30,000	6%
2003	23,000	4%

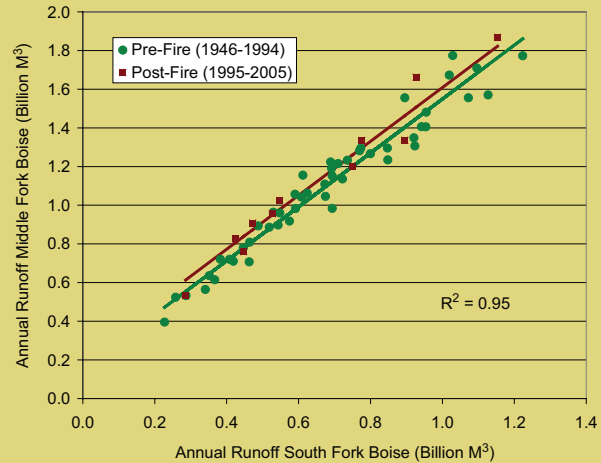


Figure T3-2: Paired watershed analysis regression for annual water yield.

A principle obstacle to researching the question has been the lack of ability to manipulate vegetation experimentally over large fractions of a major river basin. A series of fire events in the Boise River basin, between 1992 and 2003, however provide an opportunity to examine streamflow changes from a large basin (Figure T3-1). The middle fork of the Boise River has been gauged since 1912, while the South Fork was gaged in 1946. The two adjacent basins have similar size and elevations, making them likely candidates for paired watershed analysis (Table T3-1). Several fires occurred in the middle fork basin between 1992 and 2003, the largest in 1994, burning about 45% of the basin area (Table T3-2), while the South Fork saw little disturbance. Both basins had some historical logging and other disturbances, but nothing on a scale to match the coverage of the fires.

The Middle Fork experienced a post-fire increase in water yield of about 5%, which translates to about 50,000 acre-feet of water annually (Figure T3-2). Despite severe changes in peak flows in some severely burned small basins (Figure T3-3) there was no increase in peak flows at the basin scale (Figure T3-4). The seasonal distribution of the flow increases was primarily in winter and spring, although late summer low flows also increased a little (Figure T3-5). There was a decrease in early summer flows related to advancing the hydrograph because of faster melt caused by increased solar radiation. Between 1948 and 2006, mean annual Middle Fork runoff decreased 18% (Luce and Holden 2009), and the increases associated with the wildfire were small in comparison.



Figure T3-3: Steel Creek debris flood following the Hot Creek Fire.

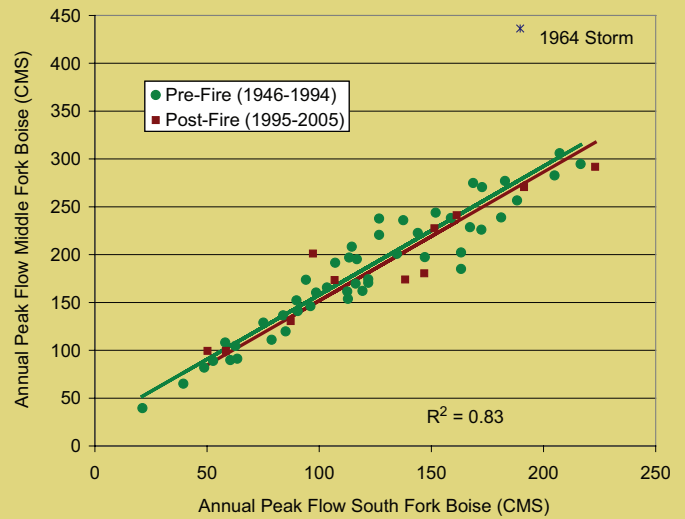


Figure T3-4: Paired watershed analysis regression for annual peak flows.

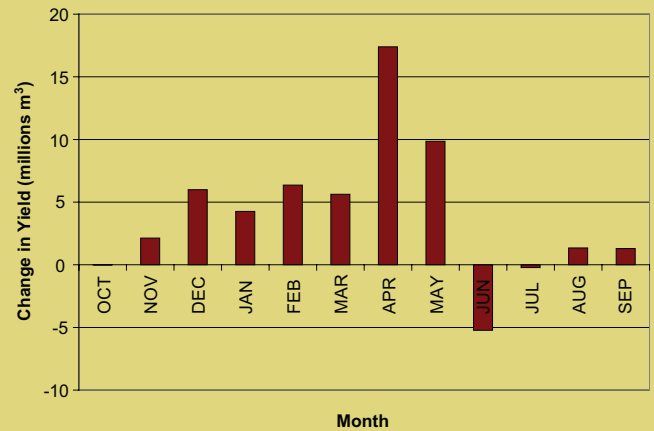


Figure T3-5: Monthly distribution of the water yield changes. Increases are shown through most of the year. The decreases in June and July are related to faster snowmelt in burned areas and relate to the much higher increases in April and May.

trees is typically attributed to the canopy, research on wind turbulence suggests that the stems could be important as well, particularly considering many branches still remain post-fire (Poggi and others 2004a, b).

The loss of forest canopy also reduces the loss of water through evaporative processes. Less precipitation is intercepted and subsequently evaporated and less water is transpired by trees, though this depends on level of tree mortality and response of other vegetation post-fire (Adams and others 2011; Guardiola-Claramonte and others 2011). Annual water yields may increase post-fire (Shakesby and Doerr 2006), as they have in many forest harvest experiments (Stednick 1996; Andréassian 2004; Brown and others 2005). In general, water yield increases would be greater in wet locations and in wet years than in drier locations and drier years. Lower evapotranspiration has

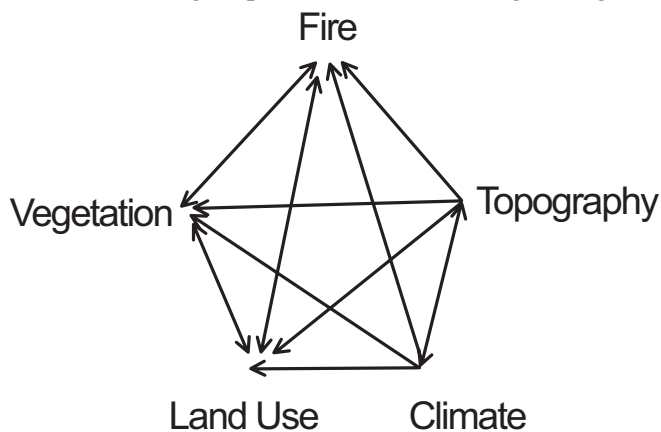
also been observed to result in higher soil moisture contents later in the dry season (Klock and Helvey 1976). Thus, we expect that some of the increased yield would benefit late season flows (see textbox on Boise River streamflow).

Fire and Climate Change: Feedback and Cumulative Effects

Fire extent is projected to increase in response to increased drought and lower precipitation combined with warmer temperatures (Littell and others 2009). Likely the effects of climate change on fire, vegetation, and streams will be synergistic. In the western United States, more large fires (Westerling and others 2006) and more widespread fires are more likely when early, warm springs are followed by warm, dry summers in the forests of the U.S. Northern Rocky Mountains and elsewhere (Westerling and others 2006; Morgan and others 2008; Littell and others 2009). Westerling and others (2006) found that fire seasons were 78 days longer 1986-2003 than in the previous 16 years (1970-1985) across the western United States, thanks to a combination of climate and fuel conditions. Based on their data, Running (2006) highlighted the 6-fold increase in area burned and the 4-fold increase in number of large fires in the same time period. Fire extent is projected to increase under projected climate changes. For instance, Spracklen and others (2009) predict that area burned will increase by a factor of 2.75 by 2050 in the Rocky Mountains and by a factor of 1.54 by 2050 across the western United States. Littell et al. (2010) predict increased area burned in many different regions across the western United States in response to changes in temperature, precipitation, and soil moisture. These projections are based upon correlations between fire extent and climate in historical records for recent decades (Westerling and others 2006), multiple decades (McKenzie and others 2004; Morgan and others 2008; Littell and others 2010), and multiple centuries as inferred from crossdated fire scars on trees (e.g., Kitzberger and others 2007; Heyerdahl and others 2008) and from charcoal in lakes (Whitlock and others 2003) and debris flows (Pierce and others 2004).

Although fuel accumulation (though more specifically fuel architecture) has been implicated in the increase in wildfire frequency and extent in recent decades (Kilgore 1973; Parsons and DeBenedetti 1979; Agee 1993; Graham and others 2004), it is part of a complex interaction of multiple variables that influence the vegetation and fire patterns we experience on landscapes today (Figure 15). Topography strongly influences patterns of burn severity in the Pacific Northwest and Southwestern United States (Holden and others 2009; Dillon and others 2011), as north-facing slopes are more likely to burn severely than south-facing slopes at the same elevation and high elevation forests often burn more severely than lower elevation forests (Holden and others 2012). North-facing slopes are often relatively moister than south-facing slopes, with soils with higher organic matter

Figure 15. The interactions among factors influencing fire patterns are complex. Adapted from Canadian Forest Service (2001). Forest fire: context for the Canadian Forest Service's science program. <http://www.nrcan-rncan.gc.ca/cfs-scf/science/context_fire/index_e.html>



and with higher biomass, which once dry can burn severely (Dillon and others 2011). Further, fires occur less often there than on adjacent south-facing slopes (Heyerdahl 2001) and biomass productivity is often higher so that when they burn, north-facing slopes are likely to burn more severely. Heyerdahl and others (2001) found that climate acts as a “top-down” factor strongly influencing fire extent, and that local factors such as topography, fuels, and vegetation influence fire “bottom-up,” resulting in local differences on contrasting aspects. Dillon and others (2011) found that topography had a greater influence on burn severity than did climate for 1,521 fires in Pacific Northwest and Southwest regions of the United States that burned 5.7 million ha 1984-2006. Likely the relative importance of fuels, weather, topography, vegetation, and climate vary greatly from place to place, and surely will be affected by land use, including fire exclusion resulting in changing fuel conditions. In years of widespread fires, fires are often large, suggesting that local fuels and microclimate have less influence on fire spread when it is especially hot, dry, and windy.

The relative role of climate/weather is likely different between forests and rangelands and among different rangeland and forest ecosystems (Collins and others 2006; Littell and others 2009) Where fine fuels are important to carrying fire, the factors affecting their abundance, including precipitation during their growing season, grazing, and wind, affect fire intensity and extent.

It is highly likely that fires and other disturbances will be an agent of climate change in altering vegetation and that effects may be cumulative. The term cumulative effects includes additive, compensating, and synergistic effects (Reid 1993). If we compare the effects of wildfire and climate change from the paragraphs above, there are several ways in which direct effects can combine with positive feedback. For instance, climate change is advancing snowmelt timing through reduced accumulation (Knowles and others 2006) while fire increases snowmelt rates and further advances snowmelt timing (Figure T3-5). There may also be compensatory feedbacks. For example, as streamflows decline in many parts of the western United States through reduced precipitation (Service 2004; Luce and Holden 2009), wildfire could result in a portion of that reduced water input reaching streams.

More complex interactions could increase some of the more severe consequences of climate change and wildfire. For instance, if warmer winter temperatures cause increased rain-on-snow flood risks in winter over more areas (Hamlet and Lettenmaier 2007), losses of forest canopy associated with fire could increase those risks. Salvage logging would likely exacerbate risks of turbulent transfer of melt to the snowpack. Increasing precipitation intensity interacting with water repellency and soil sealing processes could magnify post-fire runoff events.

Implications for Aquatic Biota

The network of streams and rivers comprises the habitats of fish and other aquatic organisms. Too little flow can pose a reduction in habitat amount, quality, and connectivity, and too much can scour or sweep organisms downstream. Timing can be important too. The decreases in low flows, particularly in the driest years, has the direct impact of reducing the volume of pools and habitat, but also reduces velocities and water surface area and therefore the delivery of food from upstream sources (Harvey and others 2006). Decreases in low flows could also cause some sections of stream to become so dry as to become impassible to migrating fish (Rieman and McIntyre 1996), which would compound the effects of water withdrawals in some situations, including groundwater withdrawal.

Higher flood flows and debris flow-related flood events have complex effects as well, depending on timing and frequency. High streamflows scour redds (Montgomery and others 1996; Tonina and others 2008) or sweep fry downstream (Fausch and others 2001) when they occur at the right time of year. Fall-spawning fish, such as bull trout (*Salvelinus confluentus*), are expected to be more vulnerable as peak flows shift from spring to winter months in historically snowmelt dominated basins, because their eggs may still be in the gravel or their fry inadequately prepared for high flows when they occur (Wenger and others 2011a). Debris flows have a much more limited footprint in the streamscape, but they typically remove all aquatic organisms from a given reach of stream, requiring recolonization. The speed with which affected reaches are recolonized will depend on the proximity of unaffected populations and the presence and abundance of migratory individuals.

C. Stream Temperature

For aquatic ecosystems, particularly for those containing rare salmonids, stream temperature is a critical variable structuring species distributions, patterns of abundance, and life history characteristics (Brannon and others 2004; Pörtner and Farrell 2008; Wenger and others 2011a). Both climate change and fire have strong influences on the energy balance of streams, primarily increasing temperatures, meaning that shifts to stream temperature regimes are among the principal processes driving changes to fish populations (Dunham and others 2003; Rieman and others 2007; Isaak and others 2010).

Climate Change

The direct effect of climate change on stream temperatures is increased incoming longwave radiation (see textbox for a description of the energy balance). Warmer air masses with higher emissivity will generate greater incoming radiation both day and night. Because water acts as a black body toward longwave radiation, the additional incoming radiation increases the temperature of the water. Warmer air masses will also increase the temperature of the forest canopy, again increasing downwelling longwave radiation. Direct warming from sensible heat transfer will likely be comparatively small (Leach and Moore 2010) and could easily be offset from increased evaporation from reduced relative humidity. Strong correlations between stream temperature and air temperature have made air temperature a proxy in estimating future stream temperature (Mohseni and others 2003; Rieman and others 2007; Wenger and others 2011b).

Indirect effects from climate change relate to changes in water availability either through streamflow or forest cover changes (see below for the fire effects). Declines in summer flows driven by declines in annual flows and earlier snowmelt (e.g., Cayan and others 2001; Luce and Holden 2009) mean that there is less water to heat in the months when the water is hottest. While the wetted width (area exposed to heat exchange) of streams will also decrease, it will not decrease as much as the depth and velocity (Dunne and Leopold 1978), yielding a net warming. Historical analyses of stream temperature also show a significant sensitivity to streamflow (Kiffney and others 2002; Isaak and others 2010; Kelleher and others 2011).

The Stream Energy Balance

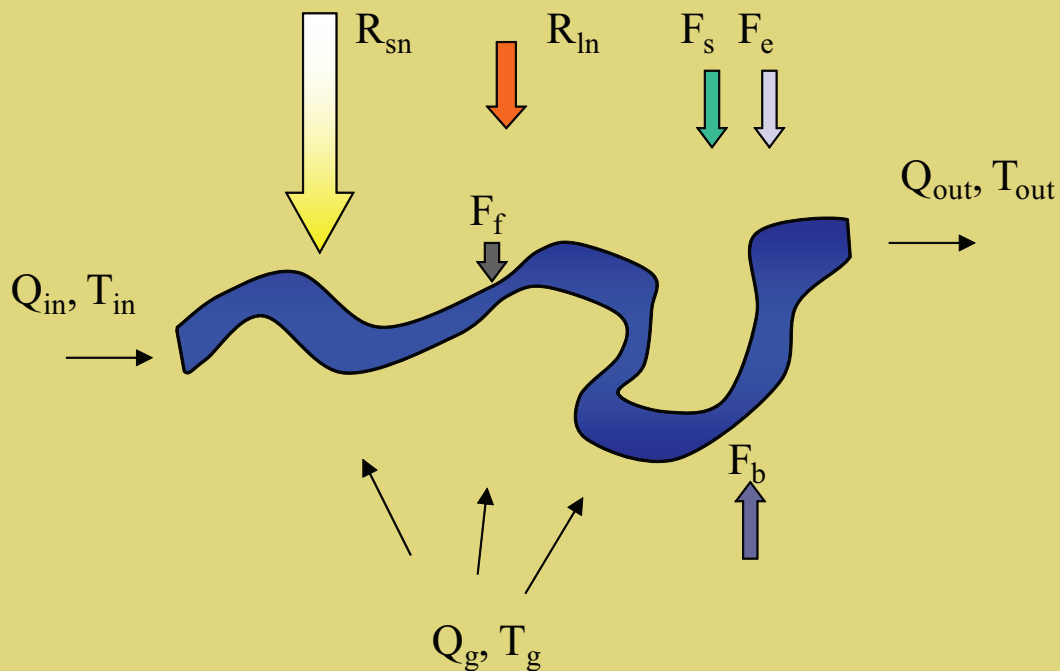


Figure T4-1: Components of the stream energy balance. Q 's are water fluxes, and T 's are temperatures of inflowing (in), outflowing (out), and groundwater (g) respectively. Together these comprise the advective heat fluxes to and from a stream reach. Radiant fluxes, denoted by R , are separated as net shortwave (sn) from the sun, and net longwave (ln). The sensible (F_s) and latent (F_e) heat fluxes are together called turbulent heat fluxes and represent the energy carried by wind in terms of cool or warm air convection on the stream (sensible) or evaporation or condensation on the stream (latent). The friction flux (F_f) is a function of the volume of water and slope of the reach. Bed fluxes include both conduction into solids of the bed and shallow groundwater exchanges (hyporheic) that are not just incoming groundwater adding to the streamflow.

The relationship of streamflow to fire and climate change is governed by the energy balance. Figure T4-1 shows the primary components of the energy balance of a stream reach. Because stream temperatures can change fairly rapidly, e.g., from one hour to the next, the diagram applies conceptually at hourly or shorter time scales; however, these fluxes are commonly summarized over longer periods (e.g., Webb and Zhang 1997). Solar radiation goes from being zero, at night, to being more than an order of magnitude greater than any other stream surface energy flux in the middle of the day (Sinokrot and Stefan 1993). Averaging over several days in one study, net radiation (short and longwave) was on the order of 70% of the incoming heat, with friction and sensible heat making up most of the rest, while radiation was 37% of the outgoing heat, with evaporation, bed conduction, and sensible heat playing significant roles (Webb and others 2008). Forest cover is a significant control on solar radiation so variations in forest cover play an important role in variation of stream temperature in forested environments (Johnson 2004; Moore and others 2005a; Dunham and others 2007). Turbulent fluxes (latent and sensible) generally oppose one another at hourly time scales because air warmed over the course of a day is typically also drier increasing evaporation; so if taken together they can be a minor component of the energy budget. Forests and streambanks can serve to protect smaller streams from the wind as well (Moore and others 2005b). Bed fluxes serve mostly to dampen and lag stream temperature responses to surface temperature forcings, and they respond primarily to the magnitude of daily stream temperature oscillations. They are more a part of the internal dynamics of a complex stream/bed/aquifer system than an external driver.

Historical trends in stream temperature show increases in many places in recent decades even without land cover changes (Langan and others 2001; Petersen and Kitchell 2001; Morrison and others 2002; Bartholow 2005; Hari and others 2006; Isaak and others 2010). While stream temperatures have been rising in concert with air temperatures, rates of warming are generally less than air temperature rates, and not all places are warming equally (van Vliet and others 2010). For example, some streams in mountains, particularly with glaciers or snowfields, show a distinct buffering due to increased snowmelt inputs from increased summer melt rates (Hari and others 2006), and greater groundwater inputs can buffer warming as well (Kelleher and others 2011).

Fire

Stream temperatures post-fire increase where vegetation shading the stream is reduced. Short wave radiation is one of the largest inputs to stream temperature, and fire can substantially open the canopy, particularly over smaller streams and with associated debris flows. Estimated increases in stream temperature due to fire range from 0.5°C – 4°C for mean temperatures and 2.5°C – 10°C for maximum temperatures (Helvey 1972; Amaranthus and others 1989; Hitt 2003; Dunham and others 2007; Isaak and others 2010). Increases depend on stream size and canopy removal, and the effects of the combination of fire and debris flow can be much greater than fire alone. In a study of small streams in the Boise River basin (less than 1000 ha) burned streams were on average 3.4°C warmer (maximum daily) than unburned streams, though with substantial variability in response, and streams that had experienced both fire and passage of a major debris flow were on average 7.9°C warmer (Dunham and others 2007)(Figure 16). Relative to biological criteria, these changes translated to about a 20% increase in probability of exceeding 20°C in burned streams, while those with a

Figure 16: Maximum summer temperatures in streams after fire. Data from 10 data loggers placed along 9 streams (Dunham and others 2007).

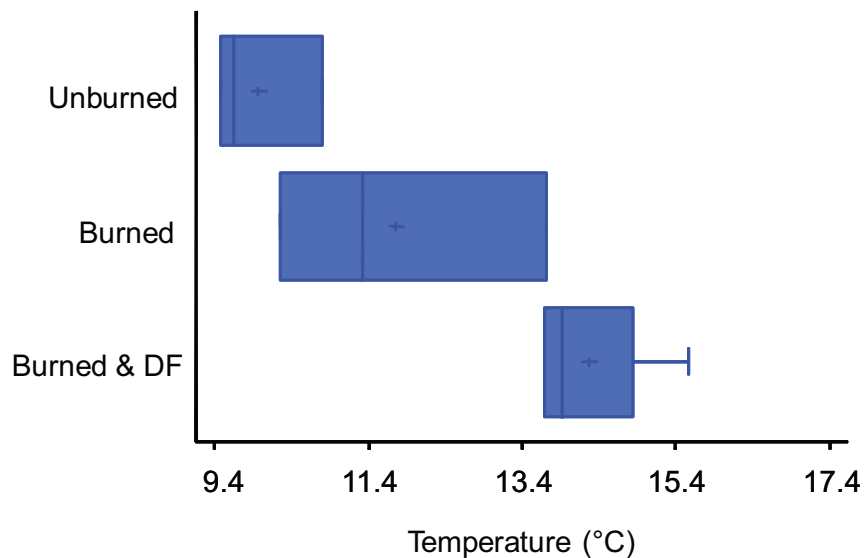
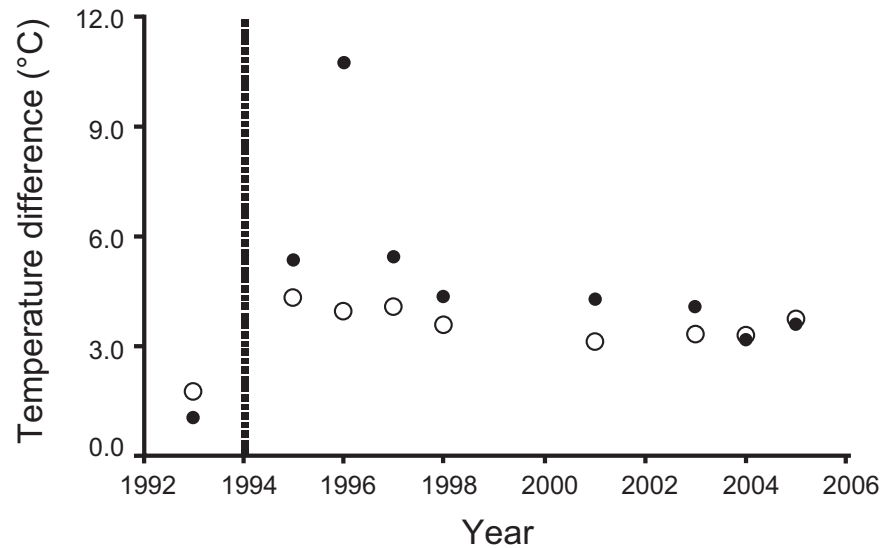


Figure 17: Temperature differences between Cottonwood Creek, which burned in 1994, and Roaring River in the Boise River basin. Open symbols are summer mean temperatures and filled symbols are the summer maximum (Dunham and others 2007).



debris flow as well showed about an 80% increase for streams between 1400 and 1600 m in elevation.

Recovery of stream temperatures over time after fires and debris flows is important to the dynamics of aquatic populations. Unfortunately there are few measurements of long-term recovery following fire. Dunham and others (2007) showed only minor recovery from about a 3°C increase in mean and maximum temperatures measured annually for more than a decade post-fire on a stream where only fire occurred (Figure 17). High solar angles during summer mean that trees and shrubs must be tall or very close to a stream to cast much shadow during periods of highest heat loading, so recovery may well take a few decades, depending on growth rates of adjacent vegetation post-fire and the size of stream.

Implications for Aquatic Biota

Aquatic biota interact with stream temperature in many ways. Poikilotherms (cold-blooded animals, like fish and many other aquatic organisms) have metabolisms that are regulated by the ambient temperature (Pörtner and Farrell 2008). This means that under warmer temperatures their metabolism runs faster with the consequences that they will need more energy (food) to survive, less of the food they consume will go to growth, and they may sexually mature earlier (Dunham and others 2007). If winter and spring temperatures increase earlier in the year, eggs will incubate more rapidly and young fish will emerge from the gravel earlier in the year. Changes in emergence timing and in growth may affect the development (or non-development) of migratory individuals from a given rearing population. Different fishes have different physiological adaptation to specific thermal regimes, and different species have tolerances for different temperature ranges (Reist and others 2006; McCullough and others 2009), which shows in the spatial and elevation distributions of fishes (e.g., Wenger and others 2011a).

Depending on the context, therefore, the ecological consequences of these physiological responses may be the outright loss of habitat suitability in stream reaches that become too warm or increased susceptibility to displacement of cold-adapted fish by relatively warm-adapted fish in stream locations where overlap occurs. For example, bull trout are generally displaced by brook trout from reaches

where the two species overlap and cutthroat trout are often displaced from entire streams by encroaching brook trout, rainbow trout, and brown trout (Wenger and others 2011a). Stream warming, through a variety of means, therefore, is predicted to shrink the extent of habitat patches for cold-water fish of conservation concern and increases the isolation of populations by pushing them farther into headwater streams (Rieman and others 2007; Isaak and others 2010; Wenger and others 2011b). At the same time, decreases in low flows and increased debris flood responses in steep tributaries may shrink habitats from above, further restricting populations and increasing the potential for debris flow disturbances. If decreases in low flows and temperature-related growth and productivity changes also decrease the number of migratory fish from these areas, the populations may increase in their vulnerability to individual fire or flood events.

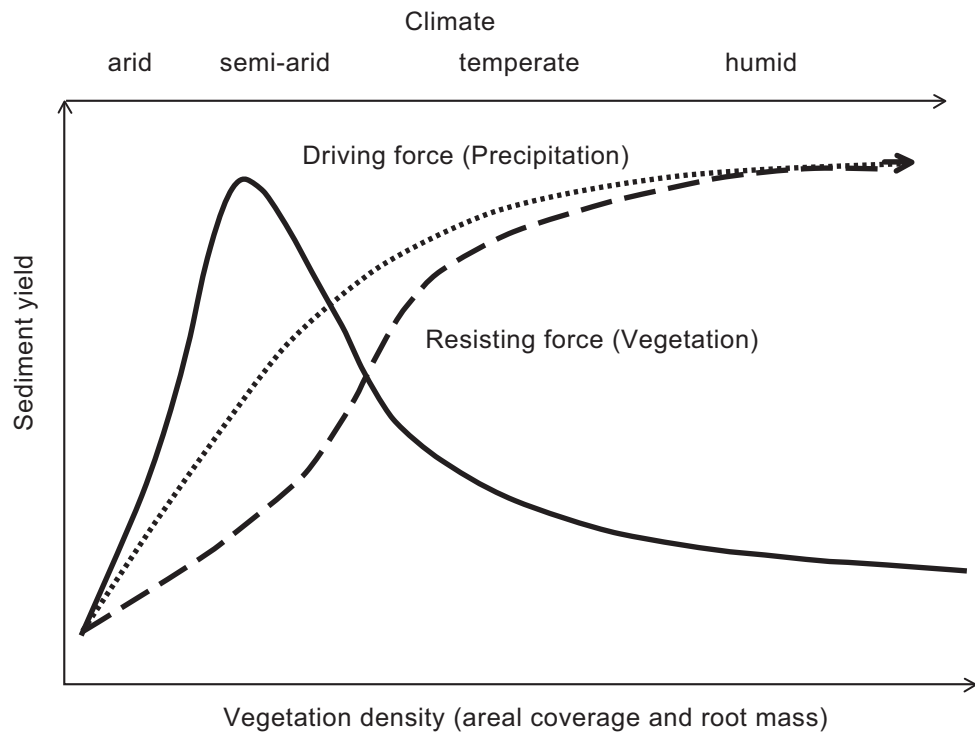
D. Geomorphology

Climate Change

Over geologic time, variation in climate has left profound marks on the landforms of the western United States. Glacially carved valleys are the most well recognized remnants of shifting climates, but we also see regionally extensive pluvial lakebeds. The connection of more contemporary climate variations to the incision and aggradation of arroyos and streams in arid and semi-arid regions has been a rich subject of research (e.g., Bull 1991), as well as subject of debate about the relative effects of climate and land use.

Some of the more direct relationships between climate and geomorphology of forested fluvial systems relate to the transport capacity of streams. Because of the strongly non-linear shape of sediment transport relationships (e.g., Parker and Klingeman 1982; Buffington and Montgomery 1997), flood flows are more important than total annual water yields. Thus, shifts in annual water yield may be less important than the potential of increased floods due to higher precipitation

Figure 18: Conceptual relationship between sediment yield and climate. The sediment yield curve (solid) is based on the empirical relationship of Langbein and Schumm (1958). The dashed line represents the resistance to erosion by vegetation while the dotted line represents the relative driving force by precipitation, with the greatest difference between the two is in semi-arid climates (Goode and others 2011).



intensity (e.g., Easterling and others 2000; Hamlet and Lettenmaier 2007) or increased probability of occurrence of rain-on-snow floods (Lettenmaier and Gan 1990; Nayak and others 2010).

A long literature on climate and landscape evolution notes that the direct effects of some climatic changes, e.g., precipitation, may be dramatically overshadowed by the vegetational response (e.g., Langbein and Schumm 1958; Bull 1991; Kirkby and Cox 1995; Tucker and Bras 1997; Istanbuluoglu and Bras 2006; Collins and Bras 2008). For example, Figure 18 (Goode and others 2011) shows a conceptual relationship between sediment yields from river basins and the mean annual precipitation. On the one hand, increasing precipitation should increase the volume of sediment removed; on the other, it supports more vegetation, which modulates the effects of precipitation. At intermediate precipitation levels, vegetation growth is interrupted by frequent disturbances, yielding a peak in sediment yield, while at lower precipitation levels, the scarcity of water dominates.

Climate is an external driver to the complex interchanges between vegetation growth, fuel accumulation, weather events, and fire frequency and severity. Paleoclimatic research links periods of drought, severe fire, and severe erosion events using tree rings, fire scars, pollen from lakebed sediments, and charcoal in alluvial fans (Meyer and others 1992; Swetnam and Betancourt 1998; Briffa 2000; Meyer and Pierce 2003; Whitlock and others 2003; Pierce and others 2004; Marlon and others 2006). At shorter time scales, it is clear that years of widespread fire are linked to severely dry and warm years (e.g., McKenzie and others 2004; Heyerdahl and others 2008; Morgan and others 2008; Littell and others 2009). As we shift toward a drier and warmer climate in the western United States, there is an expectation of greater areas burned annually (e.g., Running 2006; Littell and others 2009; Spracklen and others 2009) and the geomorphic consequences of fire.

Fire

The geomorphic consequences of fire are widely recognized; they are sometimes dramatic (Luce 2005; Shakesby and Doerr 2006; Moody and Martin 2009) and sometimes subtle (e.g., Ryan and others 2011). Hillslope and steep channel processes, such as surface erosion and mass wasting, receive the greatest attention (e.g., Benavides-Solorio and MacDonald 2001; Cannon and others 2001; Miller and others 2003; Pierce and others 2004; Moody and Martin 2009; Robichaud and others 2009b), while the disposition of channels with aquatic habitat is comparatively poorly discussed, despite more direct connections to aquatic ecology (Benda and others 2003a; Scheidt 2006; Lisle 2008). Most post-fire erosion studies focus on relatively short-term and small-scale processes, because they are acute and intense, with relevance to human life and property as well as aquatic ecology. There is, however, a growing recognition of decadal- to century-scale geomorphic dynamics distributed across stream networks and their role in evolving aquatic ecosystems (Reeves and others 1995; Benda and Dunne 1997a, b; Rieman and Clayton 1997; May and Gresswell 2003; Miller and others 2003; Scheidt 2006).

Loss of vegetative protection after fire along with alteration of soil properties increases the potential for surface erosion and mass wasting. The loss of trees reduces interception of raindrops by tree crowns and reduces root strength in the soil. Similarly, losses of trees, shrubs grass and surface organic layers expose the soil surface allowing it to be splashed and washed away more readily. Increased water repellency and surface sealing increase the runoff, as discussed earlier, and the loss of soil organic matter at the surface increases the disaggregation of soil particles allowing easier transport.

Many measurements of surface erosion from plots in many different environments demonstrate dependence on fire severity, slope, precipitation intensity, time since fire, soil characteristics, pre-fire vegetation, and aspect among other gradients of measurement. Unfortunately, the large number of methods used in estimating erosion and changes from pre-fire conditions preclude simple synthesis. Because studies are generally ad hoc after fires, systematic assessments of erosion processes over a range of fire severities, soils, and climate are lacking, so we cannot quantify how likely severe erosion events are or will be. Instead we direct readers to earlier reviews (e.g., Shakesby and Doerr 2006; Moody and Martin 2009). Rates for surface and rill erosion reported therein range from tens to a few hundred Mg/ha in the first few years following fire.

Mass wasting events, such as debris flows, directly disrupt aquatic habitat, potentially extirpating local populations and simplifying habitats in the streams where they pass. Paradoxically, these large events also provide large amounts of coarse material such as gravel, cobbles, and logs that ultimately add to the habitat complexity and quality of streams where they deposit (Reeves and others 1995; Benda and others 2003a). It is the relationship of populations to these reorganizing events, their occurrence and extent, and the recovery over time that we argue is most critical to aquatic ecology (Dunham and others 2003). This conceptualization is not premised solely on the fact that a great deal more sediment is produced from mass wasting events in small channels than from upslope areas (e.g., Santi and others 2008; Moody and Martin 2009), but it also recognizes a fundamentally different interaction between mass wasting events and aquatic populations and habitats in comparison to sediments detached and transported by water alone.

Post-fire debris flows result from two primary causes: initiation by landslides and initiation by bulking during gully excavation (Cannon and Reneau 2000; Cannon and others 2001; Istanbuluoglu and others 2002; Istanbuluoglu and others 2003; Miller and others 2003; Santi and others 2008). Climatic influences may favor the frequency of one type of initiation compared to another (Wondzell and King 2003), but both initiation mechanisms are active across steep landscapes in the West. Erosion rates from these kinds of events are typically in the range of several hundred Mg/ha (Istanbuluoglu and others 2003; Meyer and Pierce 2003; Istanbuluoglu and others 2004; Moody and Martin 2009; Cannon and others 2010). A primary difference between the two mechanisms is that the bulking debris flow events are most common during the first significant rainfall event (Cannon and others 2001), potentially tied to the durability of water repellency, whereas the window of landslide susceptibility may be on the order of a decade or more post-fire (e.g., Sidle and Ochiai 2006). Events as common as the 2-3 year (return interval) precipitation event have been noted as triggers for some of the larger debris flow events (e.g., Breidenbach and others 2004). Debris flow passage and deposition occur in distinct areas of a stream network and have greatly differing effects on habitat and biota.

The passage of a debris flow scours sediment, wood, and other biota from the stream, which become entrained in the flow. Because of the severe effects of debris flow passage to aquatic populations and habitats, a primary concern is the relative scale of debris flow tracks compared to the total habitat in a patch (see textbox on debris flow scale). If the amount of stream channel simultaneously affected by debris flows covers most of the suitable habitat in a patch, or if there is such poor connectivity to patches from which fish can recolonize, it would be difficult for the remaining fish to persist. However, if debris flows only affect a small fraction of a habitat patch at any given time, it is more likely that the patch will persist in time. Although there may be periodic dips in the population numbers

Debris Flow Scale in the Boise River Basin

Understanding how disturbance affects biota requires some understanding the nature of disturbances, particularly the most severe ones. For fishes in mountain streams, one of the most severe disturbances is debris flows. An important question is how much continuous habitat is affected during any given event.

We mapped debris flows across the Boise R. Basin from aerial photographs taken in 1969, 1979, 1988, and 1996 (Figure T5-1). The 1969 photos still showed the effects of a severe storm in winter of 1964, while the 1996 photos showed the outcome of a 1994 fire and 1995 thunderstorm (Figure T5-2).



Figure T5-1: Example aerial photo with disturbed and undisturbed streams.

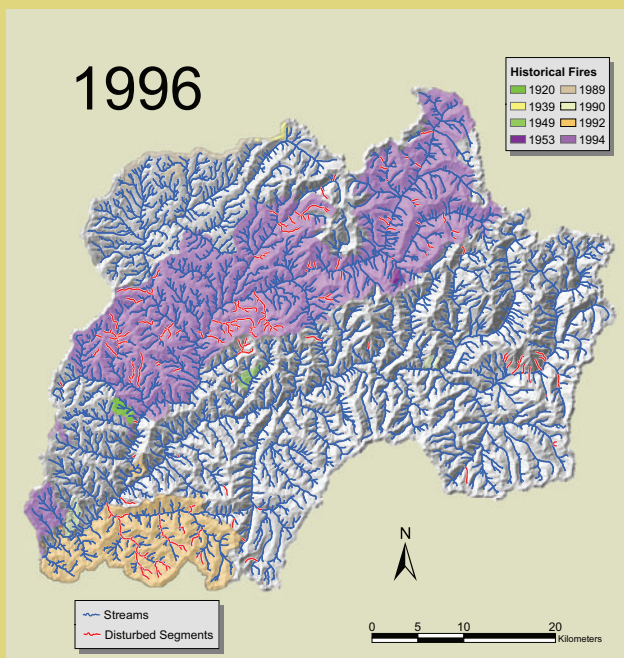


Figure T5-2: Map of debris flow affected segments from the 1996 photos.

Mapped stream segments were analyzed by calculating the probability that any two segments separated by a given distance along a stream were both simultaneously disturbed (Figure T5-3). These figures show that disturbed segments tend to cluster below 20 to 25 km network distance, and that at longer distances, they are randomly distributed.

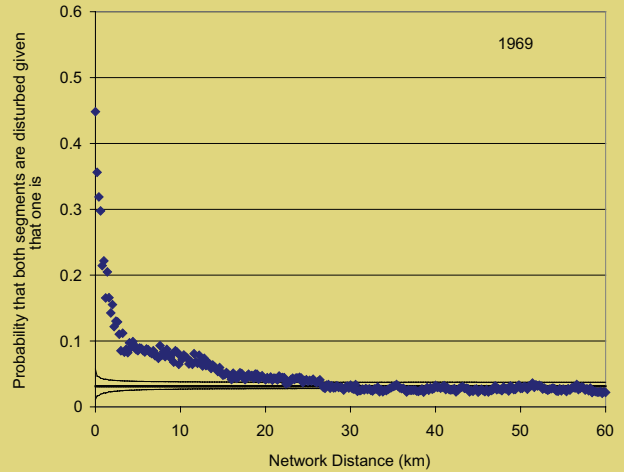


Figure T5-3: Probability that two segments separated by a given distance are simultaneously disturbed. Thin black lines show expected probability if disturbed segments were randomly distributed.

Dunham and Rieman (1999) show the probability that thermally suitable bull trout habitat is actually occupied depends on patch (watershed) size. The relationship levels off above 100 km², which corresponds to a stream network length of 20 km.

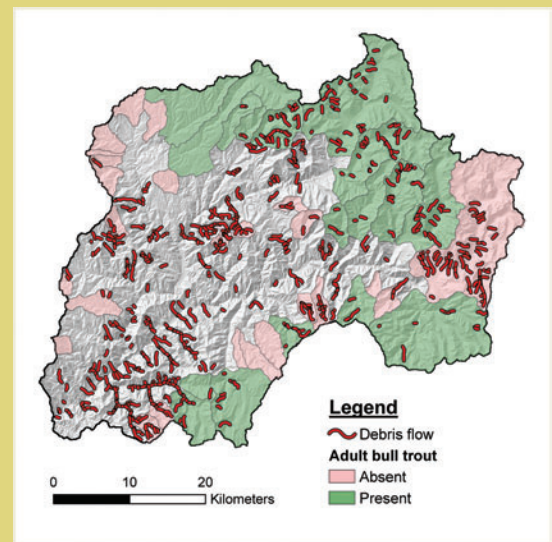


Figure T5-4: Map of all debris flows with bull trout patches from Dunham and Rieman (1999). Unoccupied patches tend to be small, on the order of debris flow affected basins.

for such patches, the capacity to reestablish within affected tributaries constitutes a key factor in resilience (Dunham and Rieman 1999). This view of size of debris flow is different from the typical volume orientation but represents an ecologically relevant perspective (Miller and others 2003). Debris flow mapping in the Boise River suggests that there are few continuous debris flow tracks with greater than about 20 km of stream length (see textbox on debris flow scale). A related way to look at the problem is by the size of drainage basins affected. From the database of Gartner and others (2005) looking at the western United States, the 95th percentile basin size was about 15 km², while the median basin size was about 1.2 km². By either analysis, debris flows most commonly impact basins smaller and steeper than are typically stable and productive fish habitats, but they occasionally affect tributary streams that are large enough to contain fish. Mapping tools considering debris flow movement and constraints can be helpful in evaluating the risk and appropriate scaling of aquatic habitats for persistence (Benda and Cundy 1990; Cannon and others 2010; Rieman and others 2010).

Debris flows deposit material when channel slopes decrease or the valley floor widens. In contrast to debris flow passage, deposition of sediment by major events appears to be important in the maintenance of diverse and high quality aquatic habitat (Reeves and others 1995; Benda and others 1998; Benda and others 2003a). Gravels, large boulders, woody debris, and soil entrained in debris flows bring in large cover, habitat structure features, spawning substrate, and nutrients. Furthermore, the large deposits form reach-scale heterogeneity in stream slope, contributing to habitat diversity. The disposition of debris flow deposits depends on the circumstances of the event and the configuration of the receiving channel. Debris flows may deposit large fans that are stable over decades to centuries (Benda and others 2003a), deposit fans that are rapidly reworked and transported (Lisle and others 2001; Cui and Parker 2005), or be lost in the flood event that initiated them (e.g., Meyer and Pierce 2003). The first type of event may have the strongest influence on aquatic habitat complexity.

Resilience is the propensity of an ecosystem to recover from an acute event (Holling 1973, 1986; Walker and Salt 2006). Part of the resilience of aquatic ecosystems is based in the life-history strategies of affected species (Rieman and McIntyre 1995, 1996; Dunham and Rieman 1999), but another aspect is the recovery of physical habitat in time (e.g., Minshall and others 1989; Reeves and others 1995; Benda and Dunne 1997a, b; Gresswell 1999; Scheidt 2006). The temporal dynamic of recovery also relates to the spatial scaling in terms of defining synchrony of disturbance (Poff and Ward 1990). If recovery takes one year, then events separated by a few years may effectively be independent, whereas if recovery takes decades, they may be effectively synchronous. Based on sediment transport theory, we suggest that basic channel form and sediment characteristics could settle fairly rapidly (1-5 years) post-flood because channel forming floods are fairly frequent (Wolman and Miller 1960). This is supported by observations of recovery of channel form within a decade after major floods (Wolman and Gerson 1978) and rapid recovery of basic channel characteristics after fire events (Potyondy and Hardy 1994). Studies examining the long-term changes in aquatic habitat following fire suggest a more complex picture relating to the supply and fate of wood proximal to the stream and tributary debris flow paths (Reeves and others 1995; May and Gresswell 2003; Scheidt 2006; also see textbox on instream large wood dynamics in riparian section). Therefore, while the basic components of a habitat are available within a few years after disturbance, recovery of optimal habitat conditions, which depend on a host of other parameters, may take several decades post-fire (Gresswell 1999).

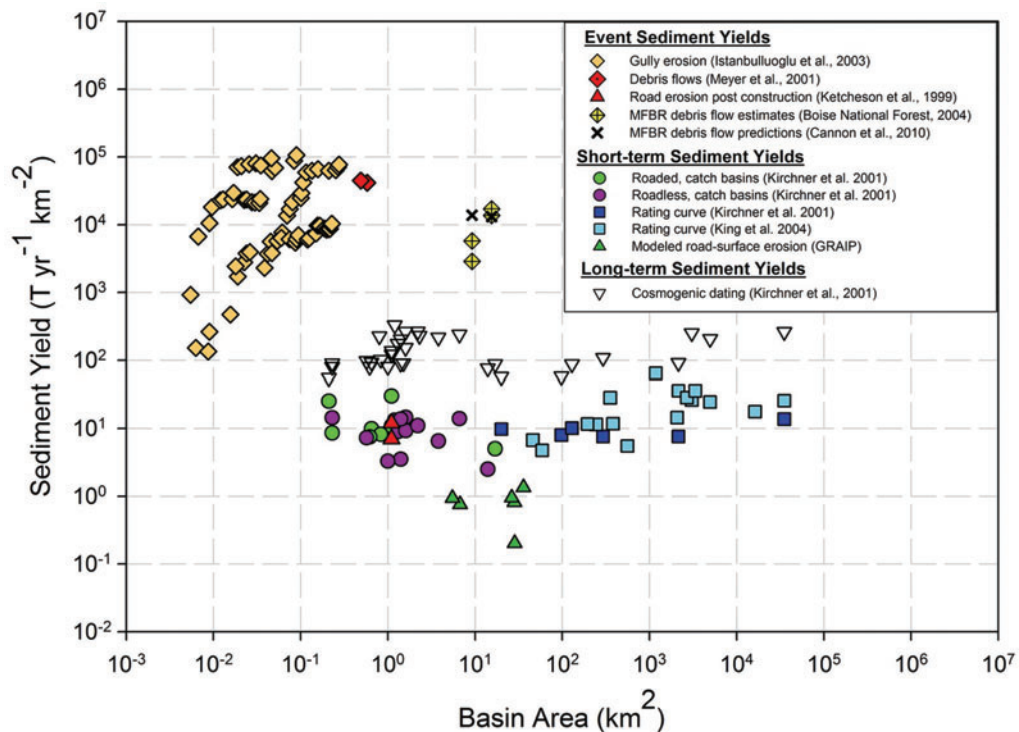


Figure 19: From Goode et al., (2011). Sediment yields for individual post-fire erosional events, long-term basin averages, short-term basin averages (~30 yrs.), and road-surface erosion. Individual post-fire erosional events include measurements of debris flow and gully tracks in the Boise and Payette River Basins (Meyer and others 2001; Istanbuluoglu and others 2003) and deposit estimates (Breidenbach and others 2004) and model predictions (Cannon and others 2010). Long-term basin averages are from analysis of cosmogenic nuclides in fluvial sediments (Kirchner and others 2001). Short-term averages for small basins (< 20 km²) are from catchbasin dams (1950s-1980s, Kirchner and others 2001) and are subdivided by the presence or absence of roads in the basin. Short-term averages for larger basins are predicted from sediment rating curves and daily stream flows (1920-2000, Kirchner and others 2001), supplemented with data from King et al. (2004) using the same methods and period of record as that of Kirchner et al. (2001). Basin-average road-surface erosion is predicted from the GRAIP model (Black and others 2010), with values updated from Prasad (2007) based on measurements of road-surface erosion from the Middle Fork Payette watershed (Black, unpub. data). Event-based road-surface erosion values are from observed, post-construction erosion (4-year average yield, Ketcheson and others 1999).

The contribution of these episodic events to basin-scale sediment yields is an important consideration with respect to the impacts on fish, vis-a-vis the idea of “pulse” versus “press” disturbance (Yount and Niemi 1990). Kirchner and others (2001) compared long-term (~10,000 years) sediment yields measured using cosmogenic ¹⁰Be isotopes to sediment yields measured over 30 years using sediment traps and suspended sediment sampling (Figure 19). They found that the long-term average rate was an order of magnitude higher than the contemporary rate, implying that sediment production was episodic in nature for these basins. Istanbuluoglu and others (2004) followed up with comparison to post-fire erosion rates to establish a magnitude and scale of the episodicity required to generate the relationship. Their conclusion was that the long-term sediment yields from forests can be explained by events as severe as the ones measured post-fire occurring on the order of a few hundred years apart, implying long relaxation periods after events.

How important are roads in this context? Some of the watersheds studied by Kirchner and others (2001) had roads; some did not. While the roaded watersheds produced more sediment, and one study of the first 4 years after construction

showed substantial sediment additions (Ketcheson and others 1999), the magnitude of road erosion is extremely small in comparison to post-fire sediment inputs in the same time frame (Figure 19). One key difference is in the “pulse” versus “press” disturbance of fires compared to roads. Observations of post-fire deposition show a rapid recovery, followed by long periods with few additions of fines (a “pulse” process). In contrast, road sediments are produced in less abundance but every year (a “press” process). In addition, fire sediments usually require a significant storm to be generated, whereas roads produce runoff and sediment in almost every precipitation event. The cumulative effect from frequent “press” disturbances on aquatic biota may far exceed the direct effect of even major “pulse” disturbances.

A great deal of effort after fires goes into the control of potential erosion using post-fire stabilization techniques. Some methods used include contour felled logs, straw wattles placed on contour, surface application of straw or engineered wood, and aggressive grass seeding (Robichaud and others 2000). The general focus of the techniques is on control of surface erosion processes, which is reflected in largely plot-scale evaluation methods (e.g., Wagenbrenner and others 2006; Robichaud and others 2008). The importance of mass wasting processes in small steep channels, however, makes it difficult to extrapolate from such studies to speculate on the broad-scale efficacy of these treatments. Because most techniques relate to the control of sediment movement as opposed to controlling water, we may ultimately expect limited performance for preventing post-fire debris flows. Two caveats to this statement are that 1) contoured stabilization methods and surface mulches both extract some water, either by ponding or intercepting and 2) hillslope-derived sediment may contribute to the bulking of debris flows. It has been noted that the effectiveness of treatments declines with return interval of the precipitation event (e.g., Wagenbrenner and others 2006).

Part II: Biological Systems

A. Forests, Climate Change, and Fire

This brief synthesis of recent fire science is focused on forest vegetation dynamics and burn severity that have implications for fish and streams. We build upon recent reviews of related topics and recently published research. We suggest reading the syntheses on related topics in other sections in this document.

We Live In a Fire Environment

Fires will continue to occur, and they will sometimes be large and burn intensely—we need to plan accordingly. Biomass accretion exceeds decomposition in most forest and rangeland ecosystems. Fire is a global herbivore (Bond and Keeley 2005), consuming accumulated biomass when fires ignite and weather conditions are conducive. As a result, every place has a fire history, though it differs from place to place (Agee 1993). Recurring fires have shaped ecosystems and species adaptations. Despite very intensive efforts at fire suppression, we have experienced extensive fires in many years with many large fires in recent decades.

Climate change and other aspects of global change means that area burned by wildfires is expected to increase to as much as 10-12 million acres per year over the next five years nationwide (NWCG [National Wildfire Coordinating Group] 2009) and to double, triple, or more in some regions of the country (Littell and others 2009; Spracklen and others 2009; Littell and others 2010), but not to the same degree everywhere. Most of the area burned in any given region results from just a few years of widespread fire, and it is in these years that climate is an important driver of fire extent (McKenzie and others 2004; Morgan and others 2008; Littell and others 2009). When many large fires burn synchronously, threats to people and property are high, our ability to suppress fires can be overwhelmed, and fires have important cumulative effects on smoke production, carbon, water, and nutrient budgets as well as habitats for many species of conservation concern (McKenzie and others 2004; Morgan and others 2008; Spracklen and others 2009).

Managers must balance the costs of fire suppression, ecological benefits and impacts of fires, fire fighter safety, protecting people and property, and the ecological realities of increasing wildland urban interface (Theobald and Romme 2007), invasive species (Brooks and others 2004), changing climate (Solomon and others 2007), and changing perceptions of risk. Doing so will require strategic fire management that integrates fuels management, fire prevention, fire use, multiple fire suppression strategies, restoration, and other management in support of effective landscape-scale fire management across lands of intermingled jurisdiction (NWCG [National Wildfire Coordinating Group] 2009). With 10,000 homes burned in wildfires 2002-2006 (Gude and others 2008) and much focus on fuels management, protecting people and homes from fires continues to be a major, and expensive, fire management goal. Yet allowing fire to play a more natural role in some locations is a goal for many federal land management agencies (NWCG [National Wildfire Coordinating Group] 2009). Managing the rising costs of fire suppression and threats to people and property is a goal of all fire managers (NWCG [National Wildfire Coordinating Group] 2009).

Changing Fire Regimes

Fire is one of many disturbances that have shaped landscape dynamics for millennia. Fire is part of the resulting natural variability to which many species are adapted and is an essential component of most terrestrial ecosystems. Yet not all fires are alike, and fire regimes vary from place to place. The degree of change in fire regime from past to present (and therefore the future) varies greatly (Figure 20). The different patterns of recurring fires by frequency, severity and other characteristics are classed into fire regimes (Table 2). Since the early 1900s, humans have significantly altered historical fire regimes in many parts of the world. People use fire, suppress fires, and otherwise change when and where and how fires burn with their direct (fuels management, fire suppression) and indirect (roads, logging, grazing, limiting vegetation management, etc.) actions. Major trends are evident. Where fires currently occur less often than they did historically, we generally see an increase in woody biomass in many ecosystems—some with native species, some with nonnative species. Many argue that this is a result of climate change, while others attribute this to very effective fire suppression and other land use; likely both sets of factors contribute (Dombeck and others 2004; McKenzie and others 2004; Morgan and others 2008; Littell and others 2009; NWCG [National Wildfire Coordinating Group] 2009). In many ecosystems worldwide, introduced annual grasses have fueled much more frequent fires than occurred in the past. It is referred to as the grass-fire cycle: the more grass, the more fire, and the more fire the more grass (Vitousek and others 1996; Brooks and others 2004). As a result, there are many ecosystems worldwide experiencing fires much more frequently than in the past (Vitousek and others 1996; Brooks and others 2004; Shlisky and others 2007).

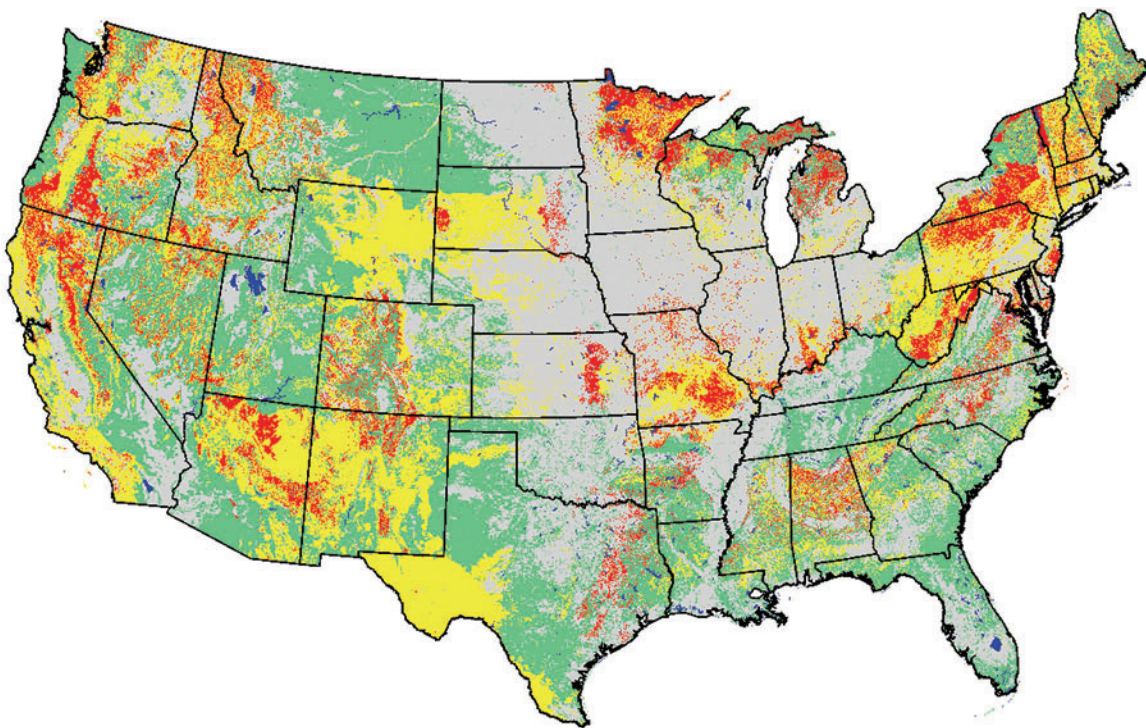


Figure 20: Fire Regime Current Condition (FRCC) class map version 2000 (from Schmidt and others 2002). Red areas show the greatest departure from historical fire regimes and green show the least. Gray areas are non-forested.

Table 2: Fire regimes can be grouped by how often fires recur and the degree to which ecosystems change when they burn (from Barrett and others 2010), http://www.fire.org/nifft/released/FRCC_Guidebook_2010_final.pdf.

Group	Frequency	Severity	Severity description
I	0–35 years	Low / mixed	Generally low-severity fires replacing less than 25% of the dominant overstory vegetation; can include mixed-severity fires that replace up to 75% of the overstory
II	0–35 years	Replacement	High-severity fires replacing greater than 75% of the dominant overstory vegetation
III	35–200 years	Mixed / low	Generally mixed-severity; can also include low severity fires
IV	35–200 years	Replacement	High-severity fires
V	200+ years	Replacement / any severity	Generally replacement severity; can include any severity type in this frequency range

The Fire Regime Condition Class is a recent national mapping effort aimed at identifying locations where departures of fire and vegetation conditions is low (class I), moderate (class II), or great (Class III) (www.frc.gov, Figure 20). Severe, stand-replacing fires are the norm in some ecosystems and therefore represent little departure, yet they represent a great departure (depending on size and other variables) in other ecosystems, including many dry forests.

Humans have altered vegetation and associated fire regimes. Humans alter the seasonality, frequency, extent, and severity of fire. The greater the degree of departure in fire frequency and severity and in vegetation, the greater the changes in biodiversity and other ecological values, and the more uncharacteristic the effects of fires will be when they occur (Shlisky and others 2007). Species may not be well adapted to the uncharacteristic fire regimes that develop when fire frequency and severity and vegetation composition are very different from historical range of variability (Keane and others 2008). More than 80% of the ecoregions examined worldwide have degraded or very degraded fire regimes (Shlisky and others 2007). Forest fire regimes have changed most where fires were historically frequent, as was the case in many grasslands and in dry forests (Agee 1993; McKenzie and others 2004). In mixed-conifer forests at moderate to high elevation, historically most fires were small and a few accounted for most of the area burned and mixed and stand-replacing fires were the norm (Schoennagel and others 2004). Departures of current from historical fire regimes in fire frequency and severity can be characterized better in some ecosystems than others, for we know relatively less about historical fire regimes in grasslands, shrublands, woodlands, and wet forests than we do in dry forests.

Some patterns emerge from an analysis of fire regimes relative to climate, topography, and vegetation types. First, the wettest and coldest forest sites do not burn often, but when they do, they can burn severely (Morgan and others 2008; Dillon and others 2011). A shift to warmer springs and warmer, dryer summers could result in more years of widespread fires, and such shifts could be contributing to the extensive area burned in many large fires in recent decades in the forests of the U.S. Northern Rockies (Running 2006; Morgan and others 2008; Littell and others 2009) and elsewhere. The degree to which current and future fire regimes

are different from historical regimes depends, in part, on the relative importance of climate and fuels in influencing fire occurrence, extent, and severity. While the relative importance of climate and fuel varies among forest types (Schoennagel and others 2004), we do not yet fully understand the implications of interactions among climate, vegetation, land use, topography, other disturbances, and fires. The complex interactions and potential for feedback among these factors lends uncertainty to predictions, particularly in complex topography.

It is one of the many paradoxes of fire that as fires have become less frequent, future fires are increasingly likely to burn more intensely and severely. People can choose to live with fire, allowing more choice about where and when fires burn, or to continue suppressing fires and suffer the effects of large fires, burning under extreme conditions that we cannot control. Integrated Fire Management approaches balance sustainable levels of effort with consequences for people and the environment (Figure 21).

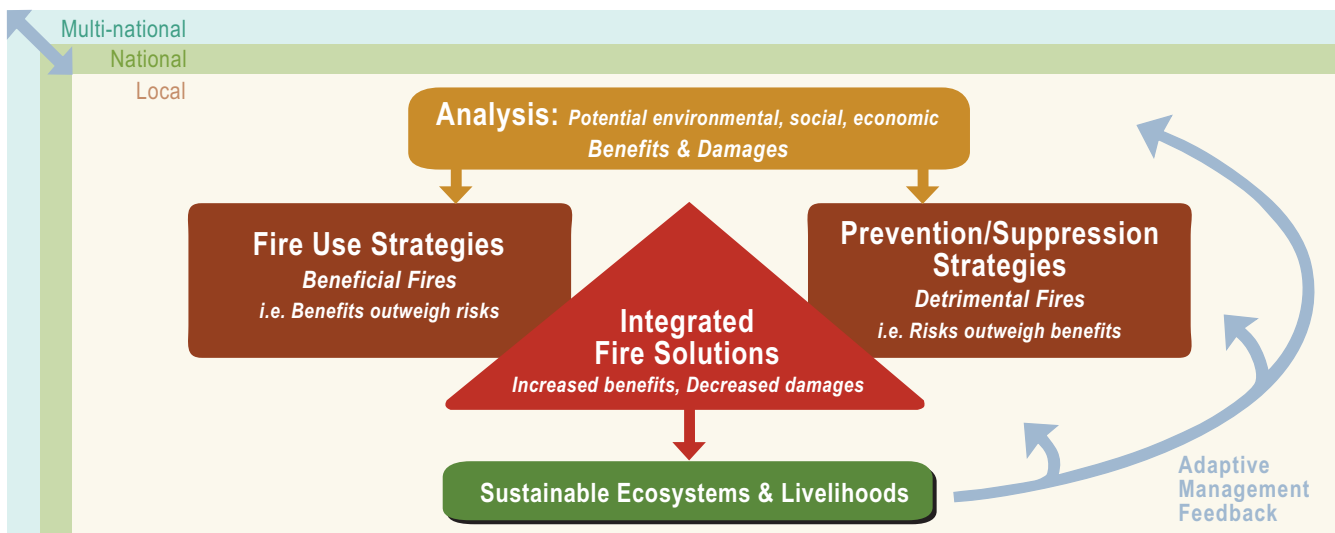


Figure 21. Integrated Fire Management encompass fire use, prevention, and suppression with an understanding of sustainable ecosystems and livelihoods (from Myers 2006).

Fire regimes, the pattern of recurring fires through time, reflect the interaction between vegetation, topography, climate, and land use. Precipitation, temperature, and soils influence where plants can grow, while disturbance, succession, and competition affect where they occur and the ongoing landscape dynamics. These factors and the amount of fuel available to burn are, in turn, major factors in determining the rate of spread, intensity, and severity of fires (Rothermel 1972). Climate and weather interact with topography and vegetation to influence fire behavior and effects. People influence vegetation structure and composition, and they suppress and ignite fires.

Burn Severity

Burn severity is an attribute of fire regimes used to express the degree of ecological change that results from a fire (Lentile and others 2006; Lentile and others 2007; Keeley 2009) (Figure 22). We now understand that fire is beneficial and often essential to most ecosystems, and ecologists and managers are often focused

on the post-fire effects when an area burns. Thus, fire severity or burn severity (these terms are often used interchangeably) is perhaps the most significant but least understood attribute of fire regimes. We focus on describing recent research on burn severity here.



Figure 22: Burn severity is used to describe the degree of change due to fire and is often based upon fire effects, including but not limited to overstory vegetation mortality, soil heating, and fuel consumption (Keeley 2009, Lentile et al. 2006, Lentile et al., 2007, Keane et al. 2008). Burn severity can be classified for forests (as shown here), woodlands, shrublands, and grasslands in the field and from satellite or airborne remote sensing immediately or one year post-fire (from Lentile et al., 2006).

There are a number of fire effects of ecological consequence, including vegetation loss, tree mortality, soil heating, and soil effects. The term “burn severity” is imprecise and carries inherently negative connotations (Lentile and others 2006; Keeley 2009). For instance, in many vegetation types, “severe” stand replacing fires are characteristic and within the historical range of variability, yet calling these severe implies that they are also catastrophic and by definition undesirable. While a fire that results in mortality of the aboveground biomass is indeed one that results in great change (and is therefore often assigned high burn severity), many plant and animal species thrive after such events; some are even dependent upon severe fires. Thus, a severe fire is not necessarily catastrophic or “bad,” and all fires will have some desirable and some undesirable effects, even for streams and fish. In some cases, the ecosystem changes resulting from an absence of fire may result in less desirable ecosystem characteristics and perhaps more severe fire effects once the area burns in subsequent wildfires.

Burn severity is a continuous measure of multiple fire effects, including but not limited to overstory vegetation mortality, soil heating, and fuel consumption (Lentile and others 2006; Lentile and others 2007; Keane and others 2008; Keeley

a)



Low burn severity (typically in places with light fuels) may be patchy and may not even show complete combustion of herbaceous species.

b)



Moderate burn severity may show black ash or blackened duff at the surface.

c)



High burn severity presents mostly bare soil with some white and black ash at the surface.

Figure 23: Soil burn severity is commonly described in classes based upon fire effects on overstory and soils.

2009). For purposes of mapping and communication, burn severity is commonly classified into four or more classes (Figures 22 and 23), including unburned, low, moderate, and high. Other classifications are possible. Such classes are often inferred from satellite imagery and field observations. In reality, burn severity is a continuous measure on a variety of variables.

When fire frequency changes, burn severity also changes. Where fires become less frequent, fuels often accumulate sufficiently between fires such that subsequent fires burn more intensely and severely. This is the case with many dry forests that historically burned frequently with many surviving large trees. Currently,

where dry forests are densely populated with small trees and ladder fuels, even large-diameter trees of otherwise fire-resistant trees, like ponderosa pine, are more likely to die in subsequent fires.

If a fire burns very intensely, the high rate of energy release may mean that foliage in the crowns of many overstory trees or shrubs are consumed, and most are top-killed. Some may resprout. A fire does not have to burn intensely to be severe, however (Keeley 2009). Fires smoldering for many months in the organic soil of a peat bog, or creeping around burning the accumulated duff and logs in a subalpine fir forests while not burning any tops of the shrubs and trees, can nonetheless result in vegetation mortality and soil effects that would be judged severe. In such fires, the flame lengths may be very low (suggesting that fire intensity is low), but the heating of the soil and consumption of organic matter is sufficient to kill many of the roots of trees, shrubs, grasses, and forbs; thus subalpine fir trees can be readily killed by cambial damage to roots and bole. Typically, fires that burn intensely are also severe as occurs when both tree crowns and dense mats of feather moss burn in black spruce forests in Alaska, or when crown fires occur and burn large areas in subalpine forests. Fires in many grasslands burn very intensely (with rapidly moving fire fronts and high flame lengths), but where the grasses are well adapted to resprout vigorously post-fire, there may be little difference between burned and adjacent unburned grasslands within just a few months or years, and the degree of ecological change as a result of the fire is small and the fire is not severe. Such rapid burning cannot endure long with sparse fuels; so subsoil heating is slight.

For soils, for streams, and for aquatic organisms, severity of fire matters. How fires burn is often more important than if an area burned. Severity is often related to fire frequency—less frequent fires are often more severe, simply because there is more fuel to burn.

Burn Severity Indicators Inferred From Satellite and Field Data

Ideally, indicators of burn severity will be ecologically meaningful, measurable in the field and remotely from air or satellite, and readily interpretable. They must be useful in describing ecosystem recovery and condition, including vegetation, carbon, water, and nutrients (Lentile and others 2009). Often burn severity is mapped from satellite imagery, such as Landsat, in order to get rapid, consistent evaluation across large areas. The Normalized Burn Ratio (NBR), differenced Normalized Burn Ratio (dNBR: Key and Benson 2006), and RdNBR (Relative differenced Normalized Burn Ratio) are widely used (Miller and Thode 2007) indices for creating the soil burn severity maps used in post-fire rehabilitation assessments (Robichaud and others 2007; Parsons and others 2010) and in ecological assessments based on one-year differences due to fire. These ratios effectively measure the relative degree of vegetation and soil char between pre- and post-fire conditions and they have been related to one-year post-fire vegetation cover (Holden and others 2005; Smith and others 2005; Lentile and others 2006; Hudak and others 2007). These continuous measures are often broken into classes, often based on field assessments made with the Composite Burn Index (CBI) (van Wagtendonk and others 2004; Brewer and others 2005; Cocke and others 2005). Unfortunately, there are few studies evaluating dNBR or CBI against quantitative biological or ecological measures of post-fire effects (Hudak and others 2007; Robichaud and others 2007; Santis and Chuvieco 2007; Smith and others 2007), especially more than one year post-fire; such studies would help users to know how effectively burn severity can be inferred from satellite imagery and how it influences subsequent vegetation trajectories. These indicators are widely applied in mapping burn

severity for individual fires and across the United States in the national Monitoring Trends in Burn Severity (MTBS, www.mtbs.gov) program (Cocke and others 2005; Epting and others 2005; Miller and Thode 2007). There are limitations to these approaches (Roy and others 2006; Smith and others 2007; Lentile and others 2009). Briefly, limitations include the following: subjective and qualitative nature of breaks between classes and field assessments using CBI exists; it is difficult to scale from the points where CBI or other on-the-ground assessments are made to the landscapes and watersheds over which assessments are needed; the spectral bands used are not ideal; RdNBR was developed to address the less-than-optimal performance of dNBR in woodlands; shrublands and grasslands have such low biomass that even if most is consumed the absolute change is not high; and the measures are mostly affected by vegetation consumption when we also want to know soil heating and fuel consumption (Roy and others 1987; Hudak and others 2007). Further, though NBR from a single post-fire image may suffice, differencing helps to address site conditions that can affect inferences from imagery, thus helping to ensure that difference is due to fire, not other site conditions. Scaling is an issue because the post-fire effects measured in the field typically reflect fine-scale processes, but also impact coarse spatial (watershed to regional) and temporal (decadal) scales. Char fraction has been suggested by Lentile and others (2009) as a potentially versatile measure of postfire ecological impact that also influences the terrestrial carbon and water cycles. However, no single indicator of burn severity will be ideal for evaluating burn severity across all ecosystems affected by fires.

Burn severity includes overstory mortality, consumption of biomass, and soil heating (Keane and others 2008). Many people assess burn severity using broad categories of overstory tree or shrub mortality. Burn severity varies with fuel and the environmental conditions before and during combustion (Ryan and Noste 1985). Assessments of burn severity based on satellite imagery are more likely to be accurate for high burn severity (Hudak and others 2007). In part, this is because it can be difficult to see soil effects through overstory vegetation canopy that is left where fires burn with low or moderate severity, and both low and moderate severity are spatially heterogeneous at fine scales. Further, we are likely more accurate in locating large patches of high burn severity.

All fires are patchy. Even when large fires burn through thousands of hectares in a few hours, burn severity is seldom uniformly severe (Schoennagel and others 2004; Keane and others 2008). Pattern is very important to biodiversity and vegetation dynamics. In terms of post-fire soil erosion, we are often most concerned with large patches of stand-replacing fire, especially if these are on steep ground and on erosive soils. Fire effects on the soil are typically very heterogeneous; even within a large patch of severely burned overstory, there are often unburned vegetation patches. Because soil effects are more uniform where overstory vegetation was largely killed, and because removal of that vegetation means that we “see” the soil from remotely sensed imagery, mapping high burn severity is more accurate than mapping of moderate and low severity burns (Hudak and others 2007).

Burned Area Emergency Rehabilitation (BAER) teams concerned about post-fire erosion focus on large patches that are burned severely, especially when those are on steep slopes with erodible soils (Robichaud and others 2009a; Parsons and others 2010). Starting with Burned Area Recovery Characteristics maps that are based on dNBR from satellite imagery (usually Landsat with its 30-m resolution), BAER teams evaluate the local conditions. Because they wish to identify those locations where fires have greatly affected soils, in the field they may look for bare, reddened soil with white ash. Soils with cover are less likely to erode. Where litter, duff, vegetation or other organic layers remain post-fire, and where

vegetation survives or rapidly recovers, soils are less likely to be displaced even in relatively high-intensity rain storm events (Robichaud and others 2007 and references therein).

Will Future Fires Be Larger and More Severe, and If So, So What?

People worry that fires will be larger and more severe as climate changes (see earlier section for a summary of recent work on fire and climate). No doubt fires will happen. Spracklen and others (2009), Littel and others (2010), and Westerling and others (2011), among others, have predicted that fires will become larger and more severe as fires respond to changing temperature and precipitation and lightning patterns. The effects will not be the same everywhere. Some regions will have more fires in some years, and in some years, fires will be widespread across multiple regions (NWCG [National Wildfire Coordinating Group] 2009).

Miller and others (2009) found that recent fires in the Sierras and Cascades are more severe than historically. They evaluated satellite images before and after fires for a 22-year period. Dillon and others (2011) found that annual proportion burned severely increased in just one of the three U.S. Southwest ecoregions and two of three ecoregions in the U.S. Northwest 1984-2007. They attributed the change to a combination of fuels and climate. More analyses like these based upon the MTBS data (www.mtbs.gov) are likely soon. Dillon and others (2011) found that topography exerts significant “bottom-up” controls on patterns of burn severity with north-facing slopes often burning severely, probably due to higher productivity and more biomass, and a low likelihood of burning except under relatively extreme weather conditions. Burn severity in the Gila Wilderness is influenced by time since and degree of burn severity of previous fires (Holden and others 2010).

Fuel Treatment Effectiveness

Managers implementing fuel treatments use and manage disturbance, taking advantage of the strong interplay between fire and vegetation. Nearly 30 million acres have been treated to reduce fuels and fire hazard on federal lands with additional treatments on private and state lands (NWCG [National Wildfire Coordinating Group] 2009; Schoennagel and others 2009), most of which have been implemented since adoption of the National Fire Plan in 2000. More are planned especially as the wildland urban interface has grown (Theobald and Romme 2007). Nonetheless, the area with fuel treatments and other active vegetation management is far outpaced by wildfire and by insects and disease, which argues for strategic planning of fuels and fire management (NWCG [National Wildfire Coordinating Group] 2009). The goals of fuel treatments commonly include reducing wildfire risks to communities and the environment, and improving ecosystem resiliency to wildfire effects (USDA and others 2006). Fuel treatments are designed to reduce fire hazard with the goal of altering fire behavior, thus easing fire suppression efforts (Graham and others 1999; Graham and others 2004) and the escalating costs of fire suppression and threats to people and property (NWCG [National Wildfire Coordinating Group] 2009). In forests, thinning from below to remove small trees can reduce crown fire hazard, but if wildfires occur before the residual surface fuels are treated (e.g., before hand piled wood is burned), tree mortality can be high. Treated areas are less likely to stop a fire, but can be useful during fire suppression. Treatments to reduce fire hazard often focus on thinning from below to reduce vertical (ladder fuels) and horizontal continuity of fuels, as well as treatments to reduce the amount of fuel available on the ground. Grazing is a common treatment in grasslands and shrublands. Mechanical treatments, such as

mastication, chipping, piling by hand or machine, and compaction, as well as burning treatments including piling and burning and broadcast burning, are all designed to reduce the amount of fuel available to burn in subsequent wildfires.

In recent reviews, Graham and others (1999; 2004; 2009) found abundant evidence that forest fuel treatments can reduce fire intensity and fire severity, and that treatment effectiveness varies with time, location, and time since treatment. In 2007, many fuel treatments were subjected to wildfires and multiple case study assessments judged them effective. That fuel treatments work to alter fire behavior, make fire suppression easier, and make fires less severe is supported by simulation modeling (Pollet and Omi 2002) and recent case studies using remote sensing and field assessments where 2007 fires burned into treated areas (Fites and others 2007; Harbert and others 2007; Murphy and others 2007; Martinson and others 2008; Hudak and others 2011). Fuel treatments challenged by 2007 fires, many of which burned under relatively extreme conditions, were generally judged successful unless treatments were not complete or homes were readily ignited by burning embers. Wimberly and others (2009) found similar results. Continued effectiveness will depend on maintenance and retreatment (Graham and others 2009). Hartsough and others (2008) review the costs of alternative fuel treatments for dry forests.

While there is general agreement that removing and reducing fuels will reduce fire intensity, not all agree that fuel treatments work. They are likely more successful in dry forests and adjacent to buildings (Graham and others 1999; Graham and others 2004; Graham and others 2009), but it is unclear how this will change with climate. Until the recent case studies, many assessments have been qualitative or based on simulation models with little empirical data. Rhodes and Baker (2008) argued that fuel treatments were unlikely to be burned. Using extensive fire records for Western U.S. Forest Service lands, they estimated that a given fuel treatment had a 2-8% probability of being burned in a moderate- or high-severity fire within 20 years of implementation. Thus, it is important to ensure that fuel treatments are ecologically appropriate, socially acceptable, and feasible as vegetation management treatments (Schoennagel and others 2004; Graham and others 2009). Further, fuel treatments seldom stop fires, though fire fighters can effectively use them in fire suppression efforts. There is general agreement that fuel treatments immediately adjacent to homes are more effective (e.g., in the home ignition zone, Cohen 2000), and that fuel treatment effectiveness will vary among forest ecosystems and with the fire behavior and weather (Schoennagel and others 2004). Further, what is ecologically appropriate, sustainable, and socially acceptable will vary from place to place (Graham and others 2009).

Bark Beetles and Burn Severity

Insects, including bark beetles and defoliators, are major disturbances that along with fire, wind, and human action have shaped forest composition, pattern, and structure. Landscapes are dynamic, and most places are in some state of recovery from disturbance. Interactions between insects and fire as agents of forest disturbance have many implications for landscape dynamics, carbon, sustainability, and resilience, but the interactions are poorly understood, especially at landscape scales. Although we generally lack good historical data on the extent of tree mortality from diseases and insects, the area affected is increasing and will likely continue to increase. Between 1997 and 2001, the five-year trend ranged between 2 to 3 million acres affected per year in U.S. forests. From 2002 to 2007, extensive tree mortality occurred on approximately 5 to 12 million acres per year (NWCG [National Wildfire Coordinating Group] 2009). Similar trends are apparent around the globe (Allen and others 2010).

There is strong agreement from observational and modeling studies that extensive tree mortality due to bark beetles, defoliators, and other agents can affect the available fuels and crown fire behavior (Schoennagel and others 2004; Jenkins and others 2008; Hoffman and others 2011; Simard and others 2011). However, there are conflicting reports in the peer-reviewed literature on the influence of insect-induced changes to stand structure on resulting fire behavior (Jolly and others 2012b) and outcomes are less certain for burn severity. Many people think trees killed by insect outbreaks are more likely to burn and burn severely (Geiszler and others 1980; Knight 1987) because foliage is drier and more flammable (Jolly and others 2012a) and there are more fine fuels on the ground and eventually more large wood (Page and Jenkins 2007; Jenkins and others 2008). Others argue the opposite (Bigler and others 2005; Simard and others 2011) because they feel that the species composition and forest structure, especially less continuous fine crown fuels, following insect outbreaks are less conducive to fire occurrence and spread (Veblen and others 1994; Bebi and others 2003; Kulakowski and others 2003). Likely, crown fire hazard is high where the proportion of trees killed by bark beetles is high and the red needles are still in the trees, but then the hazard decreases as red needles fall (sometimes called the gray stage) (Hoffman and others 2011). Tree mortality could also increase fire hazard by increasing amount of solar radiation in the subcanopy, which is not only effective in drying surface fuels (Hoffman and others 2011) but also in causing earlier snowmelt (Pugh and Small 2011). The effects of tree mortality vary rapidly with time since outbreak and likely vary with site (Page and Jenkins 2007). Fire-induced tree injury could favor insect attack of stressed trees (McCullough and others 1998), and this was evident in some studies (Bradley and Tueller 2001; McHugh and others 2003; Wallin and others 2003; Cunningham and others 2005), but not others (Elkin and Reid 2004), or no relation was found (Sanchez-Martinez and Wagner 2002).

Lynch and others (2006) found that the extent of the 1988 fires in Yellowstone National Park were related to bark beetle outbreak 13-16 years prior (but not to bark beetle outbreak 5-8 years prior), drought, and aspect. Prior mountain pine beetle-induced tree mortality increased the odds of burning by 11%. They concluded that for fires following mountain pine beetle outbreaks, the effect of the changed stand structure and composition (increased understory vegetation) resulting from canopy mortality were more important than the increase in fuels. Bigler and others (2005) found that prior stand structure that resulted from multiple disturbances, including bark beetles, affected burn severity in spruce forests in Colorado. Kulakowski and others (2003) found that areas affected by a 1940s spruce beetle outbreak burned less often by a 1950 fire than would be expected at random. It is very likely that this effect varies with drought severity, insect species and associated tree species, and the extent and timing of mortality relative to time of the burn. It is likely that these relationships vary with insects, time since outbreak, severity of the outbreak, and climatic conditions (Jenkins and others 2008). Hoffman and others (2011) found that fire intensity and crown consumption increased with level of mortality in mixed conifer stands.

Key Uncertainties

The many interactions among fire, vegetation, topography, land use, and climate, and between fire and other disturbances, will likely lead to non-linear, synergistic, and unexpected effects as climate changes. The effectiveness of post-fire management, including salvage and rehabilitation, is poorly studied, especially in streamside areas. There are many unanswered questions. How will projected future

changes in vegetation composition and structure associated with climate warming and disturbance influence future fire extent and ecological effects? At what point do landscapes become fire-limited? In other words, when do we have enough past wildfires and fuel treatments such that future fires become self-limiting? How will severity and spatial patterns of fire change as fires become more extensive with climate warming? How will ecological effects of shifts in seasonality of burning (e.g., earlier fires) influence ecological effects? Despite these uncertainties, fires are occurring and decisions about management strategies must be made before and during fires.

Fire Management Strategies

Fire management, including suppression and management of intentional fires and lightning-ignited fires, fuel treatments, and other vegetation management treatments, are widely applied, often with the goal of altering the size and severity of subsequent wildfires. These actions can be taken at local to landscape scales. Fires respond to and interact with the vegetation dynamics that are often a legacy of past disturbance.

Fire managers and scientists now widely acknowledge the critical and natural role fire plays in most terrestrial ecosystems. The last two decades have seen a paradigm shift from a suppression orientation to an acknowledgement of the need to use wildland fire as a whole-landscape management tool. This shift in thinking acknowledges that in the western United States, where the scale of land use change due to fire exclusion has been large relative to the size of stand treatments, use of larger, naturally occurring fire is essential. However, fire management is complicated by many of the factors described above, including climate change and growing western U.S. populations in fire-prone areas.

Many efforts are focused on homes to reduce the likelihood that they will ignite in a rain of embers when surrounding wildlands burn (www.firewise.org, Cohen 2000). If homes were less likely to ignite, more different fire management strategies would be possible. However, designing “fire-smart” landscapes that are resilient to the effects of fire on both ecological and social systems is challenging, both from a technical and social perspective. Increasing public awareness of the importance of fire as a natural process, and the individual homeowners’ role in facilitating fire management, will be key to the successful implementation of fire use strategies in the future.

Effective fire management at the landscape scale will require thoughtful assessment and means to take advantage of past fires, prescribed fire treatments, and local topography and other conditions to understand and manage at the landscape scale if we are to address the implications of fires for streams and fish, and for the many other landscape values. Likely, thinning or other fuels treatments alone will not be enough to alter the size, severity, and occurrence of fires. Increased use of naturally occurring fires is helping managers meet the need for fuel reductions in some places, and there is a need to resolve risks and challenges in others to open opportunities to take advantage of natural fires in others.

The fire management challenges are many. Fire organizations are under intense pressure to reduce costs and ensure the safety of fire personnel while protecting people and property, addressing smoke impacts on human health and visibility, and realigning public perceptions about fire and fire impacts. They do so through concerted efforts before, during, and after fires. Initial attack is largely successful, so the few large fires burn under very hot, dry, and windy conditions. This reinforces perceptions that fires are always large, intense, and threatening to people

and their property or other valued resources. Many of these large fires are managed for months, and when there are some large fires there are typically many large fires, severely challenging fire suppression resources and budgets to pay for them. Since 2000, fire managers have sought to provide strategic, comprehensive plans. These will be even more important in the near future given implications of projected climate change, drought and fuel conditions, demographic shifts in human society, public expectations in the wildland urban interface, budget limitations, and demand for fire suppression resources to respond to other natural disasters (NWCG [National Wildfire Coordinating Group] 2009).

Summary

It is quite likely that fires will mediate the effects of climate change on forests and associated aquatic ecosystems. How forests will respond is uncertain given that the effects of climate are both direct and indirect, but we can expect to see changes in where, why, and when fires burn. Ecological effects of those fires will vary depending on where and when fires occur. Whether and how forest species will adapt depends on how climate variability affects them and the extent, frequency, and severity of fires. Practical solutions depend on framing constructive approaches that facilitate future ecological and social resilience to those fires.

B. Riparian Forests, Climate Change, Fire

This brief synthesis focuses on characteristics that differentiate riparian areas from uplands in considerations of fire, forests, and climate change. The valued habitat functions provided by riparian vegetation are discussed, as well as how these may change with shifting climate and management actions. The role of natural and human disturbance in shaping riparian communities is described, with emphasis on the role of fire.

Riparian Vegetation, Values, and Connection to Streams

Uniqueness and Natural Variability of Riparian Vegetation

Riparian plant communities are frequently the most floristically and structurally diverse vegetation in a given region (Naiman and others 1993; Naiman and others 1998; Pollock and others 1998; Tabacchi and others 1998; Naiman and others 2005). Stream-riparian corridors are characterized by multidimensional spatial gradients that change within a watershed in response to elevation, aspect, lithology, stream size, and local and regional geomorphology and hydrology (Naiman and others 2005; Wohl and others 2007). Streamside vegetation reflects these local physical features (Baker 1989; Friedman and others 2006). Because of their transitional location at the land water ecotone, riparian vegetation may include upland, riparian, and wetland species, and a range of life forms and functional groups (Pollock and others 1998). High levels of biodiversity in riparian areas are maintained by spatial habitat heterogeneity (Pabst and Spies 1999; Sarr and others 2005).

The diversity of riparian areas is also attributed to the temporal variability in natural disturbances, such as floods, debris flows, landslides, and wildfire (Gecy and Wilson 1990; Naiman and others 2005). Hydrogeomorphic disturbances, including seasonal variability of flow and sediment erosion, transport and deposition

contribute to the shifting mosaic of physical landform patches and associated biotic communities along stream-riparian corridors (Poff and others 1997; Corenbilt and others 2009; Merritt and others 2009). Successional patterns of riparian plant community development are driven by responses to natural and anthropogenic disturbances, physical variables, and plant species attributes (Baker 1989; Merritt and others 2009). There are also feedbacks between riparian plant species and the physical environment. These involve plant features that influence sediment deposition and accumulation and lead to biostabilization of streambanks and floodplains. Riparian plant characteristics include mechanical resistance and flexibility, root anchorage ability, and post-disturbance regeneration via sprouts and seedlings that influence sediment deposition and accumulation (Petitt and Naiman 2007; Corenbilt and others 2009). Thus, the diverse composition and structure of riparian vegetation are a result of the interdependence of physical and biotic processes over time (Bennett and Simon 2004)

The natural variability of riparian plant communities can pose management challenges and is apparent in the many classifications that have been developed for national forests and states in the western United States (e.g., Hansen and others 1995; Manning and Padgett 1995; Crowe and Clausnitzer 1997; Carsey and others 2003). Most classifications are based on plot-level vegetation sampling but indicate the dependence of streamside plant distributions on elevation, hydrogeomorphic features, landscape position, and location within watersheds. These classifications have served as management tools, and may be useful in determining the vulnerability of some riparian community types to climate change. Current challenges for riparian management include: (1) the integration of existing riparian classifications with developments in landscape ecology that highlight the role of landscape position and location within watersheds; (2) prediction of changes to riparian vegetation in response to climate-related shifts in temperature and precipitation given local and regional characteristics, watershed condition, and disturbance regimes; and (3) maintenance of valued riparian functions.

Valued Functions of Riparian Plant Communities

Riparian areas cover a relatively small area in any given watershed, yet they provide critical ecological functions (Brinson and others 2002; Naiman and others 2005). They are disproportionately important for maintenance of water quality and quantity (water storage and aquifer recharge), habitat for aquatic and terrestrial biota, sediment retention, stream bank building and maintenance, and provision of services of economic and social value (Gregory and others 1991; Prichard and others 1993; revised 1995, 1998; Naiman and Decamps 1997; Brinson and others 2002; Naiman and others 2005). The focus here is on the contribution of riparian vegetation to the maintenance of aquatic habitat for native fishes, specifically: (1) provision of shade for thermal modification of stream temperature; (2) inputs of large wood for instream habitat complexity; (3) allochthonous organic matter inputs to aquatic food webs; and (4) provision of streamside habitat and stabilization of streambanks. Each of these functions could be altered at the reach scale with changes in riparian vegetation, including short-term responses to fire and longer-term responses to changing climate.

Stream temperature: Along many stream segments, riparian vegetation attenuates the input of solar radiation. Direct sunlight warms streams, particularly during periods of low flow. During winter, lack of cover can affect stream temperature by permitting radiant cooling to the sky, potentially resulting in the formation of anchor ice (Ashton 1989). Riparian and topographic shading moderates these

thermal fluctuations. Stream temperature has tremendous ecological importance for aquatic biota and for ecosystem processes such as productivity and nutrient cycling (Sweeney 1992; Allan and Castillo 2007; McCullough and others 2009). Water temperature strongly influences growth, development, and behavioral patterns of aquatic biota directly and because of its influence on dissolved oxygen concentrations (Sweeney 1993; McCullough and others 2009). Stream temperature is an important factor determining the distribution of fish in freshwater streams, and most species of concern have limited temperature tolerances (Torgersen and others 1999; Dunham and others 2007; Isaak and others 2010).

Stream water temperature varies markedly within and among stream systems (Poole and Berman 2001; Caissie 2006). Natural influences on water temperature include topographic shade, upland and riparian vegetation, ambient air temperature and relative humidity, altitude, latitude, discharge, water source, and solar angle and radiation (Poole and Berman 2001; Ebersole and others 2003). Various approaches to modeling stream temperature have been developed; in general, these either examine components of an energy budget with deterministic models, or develop regression or stochastic models based on relationships between air and water temperatures (Caissie 2006). Whatever approach is used, riparian vegetation is implicitly included in the radiation terms, since riparian shade protects streams from excessive heating or radiation. For the upper Boise River basin, western Idaho, Isaak and others (2010) developed a series of multiple regression models to determine the relative importance of input variables on summer stream temperature (means and maxima). Consistent with other studies, they found that three critical input variables were air temperature, stream flow (which together describe the time variation in temperature), and radiation (the most significant geographically varying quantity). They also evaluated the role of fire on stream temperature and found that stream temperature increases over the study period averaged 2-3 times greater within the burned portions of watersheds than basin averages, and that increases in radiation accounted for 50% of the warming. These results highlight the role of both upland and riparian vegetation in moderating incoming radiation and reducing stream temperatures, particularly following fire.

Effectiveness of vegetation in providing stream shade varies with topography, channel size and orientation, extent of canopy cover above the channel, and vegetation structure. Streams in different regions and stream segments in different parts of a basin vary in response and sensitivity to disturbance and human activities that alter vegetative shading (Poole and Berman 2001). However, stream shading by riparian and upland vegetation is one of the few factors that can be actively managed to achieve stream temperature targets, as reflected by riparian Best Management Practices and designation of riparian buffer widths (Beschta and others 1987; Belt and others 1992). With predictions of rising stream temperatures in response to changing climate and increased incidence of fire, more focus will be directed toward manipulation and restoration of riparian vegetation to increase shade (Davies 2010; Furniss and others 2010). An important aspect of prioritizing future restoration efforts will be to identify stream reaches where increasing or maintaining riparian shade could protect or extend the longitudinal influence of cold groundwater influxes (Isaak and others 2010). Potentially, extension of fuel reduction treatments into riparian areas may also reduce the fire risk or decrease the severity of wildfires along stream-riparian corridors. Although reduction of riparian fuels may reduce effective shade in the short-term, i.e., for several post-treatment years, vegetative recovery following treatment (or wildfire) may proceed more quickly and vigorously and prolong shade benefits over decades.

Inputs of large wood for instream habitat complexity: Over the last three decades, extensive literature has documented the hydrological, ecological, and geomorphic effects of instream large wood and reported on the role that large wood plays in linking aquatic, riparian, and upland portions of watersheds (Lienkaemper and Swanson 1987; Bilby and Bisson 1998; Gregory and others 2003a). Large wood strongly influences channel form in small streams, creating pools and waterfalls and affecting channel width and depth (Montgomery and others 2003). Many aquatic species use pools formed by large wood as habitat and in-stream wood for cover (Bilby and Bisson 1998; Wondzell and Bisson 2003). The presence of large wood in streams affects erosion, transport, and deposition of sediment, as well as the creation and growth of gravel bars and channel and floodplain sedimentation (Montgomery and others 2003). Dams formed by accumulations of large wood increase channel complexity and facilitate deposition of organic matter, thus providing a food source for numerous invertebrate species and contributing to nutrient cycling and retention (Bilby and Bisson 1998; Wondzell and Bisson 2003). The influence of wood in affecting stream morphology depends on the size of the stream and the size of the wood pieces (Bilby and Ward 1989; Marcus and others 2002; Wohl and Jaeger 2009). The function of LW in forming fish habitat, especially plunge and dammed pools, is strongly influenced by the location of the stream or reach within a given watershed (Richmond and Fausch 1995).

Less well documented are the different processes of wood recruitment, retention, transport and turnover, and longitudinal distribution of wood pieces and jams within stream networks (but see May and Gresswell 2003; Wohl and Goode 2008; Wohl and Cadol 2011). These are important considerations for estimation of instream large wood targets and the long-term management of streamside forests and in-channel habitat. Chronic inputs of large wood to stream channels occur as a result of bank erosion, windthrow, and mortality of individual trees from adjacent hillslopes and riparian areas (McDade and others 1990; Bragg 2000; Benda and others 2003b; Reeves and others 2003). Large pulses of wood may originate from near channel sources following fire (Figure 24), windthrow, or insect infestations, or pulses may be transported from other portions of a watershed by debris torrents, avalanches, or landslides (Bilby and Bisson 1998; Bragg 2000; Benda and others 2003b). The relative importance of chronic LW inputs vs. episodic, disturbance-related inputs varies in time and space (Benda and others 2003b) and is reflected in wood distribution at multiple scales. In one of the few empirical studies to quantify the longitudinal distribution of instream LW, Wohl and Jaeger (2009) surveyed wood pieces in 50 contiguous stream segments, each segment 25 m in length (total surveyed length = 1250 m per stream) along 12 streams in the Colorado Front Range. Their results suggested that local valley and channel geometry, i.e., valley-bottom width, gradient, and sequence of channel changes, exerted a stronger influence on patterns of longitudinal wood distribution than either time since last forest disturbance or progressive downstream trends associated with larger drainage area. They also found that the combination of forest stand age, longitudinal sequences of wood recruitment sources (hillslope and riparian), and channel geometry significantly influenced reach-scale wood loads and aggregation patterns. These findings represent one point in time; the temporal variation in LW loads, which includes disturbance-related inputs and wood movement, creates additional complexity.

Retention and transport of instream LW depends on wood piece dimensions, notably diameter and piece length relative to channel width, stream flow regime, and channel characteristics. Reported values for wood residence time in streams vary from weeks to centuries (Wohl and Goode 2008), although residence time of log

Figure 24a: Post-fire inputs of large wood (source=hillslope) to Boulder Creek, Bridger-Teton National Forest, Wyoming. Photo taken in 2007, seven years following the Boulder Fire (2000). At the time of the photo, approximately 75% of the hillslope and riparian 'recruitable wood' had entered the stream along this reach.



Figure 24b: Post-fire inputs of large wood (source = riparian) to Boulder Creek, Bridger-Teton National Forest, Wyoming. Photos were taken in 2007 (left) and 2011 (right), 7 and 11 years following the Boulder Fire (2000). In 2011, approximately 90% of the riparian 'recruitable wood' had either entered the stream channel or fallen on the floodplain along this reach.

jams is longest in small headwaters and tends to decrease with increased drainage area (Martin and Benda 2001).

In second and third order streams, however, most researchers have reported fairly rapid turnover (<10 years). In five Colorado mountain streams, Wohl and Goode (2008) found that reach-scale wood loads and logjam locations remained relatively constant during an 11-year monitoring study. Although results from other regions vary, instream LW has been shown to be mobile and dynamic, and the physical factors influencing in-channel wood distribution and loads are similar (Lienkaemper and Swanson 1987). Most published studies have presented data on chronic inputs; because tracking individual LW pieces is time-consuming and labor-intensive (Wohl and others 2010), few studies have monitored individual reaches beyond 10 years, particularly the fate of LW pieces following fire and other disturbances. Field surveys of short durations have assisted in defining natural variability in wood recruitment and storage for a few forest types, but questions about long-term dynamics, watershed patterns, and integration of disturbance processes—difficult to address based on sparse empirical data alone—have led to modeling efforts.

Instream LW dynamics have been simulated using deterministic and stochastic models that incorporate a range of recruitment, transport, and decay processes (Bragg 2000; Bragg and others 2000; Benda and others 2003b; Gregory and others 2003a; Gregory and others 2003b; Meleason and others 2003). Simulation models have been run at reach and watershed scales, using empirical or derived data on upland and riparian vegetation and terrain. Some have specifically included disturbances, notably fire, landslides and mass failure, forest harvest and insect outbreaks. Most models have been developed in the Pacific Northwest, reflecting the history of LW research and existence of empirical data for this region (Beechie and others 2000; Gregory and others 2003b). Model objectives have focused on recruitment dynamics; input variables include streamside forest attributes, rates of wood delivery to the stream, and depletion from decay, transport, and breakage. By necessity, existing models are very simplistic representations of riparian forests, and most do not address the role of channel characteristics on the distribution of wood. To date, model assumptions have not been well supported by empirical data. Despite these limitations, the development of quantitative wood supply models has highlighted the importance of riparian forest processes and improved understanding of the role of disturbance in LW recruitment to streams. Future model development and application will be necessary to predict and manage for instream LW over varying time periods, across stream networks, and with different scenarios that incorporate climate-related disturbances, including changes in streamflow regimes and fire frequency.

Conceptual models of LW distribution and dynamics are generally based on a simplified landscape view of stream networks or watersheds, classified into three dominant morphologies: high gradient, small headwaters; intermediate, 3rd and 4th order stream segments; large, low-gradient, meandering streams and rivers (Marcus and others 2002; Swanson 2003; Wohl and Jaeger 2009 and see text box on large wood dynamics). As noted above, vegetation, physical constraints, and natural hydrologic, sediment, and disturbance regimes differ markedly in these portions of river and riparian landscapes and strongly influence LW distribution and dynamics. Instream LW loads are generally highest in the headwater portions, where trees are large and small channel size and stream power limit mobility (transport-limited). In intermediate stream reaches, correlations have been documented between wood load and drainage area, elevation, channel width, bed gradients, and total stream power. Although few data have been collected over the required time periods, intermediate channels appear to display a dynamic equilibrium, where LW pieces are moved out at approximately the same rate that they enter the channel (Marcus and others

2002; Wohl and Goode 2008). In large, low-gradient streams and rivers, the spatial distribution of LW varies widely but is supply-limited due to reduced areal contact with riparian edges. This broad framework serves as a starting point for estimating reasonable LW targets and rates of chronic recruitment, due to bank erosion and mortality. Although more challenging, it may also prove useful in assessing the role of shifting climate-related disturbance regimes in the delivery and movement of in-stream wood.

Contributions to aquatic foodwebs: Organic matter in streams can either be produced by aquatic organisms (autochthonous) or enter the stream from other sources (allochthonous) (Allan and Castillo 2007). Autochthonous organic matter is generated through photosynthetic production by autotrophic organisms of the aquatic community (vascular plants, bryophytes, algae, bacteria, and protists) and is driven by the amount of light reaching the stream surface. In contrast, allochthonous organic matter originates directly from riparian or upland vegetation in the form of leaves, twigs, and other fine litter and indirectly as terrestrial invertebrates (Bisson and Bilby 1998). The input, use, retention, and transport of allochthonous organic matter in streams frequently drive carbon and nutrient dynamics and affect aquatic biota (Webster and Meyer 1997). For many low order streams in forested watersheds, the energy for aquatic food webs is largely derived from allochthonous inputs (Vannote and others 1980; Newbold and others 1982). Allochthonous plant sources vary widely in nutritional quality and require different degrees of in-stream processing and conditioning by microbes and invertebrates (Webster and Benfield 1986; Allan and Castillo 2007). In some areas, seasonal inputs of terrestrial insects from riparian areas are an important food source for drift feeding fish species (Young and others 1997); such inputs are highest from closed canopy riparian areas dominated by deciduous plant species (Edwards and Huryn 1996; Nakano and others 1999; Baxter and others 2004; Baxter and others 2005). For floodplain forests, it has been suggested that the effectiveness of riparian vegetation in providing allochthonous inputs to streams declines at distances greater than approximately one-half a tree height away from the channel (FEMAT [Forest Ecosystem Management Assessment Team] 1993). Removal of riparian vegetation by fire reduces the amount and quality of allochthonous inputs and promotes autotrophic production by increasing available light (Bisson and Bilby 1998; Malison and Baxter 2010), causing shifts in the feeding guild composition of stream macroinvertebrate communities and changes in aquatic trophic pathways that affect fish productivity (Edwards and Huryn 1996; Bisson and Bilby 1998; Bisson and others 2003).

Streambank stability: Riparian vegetation can directly affect stream channel characteristics, particularly streambank habitat and stability (Gregory and Gurnell 1988; Davies-Colley 1997; Simon and Collison 2002; Pollen and others 2004). Root systems protect stream banks through armoring (Stokes and Mattheck 1996; Abernathy and Rutherford 2001) and bind bank sediment, thus contributing to bank stabilization, reduction of sediment inputs to streams (Dunaway and others 1994), and development and maintenance of undercut banks (Sedell and Beschta 1991). There are marked differences among riparian species and vegetation types in root characteristics and their influence on bank stability (Lyons and others 2000; Simon and Collison 2002; Wynn and others 2004). Management activities, such as logging and grazing, and natural disturbances, such as fire and debris flows, can directly affect stream bank stability through alteration of riparian vegetation. Removal of woody riparian vegetation with beneficial rooting characteristics can result in erosion of alluvial streambanks. Removal of herbaceous vegetation can decrease retention and accumulation of sediment, possibly influencing floodplain soil development (Thorne

Instream Large Wood (LW) Dynamics: Influence of Disturbance and Changing Climate

The numerous roles and benefits of instream large wood (LW) have been extensively documented in the recent fluvial geomorphology, forestry, fisheries, and stream ecology literature. In smaller streams, large wood pieces strongly influence channel form, creating pools, log steps, waterfalls, and affect both stream width and depth. In second order and larger streams, channel features formed by LW, such as pools and eddies, provide critical habitat for native fish species. Aquatic species also use in-stream logs as cover, feeding and colonization substrates. Instream LW contributes to the growth of sandbars and gravel bars, affects the erosion, transport and deposition of sediment, and increases channel complexity. Due to these benefits and the consequent need to manage for instream LW, the amounts, characteristics and physical controls on instream wood loads have also been studied. Less well documented, however, are the timing and processes of wood recruitment, transport and turnover, and the ever-shifting longitudinal distribution of wood pieces and jams within stream networks. Although some aspects of LW dynamics have been modeled, models predicting instream wood supply and transport are not generally available for most regions. Challenges for managers are the assessment of current wood loads for their streams of concern, setting targets for loads and distribution over stream segments (meters to kilometers) and different time scales (years to centuries), and the incorporation of a disturbance perspective in plans for future management.

Conceptual models of LW distribution and dynamics are generally based on a simplified landscape view, which classify stream networks or watersheds into three dominant portions: high gradient, small headwater streams at the highest elevations; intermediate, 3rd and 4th order stream segments; and large, low-gradient, meandering streams and rivers (Fig. T6-1). Vegetation, physical constraints, and natural hydrologic, sediment, and disturbance regimes differ markedly in these portions of stream-riparian corridors and exert strong controls on LW distribution and transport. Instream LW loads are generally highest in the headwater portions, where trees are large and small channel size and stream power limits mobility

(transport-limited). In intermediate stream reaches, correlations have been documented between wood load and drainage area, elevation, channel width, bed gradient and total stream power. Wood load decreases downstream, although the frequency of wood jams, or accumulations, is highest along these intermediate stream portions. In large, low-gradient streams and rivers, the spatial distribution of LW varies widely, but is frequently supply-limited due to reduced areal contact with forested riparian edges. Disturbance processes are integral to the understanding and management of instream LW, and the relative importance of different disturbance types also changes with landscape position. In headwaters and many intermediate reaches, fire, windthrow, landslides, and debris flows are responsible for delivering large pulses of wood to streams. In intermediate and low gradient portions of channels, flooding, water-logging and lateral channel migration influence LW inputs, redistribution and accumulation patterns. Anthropogenic disturbances that can affect delivery of LW to streams, such as mining, timber harvest, and road construction, also differ with elevation and location within a watershed.

At the reach or stream-segment scale, simple wood budgets can be estimated through calculation of inputs and outputs, which forces consideration of the relative contribution of different disturbances for recruitment, breakage, and movement of LW pieces. The conceptual framework of watershed-level LW distribution can be combined with estimated wood budgets for longitudinal segments and thus serve as a starting point for setting LW loading targets. Management and restoration of instream LW involves linking riparian forest processes with hydrology and fluvial geomorphology; planning for impacts of climate change requires consideration of altered disturbance regimes and the responses of riparian vegetation. Indirect outcomes of climatic warming will be (1) changes in LW recruitment to streams, including greater near-term inputs following fire or blow-down, and possible decreased inputs in the long-term; (2) for some stream segments, the size and species of trees entering streams will change, with fewer large, old-growth trees; (3) changes in transport rates and seasonal redistribution of instream LW, in response to shifting flooding and drought cycles; (4) local changes, such as increased frequency of debris flows, depending on lithology and precipitation patterns.

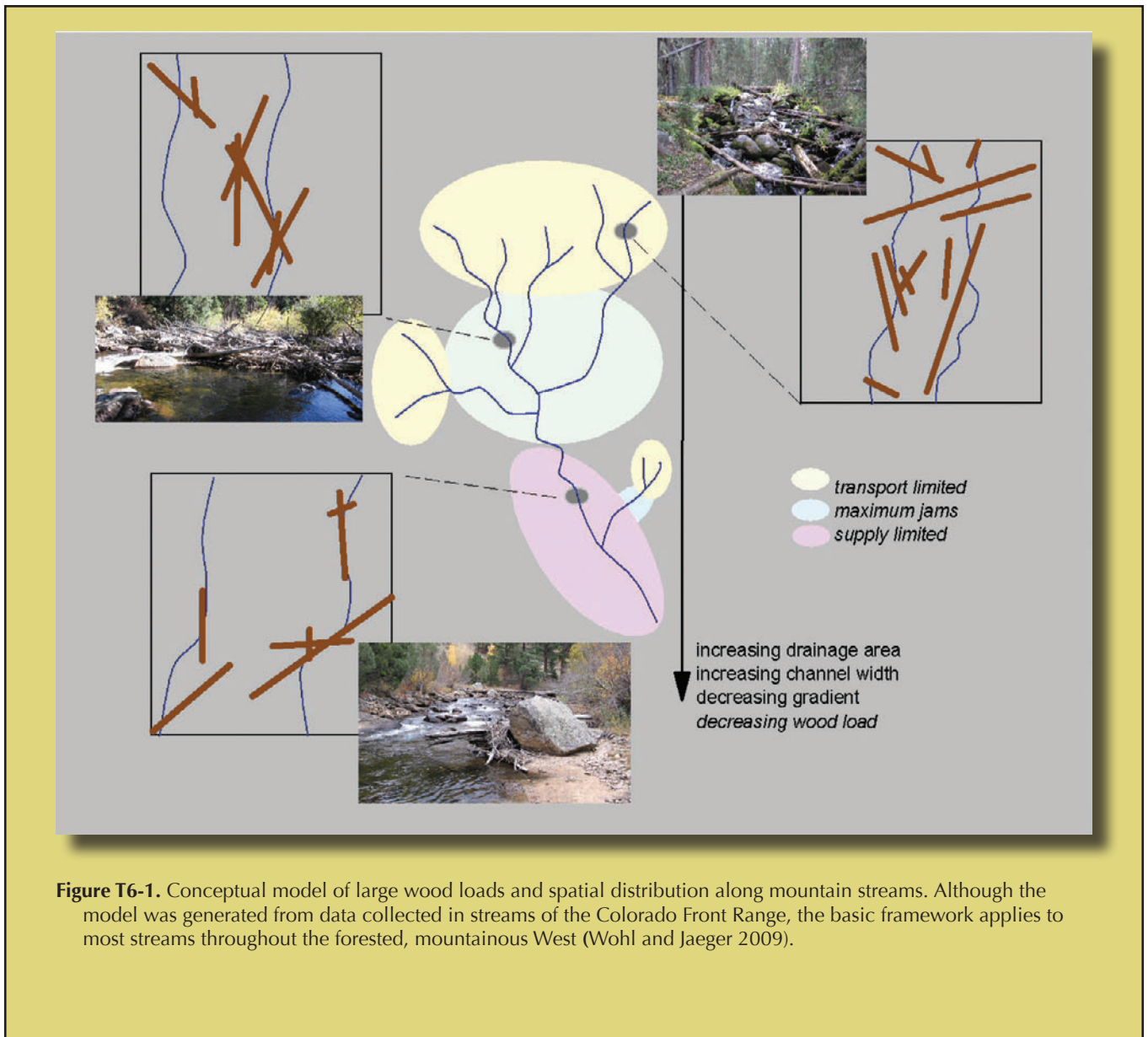


Figure T6-1. Conceptual model of large wood loads and spatial distribution along mountain streams. Although the model was generated from data collected in streams of the Colorado Front Range, the basic framework applies to most streams throughout the forested, mountainous West (Wohl and Jaeger 2009).

1990). Impacts of local alterations to riparian vegetation that affect bank stability and other geomorphic processes may have effects that extend downstream.

Disturbances in Riparian Areas

The development and maintenance of riparian environments are largely regulated by physical processes and natural disturbance regimes (Naiman and others 2005). Stream and river systems are naturally dynamic, changing at multiple spatial and temporal scales, frequently in response to episodic disturbance events. Geomorphic and hydrologic processes, including disturbances such as flooding and debris flows, have largely shaped streamside environments. Riparian plant species exhibit a range of adaptations that contribute to rapid recovery of streamside habitat after disturbance (Dwire and Kauffman 2003; Merritt and others 2009). In this section, the following natural disturbances, their interactions, and their influence

on riparian areas are discussed: flooding, fire, debris flows, insects, and beaver. A brief overview of the impacts and legacy of land use and management, i.e., human disturbance, is also presented.

Flooding

Riparian environments are intrinsically linked to the dynamics of stream hydrographs, including flooding. Stream hydrographs show the seasonal and interannual variability in flows, and they display characteristic forms depending on the local climate, particularly precipitation patterns, and the size and shape of the watershed. Hydrograph peaks correspond to flood events that inundate floodplains, scour streambanks, and transport sediment and large wood onto bars and floodplains. Many mountain streams are strongly influenced by spring snowmelt and display distinct peaks during spring runoff (Stewart 2009). Low- and mid-order streams, and the riparian environments bordering them, are sensitive to individual precipitation events, resulting in dynamic hydrographs characterized by multiple peaks (floods) over a year. Larger rivers and their riparian environments are less sensitive to individual precipitation events because the scale of the basin usually surpasses the size of the storm. Also, flow in larger rivers integrates the flow of upstream tributaries, some of which may not be flooding. Some arid-land streams are intermittent or ephemeral, without surface flow for extended periods; their hydrographs reveal seasonal floods, such as those associated with monsoonal rainfall (Stromberg and others 1993). Floods in headwaters initiate flood waves that propagate as they travel and accumulate in downstream sections. Thus, the same flood event will affect riparian environments in distinct ways depending on location within the watershed, and flood impacts will differ in high-energy portions of stream networks relative to low-gradient, meandering portions (Bendix and Hupp 2000).

Four flood characteristics are important to riparian and floodplain ecosystems: magnitude, frequency, timing, and duration. Magnitude refers to the maximal discharge associated with an individual flood and reflects the intensity and severity of the event; variations in flood magnitude within a given watershed are expressed as recurrence intervals (Gordon and others 2005). The range of flood magnitudes for a given stream segment depends mainly on climate and the upstream catchment area. Frequency is the temporal pattern of flood recurrence, either over seasons or multiple years. Timing of floods is linked directly to precipitation or snowmelt runoff patterns. Flood duration is the amount of time that the riparian area (floodplain) is flooded, either seasonally or during individual flood events. Flood duration varies as a function of topography; low-lying areas close to channels flood first and are last to drain and thus experience longer flooding duration than other portions of the floodplain. Some aspects of these four flood characteristics are changing with shifting climate and are discussed in more detail in the earlier hydrology section. Many regions are already experiencing changes in magnitude, frequency, and timing of flood events relative to the period of record.

Flooding is an integral, essential disturbance for riparian ecosystems that has both geomorphological and hydrological hydraulic impacts (Hupp and Osterkamp 1996; Bendix and Hupp 2000). Hydraulic impacts include mechanical damage, saturation, and transport of sediment, organic material, large wood, and plant propagules. Geomorphological impacts include the shaping of fluvial environments. The structure, composition, and distribution of riparian vegetation are strongly related to fluvial geomorphological processes and forms. In many cases, species occurrences can be linked directly to specific fluvial landforms created by known

flood events (Rood and others 1998). Floods can erode streambanks and undercut, topple, and remove standing riparian vegetation. Entrained wood and debris can batter riparian trees (Johnson and others 2000), and vegetation can be buried by sediment deposited by floodwaters. The mosaic of riparian vegetation can reflect the role of floods in the differential destruction of previous vegetation, distribution of substrates and geomorphic surfaces, and in the transport of propagules. Depending on post-flood conditions and the climatic context, major floods can foster the establishment of vegetation stands or reset successional processes in riparian plant communities (Rood and others 1998). There are also feedbacks; streamside vegetation physically constrains flood flows, traps sediment and floating debris, and contributes to the erosional resistance of streambanks.

Streamflow regimes have exerted selective pressures on riparian plant species, resulting in morphological, physiological, and reproductive adaptations to flow attributes (Poff and others 1997; Naiman and others 2005; Poff and others 2007; Merritt and others 2009). Many riparian plants are specifically adapted to flooding, as well as sediment deposition, physical abrasion, and stem breakage associated with flooding (Karrenberg and others 2002; Naiman and others 2005; Merritt and others 2009). For example, the reproductive phenology of common riparian woody species, including cottonwoods and many willows, is synchronized to coincide with the seasonal hydrology and rainfall of specific regions (Mahoney and Rood 1998; Rood and others 1998). Cottonwood seed dispersal coincides with the seasonal retreat of floodwaters when moist seedbeds are available for successful germination and colonization. In addition to sexual reproduction by seeds, many riparian plant species reproduce by clonal growth (i.e., vegetative or asexual reproduction); multiple sprouts can result from burial during floods and abrasion during floods can stimulate stump sprouts (Karrenberg and others 2002).

The disruption of natural flow regimes through diversions, damming, withdrawals, and levees has focused attention on the dependence of riparian species on streamflow attributes and different portions of regional hydrographs. In the Rio Grande Valley (New Mexico), water withdrawal and flow regulation, including the cessation of spring floods, has simplified the valley, which transitioned from a mosaic of multiple channels, marshes, wet meadows, and forests to a system constrained by levees bordered by a narrow width of riparian forest (Molles and others 1998). Similar examples are common throughout the western United States and elsewhere worldwide. As the U.S. population continues to grow, increasing demands are being placed on water originating or flowing through Forest Service administered lands. Managing the limited water supply to meet multiple and sometimes competing uses is an ongoing and complex responsibility. Efforts to provide water for multiple uses include defining baseline environmental instream flow prescriptions that sustain and regenerate riparian habitats and communities. Characterizing environmental flows includes a flood component and also addresses flow requirements for channel maintenance, in-channel habitat, and maintenance of water quality (Richter and Richter 2000; Rathburn and others 2009).

Fire

Wildfire has played a critical role in shaping ecological heterogeneity across landscapes of the West (Agee 1993). Fire has also influenced the species composition, structure, and environmental conditions of the riparian and aquatic communities associated with stream networks that drain these landscapes (Gom and Rood 1999; Gresswell 1999; Everett and others 2003; Skinner 2003; Reeves and others 2006; Pettit and Naiman 2007; Stromberg and Rychener 2010). Research on riparian

fire frequency and severity has primarily been conducted in forests of the Pacific Northwest (see text box on fire histories in riparian areas). However, results are consistent with observations elsewhere and indicate that most riparian areas burn either similar to adjacent uplands or less frequently and more moderately than uplands. Reviews have summarized research on the role of fire as a natural disturbance in stream-riparian ecosystems, especially in mountainous environments (Bisson and others 2003; Dwire and Kauffman 2003; Pettitt and Naiman 2007); recent work has advanced understanding of post-fire recovery in different settings (Mellon and others 2008; Jackson and Sullivan 2009; Malison and Baxter 2010).

Different scenarios of generalized fire behavior and effects in riparian areas have been proposed. Pettitt and Naiman (2007) described four cases of fire effects, post disturbance impacts, and riparian recovery based on their observations of wildfire in Kruger National Park, South Africa. The four cases were categorized by stream gradient (high or low) and amount of rainfall (high or low). Halofsky and Hibbs (2008) developed a sequence of hypotheses to test the relative effect of riparian vegetation, valley bottom topography, and upland fire variables on riparian fire severity. The relative role of these driving factors varies locally and regionally but can be used to predict how wildfire may burn along specific stream segments. Key considerations address the connection to the larger landscape and include: location within the watershed relative to precipitation regime (snow vs. rain influence, Wohl and others 2007); topography, such as aspect and shifts in stream gradient and slope relative to uplands; geomorphology, such as changing width of the channel and valley floor; and riparian vs. upland vegetation and fuel characteristics.

We present four generalized scenarios of fire behavior and effects in riparian areas and speculate about potential responses to climate change (Table 3). Variations of these four scenarios occur and different combinations may be observed in the same watershed or during the same wildfire. The relative likelihood of occurrence for any scenario is largely driven by vegetation and fuel indicators, basic topographic variables, and characteristics of the fire and fire weather.

Riparian areas burn like adjacent uplands: This scenario is most likely to occur along stream reaches where the riparian vegetation, terrain, and general topography are similar to uplands. Stream reaches that drain shrub-dominated portions of drainage networks, such as shrub-steppe ecosystems throughout the portions of the Great Basin, or stream segments that drain the lower parts of stream networks in shallowly dissected terrain with low local relief, are likely to burn as frequently and severely as adjacent uplands. Other examples occur in the upper portions of drainages at high to moderate elevations in fairly steep terrain with steep stream valleys. This scenario could also occur under conditions of severe fire weather, i.e., when a large fire carries across the entire landscape and could overwhelm both the influence of local topography and vegetation differences between riparian and upland areas.

Riparian areas burn less frequently and/or less severely than adjacent uplands: In contrast to the above, this scenario is most likely to occur where riparian conditions are distinctly wetter or more mesic than upland vegetation. It is the most commonly documented scenario in the literature, especially for forests of the Pacific Northwest (please see textbox on riparian fire histories). In forested riparian reaches, particularly those located in deeply dissected terrain with north-facing aspects that foster cold-air drainage and cool riparian microclimates, fires tend to burn less “hot” and less frequently than nearby uplands. However, even within similar vegetation associations and in lower portions of drainage networks, the relative frequency of fire scars has been found to increase linearly with distance from the stream (Everett and others 2003; Skinner 2003).

Table 3. Four generalized scenarios of fire behavior in riparian areas. Variations on these four scenarios occur and different combinations may be observed in the same watershed or during the same wildfire. Ecological outcomes are given, as well as speculation regarding potential responses to shifts in temperature and precipitation regimes. Please see text for additional explanation.

Fire Behavior in Riparian Areas Generalized Scenarios	When and Where?	Ecological Outcome	Potential Responses to Changing Climate
Riparian areas burn like adjacent uplands, i.e., wildfires burn with similar frequency and severity.	When large fires burning under severe fire weather exceed the influence of local topography and riparian / upland vegetation differences; Where riparian vegetation, terrain and topography are similar to uplands.	Impact depends on fire severity, season, extent, but generally moderate to high. Slow to moderate recovery, via seedlings and resprouting. Fire adapted species will likely survive.	Likely to increase, i.e., more riparian area will burn similar to uplands; differences in riparian / upland burn patterns will decrease.
Riparian areas burn less frequently and/or less severely than adjacent uplands.	Where terrain fosters cold-air drainage, higher humidity, cooler microclimate relative to uplands; Where riparian vegetation is distinctly different from uplands (more hardwoods, higher herbaceous component, higher fuel moisture); Where saturated soil conditions, presence of riparian wetlands, or hydrologic inputs from hillslopes influence fire behavior. When fires burn with low intensity.	Low to moderate impact. Moderate to rapid recovery, via seedlings and resprouting. Fire adapted species (esp. conifers) will persist.	Likely to decrease; with rising temperatures and shifts in precipitation patterns, more riparian areas will burn similar to uplands.
Riparian areas burn more frequently and/or more severely than adjacent uplands.	Where fuel abundance/accumulation is higher in riparian areas than uplands (due to riparian management or natural conditions); When riparian areas serve as chimneys or corridors for fire spread, e.g., where steep terrain and narrow stream valleys influence fire behavior.	Impact depends on fire severity, season, extent; can be high impact, with destruction of most of the riparian community. Slow to moderate recovery, via seedlings and resprouting. Post-fire invasive species are a potential concern.	May increase or decrease; Locally dependent on fuel characteristics, and/or physical context; dependent on severity and behavior of fire event.
Riparian areas serve as fire breaks.	Where large perennial stream and river valleys create significant breaks in fuel characteristics and continuity; Where saturated soil conditions, presence of riparian wetlands, or hydrologic inputs from hillslopes influence fire behavior; When fires burn with low intensity.	Low impact; rapid recovery.	May increase or decrease; locally dependent on physical context; dependent on severity and behavior of fire event.

Fire History, Severity, Behavior and Burn Patterns In Riparian Areas Relative to Uplands

Ecologically diverse riparian corridors are maintained by active natural disturbance regimes, and fire has likely played an important role in shaping the heterogeneity of riparian vegetation communities. In many forested watersheds, riparian vegetation is comprised of a mosaic of different successional stages similar to the surrounding upslope landscape, suggesting that fire and other disturbances historically played a similar role in shaping both riparian and upland ecosystems.

The current understanding of fire return intervals in riparian areas, however, is limited to a few studies conducted in the Sierra Nevada Mountains (California) and Pacific Northwest region of the USA (Table T7-1). Results are varied and depend on the physical characteristics (slope, aspect) and disturbance history of the study watersheds, pre-fire moisture conditions, and degree of difference between riparian and upland vegetation. In general, fires are thought to occur less frequently within riparian forests than adjacent upslope forests. This is the case for mesic forests in Klamath Mountains, California, where fire return intervals in riparian areas were approximately twice as long as intervals in adjacent upslope forests (Skinner 2003). Fire return intervals are usually less frequent in streamside areas with notably different vegetation relative to uplands, especially higher densities of understory shrub and herbaceous species. In mixed conifer in the northern Sierra Nevada Mountains (CA), Van de Water and North (2010) found that fires burned fairly continuously across the landscape and were historically common in both riparian and upslope forests. At high elevations along low order streams, where the vegetation composition of riparian areas is similar to that of adjacent uplands, streamside areas are also likely to burn as frequently as the surrounding uplands.

Patterns observed for fire behavior, intensity, and severity in riparian areas are similar to those reported for fire frequency. Fires in riparian areas may be less severe, as severe, or more severe than in adjacent uplands, depending on the local topography, vegetation characteristics (especially fuel moisture and loading), and fire weather. Several recent studies have specifically addressed

fire behavior in riparian areas, contributing to current understanding of riparian burn patterns. In different conifer-dominated forests of Oregon, Halofsky and Hibbs (2008) compared fire severity in riparian vs. upland plots following the Biscuit Fire (Klamath-Siskiyou region) and the B&B Complex Fire (east Cascades). They found that the strongest predictors for riparian overstory fire severity were upland fire severity, riparian vegetation indicators, and local topography, including stream width and gradient, and adjacent hillslope steepness. Their study sites had diverse riparian understories, with varying levels of the deciduous tree and shrub component. Riparian understories generally burned less severely than upland understories, and the authors attributed this burn pattern to high fuel moisture in riparian vegetation, and cooler, moister streamside microclimates.

These plot-based results are consistent with a geospatial analysis that utilized remotely sensed Burned Area Reflectance Classifications (BARC) to compare upland vs. riparian burn intensity for four large fires (Fisk and others 2004): the Hayman and Missionary Ridge Fires in Colorado; Rodeo-Chediski Fire in Arizona; and the Stanford Fire in Utah. Authors found that riparian areas burned 'less hot' than upslope areas. However, riparian burn values related positively to upslope burn values; i.e., the hotter the watershed burned, the hotter (on average) the riparian areas burned. Results also indicated that smaller, lower order streams burned more like uplands, while riparian areas along larger, higher-order streams burned less like surrounding uplands. These studies are place-based; more information is needed on regional variation and for a range of riparian plant associations. However, the results largely support previous anecdotal observations regarding riparian burn patterns and relations between upland and riparian fire severity.

Looking forward, climate change predictions include rising air temperatures and shifts in precipitation regimes. In the western United States, these changes will result in longer summer droughts, potentially leading to smaller differences in riparian vs. upland fuel moisture in the short-term and reduction in the areal extent of riparian zones over time. More intense wildfires are also predicted, so differences in fire frequency, severity, and intensity in riparian areas vs. uplands will likely diminish.

Table T7-1: Fire return intervals for riparian versus upslope forests.

Location	Forest Type	Riparian Fire Return Interval (years)	Sideslope Fire Return Interval (years)	Citation
Blue Mountains, OR	Dry, Douglas-fir and Grand Fir series	13-36	10-20	(Olson 2000)
Elkhorn Mountains, OR	Dry, Ponderosa Pine, Douglas-fir series	13-14	9-32	(Olson 2000)
Salmon River Mountains, ID	Dry, Ponderosa Pine and Douglas-fir series	11-19	9-29	(Barrett 2000)
Cascade Range, WA	Dry, Ponderosa Pine and Douglas-fir series	15-26	11-19	(Everett and others, 2003)
Northern Sierra Nevada Mountains, CA	Dry, Ponderosa/ Jeffrey Pine series	10-87 ¹ (8-42 ²)	10-56 ¹ (6-58 ²)	(Van de Water and North 2010)
Dry Forest Type Average		12-36	10-31	
Cascade Range, OR	Mesic, Douglas-fir series	35-39	27-36	(Olson and Agee 2005)
Klamath Mountains, CA	Mesic, Douglas-fir series	16-42	7-13	(Skinner 2003)
Mesic Forest Type Average		26-41	17-25	

¹including only fire events recorded on two or more specimens at a given site

² Includes every fire event recorded on every specimen

*Table modified from Table 1 in Stone and others (2010)

Riparian areas burn more frequently and/or more severely than adjacent uplands: This scenario has been reported by the fire control/fire management community (Barrows 1951; Countryman 1971). It has been observed where steep terrain and narrow stream valleys create more heat and serve as chimneys or chutes that promote updrafts and convective heating of the fire, causing it to carry upslope and upstream at a rapid rate of spread with high intensity (Skinner 2003). This fire behavior is most likely to occur in the middle or upper portions of drainage networks with south-facing aspects, along small perennial or intermittent stream channels. Although we are not aware of research that has quantified the vegetative conditions that influence this fire behavior, we suspect that riparian vegetation is either (1) similar to upland vegetation in stand and understory composition and fuel characteristics; or (2) contains higher levels or denser fuel loads, particular ladder fuels, than adjacent uplands (Agee 1993). If fire suppression, “hands-off” riparian management, or natural processes have contributed to higher accumulations of fuel loads in streamside areas relative to uplands, and if pre-fire moisture levels are low due to drought or season, riparian fire severity may be greater than adjacent uplands. High riparian fuel loads, especially if uplands have been harvested or actively managed for fuel reduction, can influence fire spread by serving as

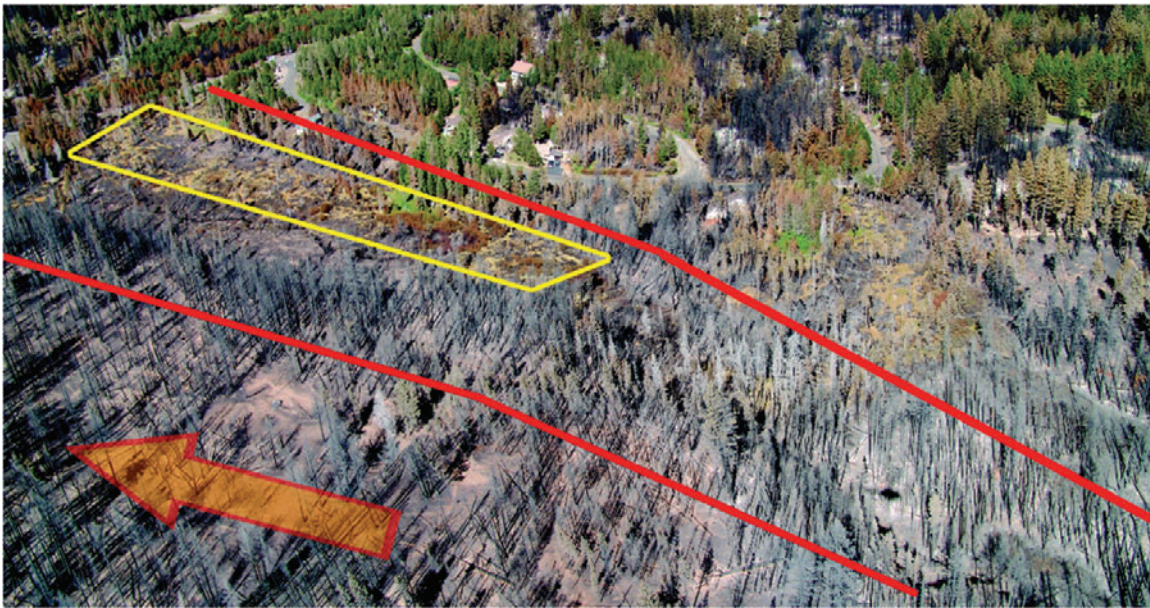


Figure 25: Stream Environment Zone (SEZ) along Angora Creek following the Angora Fire, Tahoe NF, California (2007). Dense, continuous stands of trees contributed to rapid spread rates (to the NNE) down this stream corridor. Arrow points in direction of wind and fastest fire spread (NNE). Note greater density of trees within the SEZ (roughly outlined in red). Moister portions of the SEZ (outlined in yellow) burned less severely than surrounding areas. (Photo originally published in USDA, R5-TO-025, August 2007.)

“wicks.” This fire behavior was observed during the Angora Fire, Tahoe National Forest, California, in late June 2007 (Murphy and others 2007). Prior to ignition, the Angora Creek Stream Environment Zone (SEZ, or riparian area) contained heavy dead woody fuel loadings. A retrospective evaluation of the Angora Fire behavior noted that “dense stands of trees in the Angora SEZ likely contributed to the rapid spread upslope to Angora Ridge and across the slope to the base of Tahoe Mountain” (Murphy and others 2007; Figure 25). This fire burned over 250 structures on private property, cost approximately \$160,000,000 in property loss and suppression costs, and has drawn attention to the role of riparian corridors and fuel conditions on fire behavior (Murphy and others 2007; Safford and others 2009).

This scenario is locally dependent on fuel characteristics, physical context, and the characteristics of a given fire event. However, the contributing riparian conditions may become more common with shifts in temperature and precipitation regimes. Although not well documented, riparian areas may also burn more severely in arid landscapes where frequent, low-intensity fires limit fuel abundance in uplands, while fuel accumulates in streamside areas. During periods of drought, differences in the riparian-vs.-upland microclimate and fuel moisture may be high enough to promote plant growth, stand development, and fuel accumulation in riparian areas, but not high enough to protect riparian forests from fire. This scenario is of particular concern for resource managers and fuels specialists in some locations in the Great Basin and Southwestern United States, where woody encroachment into riparian areas has increased streamside fuel loads.

Riparian areas serve as fuel breaks: This scenario is most commonly observed where large perennial stream and river valleys create significant breaks in fuel characteristics and continuity. Wide stream channels, alluvial terraces with extensive gravel bars, and large, sparsely vegetated areas with wet soils may function as fuel breaks. Other examples include wet meadows, stream segments with a high herbaceous component, and willow-dominated reaches or riparian areas

with a notable hardwood tree and shrub component. These meadow segments are frequently located in wider, lower gradient portions of stream networks that may receive significant hydrologic inputs (surface and subsurface) from surrounding hillslopes, resulting in saturated soil conditions and the presence of riparian or slope wetlands. They may be sites of past and current beaver activity that has modified the channel and flooded portions of the valley bottom. Saturated soils combined with high fuel moisture can stop the advance of fire or cause a fire to “jump” from hillslope to hillslope and not burn in the streamside area. In some cases, fire characteristics and upland conditions can influence the extent to which riparian areas function as fire breaks. If a fire is burning with low-intensity, riparian areas along low gradient, perennial streams may serve as effective barriers to fire spread.

Seasonality also plays a role in fire behavior and fire severity and may influence each of these scenarios. In mixed conifer stands of the Sierra Nevada Mountains (California), Van de Water and North (2010) found that depending on forest type, the majority of fire scars in both riparian and upland areas occurred during the late summer and fall. Later in the season, as trees become dormant, foliar moisture decreases, increasing the probability of a crown fire (Agee and others 2002). However, the ratio between the current year’s growth and older foliage influences moisture content as the seasons change. The ratio between old and new foliage depends on species and environmental factors such as elevation, site fertility, and light (Agee and others 2002). Drought cycles can also be credited for lower foliar and fuel moistures and have been correlated with increased fire occurrence. Although this correlation is stronger in uplands, riparian areas also experience more fires during times of drought (Van de Water and North 2010).

Debris Flows

In many headwaters and other steep, erosive landscapes, landslides, mass failures, and resulting debris flows are common natural disturbances. The occurrence of debris flows depends on topography, underlying geology, and soil and vegetation characteristics and is frequently associated with fire, past management activities (roads and forest harvest), and storm events. In mountainous areas, debris flows can play a major role in routing sediment and wood stored on hillslopes and in low-order channels and delivering it to higher-order channels (May 2002; Istanbuluoglu and others 2003). Because low-order streams lack capacity for fluvial transport of large wood, they can accumulate and store large volumes of sediment and wood (Swanson and Lienkaemper 1978; May and Gresswell 2003). Debris flows episodically transport and redistribute this material to downstream portions of the stream network. In mid-order streams in the Oregon Coast range, the contribution of wood from debris flows ranged from 11 to 57% of the total volume of wood in the channel (May 2002). In the Boise Basin of western Idaho, sediment delivered by debris flows has been shown to be beneficial for fish spawning habitat (Benda and others 2003a). Although considered a major hazard in mountain regions worldwide (Coe and others 2008), debris flows are natural processes that contribute to the shifting mosaic of stream and riparian habitat patches along stream networks.

The impacts of debris flows on riparian areas are not well documented (but see Johnson and others 2000; May and Gresswell 2003; Wohl 2006), although they have been anecdotally noted in the geomorphology literature. In steep stream segments, where the channel becomes the runout path, debris flows can scour riparian areas, removing soil and vegetation, including large streamside trees. During a

large flood (~ 100 year recurrence interval) in the Cascade Range of Oregon, riparian trees were uprooted and removed for nearly 1.5 km downstream of the debris flow tributary channel (Johnson and others 2000). This debris flow contained large accumulations of congested large wood, which contributed to the toppling and removal of riparian trees. At tributary junctions or along larger channels, debris flows can also deposit large volumes of sediment, burying portions of existing riparian habitat while creating new geomorphic surfaces for potential vegetation colonization and establishment (Gecy and Wilson 1990). Despite limited research, debris flows have exerted considerable localized influence on forested riparian areas in mountainous regions.

The occurrence of debris flows in relation to wildfires is of great concern throughout the western United States, particularly in steep terrain. Numerous studies have documented increased frequency of debris flows following large-scale, severe fires (Swanson 1981; Meyer and others 1992; Cannon and others 2001; Istanbuluoglu and others 2002; Istanbuluoglu and others 2003; Pierce and others 2004; Gabet and Bookter 2008; Santi and others 2008). In the Oregon Coast range, May and Gresswell (2003) found that a pulse of debris flow activity occurred following the last stand-replacement fire on mid- and upper-slope positions. In their study basins, the most recent fire in the upper slopes did not directly impact the lower elevation channels or valley bottoms, but the influence of the fire was propagated through the stream network by debris flows in the tributaries. In central Idaho and northeast Wyoming, Meyer and Pierce (Meyer and Pierce 2003) used ¹⁴C-dated geologic records to examine evidence of past debris flows and fire frequency in relation to long-term climatic reconstruction (last 10,000 years). They concluded that drought and a warming climate have contributed to severe wildfires and postfire sedimentation, both past and present, and that the incidence of fire may increase with future warming. Much remains to be learned about the frequency, magnitude, and spatial extent of debris flows in different regions, as well as the rate and direction of temporal recovery for stream, riparian, and hillslope ecosystems. This is an active area of research, particularly for physical scientists, but is becoming increasingly multidisciplinary as the impacts of debris flows on aquatic ecosystems are being investigated at different spatial and temporal scales.

Insect Outbreaks

Insect outbreaks are a recurring natural disturbance in forested ecosystems, but current beetle outbreaks (mountain pine beetle, spruce beetle, ips, and others) are among the largest and most severe in recorded history (Bentz and others 2010). Mechanisms contributing to the widespread outbreaks are complex and influenced by multiscale factors, but most insect populations are highly sensitive to changes in temperature and moisture. As noted above, air temperature is projected to increase across North America, particularly at high latitudes and elevations; associated changes in precipitation patterns will result in earlier and longer dry seasons across the western United States, with a greater frequency and duration of droughts (Seager and al. 2007; Solomon and others 2007). These climatic changes will affect the condition, distribution and productivity of forest trees species, as well as associated insect populations. An emerging literature addresses climate change influence on native bark beetle populations, which have evolved with forest tree hosts as natural disturbance agents (Jenkins and others 2008; Bentz and others 2010). Here, we briefly discuss the potential impacts of climatically caused shifts in the extent and frequency of forest insect outbreaks on uplands and stream-riparian corridors. Although most research in the western United States has focused

on native beetles in coniferous forests, it should be noted that similar trends are likely occurring with insect species that utilize and parasitize riparian hardwood species, particularly cottonwoods and willows (Kendall and others 1996).

Warming temperatures are predicted to dramatically affect insect outbreaks in forested areas (Bale and others 2002) by increasing water stress on the host trees while conferring physiological advantages to the insects. The cumulative effect of forest harvest patterns, fire suppression, and climate change, especially drought and mild winters, has already resulted in large, contiguous landscapes susceptible to bark beetle outbreaks (Jenkins and others 2008; Bentz and others 2010). Some forest types are dominated by fairly even-aged stands within the preferred size class range for native beetles; others contain a high percentage of old, large diameter, and low vigor host trees. Flexibility in the life-history strategies of some insect populations appears greater than previously anticipated and rapid genetic adaptation of insects to seasonal changes in temperature has already been documented. Warmer temperatures could disrupt climate controls on winter mortality, generation duration, and developmental and emergence timing of insects, thus increasing survival and the probability of population success. As temperatures rise, the area suitable for both adaptive seasonality and winter survival for insects is predicted to grow, thus expanding the potential range of some species as they move into new niches. Bark beetle outbreaks will vary regionally because of differences in feedbacks driving beetle populations and physiological differences among host tree species. Although a high degree of uncertainty and complexity exists, bark beetle outbreaks driven by climate change may shift some forest ecosystems beyond their natural boundaries of resilience.

Elevated temperatures are also associated with drought conditions that exacerbate tree stress. An important consequence of climate change is higher frequency and severity of droughts (Seager and others 2007), which will influence distribution of forest tree species and increase susceptibility to bark beetle attack. Using existing data for 130 North American tree species and associated climate information, McKenney and others (2007) predicted that the average range for a given tree species will decrease in size by 12% and will shift northward by 700 km during this century. Relative to current distributions, by 2060 the range of Engelmann spruce, a common riparian species and principal host for spruce beetle, is projected to decrease by 47% within the contiguous United States. Beetle outbreaks increase tree mortality rates and can result in subsequent replacement by other tree species and plant associations (Veblen and others 1991). Bark beetles are linked to their host trees and will undoubtedly influence the formation of new western North American forests, including riparian forests. Broad-scale tree migrations are predicted to occur this century. Riparian areas provide mesic refugia for some conifer species at the margins of their current distributions. As these distributional boundaries retreat and expand for Western conifer species, bark beetles may play a significant role in colonizing and killing stressed individuals at the margins. Characterization of thresholds regulating species distributions (insects and trees) may be an important component of forest management in a changing climate, both in uplands and along stream-riparian corridors.

Complex feedbacks relate to increased incidence and consequences of bark beetle outbreaks. Fire, an important forest disturbance that is directly influenced by climate change (Westerling and others 2006), can reduce the resistance of surviving trees to insect attack. Insect-caused canopy mortality alters the amount, composition, and arrangement of living and dead biomass in various fuel complexes. Currently, this is a major concern throughout portions of the western United States impacted by the recent mountain pine beetle epidemic. Relationships and

consequences of the interactions between fire and beetle outbreaks are poorly understood, complex, and spatially and temporally dynamic (Jenkins and others 2008). However, as fire and insect-caused mortality are transforming Western forests, addressing their interactions is necessary in the development and application of forest management strategies.

Beaver

Beaver (*Castor canadensis*) profoundly influence the short- and long-term composition, structure, and function of riparian environments throughout stream networks in mountainous regions of the western United States. As agents of natural disturbance, beaver both use riparian areas as habitat and alter the hydrology, geomorphology, biogeochemistry, and biota of the stream segments they occupy (Naiman and others 1998). The beaver is considered a keystone riparian species due to its extensive influence on fluvial corridors (Pollock and others 1995). Prior to their large scale removal in the late 1800s, beaver occupied nearly all stream habitat types from the arctic to northern Mexico (Naiman and others 1998); their removal is considered a major disturbance in itself (Wohl 2001, 2006). In catchments where beaver are abundant, there may be 2 to 16 dams/km of stream length and each dam may retain between 2000 and 6500 m³ of sediment (Naiman and others 1998).

Beaver cut and utilize riparian woody species to build dams in first- to fourth-order streams and in side-channel and floodplains of larger rivers (Johnston and Naiman 1990). Dams are generally built on low gradient stream segments; however, where beaver population densities are high, dams may be built in steeper gradient portions (Collen and Gibson 2001). Dams retain water and sediments, forming ponds that inundate and frequently flood surrounding trees, altering upstream and downstream riparian environments, and creating wetland habitat. The cyclic pattern of pond creation and abandonment has produced a shifting mosaic of habitat patches and left a legacy on riparian plant community composition and distribution in many stream networks (Pollock and others 1995; Naiman and others 1998). Some abandoned ponds are rapidly recolonized by riparian plants and return to pre-ponded conditions in a few years to decades. Depending on topography, soil characteristics and other factors, other ponds may develop distinct and stable wetland or meadow features that persist for decades or centuries, enhancing species and habitat diversity.

The hydrologic effects of beaver dams and dam-building activities can extend well beyond the boundaries of the pond, both upstream and downstream within the fluvial corridor. Beaver dams alter the patterns of stream discharge by decreasing current velocity and enhancing the depth, extent, and duration of inundation associated with floods. They also elevate the water table during both high and low flows for stream segments upstream of dams. On the upper Colorado River, beaver dams caused water to move around them as surface runoff and subsurface seepage during both high- and low-flow periods, and dams attenuated water table decline in the drier summer months (Westbrook and others 2006). Beaver can influence hydrologic processes during both peak flow and low flow periods, thus creating and maintaining hydrologic regimes suitable for the formation and persistence of wetlands (Westbrook and others 2006). Geomorphic effects include the retention and redistribution of sediment and organic matter, flooding and erosion of streambanks, and expansion of the extent of flooded soils (Pollock and others 1995; Naiman and others 1998). Beaver also affect plant community composition and the spatial-temporal dynamics of the vegetation through selective herbivory and

foraging practices (Pollock and others 1995). Beaver cut trees and shrubs to feed on bark, preferring trees with soft, brittle bark, including common riparian species such as aspen, willows, alder, maple and ash. Felled trees may be used in dam construction or they may be left in place, adding structural complexity to riparian zones (Barnes and Dibble 1988). Foraging by beaver significantly impacts the composition and successional dynamics of riparian vegetation, particularly where beaver densities are high (Barnes and Dibble 1988; Johnston and Naiman 1990).

Beaver activities and dams can have both positive and negative effects on fish community composition and habitat (Collen and Gibson 2001). Dams stabilize and warm stream temperature, increasing localized productivity but potentially having a negative effect on coldwater species. Spawning sites may be flooded and silted, upstream migration may be impeded, and habitat may be created for predators, with detrimental effects on desired fish species. However, the increased habitat complexity may provide refugia and cover. Because stream discharge is stabilized, channel scouring and bank erosion are decreased, organic matter and nutrients are retained, and invertebrate and fish production may increase (Collen and Gibson 2001). Reintroduction of beaver has been suggested as a possible “adaptation action” to climate change that may improve watershed resilience (Furniss and others 2010). However, the practicality and benefits of introducing or restoring beaver populations will vary according to location and should be considered in conjunction with a management plan to control their densities.

Human Disturbance of Riparian Areas: Land Use and Management

Natural disturbances and processes have influenced the development and current condition of riparian and stream habitats as briefly described above (McAllister 2008). Along many stream and river segments, however, the effects of past and present human disturbance may be more pervasive than natural processes. Human effects can be broadly considered with respect to five categories: flow regulation/alteration, water pollution, channel alteration, decreased biotic integrity, and land use (Wohl 2006). Direct human impacts on stream-riparian corridors result from activities conducted within the stream channel itself that alter channel geometry, the dynamics of water and sediment movement, or aquatic and riparian communities. Examples include construction of dams or diversions, channelization, removal of beavers, and placer mining (Wohl 2006). Less direct human impacts result from activities within the watershed that alter the movement of water, sediment, large wood and nutrients, or introduce contaminants into the channel. Examples include road-building, forest harvest, urbanization, agricultural cropping, and grazing. Human impacts frequently interact or lead to changes in the timing, frequency, or magnitude of natural disturbances. For example, activities such as forest harvest and road building can accelerate the frequency and volume of debris slides and hillslope sediment loss; and grazing can increase erosion due to changes in bank stability. Several extensive reviews have described the impacts of human disturbance and land use on streams, rivers, and riparian areas (Patten 1998; Wohl 2001; Brinson and others 2002; Naiman and others 2005; Wohl 2006).

In the context of climate change, increased alteration of streamflow is a critical human disturbance affecting the ecological integrity of many aquatic and riparian ecosystems (Furniss and others 2010). Alteration in stream flow, including the amount, timing, and duration of flow, all contribute to changes in the geomorphology, physical processes, ecological condition, and biological characteristics of the stream channel and associated riparian habitat (Poff and others 1997; Merritt and others 2009). Land use practices, such as agriculture and urbanization, have

added to the disruption of natural hydrologic regimes within stream networks (Poff and others 1997). As noted previously, reconciling the increasing human demand for water with the dependency of stream and riparian biota on natural flow attributes remains one of the most difficult challenges in the face of climate change predictions.

Land use and management has changed considerably with time and past practices have been discontinued or modified to mitigate environmental impacts. However, the legacy of human disturbance and land use continues and must be considered in current management strategies. For example, historical practices, such as removal of wood from rivers for navigation and fish passage, splash damming, tie drives, and clearing of riparian trees has resulted in simplification of stream channels and streambanks, reduction in the areal extent of riparian areas, and local decreases in amounts of instream large wood (Sedell and Froggart 1984; Young 1994). A legacy consequence of timber harvesting is the marked long term reduction in recruitment of large wood to streams in logged basins. Livestock grazing in the past has resulted in significant impacts to riparian vegetation and soils. Fire suppression in uplands and riparian areas has resulted in an increase of fuel loads within areas that typically experienced low-severity historical fire regimes (Ellis 2001; Dwire and Kauffman 2003). The legacy effects of past human disturbance influence current and future condition and potential of streamside areas; their continuing impacts must be considered when defining riparian management targets, planning restoration projects, and strategizing on climate change adaptation and mitigation actions.

Climate Change, Fire, and Riparian Values and Functions

Riparian areas are dynamic environments, influenced by strong disturbance regimes, and characterized by considerable habitat heterogeneity and multidimensional gradients. The range of riparian ecological processes, values, and functions depend on physical characteristics associated with location within the basin and stream network. The influence of climate change on riparian areas, with consequent shifts in precipitation, stream flow characteristics, and fire severity and frequency, also depends on the physical context of a given reach or stream-riparian segment. Because of their spatial position in watersheds, riparian areas integrate interactions between aquatic and terrestrial environments and can be sensitive to disturbance and management both upslope and upstream. In addition, the interactions and feedbacks among natural and human disturbances depend on location within a watershed, physical context, and land use legacy (Nakamura and others 2000; Rood and others 2007; Rieman and others 2010).

The impacts of climate change will influence different stream-riparian ecosystems in different ways. In high elevation headwaters, stream segments in alpine and treeline environments will be affected by variability in annual snowpacks and higher temperatures through the growing season. In subalpine and montane forested riparian areas, riparian vegetation may be most affected by shifts in streamside microclimates. Riparian tree species composition is commonly similar to surrounding uplands, but with higher frequency of more mesic species, like Engelmann spruce, and greater understory diversity and productivity. Although influenced by streamflow and shallow subsurface drainage that may emerge near streams, many conifer-dominated riparian areas could be characterized as micro-climate

Non-Native Invasive Plants in Riparian Areas of the Western United States

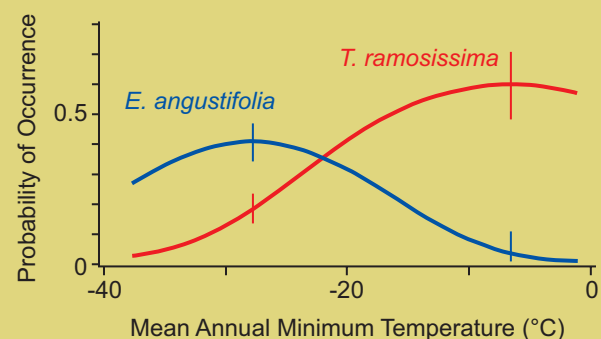
Riparian corridors are particularly susceptible to invasion by non-native plant species. The same factors and processes that foster high native species diversity along streams also facilitate dispersal, establishment, and persistence of undesirable species. These factors include: active natural disturbance regimes, resulting in frequent turnover and a range of habitats; high degree of connectivity that facilitates the movement and transport of plant propagules from upslope and upstream; and resource availability, particularly moisture and nutrients. Common management activities in riparian areas—including grazing, road construction, mining and recreation—can accelerate the spread and persistence of invasive species. Many stream valleys are also transportation corridors, which frequently serve as conduits for the movement of nonnative species. The alteration of natural stream flow regimes through damming, diversion, and groundwater extraction has directly and indirectly contributed to the increase of non-native plant abundance along streams and rivers. This has been extensively reported for low gradient, low elevation streams and rivers in arid and semi-arid regions, where native cottonwood and willow species have been displaced by saltcedar, especially *Tamarix ramosissima* and *T. chinensis*, Russian olive and Siberian elm. Extensive streamside stands of saltcedar and other nonnative woody species can result in high streamside fuel loads, potentially influencing fire behavior and frequency. Shifts in precipitation, as regionally predicted by climate change models, will affect stream flow and hillslope hydrologic inputs to streams, likely reducing the areal extent of riparian areas over time, and conferring additional advantage to invasive woody species, particularly in low gradient floodplains.

Climate change predictions also include rising air temperatures, which influence the distribution of both native and invasive species and thus affect species assemblages. The distribution of saltcedar is thought to be limited by sensitivity to frost, and

the frequency of its occurrence in the western United States has a strong positive relation with the mean annual minimum temperature (Figure T8-1). In contrast, Russian olive occurs most frequently at sites with mean annual minimum temperatures of -8.7°C or lower, suggesting a chilling requirement for bud break or seed germination. Shifts in the ranges of both species—*tamarix* moving further north, Russian olive moving to higher elevations—will continue to affect native riparian plant communities throughout arid portions of the west. Non-native herbaceous species have also invaded riparian areas; in many rangelands and forests, these are the same noxious species that occur in uplands, and include leafy spurge, nonnative brome grasses and thistles, knapweeds, toadflaxes, and knotweeds. Existing distributions of other riparian native and non-native species are also adjusting, depending on biotic and abiotic constraints, and will further complicate efforts to maintain functional, native riparian plant communities.

The basic principles of managing invasives apply in riparian areas as in uplands: minimize the potential for their dispersal and establishment; focus on prevention of spread; maintain resistant native ecosystems; and conduct incremental treatments within an adaptive management framework. However, control or eradication of invasive species is more difficult in riparian areas because use of mechanical and chemical treatments is restricted or not allowed along many streams. Prescribed fire is increasingly used to control invasives in riparian areas; use is likely to increase as more is learned about treatment effectiveness in different ecoregions. Diligent post-treatment and postfire management is critical, however, since certain invasive species (e.g. cheat grass and reed canary grass), have been observed to increase in riparian areas for several years following fire. Follow-up restoration activities may be required to reestablish native riparian plant communities in some locations. Interactions and feedbacks between fire, climate change variables, and response of invasive plant species are likely to increase in complexity, requiring expanded coordination for invasive species control at watershed and regional scales.

Figure T8-1. Gaussian logistic regression of occurrence of salt-cedar, *Tamarix ramosissima*, and Russian olive, *Elaeagnus angustifolia*, as a function of mean annual minimum temperature. Vegetation data were collected at 475 randomly selected stream gaging stations in 17 western states; temperatures were derived from weather station data recorded from 1961-1990 (Friedman and others 2005).



dependent. They are frequently cooler and moister due to spatial position in steep watersheds, cold air drainage, and topographic shading. With increasing air temperatures, riparian microclimates may warm and coniferous streamside vegetation may become more similar to upland vegetation. During wildfires, these riparian areas may increasingly burn like surrounding uplands (Table 3).

Throughout the western United States, willow-dominated riparian areas occur in broad valley bottoms, including unconfined and glaciated valleys with low slopes (<3%) in montane and subalpine settings (Patten 1998; Rocchio 2006). While floods and streamflow are important regulators of willow ecosystems, other major drivers are beaver and shallow subsurface drainage that contributes to maintenance of high water tables (Gage and Cooper 2004). Groundwater recharge can originate from deep glacial till, hillslopes with highly fractured rock, and long, slow-draining hillslopes. Typically, at higher elevations, the magnitude of hillslope discharge is higher as a consequence of snowmelt runoff and precipitation events. The relative importance of streamflow and hillslope discharge for maintenance of willow ecosystems depends on elevation, geology, season, and other factors (Westbrook and others 2006; Wolf and others 2007; Westbrook and others 2011). Climate change will affect both streamflow and patterns of groundwater discharge and may result in the spatial contraction of willow ecosystems and local loss of species near limits of their distributional ranges. The dry-down of willow ecosystems may limit their ability to serve as fuel breaks during wildfires (Table 3).

Cottonwoods are keystone riparian species, dependent on flooding for recruitment and stand replacement, and dependent on streamflow for stand maintenance. Streamflow-ecology relationships have been described for different cottonwood species, geomorphic settings, and regions in western North America, mostly in response to dams and other flow alteration (Rood and Mahoney 1990; Stromberg and Patten 1991; Braatne and others 2007; Wilding and Poff 2008; Poff and others 2009; Merritt and Poff 2010). These relationships will also prove useful in predicting potential shifts in distribution and condition of cottonwood stands in response to altered streamflow due to climate change. As noted above, a key challenge in securing sustainability of cottonwood and other riparian ecosystems is developing a framework that incorporates predicted changes in streamflow characteristics and guides the development of environmental flow standards for regional planning (Poff and others 2009).

C. Fish, Fire, Forest Management, and Climate Change

Forest streams provide some of the coldest and cleanest waters, and very high ecological, recreational, and intrinsic values are placed on the trout and salmon species that require the high quality water. Many of these fishes are now listed as sensitive, threatened, or endangered (Rieman and others 2003b), and sometimes their presence is cited as a protected value to support fire suppression or fuel abatement projects. Fire does indeed represent a challenge, as do land management and climate change, but fire has also played an important role in the development of the fish communities in the western United States. Understanding the ecological dynamics in relationships between fish and fire is an essential step in successfully managing forests and streams in a changing climate. This chapter provides a basic overview of the topics necessary to understand the dynamics and provide the logic supporting the continuing synthesis. The reader is directed to the compendium of “Advanced Topics on Fish Populations and Fire” at the back of this publication

for a richer and more thorough review of the general topic and several important aspects. Following is a list of the articles:

Advanced Topics on Fish Populations and Fire

- Fire and Fish: A Synthesis of Observation and Experience
By Bruce Rieman, Robert Gresswell, and John Rinne
- Genetic variation reveals influence of landscape connectivity on population dynamics and resiliency of western trout in disturbance-prone habitats
By Helen M. Neville, R.E. Gresswell, and J.B. Dunham
- Fish life histories, wildfire, and resilience—a case study of rainbow trout in the Boise River, Idaho
By Amanda E. Rosenberger, Jason B. Dunham, and Helen Neville
- Aquatic species invasions in the context of fire and climate change
By Michael K. Young

The relationship of fish to fire is complex, like it is for other ecological systems. Aquatic communities in the West evolved with fire, along with the forests. Similar to effects on forests, there may be severe short-term negative consequences of fire for the individuals or local populations, but those may be coupled to long-term benefits for habitat complexity, quality, and productivity. Disturbance and recovery are key processes in many ecosystems (Pickett and White 1985), providing analogs for learning. A chief lesson is that in many circumstances, the full range of dynamics from mild to severe performs important functions in ecosystem renewal and cycling. Severe fire, and severe mass wasting erosional consequences following fire, often play important, positive ecological roles in aquatic ecosystems.

Fisheries and land managers often have natural protective instincts relative to headwater streams that have our coldest and cleanest water and our rarest fishes. Decades of work have demonstrated how forest management, which is conceptually less intrusive than severe wildfire, has contributed to declines in habitat quality and populations of salmonids across the West (Salo and Cundy 1987). Even within the context of recent harvest practices where the focus is on ecosystem integrity more than timber volume, there are potential consequences of both interfering with recovery dynamics and attempting to soften the role of fire.

Reconciling these different views of aquatic systems and fishes, fragile yet also tough, can be enlightened by considering the ecological processes protecting them. Millions of years of coevolution between forests and fish have developed some measure of ecological stability through resilience, a capacity to recover from not just population reductions, but major habitat altering events. Salmonid populations have substantial resilience to disturbance from fire or flood through expression of diverse life histories (Dunham and others 2003) that may include variation in patterns or extent of migration and timing of critical events in the life cycle. Metapopulation dynamics, wherein populations at one locale are supported by individuals dispersing from others (Levins 1969), provide an additional degree of buffering for aquatic populations as they do for many other wildlife species (Rieman and Dunham 2000). Seeking to engineer stability by resisting or controlling disturbance has inadvertently undermined some of the resilience, particularly through activities that fragment or isolate habitats from one another (Fausch and

others 2009). A key difference in outcomes is that there are many opportunities over time for resilience processes to succeed, and a single failure in time does not consign a given population to permanent oblivion, whereas resistance dependent measures may have greater sensitivity to individual failures.

Even as we come to terms with this interesting dichotomy and begin to articulate more robust strategies for dynamic management, we are faced with the implications of a changing climate on aquatic systems. The nonstationary behavior of streams under future climates (Barnett and others 2008) with trends in streamflow timing (Stewart and others 2005), increasing variability in streamflows (Pagano and Garen 2005; Luce and Holden 2009), and warming streams (Isaak and others 2010) challenges most notions of stability, dynamic or otherwise. The ecological response of fish populations to the stresses imposed by fire, land management, and climate change can help us see how they may interact in the future to affect fishes. This understanding can form a foundation for management response to climate change.

The Response of Fish Populations to Fire

The immediate and short term effects of fire are commonly harmful to individual fish and even local populations, but the intensity of the effect varies. Direct heating of water by fire and dissolution of ammonia and other chemicals from smoke has resulted in fish kills (Minshall and others 1989; Earl and Blinn 2003; Spencer and others 2003), but fish also appear to simply avoid affected areas if refugia are available (Rieman and Clayton 1997). Introductions of toxic material and ash flows shortly after fire have resulted in local extirpations (Rinne and Neary 1996; Rinne 2003; Rinne and Carter 2008). Anecdotal observations of lower concentration introductions of ash and sediment have shown little immediate change in other circumstances (Sawtooth National Forest 2007). Major debris flow events in steep channels (e.g., Cannon and Reneau 2000; Miller and others 2003) almost certainly remove the fish that are present at the time.

The response of fish populations to these impacts are varied (see Rieman and others, this volume for additional detail). Some extirpations are permanent (e.g., Rinne 2003), while some locations see reestablishment of fish populations within a relatively short time (e.g., Jakober 2002; Howell 2006). Sublethal temperature increases after canopy removal have been observed to alter the growth and maturation of fish (Dunham and others 2007). Although fine sediment increases are documented to interfere with life stages that use gravel interstices (Everest and others 1987; Chapman 1988; Thurow and King 1991), the brief period of vulnerability to surface erosion post fire (Shakesby and Doerr 2006) and the large transport capacity of rivers to rapidly remove fine sediment (Lisle and others 2001; Burton 2005) seem to make fine sediments less important than other factors for fish post-fire status.

The long-term benefits of fire effects have been noted as well. The renewal of spawning gravels is cited (e.g., Reeves and others 1995; Benda and others 1998) (Figure 26). Inputs of nutrients released by fires may also provide at least a temporary boost in productivity (Spencer and others 2003; Malison and Baxter 2010). Fires are one of many disturbances that regulate sunlight coming to streams; so they contribute to maintaining a diversity of invertebrates that use both algae growing in streams as well as detritus falling from riparian forests (Minshall 2003). The legacy left behind by fire, including both the renewed material availability and the presence of fish to use those materials, is important in the net benefit of a fire.

Life History Diversity and Resilience

Life history variation or diversity, is a key concept in the biology of native fishes. Considerable interest has focused on notions of “resilience” (Dunham and others 2003; Bisson and others 2009; Bottom and others 2011) and “biocomplexity” (Hilborn and others 2003; Schindler and others 2010) that follow from diversity as foundations for conservation management and habitat restoration (ISAB 2011). There are three key points linking these ideas. First, variation in life histories within and among populations contributes to increased productivity, persistence in variable environments, and the capacity of populations to recover from major disruptions (Dunham and others 2003; ISAB 2011). Resilience is the maintenance of these functions in the face of disturbance (Bottom and others 2011). Second, the expression of any potential variation depends on the diversity, connectivity among, and access to, the full range of habitats and environmental gradients available throughout the life cycle (e.g., Healey and Prince 1995). And third, it also depends on genetic diversity underlying the potential range of life history expression and its continuing evolution. In salmonids, life history variation commonly has been classified by distinct migratory forms. Interior salmonid life histories are often characterized, for example, as individuals that remain in natal waters their entire life (resident forms), migrate as juveniles from natal streams to larger rivers (fluvial forms) or to lakes or reservoirs (adfluvial forms), to mature before returning to spawn. Rieman and Clayton (1997) argued that the existence of these distinct life histories was key to the recovery and persistence of bull trout populations that experienced catastrophic disturbance associated with large wildfires. But, life history variation can be far more than several distinct migratory or non-migratory patterns. Salmonid populations, in particular, may show strong variation in egg size, incubation time, and size and timing of alevin emergence (McGrath and others 2008), subsequent patterns of foraging and growth (Nielsen 1992), size and age at juvenile migration (Downs and others 2006), migratory route and time spent in transitional habitats (Burke 2004), size and age of maturity, seasonal timing and distribution of adult migration and spawning (Doctor and others 2010; Crozier and others 2011), and for species that do not die after spawning, the frequency of migration and spawning events. The result can be a very large number of possible combinations contributing to the potential diversity of life histories within and among populations and even within family groups. The diversity that is actually expressed results through the interactions of individuals and populations with each other and the available habitats and streams, through phenotypic plasticity and evolution. These environments and interactions are embedded within larger watersheds and landscapes that are themselves shifting, reorganizing, and evolving through time and space. Together these represent complex systems or ‘biocomplexity’ that is arguably key to resilience in the larger perspective (Hilborn and others 2003; Schindler and others 2010). In this view, life history variation is both a product of, and a necessary condition for, persistence in a dynamic and changing world.

Life history diversity, which considers the variation in life cycle stages, timing, and patterns of behaviors, is an important source of resilience in fish populations (Hilborn and others 2003; Moore and others 2010; Schindler and others 2010). Commonly identified life history categories include the range and extent of migration (from purely resident to anadromous) along with tremendous variation in the timing of life history events. In general, migratory fishes that move to large rivers, lakes, or the ocean grow to larger sizes than fish that do not migrate, commonly returning to natal streams more fecund than resident fish. Besides being able to contribute to increased population growth rates, these fish also spend less time in the smaller steep tributaries and are less likely to be directly impacted by violent post-fire impacts (Rieman and Clayton 1997). Fish that migrate away from natal streams have their own set of hazards to navigate, but in general, populations with

Figure 26: Salmon redds on recent debris flow deposit in the Middle Fork Salmon River in a location where spawning did not occur previously because of a lack of suitable substrate (photo courtesy of Russ Thurow).



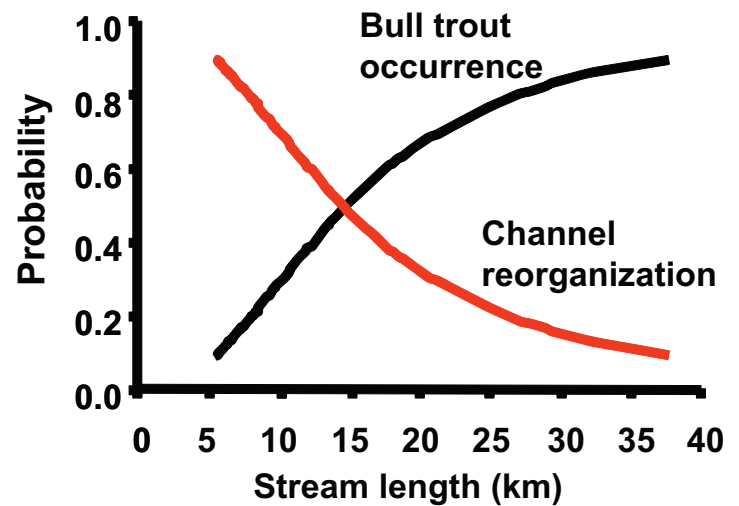
a diversity of strategies will have greater resilience with respect to a range of disturbance events.

Dispersal of fish from populations in nearby unburned streams or reaches is another important mechanism for refounding and supporting populations in burned streams that contributes to resilience (e.g., Rieman and Clayton 1997; Howell 2006). The number of proximate or interconnected habitat “patches” can be a useful indicator of this form of resilience (MacArthur and Wilson 1963; Dunham and Rieman 1999; Hilderbrand and Kershner 2000). Interconnected habitats and large patches are more resilient because they are less likely to experience synchronous disturbance from debris flows or other events and may contain some larger, more productive streams that can support less productive headwater streams. Debris flows transit through smaller streams, where their passage can be destructive to biota and habitats, but they usually deposit upon entering large streams, where the new material contributes to habitat complexity. High complexity of the stream network, e.g., having multiple branching tributaries as compared to a single threaded configuration, can also add to the robustness of the patch (Gresswell and others 2006). Patch size, complexity, connectivity, and the presence of multiple life histories also combine to produce populations that are less likely to suffer from small population size effects on genetics (see Neville and others in Advanced Topics of this volume).

The amount of habitat needed to ensure population persistence (in light of disturbance and environmental variation) is not precisely known, although available lines of evidence suggest something in the range of 20-40 km of suitable stream length for bull trout and less for other species (Rieman and McIntyre 1995; Dunham and Rieman 1999; Dunham and others 2003; Peterson and others 2008; Fausch and others 2009; Cook and others 2010). Observations of the size of occupied versus unoccupied patches that were suitable for bull trout (based on temperature) showed an increasing probability of presence leveling off above 100 km² (Dunham and Rieman 1999), which equates to roughly 40 km of suitable habitat (Isaak and others 2010). A graph comparing results of the bull trout presence/absence data and the post-fire debris flow data (1997 photos) gives a sense of the relationship between the two (Figure 27).

Plotting genetic difference as a function of distance between populations often results in a positive relationship. An analysis of genetic relationships in the Boise

Figure 27: Probability of bull trout presence within a patch based on total stream length in the habitat (based on Dunham and Rieman 1999) versus the probability that channel segments within a given distance of stream length might experience debris flows at the same time. Neither study directly measured these quantities, but this is an interpretation of how they might be compared.



River supported the notion that gene flow was much stronger at shorter distances especially under 20 km (Whiteley and others 2006). Debris flow mapping in the Boise River basin also shows an increasing synchrony of severely scoured reaches at scales less than 20 km (see textbox on debris flow scaling) in spite of much larger fire extents. These observations support speculation that the current structure and resilience of populations may emerge through the patterns of disturbance and recovery of the past.

Interaction With Land Management

If we draw from the general concept that the structure and resilience of current populations of fish reflect millennia of natural disturbances, including wildfires, but note that most local extirpations and declines have happened since the introduction of land management, including wildfire suppression, the question that arises is, “What is the difference?” This framing may place too much at the foot of land management, when issues like invasive species, introduced diseases, water diversion and management, and climate change also contribute to specific incidents. Nonetheless, this framing opens up a discussion of the contrasts between what superficially would appear to be impacts of a similar nature.

Land management comprises multiple activities that interact with streams in ways that are unique from fire. While forestry practices on individual stands, or even watersheds and landscapes, are conceptually less severe with a focus on reduced impacts to soils and dispersed impacts in space, roads are unprecedented components of managed landscapes, which have consequences disproportionate to their area (Luce and Wemple 2001). Early forest management on both private and public lands were designed to preserve soil resources both to protect site productivity and water resources (Hays 1969; Pyne 2002). Guidelines now seek to disperse canopy removal impacts over broad areas and avoid practices that result in soil degradation, even at site scales. While the positive effects of periodic mass wasting inputs were noted earlier, it is not clear that the advent of cautious harvest practices nor the reduction in burned acreages from fire suppression have substantially altered the long-term stochastic properties of these events. Rather it would appear that historical declines in aquatic species status might be more tied to the expansion of the road network (Lee and others 1997; Baxter and others 1999; Trombulak and Frissell 2000).

Roads have numerous physical effects on fish habitats including: habitat fragmentation, chronic fine sediment introduction, more frequent sediment from mass wasting, and channel constraint. Of these, fragmentation may be particularly important with respect to resilience to fire disturbances. Most obviously, fragmentation prevents migratory fish from re-founding or supporting a severely depressed population. If fragmentation prevents the expression of migratory life histories, removing culvert barriers post fire may be substantially less effective. Fragmentation by roads may lead to reduced genetic diversity, leaving populations less well prepared for shifts in conditions that could occur post fire even without the more catastrophic population resets associated with debris flows (see Neville and others in Advanced Topics of this volume). Barriers constructed to protect against non-native fish invasions can have similar consequences to road fragmentation (Peterson and others 2008; Fausch and others 2009).

Chronic fine sediment from roads reduces habitat productivity and survival of embryos and juveniles (Chapman 1988). Overall, this effect can restrict population growth rates, reducing resilience to individual events. Individual mass wasting events from roads are similar in nature to other mass wasting events. The risk of mass wasting from roads is highest in the initial decade after construction and declines over time, unless road maintenance stops, which can dramatically increase the risk. In basins where harvest was done carefully and incrementally, the serial construction of new roads may have generated essentially a chronic mass wasting scenario (see, e.g., Colombaroli and Gavin 2010)

Some contrasts of land management to fire have focused on sediment yield from harvested areas (e.g., Istanbuluoglu and others 2004; O’Laughlin 2005; Roloff and others 2005). In low gradient areas, a series of careful harvests with soil protection can produce less sediment than a single severe fire (O’Laughlin 2005). In steeper areas, long term sediment yields are similar, but the event sizes tend to be different (Istanbuluoglu and others 2004). The understanding provided above regarding aquatic ecology resilience to disturbance, however, suggests that differences in long term sediment totals may not be a useful decision variable (Luce and others 2005; Luce and Rieman 2010). Episodicity clearly has a direct influence on the consequences for fish as does the spatial distribution of synchronous major disturbances. The sediment yield studies have been faulted for failing to include road erosion impacts (Rhodes 2005). The addition of road erosion would likely contribute little to long term sediment yields (Goode and others 2011), however, considering the chronic additions would provide interesting and biologically relevant contrast.

Land management also includes fire prevention and suppression. Fire suppression practices certainly have the capacity to increase sediment loading, but they are likely minor additions compared to inputs from large severe fires. Some fire retardants are toxic to fish, and others impose a chemical oxygen demand on the water; as a consequence, fire retardant application near streams can be hazardous to aquatic systems (Little and Calfee 2002; Pilliod and others 2003; Giménez and others 2004). The introduction of diseases and non-native aquatic species from untreated pumping equipment also poses a hazard but can be managed with vigilance.

Changes in fuel loads caused by fire suppression over much of the 20th century are commonly discussed as an emergent risk for aquatic systems (Bisson and others 2003; Hessburg and Agee 2003; Rieman and others 2010). It is not obvious that such changes have led directly to extirpations historically, but the increased continuity of fuels and flammability in some forest types pose an increased risk of larger and more continuous fire in those locations, which directly relate to strategies fish have adapted to cope with fire. In explorations of the potential for forest

restoration to reduce risks in the South Fork Boise River, some intermediate sized basins showed increased persistence probabilities from reducing fuel continuity (Dare and others 2009). Many of the places with strongly altered fuels or fire regimes were directly affected by forest harvest and attendant road construction, thus they seldom coincide with current habitats of sensitive, threatened, or endangered species (Rieman and others 2000). Some of the habitats are suitable, however, so these places may represent opportunities for joint restoration of forest and aquatic habitats to more natural fire regimes (Rieman and others 2010).

Discussions of fuel change issues and fish have focused largely on forests (e.g., Bisson and others 2003), perhaps because of debate over forest management and restoration policy (e.g., DellaSala and Frost 2001). Fuel changes (and consequent fire regime changes) caused by shifts in range species, particular the replacement of sagebrush (*Artemisia tridentata*) communities by cheatgrass (*Bromus tectorum*) and other non-native brome grasses has the potential to affect many aquatic communities in the West as well. Invasive riparian species have major implications for streamside fuel structures too (see textbox on invasive species in the riparian section). Land management, particularly road management, fire suppression management, and post-fire restoration practices, have strong influences on the introduction and spread of non-native plants, which may play out as a long-term risk issue for aquatic systems because of their close coupling to the terrestrial ecology.

Interactions With a Changing Climate

Changes in climate described earlier will influence fish most directly through stream temperature increases and changes in flow regimes (Rieman and Isaak 2010; Wenger and others 2011b). Most trout and salmon are adapted to relatively cold water and typically use some of the higher elevation waters in basins where they are present. This means that there may be limited ability for some populations to shift to higher elevation streams. While temperature increases will place additional stress on populations in stream reaches where temperatures are warmer than optimal, there are some exceptionally cold streams at high elevations where productivity may increase with warming (Harig and others 2000; Coleman and Fausch 2007). Where warming air temperatures increase the probability of flood events in winter, fall spawning species like bull trout, brook trout, or the five Pacific salmon species with eggs incubating during the winter may be at greater risk. Climate change may cause additional indirect effects to populations through changes in wildfire size, frequency, and severity and alterations to riparian ecosystems (please refer to the sections on wildfire ecology and riparian ecology preceding this section for more background). Increased wildfire presence in the landscape could contribute to keeping riparian canopy less dense and stream temperatures warmer.

The combination of temperature and streamflow changes will reduce the size of headwater patches of the species adapted to the coldest temperatures (Figure 28). Patches may also effectively shrink from above in locations where streamflows are declining and streams become too small, and in places where increased rain-on-snow inputs are driving more frequent mass wasting in steep headwater channels. Similar changes may also reduce connectivity within and among habitat patches, with barriers being imposed by reductions in low streamflows (Luce and Holden 2009; Leppi and others 2011) or high temperatures.

Different species have different sensitivity to changes in temperature and flow regime, and habitat suitability models can inform expectations for shifts in fish distributions related to climate change (Rieman and others 2007; Wenger and others 2011a; Wenger and others 2011b) (Figures 29, 30). Contrasting and complementary

Figure 28: Decline in patch size and consequent probability of being occupied starting from (a) current conditions with (b) decreases due to warming temperatures [red areas] and (c) low flow decreases [dashed lines] and debris flow risks [yellow lines].

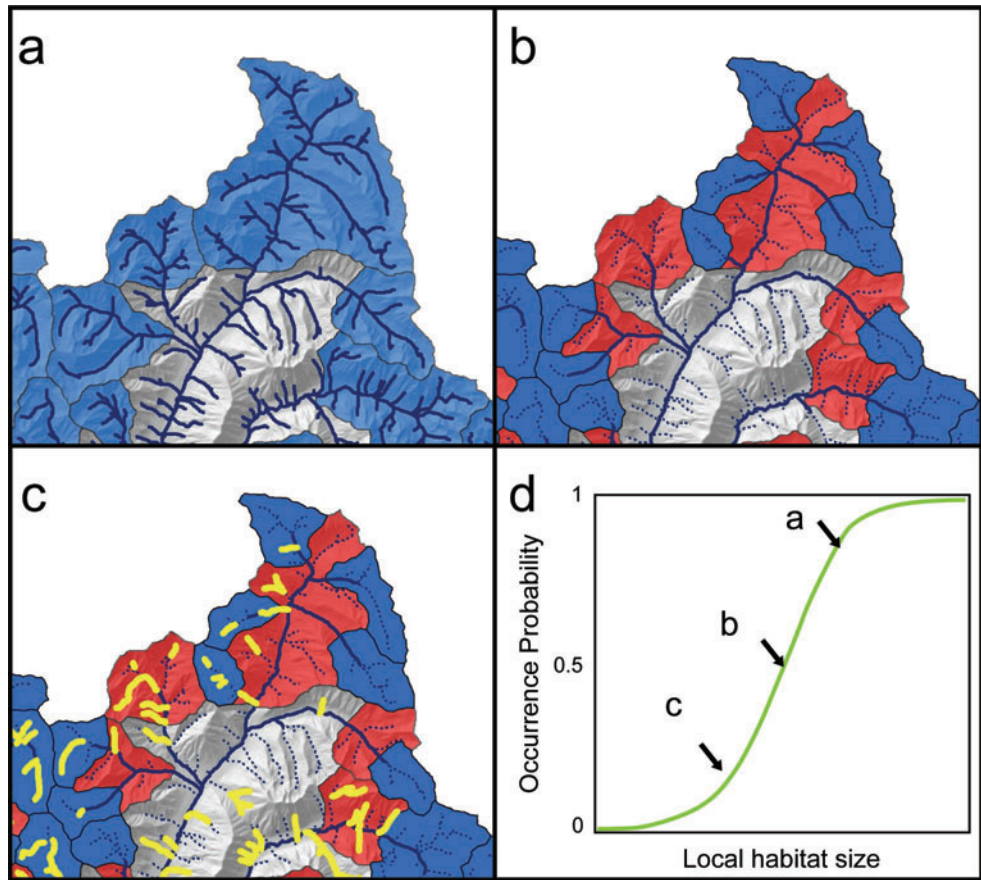
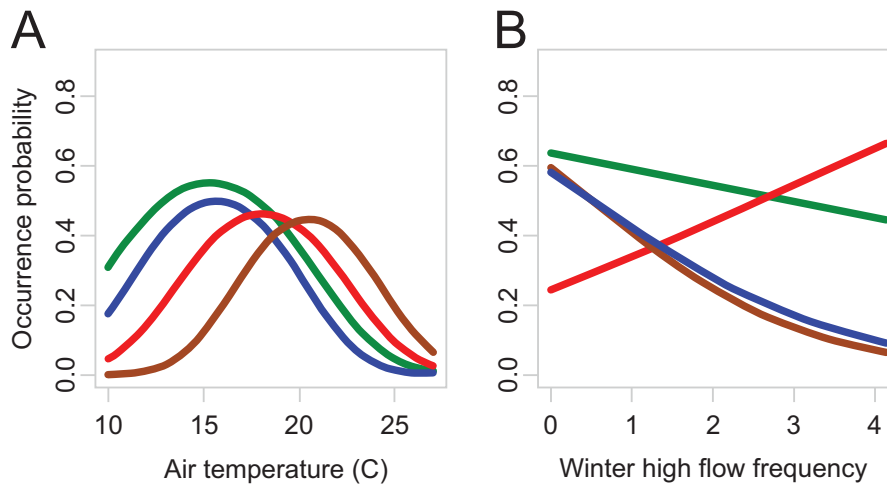


Figure 29: Comparative sensitivity of four trout species to stream temperature A and the frequency of winter high flows B. Green = cutthroat trout; blue = brook trout; red = rainbow trout; brown = brown trout. (from Wenger and others 2011b)



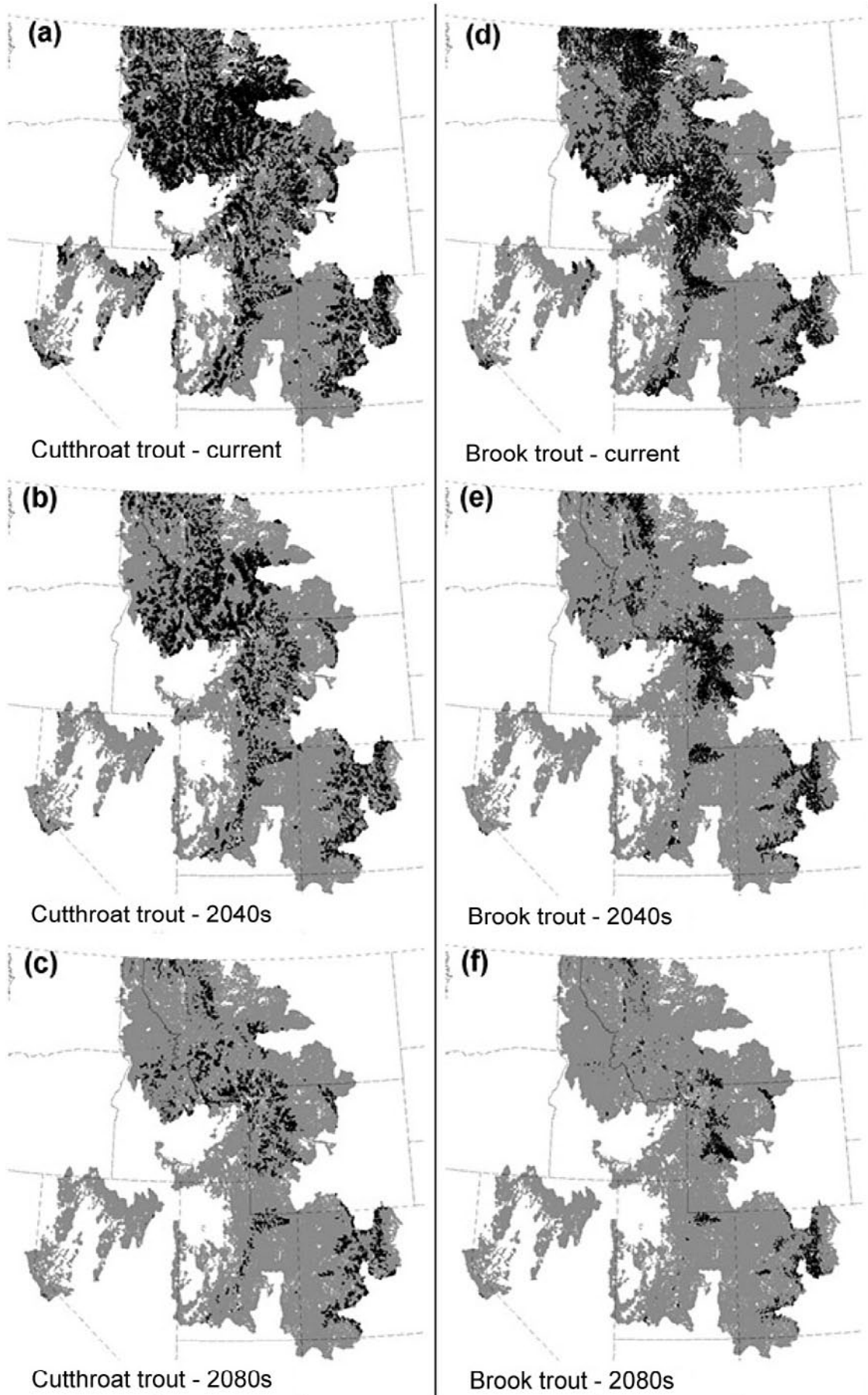


Figure 30: Modeled trout distributions under present and future (A1B) climate change scenarios. Gray streams are relatively unsuitable and black streams are relatively suitable (from Wenger and others 2011b)

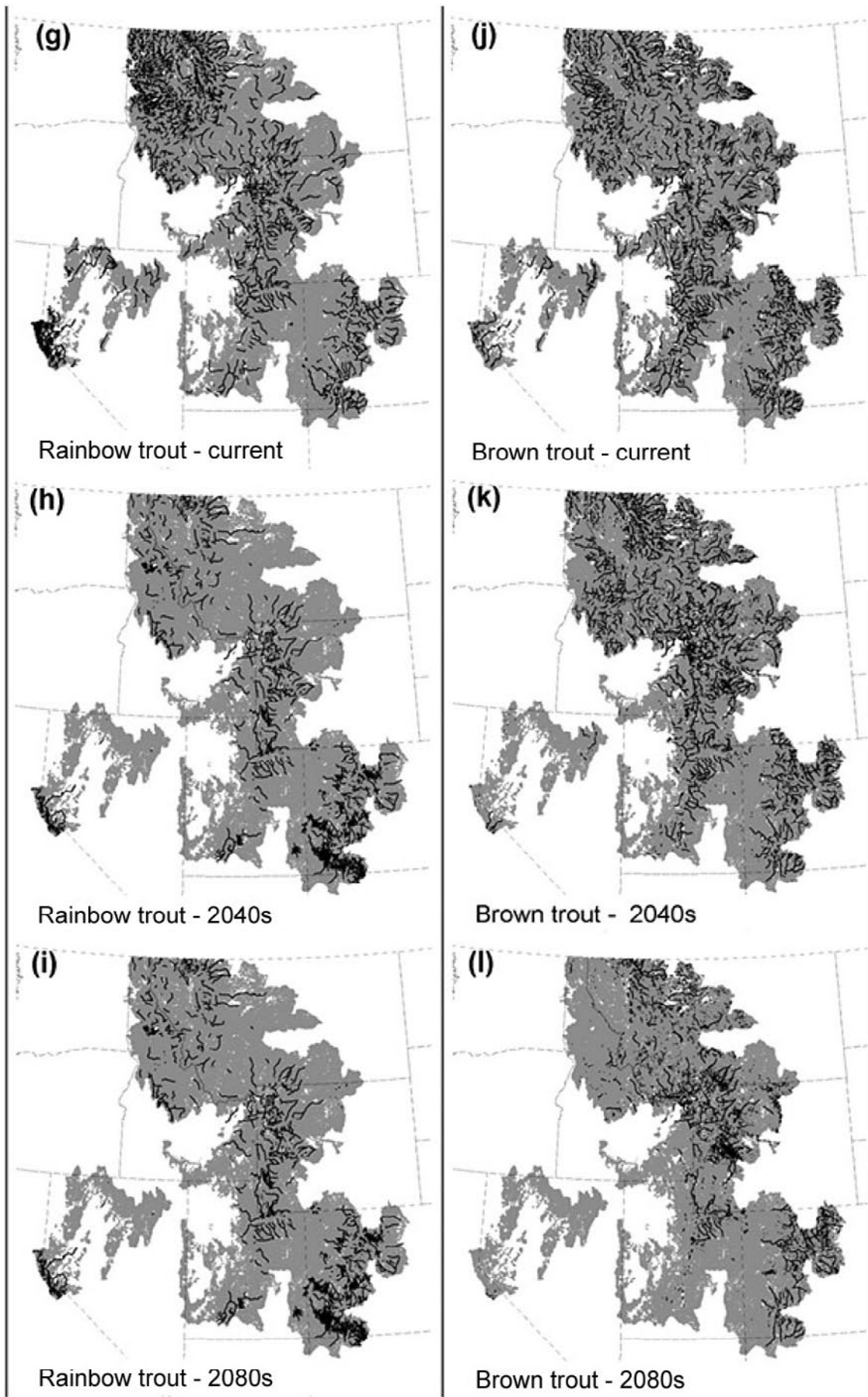


Figure 30: Continued.

effects of different processes on different species creates a complex set of potential responses. For example, changes in winter flood frequency may be less important for bull trout if temperature excludes them from habitats where flood frequency is increasing; and spring-spawning cutthroat that would otherwise be negatively influenced by temperature changes may actually benefit from increased winter floods that reduce competition with fall-spawning brook trout.

Hydrologic changes will likely precipitate changes in water management (Barnett and others 2008), which in turn will have consequences for many aquatic systems. In the Southwest, where water withdrawals are a common challenge for fish (Rieman and others 2003b), projections for decreased flows and increased demands will likely exacerbate current constraints. Summer flow declines projected for other locations may have similar consequences where water withdrawals are a substantial portion of summer streamflow now. Shifts in snow accumulation and melt have been cited as reasons for proposing new reservoirs, expansion of existing reservoirs, and altering management of others (Goode and others 2011). Changes in reservoir management have potential to affect migratory life histories using the reservoirs as well as those populations using cold tailwaters below the reservoirs. Because migratory life histories using reservoirs can have such a dramatic influence on post-event recovery for populations affected by fires and debris flows, the implications of climate change for reservoir management may be important for fish in many locations in the western United States.

Connections to Conservation

The nexus of the four primary stressors for fish—fire, land management, non-native invaders, and climate change—poses a difficult challenge. Land and aquatic managers have critical questions about what to do, and where to do it, and a primary concern is often the conservation of native fish populations. The next section of this report discusses the complexity of integrating the joint conservation concerns of terrestrial and aquatic systems together, but it is worthwhile here to reiterate important concepts related to conserving trout populations. Because the resilience of fish populations to fires is strongly influenced by diversity in life histories, the extent of habitat networks necessary to support persistent populations serves to focus the discussion.

Activities that increase connectivity among suitable habitat patches and existing populations and building or rebuilding local populations in and around large patches are likely to increase robustness to disturbances and species persistence probabilities. In some contexts, however, concerns may exist about increasing connectivity for invasive species as well, and such decisions may need to consider multiple local conditions (Peterson and others 2008; Fausch and others 2009). Under certain circumstances, climate change may reduce threats of invasion by some species (Wenger and others 2011a; Wenger and others 2011b). The most common specific activity for increasing habitat size and connectivity is replacement and removal of road culverts that act as barriers to fish movement. Restoration of local habitat quality to mitigate stresses that constrain population productivity can also encourage more migratory behaviors as well, because productivity of natal habitats is important to migratory life histories (Lucas and Baras 2001).

Management options for ameliorating warmer stream temperatures include maintaining or restoring instream flows or reservoir design and operation, especially on larger reservoirs that stratify during the summer (Neumann and others 2006; Olden and Naiman 2009). Temperature management through canopy manipulation is generally not a reasonable approach for affecting significant portions of

a stream network, particularly larger streams, and is ultimately vulnerable to wild-fire. Exceptions might occur in short sections of streams flowing through meadows if these areas have been significantly degraded by livestock grazing and riparian vegetation and bank structure are substantially altered.

Priority areas for activities that improve resilience to events may not be the same places as priorities for conservation of rare aquatic species. For example, natural areas, like designated Wilderness areas, commonly serve to house “stronghold” or “core” populations of trout, and maintaining representative core populations is a key part of a conservation strategy. Improvements to long-term species outlooks will also be achieved through restoration of areas that can support or be supported by the core populations; so most opportunities for improvement in persistence will likely be in areas partially degraded by historical land management. Priority areas for investment of restoration funding would be places that are most robust to climatic changes (e.g., those that could be made large for the future as well as the present). Places that are large might have a lower priority than places of intermediate size that can be expanded, because greater gains in net persistence would come from making a questionable patch into a core area. While there is an inclination to grow large patches to greater size, there are also benefits to having multiple large patches with somewhat greater geographical separation. A key issue, therefore, is developing more precise definitions of patch sizes needed for persistence by different species and understanding how alteration of disturbance regimes from interactions of fire and climate may affect these patch sizes in the future.

Over time there will also be discussion about the values that we want to conserve (e.g., Rieman and others 2010). Values range from just having some fish in a creek to representation of rare genotypes or phenotypes that might represent important evolutionary legacies. Between, there are values associated with economics (having enough of the right kinds of fish to attract fishing) and ecological functions. To some extent the range in value connects to the potential to substitute other fishes or other processes for maintaining fish populations (e.g., hatcheries). Ultimately, however, retaining genetic and phenotypic or life history diversity will be a particularly important conservation goal related to the changing climate because it offers the primary base from which evolutionary adaptation can take place (Haak and others 2010; Williams and others 2011), and see Rosenberger and others in Advanced Topics of this volume.

Part III: Management Actions and Decisions

Managers of forest, wildlife, and water resources have heard the call for increasing resilience of ecosystems (Walker and Salt 2006; Millar and others 2007; Baron and others 2009; Heller and Zavaleta 2009; Joyce and others 2009; Keane and others 2009; Palmer and others 2009; West and others 2009; Turner 2010). The primary question managers have is what to do about it. The question has three distinct aspects: 1) understanding the actions that could be taken, 2) how to decide which actions to apply, and 3) when and where to apply them. The general set of actions available around fire effects on aquatic systems has a fairly limited scope, although there are many variations on the key themes. While there is limited information on the effectiveness of many actions, there is an understanding of the mechanisms by which they affect risks. Making decisions about solutions has been difficult, but advances in understanding of aquatic ecosystem response to fire supports new ideas in framing key decisions (Bisson and others 2003; Rieman and others 2010).

A. Actions

Action Choices

The choices available to managers to reduce risks associated with wildfire are somewhat limited in their general nature, although there may be many variations in details available for each to increase local suitability. The general classes are:

1. Fuel treatment
2. Aquatic and/or riparian habitat restoration
3. Hillslope restoration
4. Fire suppression
5. Post-fire stabilization

Fuel treatments include a large range of potential activities ranging from carefully managed mechanical removal of specific fuels, to various levels of directly applied fire, to wildfire itself (see the section on Forests, Climate Change, and Fire earlier for more discussion on fuel treatments). The most common goal of fuel treatment programs is to reduce fuels, particularly near human infrastructure, to alter fire behavior and intensity, aid fire suppression, and reduce burn severity (Graham and others 2004). A more general ecological goal may be described as keeping the fuel levels consistent with the type of ecosystem: for example maintaining few ground and ladder fuels in a ponderosa pine forest. One objective of managing fuels is to reduce burn severity of future fires and consequent effects on vegetation and soils, with potential reductions in risk to aquatic ecosystems from thermal and sediment impacts (Hessburg and Agee 2003). Fuel treatments are also used for terrestrial habitat restoration and control of invasive plant species, particularly in riparian areas (Stone and others 2010).

Aquatic habitat restoration is directed at increasing diversity and complexity of aquatic habitats, which generally supports more productive and resilient populations of fishes. Examples of activities include adding large woody material to form pools and provide cover, reconstructing meanders to deepen pools and provide more hydraulic variability, riparian planting to improve shade, or road

culvert replacement to allow migratory fish passage. The specific activities are quite diverse but share the general approach of reducing risks to aquatic systems by improving some aspect of the in-stream and near-stream habitat. As might be discerned from the section on aquatic ecosystems, restoration of connectivity may be critical for many populations, although the risk from invasive species must be considered.

Hillslope restoration consists of a substantially more restricted tool set. In most forest ecosystems, it refers to road repair, upgrade, or decommissioning with the goal of reducing erosion and mass wasting. In a broader sense there are related activities applied to mined, overgrazed, or logged sites. We distinguish these pre-fire activities from post-fire stabilization (see below), which is focused on preventing (resisting) losses to populations. Reducing impacts from sediment over the longer term before fire occurs helps to build populations and communities that are more diverse and productive and, thereby, more resilient to fire effects.

In the absence of strategic planning and implementation of other restoration and enhancement actions, fire suppression becomes the default activity for reducing risks to both forest and aquatic resources. Although there are conceptually short-term reductions in risks associated with putting out the fire, the continued presence of fuels could lead to a longer-term hazard. Thus the choice not to suppress a fire, within the context of a broader plan, can be seen as a fuel management strategy. Fire suppression has been very successful, but the few fires that escape initial attack can burn intensely and severely. Because conditions will eventually occur wherein fire suppression won't be successful, suppression should be only one tool in the "tool box" that is a broader plan for fire management for long-term ecosystem resiliency.

Emergency post-fire stabilization practices are done after fires to "suppress" post fire erosion events. Depending on agencies involved, these measures may be referred to as Emergency Stabilization and Burned Area Rehabilitation (ESR) or Burned Area Emergency Response (BAER). Both programs separate the short-term stabilization activities from the long-term rehabilitation activities; the latter generally have a goal of preserving ecological integrity of hillslope vegetation. The nature of stabilization activities is diverse, but the most commonly applied measures focus on restoring strength to the soil to keep soil particles in place. Runoff control strategies are mostly experimental. In general, emergency stabilization is authorized for protection of human life or property, although it can be applied for protection of special resources as well, including threatened or endangered aquatic fauna. As with fire suppression, it is done in a preventative fashion; so it does little to build resilience in the ecosystem, and aquatic ecologists note that periodic influxes of gravels and nutrients from erosion and mass wasting are important to aquatic habitats in the longer term (Reeves and others 1995). Most stabilization practices offer protection only for relatively common storms.

Effectiveness of Actions

It is worth discussing the understanding and uncertainties of the different choices to clarify expectations for different types of actions. The above listed actions have been evaluated to some degree, but generally for only the most proximal goal of the action—for example, changes in fuel loading for fuel treatments, improvement in number of pools for aquatic habitat restoration, decreased fragmentation for culvert replacements, and reductions in sediment from road decommissioning or post-fire treatments. The more distant, long-term goals of protecting threatened aquatic populations or protecting forest ecological processes are much more

difficult to evaluate because of the large mix of influences on those outcomes. The evidence for these actions improving resilience is derived primarily from mechanistic logic. The complexity of interactions from multiple influences, however, has left questions about even something as seemingly direct as habitat improvement (Bash and Ryan 2002; Palmer and others 2005).

Many forest and stream ecosystems in the western United States contain species that have been present for at least a few million years and that have occupied more or less the same areas as they do now for the last several millennia despite historical fluctuations (Dunham and others 2003; Keane and others 2008). Such persistence in the face of natural dynamics supports an expectation that reliance on natural adaptations is a reliable conservation approach. Unfortunately the reality is that conditions now and expected in the near future depart substantially from those of the last few hundred thousand years. As noted earlier in this document, air temperatures are increasing in ways that are unprecedented, and temperature is important to both forest ecosystems and aquatic species. Although hydrologic cycles are changing with more uncertainty, they are changing nonetheless, and for the most part changes have been in a direction less favorable to present ecosystems. In addition to climate changes, there are many changes in the last century from invasive terrestrial and aquatic species, changing fire regimes, and displacing native species to less favorable habitats. On top of these pervasive changes, affecting wilderness and developed areas alike, the dams and other water management infrastructure have imposed significant, and usually irreversible, constraints on the naturally adapted system.

One simple classification of actions related to fire is between those that are done prior to fire to build resilience or resistance, and those that are done in response to a fire to prevent or reduce harm (Dunham and others 2003). The effectiveness of pre-fire solutions in building resilience depends in part on the potential for managed systems to operate within the limits of the adaptations. Changes in forest structure and climate that alter the spread and intensity of fires, changes in connectivity of habitats caused by infrastructure or invasive species, and changes in habitat quality affecting productivity all impose limits on a general strategy relying on natural resilience, meaning that solutions depending on this strategy need to address, or at least evaluate, multiple potentially limiting factors.

Actions focusing on responses to emergencies can be considered less reliable. Large wildfires, for example, are usually escaped suppression efforts. Similarly, most post-fire strategies include a combination of protection of human infrastructure and temporary evacuation of people to increase reliability of protection of human life and property. In some rare instances, fish have been evacuated for protection of a small population. If a fish population's persistence depends on successfully suppressing fire or its effects in the short-term, it is necessarily at greater risk than a population that has the capacity to weather a fire event and then rebound post-fire.

Some assessment of pre-fire treatments has been done. Fuel management, for example, does not result in fully controlled or completely "tame" wildfires, nor is it 100% effective. It is important to recall that some severe fire effects are desirable for gravel, nutrient, and energy inputs to streams for long-term maintenance of aquatic systems. The implementation of aquatic habitat restoration is usually more directly controllable, e.g., the number of additional logs or constructed pool features is specified in a construction contract. What is less well understood about aquatic habitat restoration is whether productivity is actually increased, or whether fish move from poorer quality habitats nearby, with no net gain in production from a stream (Bash and Ryan 2002; Palmer and others 2005). Road decommissioning

assessments generally show substantial improvement, but remediation of steep slopes is problematic, and it is difficult to entirely erase road impacts (Luce 1997; Madej 2001; Switalski and others 2004).

The effectiveness of post-fire stabilization and rehabilitation has received more scrutiny. Although the intention of the treatments is to protect human life and property by reducing the probability of severe erosion events, the reality is that only the smallest events are prevented or reduced, and larger events overwhelm treatments (Wagenbrenner and others 2006). While smaller erosion events can represent a threat to some isolated fish populations in small streams (Rinne 2003), many populations can rebound from events that do not completely displace them (Rieman and others 2003a). As a consequence, many post-fire stabilization treatments probably benefit homeowners more than fish. Post-fire removal of migration barriers is less beneficial than doing it beforehand, because pre-fire removal allows for development and dispersal of migratory life histories from the stream itself, which is a more reliable source of recolonization (Dunham and others 2003). Post-disturbance removal would at least make the stream available for reestablishment from dispersing fishes.

Hopefully, the reader takes away the message that no single approach will be adequate to guarantee fish persistence in any single location, much less across the diversity of situations in the West. A key concept is reliability of the approach, and reliability analysis can be a useful approach in thoughtfully and efficiently deciding which steps to take where.

B. Framing Issues and Decisions

The Broad Scope of Debate

A primary issue facing land, wildlife, and water managers is to understand what can be done to improve the prospects of fishes, particularly threatened, endangered, or sensitive species and stocks, in the face of wildfire and a changing climate. Sometimes this goal has been framed in a way that is competitive with the health of the terrestrial landscape, usually by way of recognizing the threats that forest management poses for aquatic ecosystems (Rieman and Allendorf 2001; Bisson and others 2003). The challenge that all resource managers face is in developing resilient and resistant landscapes encompassing both streams and forests.

Managers from different disciplinary backgrounds, often in agencies or departments with differing missions, have developed tools and approaches to supporting those ecosystem components with which they are most familiar or have the most control. Thus, foresters have focused on forests, while civil engineers develop strategies to maintain water supplies, and fisheries biologists find ways to maintain aquatic ecosystems. Some solutions that are optimal for one resource may be less optimal for another, or even harmful to it. As a consequence, from disciplinary perspectives, concerns of other disciplines are sometimes viewed as constraints.

Forest managers have focused on fuel reduction through manual thinning or application of both intentional and unintentional fire. There is, however, a great deal of concern about effects of management intervention (e.g., DellaSala and Frost 2001; Rhodes and Baker 2008). Direct vegetation management represents a continued and in some cases additional threat to aquatic systems through management-related disturbances, including roads. However, even natural fuel treatments through fire use can be controversial in some circumstances (Holden and others 2010). Solutions are commonly suggested with somewhat universal

framing, e.g., we need to thin the forests to prevent catastrophic or uncharacteristic wildfire or thin forested riparian areas to prevent damage in the event of a fire (O’Laughlin 2005; Roloff and others 2005). A primary framing for foresters is in suggesting that the short-term impact produces long-term benefits.

Aquatic managers have focused on erosion prevention, usually road related, as well as some aquatic habitat restoration. Road decommissioning or related restoration techniques can be helpful in reducing chronic sediment loading (Switalski and others 2004). Removal or replacement of road culverts to improve fish passage has been viewed as an important activity to increase the connectivity and size of habitat patches (Clarkin and others 2005). Aquatic restoration also includes actual habitat manipulation to add wood, for example, even though the benefits of such activities are less well understood (Palmer and others 2005). Introduced species are a critical issue in many areas, and consideration of the relative benefits of connectivity or habitat restoration to invasive species may be important in some cases (Peterson and others 2008).

Forest and aquatic managers, alike, have drawn heavily on strategies depending on fire suppression and suppression of the effects of fire through post-fire stabilization. Although fire suppression as a general approach to ecosystem management is not widely supported, it is accepted in situations where there is a threat perceived to a valuable resource, e.g., people and their property or endangered fishes. Post-fire stabilization is generally not perceived as negative for terrestrial systems, except for practices that introduce invasive plants (e.g., Monsen and Shaw 2001; Shaw and others 2005). These approaches are primarily applied for high value or irreplaceable resources, but cost and effectiveness are critical issues. Treatments are not completely reliable, and there is some irony that the cost and effort is great enough that one might be led to expect they provide comprehensive protection as opposed to a last ditch effort.

Managers of municipal watersheds have often had to work closely with forest managers to protect city water supplies from fine sediment due to timber harvest within forested municipal watersheds. The recent increase in frequency of large fires and the potentially severe impacts of wildfire on water quality have made some water managers proponents of fuel treatments and aggressive fire suppression and post-fire treatments within watersheds (Graham 2003). Although an additional perceived benefit of the fuel treatments is increased streamflow, research does not support the hypothesis, particularly for more fire-prone forests where fuel treatments are ecologically recommended (Troendle and others 2010).

Increased withdrawal of water may be an impending issue with respect to fire and fishes in a changing climate. There is concern about the potential need for increased irrigation in a changing climate to satisfy higher evaporative demands and longer growing seasons. Stream segments dewatered for irrigation may pose critical barriers to migration, particularly in more arid parts of the West (Rieman and others 2003b). In some cases, technological fixes may be available to shift withdrawal locations, but more commonly there may need to be discussion of water rights for instream uses. In locations where climate change is driving deeper droughts or lower summer streamflows (e.g., Luce and Holden 2009; Leppi and others 2011), water diversion issues may become more severe and urgent. Similarly, the construction and operation of dams for water storage to offset timing shifts in streamflow could impair migration.

Despite a desire for a blanket answer covering a range of climates and landscapes, solutions depend on a complex set of contexts. None of the tool sets is without controversy or consequence to other resources. Forest management may threaten aquatic systems, particularly through roads, but road deconstruction could

limit future forest management options and may reduce fire suppression success. While we protect the forests for water quality and supply, withdrawal or storage of that water for use may impact fisheries, and affect riparian conditions. Many of the solutions may have high and unpredictable costs as well, adding issues of economic efficiency to an already complex ecological problem.

The number and dimension of issues impinging on decisions about fire is high, and the decisions can be difficult and, sometimes, overwhelming. Multiple competing interests and issues can create an impasse that could in itself yield an outcome that is optimally detrimental. While there are no magic bullets for cutting through all of the different considerations, there are ways of looking at the problem that can simplify some aspects.

What is important is setting general principles that help to: 1) build frameworks and logic for broad decisions, 2) simplify the issues for managers and interested publics who have diverse, often non-technical backgrounds, and 3) suggest process and perspective to help solve problems and puzzles where the “knots” are hardest.

Simplifying the Frame

There are a few critical ideas that are helpful for simplifying the complexity. In part they help build a hierarchy by noting overarching priorities, and they point to interesting features of the problem itself that reduce conflict. We summarize them as five general principles to apply to aquatic-terrestrial planning for fire:

- 1) Holistic approaches are required,
- 2) Spatial arrangement has relevance,
- 3) The system is dynamic,
- 4) Sustainable solutions are needed, and
- 5) Timing may be critical.

Perhaps these are more reminders than principles to people well versed in natural resources management; nonetheless, they provide guidance to sort through the myriad choices presented to us.

The need for a holistic approach has already been stated and would seem to contribute to the complexity described above. It is repeated here as the first principle because a clear expression that there is only one ecosystem to manage helps immediately deemphasize solutions that harm one component of the ecosystem to preserve another. Although common usage of “ecosystem” (including our own) treats different locations within a watershed along lines of scientific discipline (e.g., riparian, aquatic, forest, rangeland, and terrestrial ecosystems), the interconnectedness of these parts is an important feature of the fish-fire-forest problem. Solutions treating just one aspect of the ecosystem may be considered under particular circumstances; however, if such circumstances are limited in space (see next principle), there may be alternatives that are more broadly beneficial or solutions to one problem that are benign to other ecosystem components.

The spatial arrangement of forests and aquatic habitats at risk has a profound influence on reducing apparent conflicts. Flows of energy and material through the landscape control the degree of interaction between land and water, and thoughtful mapping and zoning may be applied with these concepts in mind to reduce conflict compared to more generalized application of solutions (e.g., Cissel and others 1999; Dellasalla and others 2004; Rieman and others 2010). For example, forest management or fire, either one, would do little to impact fishes upstream, unless, for instance, a culvert blocking upstream fish passage were placed. By

systematically mapping where restoration may be needed to help either forests or aquatic habitats, there are opportunities to highlight large areas where no work is required, places where only aquatic work might be required, and places where forest work would not affect sensitive aquatic habitats. That remaining portion of the landscape where forest work could degrade aquatic habitats would then become the focus. The joint spatial alteration of fish and forest habitat through historical forest harvests represents some further opportunity to improve conditions in the same places for both aquatic and terrestrial ecosystem components (Rieman and others 2000; Dellasalla and others 2004; Rieman and others 2010). Although work required to restore these forests might be directly affecting already degraded aquatic habitat, identification and remediation of the causes of that degradation in concert with the forest treatments could produce net benefit for both ecosystem components.

The fact that the system is dynamic favors solutions seeking to build resilience over those trying to protect against dynamism. If the motivation behind a forest treatment is to make fire more manageable (e.g., more easily suppressed) than it might be under natural variations in fuel load, then there may be negative consequences for ecosystems. Both the forests and the aquatic habitats are adapted to fire and have co-existed successfully with fire for a few millennia. Although climate change has already altered fire regimes in some locations substantially compared to the 20th century, past variations in fire synchrony and associated mass wasting have matched current levels within these millennia (Kirchner and others 2001; Meyer and Pierce 2003; Whitlock and others 2003; Pierce and others 2004), implying that adequate biological mechanisms exist to survive widespread and severe fire and its consequences. Where historical anthropogenic effects have impaired the resilience of aquatic or forest systems by altering the spatial structure or connectivity of habitats, risks are higher.

Sustainability relates to the level of external effort, as energy and materials, required to maintain system processes. Solutions requiring persistent large outlays to maintain a particular condition through a combination of fire suppression and thinning, for instance, would be expensive and probably impractical for long-term application except where very high values, like homes or other infrastructure, are protected. If we look across the broader landscape, there are insufficient resources available to public land management agencies to correct current issues immediately. For example, there is a \$4.5 billion backlog in road maintenance, some related to water quality impairment (U.S. Forest Service 2011 Budget Justification). Sustainable restoration practices require prioritization of the most important issues, e.g., determination and targeting of the most critical places and the treatments with the most effect on desired outcomes. As work is done and the dynamics of disturbance play out across the landscape, periodic reevaluation may be beneficial.

Timing is critical because disturbance is imminent in a dynamic landscape. Given both the high technical complexity of designing landscape-scale solutions to persistence in a dynamic environment and the huge challenges in convincing a diverse public that it is all in their best interest (i.e., in the interest of their pocket books, their houses, their safety, and the environment), it is tempting to put off concrete decisions and actions until a wildfire provides a seemingly unquestionable mandate. There is a certain degree of hubris in waiting for a fire to occur before acting, however, and it is increasingly recognized that both the forest and streams could suffer in the aftermath of such an event without some preparation. It is also expensive and dramatically limits the scope of choices available to managers. One need only go as far as one's own dentist or doctor to hear the benefits

of preventative care. It applies for forests and fish as well. For example, the existence of migratory stocks provides one of the stronger guarantees that a population will persist despite a short term setback; however, migratory stocks must be available prior to the occurrence of a major disturbance (Dunham and others 2003). Likewise, fuel treatments maintaining ecologically appropriate fuel levels, vertical structure, and spatial patterns create greater opportunities for managers to use natural ignitions to continue to maintain the situation. Perhaps some would note this principle looks redundant since a strategy relying on fire suppression and post-fire stabilization alone would allow fuel buildups that reduce forest resilience, imposes a static conceptualization for the forest and aquatic habitats, and requires substantial resources. In other words, it conceptually violates the first, third, and fourth principles. We specify the timing as a unique principal for emphasis and clarity. We accept that in the short-term, there may be no reasonable alternatives to suppression of fires and post-fire erosion in some locations. Prioritization may place restoration work in other areas first, for example, or physical isolation may preclude major improvements in fish migration.

The point of these five principles is that even though there are seemingly conflicts between management actions for different disciplines, there are also parallels and complements in process. Taking advantage of complementary processes in planning requires understanding the fundamental behaviors shared across resources and acknowledging realistic constraints on managers. Individually, the five principles look like truisms. Taken together, however, these five principles allow for a first order evaluation of most proposed actions or strategies.

Applying the Principles

Most individual proposed activities, e.g., a road decommissioning, a culvert replacement, or a fuel-reduction project, would fail screening by the five principles if designed outside of a more comprehensive plan that describes the spatial arrangement and sequencing of projects to reestablish dynamic ecosystems of multiple resources in a financially sustainable way. For example, an individual NEPA analysis stating that a particular pile and burn project would reduce the risk of wildfire and therefore sedimentation in the stream, cannot really address whether the sediment input from that site has any relevance to fish populations (negative or positive), or whether a different project would better achieve goals of sustainability or restoration of dynamic processes. Thus, there are two reasons why a stand-alone proposal of this fuel treatment project would be inadequate: 1) there is no context of spatial or temporal prioritization (e.g., watershed analysis or cumulative effects analysis), and 2) it suggests an inappropriate scoping (e.g., a belief that sediment load is the primary issue to address for streams). Although fuel treatment projects are usually proposed with benefits to vegetation in mind, when threatened or endangered fishes are potentially affected, a benefit (or at least a lack of risk) to them must be shown. Simply scheduling treatments in areas without threatened or endangered fish to avoid the regulatory problems, however, equivalently misperceives the value of an integrated plan.

The first point about lacking a larger contextual relationship to other projects is well recognized by land managers and has resulted in technical planning initiatives that are spatially comprehensive, such as watershed analysis (FEMAT 1993), fire management plans (NWCG 1995), and transportation planning (e.g., Forest Service Handbook 7709.55). Although all of these are carried out using interdisciplinary teams, the very fact that there are different kinds of plans (along disciplinary lines) reveals a lack of interdisciplinarity in their development or inception. A

brief reading of such plans or the manuals for their development shows a series of descriptions of issues with one particular resource, the proposed solutions, and comments on the impacts to other resources. Such plans are commonly tiered to more comprehensive area planning documents (such as Forest Plans in the Forest Service or Resource Management Plans in the BLM), but even those tend to have a series of chapters with resource-specific guidelines. Considering that there is probably more effort involved in developing four or more plans for any given land area (and trying to roughly tie them together each time) than a single complex intertwined one, it is not much of an assumption to believe that the issue is not so much a lack of will as a lack of a well stated frame to build them on. Although the hypothesis that building consensus among multiple individuals from diverse disciplinary backgrounds is difficult, it cannot be ignored.

The five principles can serve as a frame for interdisciplinary planning across the issues of fire, fuels, roads, and aquatic habitat. Above, we noted how each resource could be viewed as conflicting with another; however, when viewed through the frame provided by the five principles, the importance of the complex interrelationships among them becomes more apparent. In short, although there would seem to be many potentially competing needs, in reality, only a few would address problems holistically and sustainably, and these would acknowledge disturbance and recovery in patches over the landscape.

Viewing the problem in this way leads to steps that can shed light on the complex relationships and help diverse teams decide which actions to take:

- 1) Identify resource specific needs, limitations, or vulnerabilities (multidisciplinary step)
- 2) Identify where they are in conflict (and conversely not)
- 3) Identify where sequencing (order of multiple activities) could ameliorate conflicts
- 4) Prioritize and schedule non-conflicting tasks
- 5) Creatively solve remaining issues—describe and quantify risks and means to obtain feedback to guide future management

The first step is familiar, identifying the needs of the ecosystem in recognition of a dynamic system and the need for sustainable solutions. Some may frame this step in terms of ecosystem restoration, sometimes as restoration of process. It can also be framed as a vulnerability analysis, or alternatively as a reliability or persistence analysis. One approach is to identify what makes the forests or aquatic resources vulnerable to fire and suggest what steps could provide resilience or resistance to those vulnerabilities. It is a multidisciplinary process, with specialists in each discipline using their understanding to clearly articulate the specific aspects of current conditions that could result in an anomalous outcome from wildfire. This could be seen essentially as a diagnostic examination: for forests determining where they are at risk for rapid spread and or homogeneous severity, and for aquatics determining what would maximize survivability following fire.

Although this step is primarily multi-disciplinary, there should be clear recognition that a holistic solution is being sought. Overstated “needs” for individual ecosystem components can lead to unnecessary conflicts being identified in the second step. Each team member must recognize that there is only one ecosystem, and that while there may seem to be tradeoffs between individual resources in an ecosystem, the objective of the interdisciplinary team is to find the balance that best allows the ecosystem to thrive without substantial ongoing external investments. There is potential at the stage of identifying needs for interpersonal skills

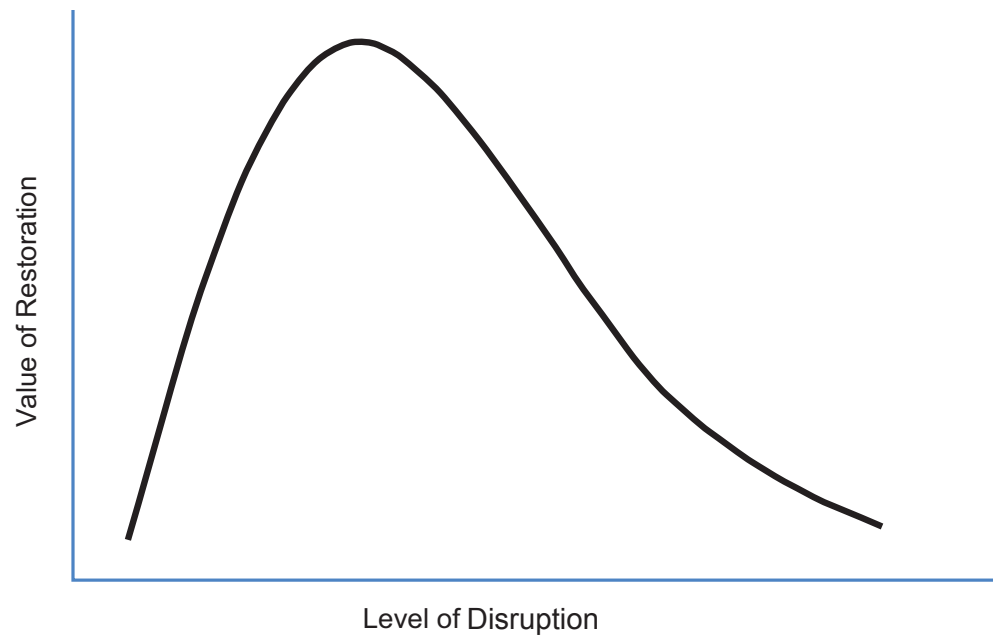
to override objective assessments. There is a need for a combination of personal determination, restraint, and leadership by the team members to separate interpersonal conflict from real resource conflicts.

Identification of conflicts or agreement and complementary needs is the key interdisciplinary step. As outlined in the general framing that is commonly experienced, there may be some general expectations of conflict a priori about managing forests in places with sensitive fish species. Unfortunately, this expectation is uninformative for decisions, and it is important to note where activities for the benefit of one component do not impair another, for example, identifying where impacts from forest management practices would affect areas downstream of sensitive life stages of fish, or where a road removal would not reduce access for fuel treatments. Rieman and others (2000) provide an example where impacted forest and stream ecosystems tend to be in the same places and superficially would appear in conflict, e.g., impaired fish populations being particularly sensitive to additional management. Recognition, however, that such sites may represent opportunities for more comprehensive treatment could reveal more opportunity on the landscape than conflict. Such recognition could only occur in an interdisciplinary process. Misidentified and overgeneralized identification of conflict can also impede finding solutions, and management of personal and interpersonal factors is important in this step.

Roads are a primary source of conflict in management of public lands so warrant additional discussion. Roads are not a resource; they are a tool to manage, access, or benefit other resources. Roads only have value for the resources they access. In this framing, roads do not have “needs”; however, if they serve to more sustainably manage a resource or obtain the benefits of a resource, the roads could be viewed as a “need” for that resource. More objectively, they represent a value with respect to a particular resource, e.g., a recreational resource becomes more accessible, or a mineral resource becomes economically viable. The environmental and financial costs of the road could then be objectively evaluated against such a value. A key point to consider is what could be substituted for the road when determining its value. A well designed and maintained road could ultimately be more sustainable than other forms of access, depending on frequency of access and hauling requirements and the topography to be crossed. Roads are a shared value when they are used to access multiple locations, perhaps with different ownership and different land uses, so coordination of road systems is often needed.

Some conflicts could be reduced through sequencing of treatments. Most fuel treatments require roads to be practical. Fortunately, fuel treatments are likely most needed in places that have roads now (Rieman and others 2000). Although those roads may be the most practical means for a fuel treatment now, if the fuel treatment can be maintained in the future through means not requiring roads (e.g., wildfire use), those roads can be decommissioned. If they might be needed in the future, and values warrant ongoing costs, they can be upgraded to be resilient to storms. This kind of coordination allows both improvements to terrestrial and aquatic components. Note, however, in any arrangement whereby future risks are reduced following a short period of increased risk could be self-defeating. For example, in the case of a species extirpation related to a temporary risk increase, no amount of future risk reduction would be of any benefit. If the increase in risk is more than nominal, and there are no additional measures that can be taken to reduce the risk, the proposed activity would need to be identified as potentially conflicting and set aside for creative thinking about how to quantify and manage through the uncertainty.

Figure 31: Most benefits from restoration are likely to be found from efforts in places only mildly or moderately degraded.



Prioritization among the non-conflicting actions requires identifying which treatments provide the greatest return in terms of the objectives for the ecosystem, again a thoroughly interdisciplinary process. Various criteria are available for prioritization and, ultimately, monitoring progress. Two of the biologically critical ones are persistence of sensitive aquatic stocks and vertical/spatial structuring of forest stands. Elements overlap, but there are also many independent aspects. The key question for managers may be to determine which action sets (including those with sequencing) most affect persistence and improve forest structure. It has been suggested that places with some intermediate levels of historical alteration/disturbance may be the most productive for restoration results (e.g., Dellasalla and others 2004) (Figure 31). These places are not necessarily so degraded that connectivity can only be gained in increments of several meters, nor are they in places where improvements represent icing on the cake. Locations where persistence probabilities for fish species are already high would be lower priority than places where fragmentation has put species at significant risk. Places where the original genetic diversity values are lost are lower priority than preserving those currently at risk, but still extant, although eventually reestablishment of extirpated populations could be a sustainable approach to conserving the remaining diversity.

In places where conditions prohibit improvement in the condition of some ecosystem components without harm to other components, more creativity will be needed. A combination of historical practices, changing climates, and introduction of species has resulted in many locations where conflicts are real. In such places, risks for both forests and fish are likely high, with potential for severe effects from wildfire or loss of a local fish population. In these circumstances, choices must be made, and even deferral is a decision with consequences. These places may show conflict because the sensitivity of the ecosystem is highest and risks are greatest, and these are the places in greatest need of decisions and actions.

We also note the utility of the first four steps in helping with the tough “knots” that are left over. Within the map of non-conflicting treatments developed from the first four steps is a list of practices or activities that will alleviate stresses on the most stressed parts of the ecosystem. For example, restoring structure of forests in one area ultimately benefits places nearby by altering fire spread and fire risks

from off site. Similarly, restoration of fish migratory patterns in larger river systems could be a critical step in ultimately providing stronger persistence for what is now an isolated habitat.

Even the most creative solutions will require accepting risk and impacts, at least in the short term, to reduce longer term risks. As noted earlier, the balance between short- and long-term risk is important to consider. If a short-term high-impact action can be mitigated through short-term high costs to effect a change that makes a situation more sustainable in the long term, high costs may be justified. A small, isolated fish population, with unresolvable downstream invasive species issues, for example, poses a difficult problem. One can reduce fire risks upstream with forest management, but risks from the management and roading could pose more certain risk to this isolated population. Potentially, however, helicopter access and hand work on finer fuels could dramatically reduce management related impacts. The cost would be high in the short term, but if it allowed wildfire use in the future to maintain fuel levels, it may be more sustainable in the long term. If such steps produced an anomalous fuel condition for the forest (e.g., thinning a mixed fir forest), but the condition could ultimately be maintained with less expensive broadcast burning, it would still represent a more sustainable and likely more reliable way to conserve an isolated population than relying on suppression alone.

Decisions about values include more than forests and fish. Recreation access uses roads, and there will come a question as to which is more important: that particular access or the aquatic resources kept at risk by the road. In streams facing the potential of invasion from native species, the choice to place a barrier or not involves tradeoffs in risk between the invasion and potential for loss of genetic diversity in some places versus the risks associated with isolating a population, and people will need to decide between the different values the fish provide in that stream (Rieman and others 2010). Dams for water supplies are sensitive to sediment from management, thus limit options for land management. They are also sensitive to the major water quality changes from fire, such as metals, nitrates, and sediment. Fish are also sensitive to these and are made even more sensitive by the fragmentation associated with the dams. Two choices are to “engineer” the ecosystem tightly and dampen disturbances, which is expensive to the land managers, or to design a more reliable water supply system (e.g., with other intakes or ground-water reserves for temporary use) that could operate within the dynamic landscape. Both choices are expensive, but the costs are borne by different parties.

The difficult conflicts will require complex and creative thinking. Ideally we want to go back to the five principles to develop a holistic solution that can be sustained in time recognizing landscape dynamics and using space and time to build strategies. A variety of constraints will impinge on some of these principles, and sometimes the best solution does not follow all of the principles in the short term. There are too many stories to analyze here, but the sample should demonstrate the reality and complexity of the situations, but also their potential rareness in the general landscape. If we can work in the places that are not difficult first (e.g., where the fish are resilient to management effects or where the management can help fish habitat), perhaps we can lessen the stresses on the difficult places too.

Evaluating Risks

Balancing tradeoffs or optimizing for priorities are inherently quantitative in nature; even in a world of dynamic and stochastic processes and events, there are gaming theories that could be applied to rationally decide the most astute course of action (Bishop 1978). Such rational decisions, however, require objective

valuations of alternative outcomes, which are rarely agreed upon with respect to natural resources, and in any event are not technically allowed as a basis for decisions on endangered species. Where the decisions bear on traditionally valued resources like trees or homes, or even where they bear on replaceable populations of fishes, there is the potential to apply rigorous tools. When the mix of values involved includes officially protected species, the decision frame is dramatically reduced to evaluate whether an action increases or decreases risks to the species in question.

As a consequence, the risk to these species becomes an overriding factor in the decision making, and quantification means evaluating risks of alternative actions. Technically, risk is defined as the probability of loss multiplied by the value. If we are only evaluating increases or decreases in risk for a given resource, however, we can assume the value remains fixed, if immeasurable, and focus on probabilities of loss or persistence. Within a stochastic framework, probability is commonly expressed in terms of time, e.g., a probability of 0.01 yr^{-1} would be a 100 year event. Since we know that fire events are a limiting condition for fish, and because fire severity/size characteristics are rarely expressed in this temporal probability framework, one could further limit analysis to whether a population persists in the eventuality of a fire. For a management action with a short-term impact, the equivalent question is whether the population would survive the action and its consequences. The question is whether the reduction in future risks from the management action outweigh any temporary increases in risk with the management action.

With respect to fishes and fire, the key concepts that apply to long term persistence of a local population are metapopulations and life history diversity. Metapopulation theory addresses the spatial interactions between patches of habitat and patches of disturbance, examining how local populations that are lost to disturbances can be refounded from other nearby occupied habitat patches (Rieman and McIntyre 1995; Rieman and Dunham 2000). Life history diversity relates to whether there may be migratory individuals from a particular stream that would be elsewhere (downstream in a reservoir or the ocean) when the fire-related effects occur (e.g., Hilborn and others 2003). Both means of repopulating after disturbance depend on either a large local population or a well-connected population. Essentially, these are populations where disturbances cannot simultaneously affect all potential refounding sources, including fragments of a given habitat patch, other nearby populations, or downstream migrants. The primary metrics that emerge as critical control on persistence are the 1) sizes of habitat “patches” and the 2) connectivity of those patches. Productivity within patches can be important too, in that it interacts with their size and connectivity. If habitat quality is low, reproduction in undisturbed habitat will not be effective to reinforce losses elsewhere. Also if productivity is low, migration may not occur, or could be less successful because small migrating fish are more susceptible to predation.

Sometimes, the view that sediment is an impact to aquatic biota is overgeneralized, and the comparative “risk analysis” is boiled down to a comparison of total sediment mass from fire versus that from management (Istanbulluoglu and others 2004; O’Laughlin 2005; Roloff and others 2005). Unfortunately this oversimplification of the ecological response of streams to sediment inputs leads to 1) stalemate in some circumstances (in areas with steep slopes), 2) negative decisions (when chronic road sediments are less than the total long-term pulsed sediment), or 3) low efficiency decisions (e.g., to rely on post-fire stabilization). A contrast of assessments done for relatively low gradient slopes (O’Laughlin 2005) to one on steeper slopes (Istanbulluoglu and others 2004), for instance, shows that while the

total sediment over the long term is higher from fire than forest management on low gradient slopes (considering surface erosion only), there is little difference in estimated sediment mass between fires and forest harvest for steep slopes where mass wasting might occur (Luce and others 2005). Such an analysis would show no improvements for environments with steep slopes, providing little guidance. More problematically, this kind of decision criterion would preferentially select frequent fine road sediments inputs over infrequent but more severe inputs from fires, which could have negative implications for aquatic habitat in the long term (e.g., Reeves and others 1995). Modeling the aquatic habitat response based on total sediment loads does not recognize the dynamism of the coupled forest-stream ecosystem but instead drives management toward a low sediment, low variability state, that cannot persist without large energy and financial inputs, such as fire suppression and post-fire stabilization.

An alternative model is to consider the length of aquatic habitat simultaneously affected by debris flows, temperature, and sediment effects within a given habitat patch, and its connectivity to migration corridors. Decisions under such a model are likely to emphasize resilience options. Within this kind of persistence model, one could describe the effects of temporary sediment loads on population productivity versus the effects of wholesale habitat loss. Such a model is also more likely to reflect the benefits of careful timing and location of various treatments (e.g., forest thinning, road decommissioning, or culvert replacement) to avoid synchronous risks over a patch or stream segments. More importantly, however, it would clarify which measures do the most to increase connectivity, productivity, and size of population units.

Persistence analysis should also address solutions requiring sustained external inputs. Sustainability relates to how reliant a strategy is on future financial and energy inputs. Reliability depends both on whether a particular effort-intensive option is effective and whether or not it gets done. There is a certain irony to managing systems that can survive a catastrophic wildfire but not a recession. It can be difficult to show performance and accountability for “disasters averted” (e.g., extinctions), particularly when those disasters are rare in time. This makes it difficult to maintain funding for approaches that require continuing or repeated financial outlays. This concept is relevant to strategies incorporating fuel treatments, fire suppression, and post-fire stabilization as elements. If the fuel treatments promote resilience to natural fire events and allow for more passive management of the system in the event of fire, they are sustainable. If, however, fuel treatments are done to reinforce suppression efforts, e.g., an increase in resistance to fire, then it is likely that such treatments would require sustained efforts to maintain. The uncertainty in future funding (and therefore implementation) for a proposed program could be directly incorporated into a formal risk/reliability analysis estimating persistence probabilities.

Addressing Uncertainty

One of the values of a more formal and quantitative assessment of risks is in clarifying and defining uncertainties. Disagreements about the perceived risks associated with either fire or forest management can be strong sources of debate about endangered species. Discussion about the acceptability and relative tradeoffs between particular risks can be long, expensive, debilitating, and difficult to resolve (Rieman and others 2010). By placing decisions within a risk analysis framework, two different aspects of uncertainty are separated: those related to chance or luck (e.g., future weather) and those related to what we know about the system (e.g., the

probability of different future weather events). The more clearly and specifically the risks are defined, the easier it is to determine the degree to which uncertainty about outcomes affects the balance of risk. For example, although there may be wide error bars on the amount of fine sediment reduction that may be achieved from post-fire stabilization, we also know that fish populations are more sensitive to rare mass wasting events. Because the post-fire stabilization has little influence on debris flow events, the uncertainty about performance of treatments on fine sediment loads has little leverage on the outcome for fish.

Costs of uncertainty are manifested in a variety of ways. The most common is unresolved disagreements, which are ultimately reflected in agency budgets and morale. Incorrectly applied action or inaction (sometimes related to unresolved disputes) can also result in local extinctions of threatened and endangered species, which is potentially the worst outcome. Alternatively we may see only a loss in other values, such as recreation, grazing, or timber, derived from forest and stream ecosystems. Wasted restoration efforts may seem like an economic issue, but they may also represent continuing risks in other places. None of the outcomes from poor or incomplete information are positive, but some are more severe than others. Discerning the places and circumstances where ignorance is relatively benign is an important step forward in generating potential options for decisions. Those topics where not knowing something can generate severe consequences represent a clear priority for generating information.

This synthesis illustrates that we have a general understanding of the relationship of some fishes to wildfire in semi-quantitative terms (Bisson and others 2003). One key uncertainty is the scale of habitat patches required for persistence in time (Rieman and others 2010). While observations of currently occupied and unoccupied habitats provides some guidance, actual testing of the fish population response to fire is lacking. Although uncertainty has been expressed over the degree to which various fuel treatments affect the severity and spread of fire (e.g., Rhodes and Baker 2008; Stone and others 2010), there is agreement that disturbance patch sizes of natural forests tend to be smaller than in regulated forests or spatially homogeneous forests (Miller and Urban 2000; Hessburg and others 2007; van Wagtenonk and Lutz 2007). Disturbance patch size may be important to some fish populations, but less so to others. For instance, populations in very small patches can be affected by small disturbances, so they are relatively insensitive to the distribution of fires beyond a nominal size. A better understanding of which fish populations benefit more depends on the relationship of patch scale to long term persistence. The degree to which stream heating occurs after riparian thinning, its recovery rate, and the degree to which thinning reduces losses in fire remain unanswered questions that affect decisions for specific projects (Stone and others 2010). Perhaps the more important question is how stream heating would affect patch size or geometry, which depends on the distribution of fishes, and the degree to which they are constrained by temperature. More broadly, and we suggest more problematically, the lack of information about the current status and condition of fish populations and diagnostic information about their limiting conditions at local population scales most strongly limits risk assessments.

Risk reduction is not just about performing some action in the landscape, it is also about making fewer mistakes, which will require better information. Inventory, monitoring, traditional research, and adaptive management are all important in gathering the information that will be most important to decisions. In applying the five principles, we outline a procedure that identifies areas where activities are not in conflict, offering ideal opportunities to learn from earlier actions in local areas. Analysis of fish populations and habitats in the wake of large fires would

also address some of the key uncertainties, particularly related to scale of sustainable habitats. Although members of interdisciplinary teams who are charged with assessing risks under various alternatives recognize the value of inventory, there seems to be less understanding of the value of such information to those providing funding. There is, in general, much more support to perform actions than to design them, which suggests a need to better estimate the value of inventory and monitoring efforts for informing actions.

Finally, we should note that what the climate will do in the future may be one of the largest uncertainties before us. Although it is clear that temperatures will increase, much less is known about precipitation trends. Fluid dynamic theory consistently predicts increases in precipitation at high latitudes (>50 degrees) and broadening of the arid sub-tropical (~30 degrees) zone. Between, there is large uncertainty (Solomon 1986). Within the western United States, for example, there is substantial uncertainty with respect to changes in orographic precipitation from mountains (Dettinger and others 2004; Kirshbaum and Smith 2008). One of the clear needs that emerges from evaluating risks is the need to pay attention to what changes actually occur and what the responses are to those changes. These sensitivities provide important context to which habitats will be most resilient in the face of fire in the future.

C. A Changing Climate

In contrast to the comparatively immediate and substantial effects of fire, changes in weather statistics associated with climate change seem subtle or gradual, although persistent. An important question is how to think about the consequences of climate change (the more gradual but steady effect) on how forests and aquatic ecosystems respond to and could be managed with respect to fire. The climate change adaptation literature has already noted the importance of increasing resilience to temporary disturbance in ecosystems (Millar and others 2007), which begs the question of what more can be done than what has been suggested for increasing the resilience of forests and stream ecosystems to fire (Bisson and others 2003).

Two primary issues are important to discuss: 1) the altered contexts of forest and stream ecological dynamics, and 2) an increased urgency, particularly for restoration of historically disrupted habitats. In earlier sections we saw how climate change impacts are constraining habitat patch sizes and reducing habitat connectivity for fishes. Given that size and connectivity are critical parameters for the resilience of fish populations to fire, climate trends are causing changes that oppose what managers would like to create. Climate, for example, has contributed to the increased number of large fires in the western United States (e.g., Westerling and others 2006; Morgan and others 2008; Dillon and others 2011; Holden and others 2012). The rapidity of the changes is challenging ecosystem responses, and places where resilience has been impaired by management, such as through isolation of streams (Dunham and others 2003) or homogenization of forests (Hessburg and others 2007), now have compounded risk factors.

Climate changes are also reducing the effectiveness of resilience adaptations under natural conditions, suggesting that strategies for managing fire that assume stationarity in climate could be risky. Millar and others (2007) have suggested measures for adaptation to climate change, including building resistance to the changes and taking measures to help transitions occur more manageably, as opposed to catastrophically. They offered suggestions to essentially embrace the coming changes, some of which seem controversial, e.g., facilitating range shifts.

Some suggestions are derived from basic principles in conservation biology and can be revisited in terms of resistance, resilience, and facilitation for the future. Their key points are [with editorial license for emphasis]:

1. Experiment with [networks of] refugia
2. Realign disrupted conditions
3. Increase redundancy
4. Expand genetic diversity guidelines [introduce individuals from other parts of range]
5. Facilitate migration [but applied within current ranges]
6. Promote connectivity [but reduce contagion of fire, disease, and pests]
7. Manage for [and take advantage of] asynchrony

In Millar and others (2007) some of the ideas presented were offered in the context of helping make transitions to new habitats successful, but all of these steps could be equally well applied in resisting effects of climatic changes on loss of local representation of species and promoting resilience. If such steps delay the consequences of climate change, they allow more time for decisions about species, adaptation of species, or successfully reducing carbon loading in the atmosphere. How these ideas are emphasized relative to one another and how they are functionally implemented may look different in aquatic and forest ecosystems.

Climatic changes can be viewed simplistically as shifting suitable habitat ranges higher in elevation and farther north. Elevation-wise distribution shifts can be accomplished more easily than latitudinal shifts, owing to much shorter distances for a given temperature change and the lack of any need to cross potentially large unsuitable terrain. This latter issue can be particularly troublesome for fishes, who are constrained to move within waterways, many of which are blocked by dams or geologic features. In general, remaining aquatic habitats for cold-water fishes will be higher in watersheds with steeper channels that are more prone to post-fire debris flow disturbances. They will also be smaller streams with greater vulnerability to drier dry years and lower summer flows (e.g., Barnett and others 2008; Luce and Holden 2009; Leppi and others 2011). Even species that are less temperature sensitive may have reduced net productivity and fewer migratory individuals if temperatures warm (Dunham and others 2007). One of the primary effects of climate change is to pinch populations into increasingly small, isolated, lower quality, and dangerous habitats. One of the key weaknesses of refuge based conservation programs is that a refuge can be lost to an individual event (Williams and others 2011). Recognition of this condition leads to a generally pessimistic view of the fate of species and connected ecosystems and could cause one to question the utility of suggestions about expending effort on building resilience. We would like to counter that pessimism.

The changes occurring to ecosystems are diverse in nature, and not all habitats are changing at the same rate. Some will change faster than ecohydroclimatic models would suggest, while others are changing more slowly. There are several examples describing mechanisms of fine grain heterogeneity providing sustained water flows or cooler temperatures, at least temporarily (Luce and others 1998; Baxter and Hauer 2000; Hari and others 2006; Lundquist and Cayan 2007; Tague and Grant 2009; Millar and Westfall 2010; Holden and others 2011). These areas are sometimes referred to as climatic refugia or microrefugia (e.g., Noss 2001; Dobrowski 2011). While sometimes these are cooler or moister habitats now, the important point is that habitats in future microrefugia should be relatively insensitive to regional climate changes. For fish, we might expect spotty distributions in

the future related to local phenomena of cold air drainage, snow drifting, alluvial valley fill aquifers, aquifers in volcanic terranes, glaciers, or related phenomena. Some of these would be less enduring than others. The remaining patches created by these kinds of processes will tend to be small and isolated, thus at extreme risk to fire or other disturbances. They may also be at higher risk to small populations effects related to genetic diversity (e.g., Ellstrand and Elam 1993) (and see Neville and others in Advanced Topics of this volume). Forests may see similar local variations in topography and microclimate that help protect against fire, drought, and pests. Some of these variations in geology and topography provide the complex ecological landscapes we now see. Fires during extremely dry conditions promote homogenization of landscapes, with most of the area prone to severe fire regardless of local conditions (Dillon and others 2011). Fires during less extreme conditions promote heterogeneity because they are more responsive to local microclimate and fuel conditions. By consuming fuels and promoting heterogeneity in fuels, they also reduce the likelihood of later homogenizing events. Thus using wildfire during conditions that might normally allow suppression could be an important part of an adaptation strategy.

A Thought Experiment

If we consider the situation of heterogeneous, fragmented habitats in an idealized sort of way, we can examine how the principles outlined above might work together. Climate change can be viewed as creating disconnected networks of small refugia. If we identify the anticipated network of refugia and potential refugia, we can take steps early in the process to reinforce them with appropriate realignment (a term meaning restoration but recognizing that some aspects cannot be restored) of conditions disrupted through historical management and land use. We could also reintroduce individuals to some of the small suitable but currently unoccupied habitats, presumably that have been used in the past but may be currently unoccupied as a result of recent disturbance and migration obstacles or barriers. Recolonizing these habitats now would increase the redundancy available in the refugia network. Because these sites are not currently occupied, we could introduce individuals from various populations farther south or at lower elevation that may already have some adaptation toward future conditions in the unoccupied habitat. Manually maintaining (and controlling) gene flow among the habitats and providing restocking of extirpated patches as they occur is a related extension, wherein connectivity between habitats would be provided through management intervention where the option of simply opening or maintaining passage has been preempted. While more expensive, providing artificial connectivity between populations also offers control over spread of some introduced species and contagions.

This idealized approach draws on the idea of asynchrony in disturbance. Asynchronous disturbances mean that some places are disturbed while others are not, thus forming a foundation for resilience. If one views a snapshot of a dynamic asynchronous landscape, it will look heterogeneous. A snapshot of synchrony, on the other hand, looks like uniformity. Some tree species are well adapted to large conflagrations having seeds that are resilient to fire (e.g., serotinous cones on lodgepole pine or jack pine), producing large even-aged stands that themselves encouraged spread. Unfortunately, such an adaptation is ill-suited for a climate that more frequently yields severe fire weather conditions. Creative fragmentation of forest landscapes can help maintain current stands of such species as islands within patches of more frequently burning uneven aged stands. The trick is finding

the places that burn less frequently and encouraging reproduction of fire-sensitive species in those locations.

Futures with dotted refugia of climatically misfit forests and fishes are unlikely to evolve naturally, and hopefully the reader recalls that this is a thought experiment within an idealized example. Such networks require planning and intervention, and some cost to maintain. The level of effort will depend on how much area will be manipulated and the degree and nature of intervention involved. We lack the capacity to keep whole landscapes from transforming, however, by taking advantage of natural sources of heterogeneity, there is some capacity to build small reserves for specific genetic resources. Tradeoffs between values and costs are an important consideration.

Unless substantial changes are made in greenhouse gas production, all of these steps may ultimately fail in any given location. One should recognize that gradual step wise changes could help bring (and build) genetic variations that are better adapted to warmer climates to more suitable ranges in a slow and orderly fashion, without placing species currently in those locations at increased risk from longer distance introductions through large scale facilitated migration. The gradual nature of the process we describe is also more adaptable to take advantage of (or respond more quickly to) incorrect climatic projections or temporary reprieves created by low-frequency oceanic temperature cycles (e.g., such as described by Mann and others 1995a; Mann and others 1995b; Jain and Lall 2001). There are characteristics in this process that may also allow taking advantage of differential effects of climate change on native versus non-native species, if we have information about the relative pressures (Bradley and others 2009; Wenger and others 2011b).

This was an exercise in idealization, but the important idea is that the situation is not without hope; there may be substantial capacity to slow or delay the most severe consequences of climate change, e.g., species extirpation or extinction. It is also useful to remember the generalized principles: 1) taking advantage of natural heterogeneity, 2) restoring historical degradation, 3) connecting refugia, 4) building redundancy and representation of genetic variability, and 5) developing flexibility around disturbance dynamics.

Integration With Water Resources Management

Climate change is driving changes to human demands on wildland watersheds. Water is increasingly more valuable across most of the West, and there are increasing demands to store water on public lands and draw water from rivers. Some of the uses will be the traditional irrigation uses, but an increasing amount of water use may go to producing energy to offset greenhouse gas related sources. Again there will be the tug of war between positive and negative effects for aquatic biota.

Some of the more simple principles may apply. For example, many requests for infrastructure additions may not coincide with the most sensitive fish habitats (parallel to Rieman and others 2000). Raising the height of an existing dam a small amount can provide additional storage for comparatively little expense and alters little additional habitat. Another example is restoration of historical impacts to meadows to increase shallow aquifer storage and release during low flow periods (Loheide and Gorelick 2007). Keeping ideas about the distribution of impacts over the landscape in mind can be helpful in reducing additional impacts from water resource development.

An important issue for aquatic, riparian, and water managers is how climate change may alter precipitation amounts. A principle reason for increased requests for additional impoundment in the West is the changing snowpacks. Warmer

temperatures melt snowpacks sooner, and the storage that the snow provides to hold water until it is needed in the spring and summer will be less available (e.g., Barnett and others 2005). Unfortunately, storage in reservoirs or meadows also increases evapotranspiration. In places where precipitation is rising or remaining constant, this may pose little difficulty. However, where precipitation is declining, additional storage could exacerbate the problem. Most irrigation related reservoir systems in the West do not have multiple year storage, and in extreme cases, the reservoirs could be dry or very low during years with the greatest need. Even where multiple year carryover is available, water allocated to its limit could render storage facilities ineffective if not carefully managed (e.g., Barnett and Pierce 2008, 2009; Rajagopalan and others 2009).

Reservoir operation will become both more important and more difficult, and improved information and forecasts about incoming flows will become increasingly necessary. Reservoir operations have potential impact (or utility) for downstream fishes and riparian vegetation and for upstream migratory fishes that use the reservoirs as part of their life cycle. Reservoir management can also reduce the need for storage increases. Some dams release water from deep in the reservoir, and thus can provide cool water for tail stream fisheries. If these releases are too cold or too strong, however, the quality and utility of the habitat as refuge can be limited. Upstream migrating fishes can be affected by channel processes and abandoned channels in the drawdown areas in the upper reaches of reservoirs. They can also be affected by food and temperature relationships within the reservoir. Because of the multiple complex, interlaced, and sometimes competing demands for reservoir operation, improved information about inflows, particularly mid- to short-term forecasts, can be useful for optimizing storage near the end of the melt season, which could reduce the need for additional storage facilities. Unfortunately, if contemporary trends of increasing interannual variability in flow continue, designing operations to benefit aquatic and riparian biota may be challenging. It is likely that improved data on snowpack, soil moisture, and precipitation data could be used to substantially improve forecasts relevant to dam operations, however, and the increasing value of that information should be assessed. What may have seemed too expensive in the past may be more reasonable with increasing value of information.

Municipal water supplies may be more vulnerable in the future to wildfire. One challenge will be deciding between fuel management activities, with expected erosion and pollution from forest roads, versus risks from wildfires that are less easily controlled. Either choice carries risks and solutions with different costs. Alternative water sources can be useful because impacts of post fire erosion events to water quality can be relatively brief. Developing a multi-tiered water sourcing scheme could improve reliability of water supplies while allowing more flexibility in fire management. For example, the San Francisco water system, with a range of back-ups from large cisterns within the city, to reservoirs close to the city, to groundwater reserves further south and finally a major reservoir proximal to the city is an example provided for engineering reliability analysis where failure of the system post earthquake can be dire (Scawthorn and others 2005). Relying on forest managers, fire fighters, and federal emergency stabilization engineers to prevent water quality impairment to municipal water resources effectively transfers costs of providing reliability away from the water supply agencies (and their rate payers). Consequently, accounting of costs to different entities may be important in clarifying the relative benefits and costs of different strategies.

Cutting down forests to release more water has been a subject of interest for water and forest managers for as long as forests have been scientifically managed

(e.g., Bates and Henry 1928; Andréassian 2004). The opportunity to both reduce fire risk and improve water yield through forest thinning makes the idea all the more tempting. The actual performance of thinning in increasing water yields, however, is not well established (e.g., Troendle and King 1987; Troendle and others 2010). In high elevation forest environments, increases in snowpack due to reduced canopy have been documented (Wilm 1944; Wilm and Dunford 1948; Troendle 1983). However, success has not been unequivocally determined for thinning (e.g., Wilm and Dunford 1948; Troendle and King 1987; Ffolliott and others 1989), because it is thought that the residual stand can develop rapidly to utilize the available water. An additional concern is that drier forests and drier years produce less additional water from forest harvest (Troendle and King 1987; Brown and others 2005; Ford and others 2011). As a consequence, increased water yields are easier to make in places and years they are least useful. Also, almost all of the work has been done on experimental watersheds smaller than 10 km², where treatments are done on complete watersheds. The example from the Boise River fires may be germane to many circumstances (see textbox on Boise River hydrology in the hydrology section). With roughly 45% of the basin in moderate to severe fire, there was a statistically significant increase in water yield of 5%. The additional 50,000 acre-feet is substantial; however, the 200,000 acres of removed canopy represent more area than most forest managers can realistically hope to treat and maintain in a condition with low canopy within current budgets.

There are numerous complexities when water resource management is considered in decisions about fire, forests, and aquatic ecosystems. The intention here is not to provide a full summary, but rather to build awareness about it because water is a critical resource to forests, fish, AND people. The values to people will likely have heavy weight where there are conflicts with other values. There are some activities, particularly investment in information infrastructure (like weather and streamflow gaging stations) that can help better balance between the needs of these resources.

Decision Making for an Uncertain and Dynamic Future

The uncertainty of elements of climate change raises questions about what climate change means for decisions regarding future management of risks to forests and fish. Is it all too uncertain to make any plans? Does knowing how the climate is changing make a difference to decisions about fire and fish?

Optimizing forest management has long been discussed with respect to timber harvests as framed in the maximum sustainable yield and normal forest concepts (e.g., Hawley 1921). Formalized mathematical optimization under constraints of multiple uses and other resource values has been used in operational forest planning for several decades now (Johnson and others 1986). Within such optimization techniques, however, are assumptions about certainty of outcomes. Some have promoted robust decision making (Regan and others 2005) as a better model for planning under climate change because of the focus on reliability and prevention of failure by considering worst case analyses. However, even implementations of robust decision making approaches still require more information than land managers might have.

The classic minimum path length (or cost) problem provides an illustration. Figure 32a provides the common framing with a set time (cost) for each path. Optimization algorithms can be applied to this and significantly more complex networks to find the shortest path, which the reader can find for this simple figure. Figure 32b frames the problem in a way consistent with robust decision making,

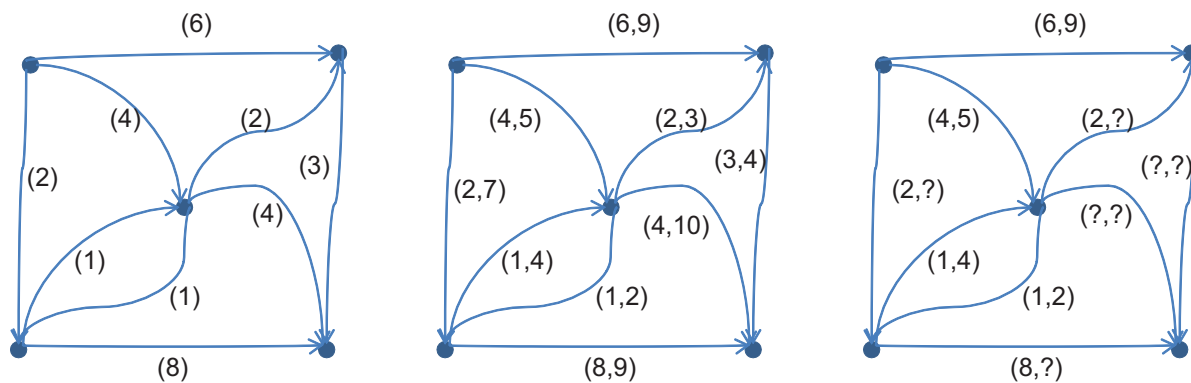


Figure 32: Conceptual path length diagrams for (a) classic optimization, (b) robust optimization, and (c) dynamic decision making. In (a) path lengths are given with uncertainty, in (b) and (c) path lengths are given across a range; however, some of the path characteristics are unknown in (c).

which essentially considers the “worst” case, i.e., finding the path with least cost if everything goes wrong. In this case, the range of potential times is bracketed. In this sense, robust decision making takes uncertainty into account in a way that focuses on preventing failure, and a less uncertain path becomes optimal, even though the original path could sometimes be faster. This is very similar in some ways to the first problem. The third Figure, 32c, shows something that might be a more common situation for decision makers, uncertainty that is poorly characterized or so uncertain as to be uninformative. A temptation might be to select the path with the most information (least uncertainty), but this is no more rational than selecting any other path and is essentially built from beliefs about what is behind the question marks. Neither classic optimization algorithms nor robust algorithms can solve this problem a priori, and decisions must be made as events unfold, a task described as dynamic decision making (Brehmer 1990).

Why might there be question marks on some paths? The cost may depend on previous paths taken, on the outcome from a previous uncertain path, or environmental events that took place in the time to cross one path. That is, answers to some of these questions may be unknowable until the first step is taken. A classic example of dynamic decision making provided by Brehmer (1990) is of forest fire management describing three criteria for dynamic decision making: 1) a series of decisions is needed, 2) the decisions are interdependent, and 3) the context or environment for the decisions changes autonomously as well (Edwards 1962). In short, they are the generally common situations associated with complex natural resource management decisions.

In the framing of dynamic decision making, information resources and the flow and processing of information become paramount concerns. The principal processes associated with improving performance are

- 1) Feedback (collecting information on environment and outcomes of decisions)
- 2) Feedforward (predictive modeling)
- 3) Cognition (collating information, assessing conditions, and making decisions)

Studies on dynamic decision making examine strategic combinations and characteristics of these three processes to improve learning and performance (Gonzalez 2005). Several emergent ideas seem generalizable across a number of fields of endeavor. Lag in feedback dramatically decreases learning and ultimate

performance (Brehmer 1990). Feedback-control alone does not work well, but is an important component in feedforward and cognitive support tools. Although it is well recognized that retrospective examination of outcomes as a stand-alone feedback strategy is inefficient and produces suboptimal results (Gonzalez 2005), it can become a preferred mode of control because it requires less cognitive effort and simpler task models. Decision makers may be unlikely to recognize that they could be doing better if they attain normative success with feedback-only controls, in part because a task model developed from a feedback-control strategy has difficulty describing improved performance from alternative strategies (Brehmer 1990). People are the decision makers and how they take in information, even the format of the information, can affect how rapidly they learn and improve and their ultimate performance (Atkins and others 2002). There are strong parallels between DDM principles and the principles involved in high performing organizations and increasing safety in complex and demanding tasks such as wildland fire fighting (Weick and Sutcliffe 2001; Black and others 2008).

Dynamic decision making theory describes how people learn to make reasoned decisions on complex problems with only partial data within constrained time periods. At the heart of it, it is about improving learning about complex situations. Why is learning important, and how is it related to climate change, forests, fish, and fire? Because adaptation is about learning. Evolution, a quintessential example of adaptation, is strictly about learning and encoding the information into genetic material (e.g., Williams 1966). Research helps us understand the kinds of tools people can best use to learn efficiently and ultimately make better decisions. Application of the ideas to climate change has captured improvement in agricultural performance at different time scales from different sources of information (Risbey and others 1999). Any visit to a modern wildfire incident command also shows an example of bringing in multiple sources of information to increase the reliability, efficiency, and degree of control offered by fire management operations. By expanding on concepts already applied in fire management and including information and relationships relevant to aquatic ecosystems, better decisions can be made for fires as well as in pre-fire planning (Dunham and others 2003; Rieman and others 2010).

Application to problems with fish, forests, fire, and climate change would suggest several key strategic components. The recent history of research on dynamics in natural systems has already taught us the value of diversification as a long-term structural/strategic approach in developing resiliency (e.g., Dunham and others 2003 for fishes). For shorter time scales where human intervention may be necessary, several information resources require some development. Better information about the conditions, distribution, and utilization of habitats for aquatic species is a key piece, including information about riparian canopies and stream characteristics. In essence, a better inventory of the aquatic/riparian habitat and species is needed. In comparison, we currently have substantial information about upland forest condition. More information about the temporal variability, at both interseasonal and interannual time scales, for precipitation, temperature, snowpacks, stream flow, and stream temperature will be critical in forecasting habitat changes. The lack of precipitation information at higher elevations (e.g., Mote and others 2005), where most of the projected habitat for the most sensitive species lies, makes it difficult to identify relationships that would help predict the most resilient habitats.

Anticipating changes over both long (decades) and short (weeks to months) time scales will be helpful. While the use of GCM projections in estimating future climates is an obvious, if fuzzy, tool, there may also be utility in data that can provide season-ahead forecasting to support fire management. Since we know that fire will be a critical mediator of climate associated impacts to forest and aquatic systems,

improving management and direction of fires, both intentional and unplanned, may ultimately be one of the most important tools for mitigating the potential impacts of climate change. Spring time information about fine spatial scale snowpack and soil moisture information can also help make planned fuel treatments more successful (Holden and Jolly 2011). Avoiding or repairing short term acute impacts may be valuable in managing a network of small refugia. This means that there is value to climate data for forest managers, in a way that may be parallel to farmers and others whose livelihood depends on weather (Risbey and others 1999; Hansen and others 2011).

Although there is a need to improve information resources about habitats and climate to improve control, much of the general approach already outlined for activities is aligned with general expectations of climate change. Current management interventions should continue to be the common sense application of habitat realignment to mend disruptions from historical management that was focused on generating wood products. Probably the key step to take at this time, however, is gathering information on habitats, stands, streams, and aquatic stocks to understand variable sensitivity to the effects of climate. Managers, specialists, and scientists should continue to anticipate new issues and new constraints (maybe new opportunities) that may require action to conserve special habitats. Perhaps the greatest management focus will be determining priorities. The rare components with greatest value will likely be afforded management attention and resources. If we use current management to encourage heterogeneity in landscapes, we can allow vegetation and other habitat changes to occur in some places while focusing management efforts on those locales and circumstances that are least sensitive to climate change. Within this context, however, we should avoid current refugia that are at limits of distributions and invest in the places that will serve as future refugia.

Finally there is a need for courageous leadership. Courage is needed because challenging goals must be pursued, with conviction. Learning is only accomplished in the context of a challenge. As Brehmer (1990) noted, if we feel we are doing acceptably well, there is no incentive to undertake the additional effort to collect and analyze data for improved decisions. If we only go so far as to envision an “acceptable” future, we may be fortunate enough to realize just that. Only if we envision a future where we have diverse species representation in many of the same general areas it is now, are we likely to obtain an outcome resembling that future. To accomplish such a task is a substantial challenge; to accomplish it within reasonable financial constraints will be an even greater challenge.

Next Steps

Fire is an important ecosystem process for forests, riparian areas, and streams. It is an agent of renewal and redistribution, and the biota that live with it have adapted a fine balance between the strain on individuals and local populations and the benefits that flow from renewal. Despite the use of the word “disturbance” to describe fire as an event, we need neither classify fire as “good” or “bad”; it need only be acknowledged as part of the diversity in nature. There are biota and ecosystems that depend on fire, and they do not often occur in places without fire. While the austere aesthetics of a recently burned forest are usually considered to be an acquired taste, there is a well noted appreciation for the beauty and simplicity of the landscapes, species, and ecosystems shaped by fire.

Biota have learned about fire through evolution. The signature of fire is encoded in their DNA and some of the resulting species, like lodgepole pine and ponderosa pine, have unique morphologic features that are classroom examples of fire adaptation. Many species have also learned to cope with the inconsistency and unpredictability of fire through development of a diversity of phenotypes adapting in slightly different ways. Some characteristics that are adaptive to fire may or may not have evolved in response to fire, but their representation in populations is reinforced by fire. For example, migratory life behaviors in fish have other major evolutionary benefits, like productivity and fecundity, but those benefits come with the cost of higher risk of predation or other incidents during migration. In so far as fire provides disbenefit to species that do not migrate, it selects for those species that do.

Much of that learning is now being put to the test. The way that fire operates in the landscape, as defined by relationships between frequency, size, and severity, is changing. Those relationships are fairly direct outcomes of weather, and trends of warming and drying in the spring and summer lead generally to an expectation for greater frequency, size, and severity of fires. Although global circulation models cannot model the effect well, other theoretical support and historical observations remind us that interannual variability is changing, and we are likely to see more extremes. The warmest driest summers are becoming warmer and drier even more so than average summers. The big and the severe fires will still be an outcome of extremes in weather, just as they are now. Shifts in the driest summers and the hottest days will be the most informative to predicting changes. These kinds of conditions conspire not just to produce the most flammable fuels, but also to make the atmosphere the most unstable and most prone to strong local winds.

Ecosystems are being affected directly by changes to climate as well, and some of the changes may reduce the effectiveness of natural adaptations to fire. Stream temperature changes, for example, may push thermally sensitive fishes into smaller, more isolated habitats that are then more vulnerable to post-fire debris flows or droughts. Trees and other plants more stressed by heat and water deficits are less able to fend off disease, pests, and invasive species, increasing their vulnerability to fire as well.

Fire will be one of several agents through which climate will change ecosystems, and losses in individuals and changes to microclimate associated with fire could be the final step in some local extirpations. For populations and ecosystems of conservation concern, the outcomes of fire will become more and more important. Many of the historical and recent challenges for land managers around fire have been prompted by the interaction of land management with wildfire and

its effects (Miller and Urban 2000; Bisson and others 2003; Hessburg and Agee 2003; Rieman and others 2010). The challenge for managers now is to blend their understanding of fire under natural variation (highlighting natural resistance and resilience) with the consequences of land management activities on responses, and an awareness of how climate change will further alter both the dynamic fire events and the response for forest and stream ecosystems.

Uncertainty about future conditions and events may seem like the greatest impediment to developing adaptation approaches. Although many consequences of a changing climate are fairly certain, like temperature increases and the subsequent effects on snowpack declines, earlier springs, and drier summers, precipitation related consequences are less certain, with uncertain sign (wetter versus drier) in some places. Precipitation is the largest term in the water balance, and uncertainties in precipitation have major implications with respect to potential adaptation, such as reservoir operation or expansion (Barnett and Pierce 2008, 2009; Rajagopalan and others 2009). These uncertainties also have substantial impact on future fire occurrence and size (Holden and others 2012). Substantial uncertainty surrounds how changes in extremes will transpire, as they are not well represented in GCM simulations. If we see more extreme events, that may affect ecosystems profoundly even if the average conditions change little. Besides these general uncertainties about future climates, there are uncertainties related to individual disturbance events. Although there is a growing acceptance that many kinds of forest disturbance are more likely, there remains the substantial uncertainty for a particular location about if and when, and then how big and how severe.

Preparing for climate change has two primary dimensions: preparing the landscape and preparing the managers. An important tactic for promoting landscape stability in the form of improved resilience and resistance is restoration of management-derived impacts. There are also tactics oriented toward anticipation of future conditions, such as thinning to adjust to future water balances. Even mild anticipatory actions yield some controversy, and stronger forms of facilitation (like introducing species in further north areas) are not widely endorsed in the literature.

Far, far less has been said about preparing managers or agencies for climate change. Ironically, people may be the most adaptable part of the system—at least in so far as managers recognize themselves as important and effective agents within ecosystems. Predictability is a key issue for adaptation. In so far as we can really see what climate change is going to do with some accuracy, we can start to prepare the landscape itself. If we are wrong, though, we are really just meddling or tinkering. For example, preparing for reduced water budgets does not prepare for greater variability, where one to a few wet years can undo efforts to reduce vegetation only to have thickly vegetated stands exposed to extreme dry conditions.

Given the large uncertainties in long-term climate projections at regional to sub-regional scales, an important strategic concept is responding intelligently and in a timely manner. This does not mean sitting around and waiting for something to happen! Effective responses result from accurate anticipation. Even if we cannot anticipate the details of climate and disturbance processes 40 years from now, there are some envelopes we can draw. As the distance to the time window for projection decreases, the envelope of possibilities grows narrower in some dimensions. Although uncertainty for particular events and locations will exist up to the moment lightning strikes or a debris flow happens, we have a general recognition that they will ultimately happen in many landscapes.

Natural resource management agencies have dealt with this nature of uncertainty for as long as most have existed. Fire suppression organizations provide several idealizations and lessons that can be applied to the context of land management

in a changing climate. Over the decades of professional wildland fire suppression development, there have been some general indicators of performance that are somewhat universally accepted: not losing important places to fire and not losing people to fire or accidents. To carry out their tasks better and more safely, they have developed weather and fire behavior forecasting skill at both seasonal and incident time scales (e.g., the National Fire Decision Support Center). They have also identified the critical points in their landscapes, analyzed how to defend them, and, where necessary or beneficial, actually prepared the places that put them or their charges at highest risk. While there is an impression that fire managers mostly do their work during fires, it is only when they are most visible. There is also a great deal of time preparing gear and equipment, preparing themselves, preparing forecast models, and preparing the critical locations in the landscape.

Among major disturbance agents (insects and disease, fire, climate change), fire stands out as being most potentially influenced by human decisions because we can, up to a point, choose when and where fires burn. We can choose, for example, to allow a fire to burn or we can attempt to suppress it. While successful suppression eliminates potential short term ecological impacts, it may defer the “external” costs of fire management until a wildfire burning under more extreme climatic conditions occurs that is difficult or impossible to suppress. Given general uncertainty about how a fire is likely to burn and the potential post-fire effects or “severity” of a fire, it is not surprising that managers choose to suppress many fires. In response to the short-term uncertainty fire managers face, fire, ecology, and hydrologic researchers are building new tools to rapidly provide managers with information about risks on relatively short time scales. For example, predictions of the potential for fire spread that account for topographic influences on snowpack and fuel moistures may be used to better manage wildfires in regions of complex terrain (Holden and Jolly 2011). Having spatial information on fuel moistures available to the fire spread models reduces the uncertainties, and therefore the costs of choosing not to suppress when conditions actually allow such a choice (see text box on modeling topoclimatic variation).

Natural resources management, as a profession, has developed an eye toward the long view. The general principles of cyclic dynamics in ecosystems have long been understood and have been used as an underpinning for developing scientific ecological management of forest ecosystems. Death and regrowth over time scales longer than human lifespans and over landscapes of millions of acres are not new ideas at all, but formed the conceptual basis for the conservation movement well over a century ago. It paralleled a realization that forests were not just a place to take wood from once and move on, but places for cultivation of sustainable goods and services. The profession has thoroughly embraced principles of robustness in management, exploring stability through resilience and resistance for various aspects of ecosystems and coupled social and economic systems (reference the Sustained-Yield Forest Management Act of 1944 from about the same time as Smokey Bear Act). It may sound anathema to suggest that we turn to management techniques associated with short time scale management, but the lessons of the “surprises” associated with climatic change over the last few decades has been that long term persistence and conservation of species may, at some point, consist of surviving from one decision or one crisis period to the next. The learning model of the wildland firefighting community may provide a good example of how to improve other land management approaches in a dynamic and partially unpredictable future.

Modeling Topoclimatic Variation to Improve Wildland Fire Management Decision Support

Topoclimate

Regional climate is mediated at finer scales by topography, particularly in areas of mountainous terrain like the western US. Three primary topographic features influence topoclimate. Surface air temperatures generally decrease with elevation as a function of the environmental lapse rate. Incident solar radiation (governed by aspect and shading by adjacent mountains) is generally greater on south vs. north facing slopes, increasing air temperatures, decreasing humidity and increasing the surface temperature of coarse woody debris and soils. The difference in radiation loading on south vs. north facing slopes influences the timing of snowmelt, which is translated into variation in fuel moistures and soil moisture deficits later in the season. Nighttime air temperatures are strongly influenced by topographic dissection. Depending on atmospheric conditions, Cold air drains downhill, accumulating in valley bottoms and basins. Nighttime air temperatures in valley bottoms can be 10 degrees celsius colder in valley bottoms than areas above the valley floor. Topographic position also effects wind speeds and valley bottoms tend to be sheltered compared to ridgetops. Wind has a strong influence on evapotranspiration, because it strips still boundary layer air from the surface of leaves, fuels and soil, accelerating drying.

Modeling topoclimatic influences on fuel moistures/water balance

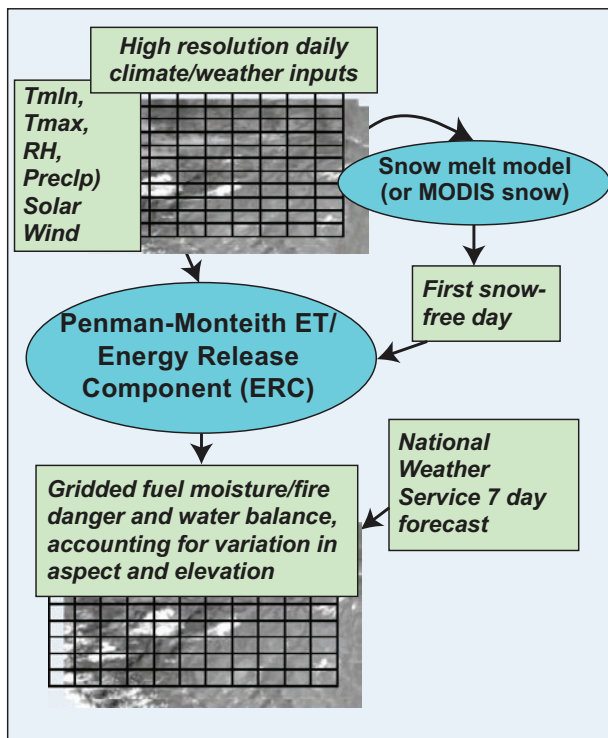


Figure T10-1. A simplified framework for incorporating snowmelt timing and topoclimatic variation into models of fuel moisture and fire danger indices.

Figure T10-2. Modeled snow-free date by aspect. Notice the early snow departure on west- and south-facing slopes.

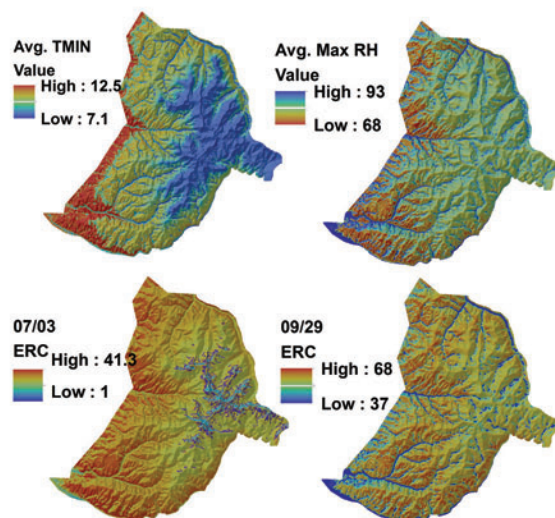
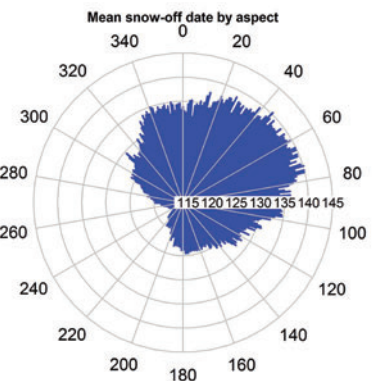


Figure T10-3. Average minimum temperature and maximum humidity across the 2009 fire season combined with delayed snowmelt timing lead to complex spatial patterns in fuel moisture and Energy Release Component (ERC).

Can We Use High Resolution Topographically-Informed Fire Danger Models To Aid in Management of Wildland Fires?

Topoclimatic variation has a huge influence on the evolution of fuel moistures during the fire season. Delayed snowmelt timing and cooler, wetter conditions on North slopes would limit fire spread. By taking advantage of topographic variation in fuel moistures and fire danger in areas of complex terrain, we could take advantage of wildfires early in the fire season and allowing more fires to burn while using terrain as a fire break. Importantly for aquatic ecosystems, we see that the combination of cold air drainage and high nocturnal relative humidity in valley bottoms leads to much lower fuel moistures at peak fire danger in valley bottoms. This suggests that some riparian areas could be buffered from severe fire behavior and effects by local topoclimatic variation.

The suggestion for the next steps in adapting to climate change, and particularly with reference to robust management of forests and streams under dynamic conditions, is for natural resource managers to do what they are already doing a little better. Most real world decisions have time pressure stemming from some degree of unexpectedness. A few examples:

- A large beetle kill occurs; should salvage harvest be done or not?
- A flood washes out a road; should it be restored, decommissioned, or abandoned as is?
- A local population of an endangered species disappears after a fire/debris flow, and the creek goes dry or a severe heat wave passes. Reintroduce, wait, abandon?
- Invasive brook or rainbow trout are found in a native population of cutthroat or bull trout. Take action or ignore?
- A new dam is proposed or new reservoir operating rules are proposed. Propose conditions?
- A new mine is proposed; is it consistent with species conservation plans?

Most readers should recognize these as day-to-day business for a natural resources management agency, and many may also recognize that these kinds of decisions will increase in number or complexity as climate change advances. At the same time, almost every one of these issues is handled as a separate emergency, with a separate wind-up time, separate meetings, and separate teams to handle it. An important, introspective question, is to reflect on how well prepared you are to handle such questions on any part of the domain for which you have responsibility to help with these questions. Could you really handle most of these well and within the time constraints imposed by natural processes or other agencies? If you are a line officer without detailed technical knowledge, do you know you can, or just believe? How can you check your answer?

Reconciling the long view with the short term is mostly about growing a commitment to having no more “surprises.” Events and decisions like the ones listed above are commonplace at the scale of the western United States, but may be so rare on a given agency field unit that experience and data are limited for those who must ultimately write the technical analysis documents. As a consequence, valuable time is lost to building expertise and collecting the new relevant data. While timber inventories are pretty common (and supported by the USFS Forest Inventory Analysis Program), information about riparian forests and aquatic species distributions is rare, piecemeal, and incomplete (see, for example, trout distribution data used by Wenger and others 2011a; Wenger and others 2011b). The ability to make rational (and defensible) decisions quickly about salvage opportunities in the absence of such information is severely impaired. By the time necessary information is collected, the opportunities may have expired, and decisions without information could lead to appeals with an equivalent result. We could complain in defense that the fire, or outbreak, or other event was a surprise ... but in a profession dedicated to the long view, that answer is not, ultimately, satisfying.

Not having “surprises” is not about perfect prognostication; it is about recognizing the potential for many different events or outcomes to happen, possibilities rather than expectations. For land and aquatic resource managers, it may mean needing to consider and evaluate sensitivities to events that may not happen during their tenure. It also means keeping time and room in their budgets to prepare for and handle increasingly common events. While the business of the fire-fighting is essentially contracted out from the perspective of a field management unit, other

events like the ones listed above and various other consequences are handled locally. Being nimble and prepared for decisions and action in response to a range of disturbances across districts and forests will consist largely of having current knowledge of the resources to be managed, so that when the time frame becomes tight, the best decisions can be made.

Geographic information is probably the most common information need identified in the preceding chapters. For example, for prioritizing forest and stream restoration efforts several maps are necessary:

- Mapping of fish habitat networks and distributions
- Mapping fuel status and potential fire severity based on fuel loading
- Mapping riparian vegetation cover and status
- Mapping of current aquatic habitat conditions
- Mapping road sediment and fragmentation issues
- Mapping streamflow patterns (e.g., flood timing, low flow magnitudes) and sensitivity
- Mapping stream temperature patterns and its climatic sensitivity
- Mapping the debris flow risks

The value of geographic information for decisions on fire and forest fuel management is well recognized and substantial outcomes for providing the information to managers were realized in the Landfire project (<http://www.landfire.gov>, and see Textbox on Mapping Future Fire Severity). The success of this kind of project relates to a strong need in a major program (fire and fuel management), persistent crises related to drought and fire occurrence, major advances in remote sensing science in the preceding decades, and a program dedicated to improving and collecting forest inventory (USFS Forest Inventory Analysis). Developing a parallel information base for aquatic and riparian management will benefit from an increasing capacity for remote sensing of stream and riparian characteristics (see, e.g., McKean and others 2008; McKean and others 2009 for geomorphic information; Isaak and others 2010 for stream shade information). Inexpensive data collection of stream temperature data may provide a wealth of spatial data across large areas, which will be particularly important with respect to climate change impacts (Isaak and others 2010; Rieman and Isaak 2010). Also, new models are emerging that simulate the genetic and habitat connectivity of fishes (Landguth and others 2011). Unfortunately water, riparian, and aquatic biota are not the primary missions of any land management agency, nor do they carry the same broad sense of urgency or crisis that fire does. The challenge will be in articulating the benefits of rapid access to both terrestrial and aquatic information for more punctual and better decisions.

Fine scale mapping of the heterogeneity of forest and aquatic habitats is a growing enterprise that will be useful in improving decisions as well. Data from GCM outputs, even some of the higher resolution downscaled projections now available, are inconsistent with the scale at which land managers ultimately carry out treatments. Tree harvest, thinning, and regeneration occurs at the scale of stands or hillslopes, yet most projections of tree species range shifts are made at 1 km scale or greater, ignoring the interactions between topography and climate and their influence on species occurrence. For example, valley bottoms, where streams occur, often experience remarkably different climates than areas upslope because of nocturnal cold air drainage, shading from adjacent terrain, and sheltering from wind. Such microclimates can be intensively sampled using inexpensive thermistors

Mapping the Probability That Fires Will Burn Severely

Assessing the potential ecological effects of large fires is crucial to effective management before, during, and after fires. Great tools exist for assessing post-fire effects, including the burned area reflectance classification (BARC) maps produced by the Burned Area Emergency Rehabilitation (BAER) using satellite imagery (Parsons and others 2010) and field methods. Managers also need tools for easily and quickly forecasting the potential severity of future fires. Now, managers can use the maps, models, and other tools developed by scientists with the FIRESEV project (<http://www.firelab.org/research-projects/fire-ecology/128-firesev>) funded by the Joint Fire Science Program to aid management decisions at multiple planning stages, including pre-fire fuels treatments and strategic management of active fire incidents.

We can map where future fires are most likely to burn severely based on statistical models of how climate, fuels, topography, and current season climate and fire weather influenced burn severity for fires that burned 1984-2006. Dillon and others (2011b) analyzed more than 7000 fires mapped by the Monitoring Trends in Burn Severity project (www.mtbs.gov) to develop statistical models based on past fires. They are currently extrapolating those models across each of 15 different ecoregions across the western United States to predict the likelihood of high severity fires in areas not yet burned. Dillon and others (2011b) are working with a very large random sample of pixels and an approach called Random Forests, thus expanding the methods Holden and others (2009) used to map the probability of severe fire occurrence based on topography and vegetation on the Gila National Forest. The resulting dynamic predictive maps will soon be available with 30-m spatial resolution for all lands across the western United States. Accuracy will be assessed based on field data and more recent fires. Though they will be imperfect, these spatially explicit predictions about the ecological effects of current and future fires can be incorporated into decision making. Managers will be able to access these maps directly off the Internet for immediate use. Or, they can use other predictive modeling tools, including the Wildland Fire Assessment Tool (WFAT, www.nifft.gov) with local data to predict the potential for severe fire. WFAT provides an interface between ArcGIS, FlamMap3 (<http://www.firemodels.org/index.php/national-systems/flammap>) algorithms and First Order Fire Effects Model (FOFEM, <http://www.firelab.org/science-applications/fire-fuel/111-fofem>) algorithms to produce predicted fire behavior and fire effects map layers. This tool is useful to managers in prioritizing fuel treatments on the basis of predicted fire behavior and effects and in assessing the effectiveness of fuel treatment proposals in a geospatial context. Dillon and others (2011b) expect that the map layers and related tools can also be incorporated into existing decision support frameworks such as the Wildland Fire Decision Support System (WFDSS, https://wfdss.usgs.gov/wfdss/WFDSS_Home.shtml) and the Rapid Assessment of Values at Risk (RAVAR, http://www.fs.fed.us/rm/wfdss_ravar/ravar/index.shtml).

In the process, scientists are learning why fires burn more severely in some places than in others. Cross-regional analyses of trends in burn severity are now possible, thanks to the MTBS data. Dillon and others (2011a) think that while area burned is greatly affected by climate (Littell and others 2009), local topography and fuels are relatively more important to the ecological effects of those fires, though this varies across vegetation types and ecoregions. Indeed, Dillon and others (2011a) found that topography strongly influenced burn severity. Including climate improved predictions some and more so for those years when fires were widespread and extensive areas burned. Climate and weather as “top down” influences on wildland fire (e.g., through fuel moisture, temperature, wind) that affect where and how fires burn at a broad scale (Dillon and others 2011a). In contrast, topography and fuels are “bottom up” controls that interact with climate and weather to alter fire behavior and effects locally. Topography is often a strong driver of general vegetation distribution, which in turn influences the distribution of fuels and patterns of severity.

Through the Fire Severity Mapping System, managers can access and create map products when and where they need them. By integrating LANDFIRE data layers, fire effects models, and new techniques for analyzing satellite-derived burn severity data into one comprehensive computer modeling package, managers can predict fire hazard and fire severity over a wide range of spatial scales. The system includes the digital maps as well as simulation models and other analysis tools to support decision making during wildfires, for targeting post-fire rehabilitation efforts, and for long-term planning. This complements existing tools widely used.

(< \$20 USD), to build models relating broader climate patterns to local temperature patterns (Holden and others 2011). Further information on terrestrial ecosystems at finer scales will develop from the use of Light Detection and Ranging Data (LiDAR) to map the three-dimensional spatial distribution of aboveground biomass (Hudak and others 2006) over heterogeneous topography. Distributed temperature measurements along fiber optic cables allow the identification of upwelling groundwater creating thermal refugia in streams (Selker and others 2006). Understanding the role of fine scale heterogeneity in the landscape process is an emerging field of work that will better inform how micro-habitats influence the potential presence of species in an area and how fire will spread over a landscape.

Doing a better, more efficient, job of natural resources management will also include increasing use of seasonal scale climate forecasts to program activities and task resources. Such forecasts are commonly made based on knowledge of the states and anticipated states of climate phenomena such as El Niño, the Pacific Decadal Oscillation, or the Atlantic Multidecadal Oscillation. Forecasting at seasonal scales (e.g., winter to summer) has been embraced by the fire community and has been applied to positioning for upcoming fire seasons (Wells 2007). Water resource managers and farmers are also common users of seasonal forecasts. The importance of seasonal-scale weather forecasting is not as well recognized in either terrestrial or aquatic ecosystem management. Because the information is already being generated and provided, however, the diligent resource manager may want to consider ways in which it could improve the bottom line. Some examples may include:

- A forecast for a wetter summer would imply relatively low risk of later extreme fire danger when deciding not to suppress a fire early in the summer.
- Summers with greater impending fire risk might be a time to prioritize having field crews available to assess riparian or aquatic conditions after fires.
- Those summers will likely also need some effort in preparing post-fire management plans.
- Wet summers are a good time to prepare to gather the information that will be used in the dry summers.
- Understanding the triggers that water managers will have for altering reservoir schedules and helping guide appropriate or provide monitoring of migratory fish to ensure their success.
- An impending flood-prone winter may trigger a response to survey culvert conditions and perform the kinds of temporary maintenance that can reduce flood impacts.
- In places with distributed small and isolated habitats, an impending fire-prone summer may trigger a response to survey current conditions and make any short term preparations (e.g., collect individuals for offsite protection).

Any skill developed in using seasonal scale forecasting to improve work flow, productivity, and conservation successes will translate into improved skill in using long-term climate projections as well.

The informational perspective has been a long-standing foundation of conservation biology. Aldo Leopold (1966) described keeping all the parts as the “first precaution of intelligent tinkering.” It has been more formally described as retaining genetic representation and diversity across the landscape. Retaining redundant

examples is particularly important in the context of disturbance. The principle operating behind this is that evolutionary adaptation is a learning process. The learning occurs as a range of “hypotheses” are tested against the environment they are in, leaving behind a range of comparably suitable outcomes. Learning in this way can only occur from among the phenotypes presented; thus species with more limited diversity have less capacity to learn in the short term. Random innovation through mutation is a possibility, but learning is much slower than if relative advantages exist from within existing genetic makeups. Essentially, species with a greater diversity of phenotypes are more likely to carry information already that allowed them to survive in analog climates or circumstances in the past.

The details may vary but one fundamental principle emerges from the discussion: The distinction between winners and losers in a changing climate will largely hinge on who has the best information.

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Advanced Topics on Fish Populations and Fire

Fire and Fish: a Synthesis of Observation and Experience

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Introduction

The effects of wildfire on aquatic systems and fishes occurring in them has been linked to the *direct* or immediate influence of the fire on water quality and the *indirect* or subsequent effects on watershed characteristics and processes that influence water quality and quantity, stream channels, and aquatic biota (Gresswell 1999). Early research linking fire and aquatic systems focused on effects to soil, erosion, and water yield and quality, with a relatively limited temporal and spatial context (see Gresswell 1999 for a review). Wildfire generally was perceived as a destructive force threatening aquatic resources (e.g., fish populations) and related values (Rieman and Clayton 1997; Kaufman 2004).

Subsequent efforts have integrated physical and ecological processes related to fire, and these concepts have been temporally categorized as short-term (i.e., <1 year), mid-term (1-10 yr), and long-term (10s to 100s of years) (e.g., Minshall and Brock 1991; Minshall and others 1998, 2001; Mihuc and Minshall 2005). Concomitantly, spatial context is critical because heterogeneity in fire severity, stream or watershed characteristics, and ecological communities constrains subsequent events and ecological responses. A broader perception includes the role of wildfire as a fundamental agent of disturbance potentially shaping heterogeneity, diversity, and productivity in aquatic ecosystems (Reeves and others 1995; Gresswell 1999; Bisson and others 2003).

The effects of fire and fire related management have been of particular importance to those interested in, or responsible for, management of native fishes. Fish and associated fisheries often hold particular social and economic importance. The sometimes dramatic short-term effects of fire and postfire disturbance on stream channels, water quality, and mortality of individual organisms can be readily apparent. As a result, attempts to influence fire and its effects on aquatic systems and fish populations before and, particularly, during and after the fire, have consumed considerable resources, time, and energy, and engendered substantial debate (e.g., Dunham and others 2003; Rieman and others 2003; NMFS 2007; Rhodes and Baker 2008; Rieman and others 2010).

It is our intent to summarize the known effects of fire-related processes in forested biomes of the western United States, briefly review existing knowledge regarding direct and indirect effects to fish, and consider the implications for fish populations. We integrate earlier summaries (e.g., Gresswell 1999; Dunham and others 2003; Rieman and others 2003) with more recent information and conclude with a final synthesis, including implications for conserving or restoring the resilience of fish populations to wildfire.

Fire Effects

Direct Effects

Wildfires consume flammable materials, and in the process, shade and cover provided by vegetation and woody debris is altered (Gresswell 1999). If a severe fire burns near or across a stream, water temperature can increase substantially (Hitt and others 2003). Dissolution of smoke, ash, and volatile compounds can alter pH and concentrations of trace metals, nutrients, and other chemical constituents in streams (Cushing and Olsen 1963; Minshall and others 1989; Spencer and Hauer 1991; Earl and Blinn 2003; Spencer and others 2003).

Direct physical effects can produce mortality of aquatic organisms including fishes, amphibians, invertebrates, and periphyton (Rieman and others 1997; Gresswell 1999). The causes of mortality have not been definitively identified, but potential mechanisms include rapid increases in temperature and accumulation of toxic chemicals (e.g., ammonium, trace metals, and cyanide) (Minshall and others 1989, 1997; Spencer and Hauer 1991; Barber and others 2003).

The observed spatial pattern of direct effects seems to depend on the extent and severity of fire and size of the watershed (Gresswell 1999). Minshall and Brock (1991), for example, suggested that fire would not directly influence temperature in third order or larger streams. Furthermore, high variability in postfire distribution of fishes in small streams appears to be associated with the heterogeneity of riparian fire severity (Rieman and others 1997). Mortality or displacement of fishes may be extensive, extending for several kilometers, but effects are often incomplete or patchy within and among stream reaches influenced by fire (Minshall and Brock 1991; Rinne and Neary 1996; Rieman and others 1997).

More recently, Howell (2006) found evidence of high to complete mortality or displacement of steelhead (*Oncorhynchus mykiss*), Chinook salmon (*O. tshawytscha*), and brook trout (*Salvelinus fontinalis*) in several moderate to severely burned reaches of three small (<2.6 m width) streams in eastern Oregon. Fish persisted at normal densities in stream reaches immediately downstream of those experiencing more severe fire. Based on observations from streams in northwestern Montana, Jakober (2001) reported that direct effects of several large fires were observable the following year. For example, the majority of fish in a drainage located in northwestern Montana were killed during a fire in 1996, and bull trout appeared to have been extirpated from headwater reaches of two small streams (Jakober and Dentino 2003). In contrast, fish distribution and density were relatively unaffected 1 year after fires that burned in Lolo National Forest during 2000 (Jakober 2002). In the southwestern United States, direct mortalities of fish appear to be less common than those following subsequent hydrologic events (Rinne and Jacoby 2005).

Direct effects may also result from fire suppression activities and the use of fire retardants (Gresswell 1999). Sodium ferrocyanide, a component of a commonly used fire retardant, is known to be toxic to aquatic organisms (Little and Calfee 2002; Pilliod and others 2003; Angeler and Moreno 2006), but the temporal and spatial extent of these effects have been poorly understood. In fact, Crouch and others (2006) recently found no evidence that retardant increased the levels of any chemical constituents above those from wildfire alone. The authors suggested that ferrocyanide was elevated through pyrogenic sources (Crouch and others 2006). Nonetheless, the Forest Service eliminated use of retardants with sodium ferrocyanide in 2007 because of the potential toxicity to aquatic organisms (USDA 2007). The replacement, known commercially as PhosChek, was less toxic in laboratory

studies (Little and Calfee 2003). The Forest Service concluded there was little risk for direct mortalities from PhosChek under prescribed operational use (Comas 2007; USDA 2007), but that conclusion remains controversial (NMFS 2007).

Indirect Effects

The indirect effects of wildfire on streams will be influenced and constrained by direct effects, the subsequent supply of materials (e.g., water, sediment, woody debris, nutrients, biological propagules) and energy (insolation), stochastic events (e.g., storms), and the interactions of these factors on physical and biological succession (Gresswell 1999; Dale and others 2005; Milner and others 2007). We provide only a brief overview focused on water, sediment, wood, stream temperature, and stream food webs as an example of the dynamic, and often complex, nature of the interactions influencing habitats for fishes.

Water and Sediment.—Changes in supplies of water and sediment are commonly observed after wildfire (Legleiter et al 2002). Short- or mid-term changes reflect alteration of vegetation and soils associated with severity and extent of the fire, geology and geomorphology, and the duration, intensity, and timing of postfire precipitation (Swanson 1981; Rinne and Neary 1996; Wondzell and King 2003). Peak flows and erosion can increase substantially, but the magnitude of response is related to factors pertaining to the watershed and weather (Robichaud and others 2005). For example, hydrophobic (water repellent) soils have been linked to floods and increased erosion in some cases, but effects are influenced by fire severity, soil texture, and vegetation type. Mass failures, debris flows, and flood catalyzed by fire have been important in the history of many watersheds and may be primary drivers in the long-term sediment supply for those systems (Reeves and others 1995; Benda and others 2003; May and Gresswell 2003; Meyer and Pierce 2003; Moody and Martin 2009). Dramatic increases in erosion that follow some fires tend to decline within 10 years as vegetation is reestablished (McNabb and others 1989; Burton 2005; Luce 2005; Robichaud and others 2009); however, the process may extend as root strength fails in fire-killed trees. The fire-hydrologic interaction has been characterized as an episodic pattern of disturbance and recovery that contributes to important variation of stream conditions in space and time (e.g., Reeves and others 1995; May and Gresswell 2003; Miller and others 2003).

Temperature.—Water temperatures commonly increase following fire, sometimes by several degrees, but increases are not universal (Gresswell 1999; Dunham and others 2007). Postfire increases in water temperature have been associated primarily with loss of forest and riparian shading (Gresswell 1999; Isaak and others 2010), but channel simplification, topographic shading, hyporheic flow, and the hydrologic changes accompanying fire can both accentuate and ameliorate these changes (Amaranthus and others 1989; Dunham and others 2007). Subsequent declines in stream temperature have been predicted with recovery of riparian canopy (Rieman and Clayton 1997), but recent work in the Boise River basin suggests that elevated temperatures can persist for one to two decades in some cases (Dunham and others 2007). Changes might even become permanent if fires initiate a transition to new vegetation communities associated with climate change (Isaak and others 2010).

Woody Debris.—Wildfire can play a key role in the recruitment of woody debris to streams (Reeves and others 1995; May and Gresswell 2003; Miller and others 2003). Wood is important because it controls channel morphology, sediment and water routing, and the heterogeneity of structure that may be important habitat for fishes and other organisms (Keller and Swanson 1979). In contrast to the

fire-related hydrological changes that can attenuate within 10 years, changes in the recruitment of woody debris can extend for decades (Reeves and others 1995; Gresswell 1999; Scheidt 2006). Postfire accumulation of wood is related to the prefire forest, severity of the fire, and processes associated with wood routing and storage in the channel (May and Gresswell 2003; Scheidt 2006). Woody debris may even decline following fire if a large proportion of the vegetation (including instream debris jams) is burned completely, or if remaining wood is transported out of the system during periods of elevated discharge (Swanson and Lienkaemper 1978). Mid-term accumulation may occur with toppling of fire-killed trees, undermining of riparian trees with increased flow and bank erosion, or the transport of wood from upslope via debris flows or avalanches (Swanson and Lienkaemper 1978; Scheidt 2006). Longer-term recruitment may depend on the rate that mature trees develop. The interaction of processes controlling the supply, accumulation, transport and storage of woody debris can lead to substantial variability within and among individual streams, or across time (e.g., May and Gresswell 1993; Young and others 2006). The capacity of a stream system to store and continually rework stored materials, however, may also lead to relatively stable or uniform conditions in some systems (Scheidt 2006).

Food Webs.—A handful of studies before 1989 provided a limited perspective on wildfire and responses in biological communities and food webs in streams (Gresswell 1999). In general, it is clear that fire and subsequent effects can disrupt invertebrate communities in segments of small streams, and that at longer time scales, effects depend on the severity and extent of the fire, subsequent hydrologic disturbance, and the characteristics and recovery of riparian vegetation. There is a substantial body of literature that initially defined the anticipated responses of stream communities and food webs to the extended effects of wildfire (e.g., Minshall and others 1989, 1997, 1998, 2001, 2004; Robinson and Minshall 1996; Minshall 2003; Earl and Blinn 2003; Mihuc and Minshall 2005). The generalized prediction emerging from this work can be summarized as an ecological succession in a temporal frame (Minshall and others 1989; Gresswell 1999; Dunham and others 2003; Minshall and others 2004).

The short-term effects of postfire disturbances are related to the biophysical template following the fire, and responses can be dramatic, varying from virtually undetectable to the complete loss of invertebrates and algae. Where disruption is great, invertebrate and algal communities are often recolonized quickly, but abundance and diversity may continue to vary (Earl and Blinn 2003; Minshall and others 2004).

In the mid-term (1-10 yr), diversity in aquatic communities tends to increase and productivity can be high with increased sunlight, stream temperatures, and nutrient flux. Where the riparian canopy is substantially reduced, foodwebs are expected to shift from largely allochthonous to autochthonous sources of carbon, with a concomitant response in trophic guilds of macroinvertebrates (or shifts feeding strategies by individual species; Mihuc 2004) and detrital respiration. Recent research has provided additional evidence that severe wildfire can stimulate primary production and a shift to primary consumers that support a greater biomass of predatory insects, potentially fish, and even consumers in linked riparian communities (Malison 2008; Malison and Baxter 2010).

In the longer term (10-300 yr), watershed processes are anticipated to interact with succession in terrestrial and riparian vegetation. Influx of wood generally leads to organic litter accumulation. Trophic pathways are anticipated to shift from autochthonous toward allochthonous carbon sources as riparian vegetation increases and the canopy closes (Minshall and others 1989). Recent research has

generally supported these predictions and extended the perspective (Minshall and others 2004; Robinson and others 2005), but changing climate may constrain future riparian communities leading to unanticipated outcomes.

The long-term effects of wildfire on macroinvertebrate communities and food webs remain speculative, largely because direct evidence from empirical studies of that temporal extent simply does not exist. The results clearly depend, however, on the myriad interactions between disturbance, terrestrial succession, and watershed process that directly and indirectly influence the legacy of materials and linkages within, and between terrestrial and aquatic systems (Dale and others 2005; Milner and others 2007, 2008; Pettit and Naiman 2007).

Fish Population Responses

Stream environments can change quickly during or following a fire (e.g., a single flood event) and may be catastrophic in ecological terms (including local extirpation of organisms in individual stream reaches). Concomitantly, indirect effects of wildfire often initiate or constrain processes, and responses can extend for decades and even centuries. Wildfire clearly plays an important role in supply of food and materials and the heterogeneity of channel conditions that contribute to the mosaic and productivity of habitats (Reeves and others 1995; May and Gresswell 2003; May and Lee 2004). Although direct and indirect effects of wildfire can induce fish mortality, the long-term consequences for fish populations and assemblages will ultimately depend on the legacies of material, biota, and the associated physical and ecological processes that shape them. Understanding these changes ultimately requires a population-level perspective.

The response of fish populations to the direct and indirect effects of wildfire has been a focus of considerable interest in the last two decades. Because fires are hard to predict, most of the early research was opportunistic. In some cases, wildfires burned in watersheds where pre-existing population data facilitated before-after comparisons (e.g., Novak and White 1990; Rinne and Neary 1996; Rieman and others 1997), but inference was also gained from comparisons among burned and unburned streams, or the temporal trajectory of populations associated with the immediate effects of fire (Minshall and Brock 1991; Rieman and others 1997). Initial research focused principally on description of changes in distribution and abundance of extant populations or segments of populations (Rieman and others 1997; Gresswell 1999), but subsequent syntheses considered a broader context of population and ecological processes (Rieman and Clayton 1997; Dunham and others 2003; Heck 2007).

Although historic information suggested that direct effects of fire could produce substantial fish mortality (Minshall and Brock 1991; Rinne and Neary 1996; Rieman and others 2007), we know of no examples of population extirpation associated with immediate effects of wildfires. In general, population level implications of wildfire appear to depend on longer-term processes.

Reductions in abundance, contraction in distribution, and even local extirpation have been reported with the indirect effects of large fires (Bozek and Young 1994; Rinne and Neary 1996; Gresswell 1999; Rieman and others 1997; Rinne and Carter 2008). Extreme effects, including local extirpations, have most often been observed in the southwestern United States (Rinne and Neary 1996). Negative consequences are related to diminished water quality associated with postfire ash flows, loss of habitat connectivity during periods of drought and intermittent flows, violent postfire flooding, and loss of food base and habitat (Rinne 2003, 2004; Rinne and Carter 2008). Fish populations in the Southwest may be especially

vulnerable because the spring and early summer fire season is followed by monsoon in late summer (Rinne 2004). When heavy rains follow severe fire, and heavy ash or “slurry flows” have been reported (Rinne and Neary 1996).

The potential for a severe disturbance is coupled with the fact that many of remaining native fish populations in the Southwest are limited to small streams isolated from any source of immigration (Brown and others 2001). In fact, many fishes in the region have been formally listed as threatened or endangered under ESA, and therefore, there is substantial concern that wildfires could negatively influence persistence. In some cases, rescue efforts have been initiated to capture surviving fish during, or immediately following, a fire and move them to a secure environment (e.g., hatchery) until habitat conditions in the burned watershed stabilized (Brooks 2006; A. Unthank, USDA Forest Service Regional Office, Albuquerque, NM personal communication).

Despite the potentially deleterious consequences, population collapses have not been ubiquitous in the aftermath of even very large fires. In some cases, fire had little apparent effect (Gresswell 1999; Riggers 2001; Jakober 2002). In others, abundance was seriously depressed, and reach-level extirpation occurred; however, populations rebounded relatively quickly (e.g., 1-6 years; Novak and White 1990; Rieman and others 1997; Jakober and Dentino 2003). The emerging context of disturbance ecology and metapopulation dynamics and documented population resurgence resulted in a hypothesis of recovery strongly mediated by the expression of migratory life histories and by dispersal from refugia not influenced by the fire and subsequent events (Rieman and Clayton 1997; Rieman and Dunham 2000; Dunham and others 2003).

Recent research has sustained this view. In an extensive study of 32 watersheds that burned in large fires on the Boise National Forest between 1986 and 1994, Burton (2005) found that even where reach or broader-scale extirpation of fishes had occurred, recolonization through dispersal was complete. Howell (2006) reported rapid (within 4 years) recovery of rainbow and steelhead populations through immigration from habitats not influenced by postfire disturbances. Debris flows actually provided new habitat for Chinook salmon by destroying a culvert and promoting access to new areas above the former barrier (Howell 2006). Similarly, fire-flood generated debris fans in the Middle Fork Salmon River created new habitat used by spawning Chinook salmon within months of the events (R. Thurow USDA Forest Service, Rocky Mountain Research Station, Boise Idaho, personal communication of unpublished data). Examples of rapid responses now extend beyond the North American continent, and recovery of native fishes within 24-36 months of extirpation following extensive postfire sediment flows has been documented in Australia (Lyon and O’Conner 2008).

In fact, evidence that fishes exhibit resilience to fire is unmatched by recovery from anthropogenic disturbance (Neville and others 2009). Dunham and others (2007) contrasted two levels of postfire disturbance (burn-and-debris-flow and burn-only) with unburned streams in the Boise River basin, but severity of disturbance did not have substantial influence on the distribution of either rainbow trout (*O. mykiss*) and tailed frogs (*Ascaphus truei*) within 10 years following fire. It appeared that even if effects from wildfire (e.g., elevated summer water temperature) continued for a decade or more, human activities that reduce the capacity of organisms to respond to disturbance may be a greater threat to persistence than wildfire (Dunham et al 2007). Similarly, Neville and others (2009) found no evidence of genetic bottlenecks associated with fire related disturbance in 55 streams affected by the Boise fires, but genetic diversity decreased in relation to human caused migration barriers (i.e., impassable road culverts).

The potential spread of nonnative species facilitated by fire-related disturbance is another important issue for some dwindling native fish populations (Dunham and others 2003). Because stream temperature can affect the relative distribution and interaction of native and nonnative species (e.g., McHugh and Budy 2005; Rieman and others 2006; Benjamin and others 2007; McMahan and others 2007), it has been suggested that warming of streams affected by wildfire may lead to expansion of nonnative species (e.g., brook trout or brown trout *Salmo trutta*) and concomitant contraction of native species, like bull trout *S. confluentus* and cutthroat trout (Dunham and others 2007; Isaak and others 2010). Results of a recent study in the Bitterroot River watershed where bull trout and westslope cutthroat trout co-occur with nonnative brown trout, brook trout, and rainbow trout suggested that native salmonids were resilient to fire (Sestrich and others 2011). Although nonnative trout invaded 4 of 17 stream sections within the burned area, the invasion rate was similar in sections that did not burn. Habitat conditions declined, but native trout populations actually exceeded pre-disturbance levels 3 years following the fire (Sestrich and others 2011).

To this point, the emphasis has been on dispersal and migration as key mechanisms behind population-level responses of fishes to fire, but research examining sublethal effects of habitat changes provides some evidence that phenotypic plasticity and adaptation to the effects of fire are mechanisms that may contribute to population resilience as well. In one recent study, habitat characteristics and growth of coastal cutthroat trout (*O. clarkii clarkii*) were evaluated in severely and moderately burned watersheds and an unburned control (Heck 2007). Stream temperature increased following the removal of riparian vegetation during the fire, and growth of fish in burned watersheds was greater than in the unburned control. Increased water temperatures and a longer growing season may have triggered a bottom-up response resulting in an increased growth of cutthroat trout, but longevity decreased (Heck 2007). Apparently, survival in these headwater streams was limited by the maximum size of the individual trout, and faster growing individuals reached the critical point at an earlier age. In a study of nine tributaries of the Boise River (i.e., three each of reference, burned, and burned with channel reorganization), there was a direct relationship between growth and maturation rate, and level of disturbance 10 years following fire (A. Rosenberger, University of Alaska, personal communication of unpublished data). Such adaptation could be critical to persistence of some isolated populations (Letcher and others 2007).

Summary, Implications, and Conclusions

Wildfire can have dramatic effects on streams and on populations of native fishes. Mortalities have been associated with the direct effects of severe fires, but these consequences have been most frequently observed in relatively small streams and over limited extent (1-2 km). Indirect effects linked to the physical and ecological process occurring after a fire can extend over 10s to even 100s of km of stream, however, and effects may last for decades, and potentially even centuries. Wildfire and associated disturbances have been, and likely will continue to be, a proximate cause of extirpation for small populations of native fishes. In general, however, the effects of wildfire are far less dramatic. In some cases, the effects of wildfires have been difficult to measure, and in others, populations that were initially depressed have rebounded dramatically, even increasing in abundance or extent relative to prefire conditions.

Gresswell (1999) suggested that over much longer time scales (decades to millennia), fish were well adapted to disturbances occurring at the frequency of wildfire. Recent evidence suggests that effects of disturbance in individual habitat patches are greatest on individuals and local populations that are least mobile, and early speculation to that effect (Warren and Liss 1980; Rieman and Clayton 1997; Gresswell 1999) has been supported by recent findings. Because habitat fragmentation is directly linked to the ability of fish to move among disparate portions of a stream network, this factor alone may be much more critical to the persistence of western salmonids than disturbance related to wildfire. There is a growing body of evidence that watershed characteristics including size, complexity (number and arrangement of tributaries), connectivity, and management history play a major role in the population structure and resilience at multiple levels of biological organization across disturbance-prone landscapes (Wofford and others 2005; Neville and others 2006; Guy and others 2008).

Clearly, context shapes the effects of wildfire on populations and communities of native fishes. Fish are most vulnerable to both direct and indirect effects of fire where populations are restricted to relatively small areas of habitat, and risk is greatest in isolated stream segments or small networks in steep, confined drainages where severe fires are likely to burn a large proportion of the headwaters and riparian corridor. Where populations are relatively large, have access to diverse, well-connected habitats and/or the capacity to adapt to changing environments, vulnerability is lessened; in many cases, the capacity and even the productivity of habitat can even be improved following wildfire (e.g. Malison and Baxter 2010).

The differences in context are evident among individual populations and watersheds across the landscape. Some populations within a basin may be highly vulnerable to any form of disturbance, but others are much less so (e.g., Guy and others 2008; Peterson and others 2008; Neville and others 2009). Populations of Chinook salmon, bull trout, and other species that migrate over large expanses of habitat in the Pacific Northwest and northern Rocky Mountains have fared well in response to the major fires of recent decades. Managers in these systems may tend to worry less about wildfire than the more chronic effects of watershed development. They may even see wildfire as beneficial for creating or expanding new habitats (e.g., Howell 2006). In contrast, managers concerned with remnant populations of native fishes isolated by invasive species and dewatered stream channels in the central Rocky Mountains and the Southwest may view the next wildfire as the catalyst for local extirpation, or even species level extinction (e.g., Brown and others 2001; Rinne 2004; Brooks 2006).

Large disturbances will likely continue in all of these regions. The watersheds of the Rocky Mountains and interior west have been shaped by disturbances linked to wildfire, large storms, and the flux, routing, and storage of water (in groundwater, snow and ice), sediment, and wood. On evolutionary time scales, large fires, mass erosion events, floods, and even local extinctions have likely been common (e.g., Bennett 1990; Reeves and others 1995; Meyer and Pierce 2003), and yet many species and populations of fishes have persisted and adapted (Gresswell 1999). Management might alter the frequency and magnitude of particular disturbance processes, but it likely cannot eliminate large and intense watershed events (e.g., Kirchner and others 2001; Istanbuloglu and others 2002). In fact, frequency and/or magnitude of some disturbances may even increase with changing climate (e.g., McKenzie and others 2004; Westerling and others 2006). Furthermore, the extent of thermal habitats for coldwater salmonids and the connectivity among refugia are expected to decline, further increasing the vulnerability of some populations

(Rieman and others 2007; Williams and others 2009; Haak and others 2010; Isaak and others, 2010).

So how can managers respond to the challenges posed by changing fire regimes and the conservation of native fishes and diverse aquatic ecosystems? In general, options can be categorized as management before, during, and after fire (Dale and others 2001; Dunham and others 2003), but a framework for monitoring and adaptation is critical in all cases (Dunham and others 2003). Management before fire includes maintaining or restoring the resistance or resilience to disturbance before the next disturbance occurs. Conditions contributing to resistance and resilience in populations emerge repeatedly in the discussion above and elsewhere (e.g., Gresswell 1999; Dunham and others 2003; Bisson and others 2003; Rieman and Isaak 2010). Conservation or restoration of relatively large networks of habitat and the physical and ecological processes that maintain them, the broad expression and adaptation of life histories, and the potential for connections within, and among populations are all critical. In essence, resilience will depend on a spatial and temporal structure and diversity in populations and habitats that can absorb or benefit from the effects of fire.

In some cases only remnant habitats exist and reconnection or expansion to support resilience and adaptive potential is not possible or desirable (e.g., the potential invasion of nonnative species). The only alternative may lie in attempts to change the character of disturbance. For example, by reducing the extent or grain of severe fire, even small populations may have some chance of persisting. Considerable discussion has focused on the management of fuels through prescribed fire or thinning as a tool for altering the severity of fire, if not its extent (Reinhardt and others 2008; Rieman and others 2010). Continued debate about the potential utility of such efforts is related to the tradeoffs between the potentially damaging effects of management relative to the effects of the fires (Rhodes and Baker 2008; Rieman and others 2010) that may occur with, or without that management (Littell and others 2009).

If the conditions supporting the resilience and adaptation of populations cannot be created before a large fire occurs, managers are left with limited options. They might attempt to suppress or mitigate effects during or after the fire. Although extraordinary measures could make the difference between extinction and persistence (e.g., Brown and others 2001), there is some question whether postfire watershed remediation can even influence the kinds of events that might actually threaten populations (Backer and others 2004). Regardless of their efficacy, these actions can be extraordinarily expensive and necessarily will be limited in the extent or number of populations, streams, or watersheds that might be considered. Concomitantly, there are other activities (e.g., salvage logging) that can be detrimental to particularly vulnerable populations in the postfire environment (Beschta et al 2004; Karr and others 2004; Reeves and others 2006).

Clearly, better information is needed to understand where and how the conditions in forests, watersheds, and native fish populations transition from those that may benefit from and are resilient to the effects of wildfire, to those that are vulnerable to those effects. At what point is a population too small or too isolated to persist? Can we actually influence the character of fire to alter that balance? How do we weigh the threats and benefits of aggressive management? Peterson and others (2008), Dare and others (2009), and Rieman and others (2010) offer some tools that may help refine this discussion and offer some approaches that can begin to answer these questions. Further collaboration between researchers and managers could extend this process. In the interim, the basic concepts of resilience outlined repeatedly throughout this review provide a critical foundation for the future.

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Genetic Variation Reveals Influence of Landscape Connectivity on Population Dynamics and Resiliency of Western Trout in Disturbance-Prone Habitats

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Introduction

Salmonid fishes have evolved and persisted in dynamic ecosystems (Waples and others 2008) where disturbance events vary in frequency, magnitude, timing, and duration (Gresswell 1999; Dale and others 2001), as well as the specific nature of associated effects (e.g., changes in thermal or flow regimes, geomorphology, or water chemistry; Reeves and others 1995; Benda and others 2004; Bisson and others 2005). In the western United States, one of the major drivers of disturbance in stream ecosystems is fire (Reeves and others 1995; Rieman and Clayton 1997; Gresswell 1999). Although there is a growing consensus that fish populations can ultimately benefit from the productive and heterogeneous habitats created by fire (Reeves and others 1995; Benda and others 2003; Minshall 2003; Rieman and others 2003), to persist they obviously have to withstand the immediate and shorter-term effects of fire, which can reduce or even extirpate local populations (Rieman and Clayton 1997; Brown and others 2001; Burton 2005; Sestrich 2005). Movement among interconnected stream habitats is thought to be an important strategy enabling persistence during and following fire, and there is mounting concern that the extensive isolation of salmonid populations in fragmented habitats is reducing their resiliency to fire (Gresswell 1999; Dunham and others 2003; Rieman and others 2003).

In spite of this concern, there are few direct observations of salmonid responses to fire. In fact, guidance is based largely on a broader understanding of the influences of landscape structure and disturbance in general on salmonid fishes (Dunham and others 2003; Rieman and others 2003), and there is considerable uncertainty about how best to manage for salmonid resilience to wildfire. Studies are limited by the difficult logistics of following fish responses in the face of unpredictable events such as wildfires. Therefore, BACI (Before-After-Control-Impact) study designs are nearly impossible, and replication is similarly challenging because fires are often low-frequency events. Furthermore, conventional ecological study approaches (e.g., studies of fish distribution, abundance, life histories, and movement) are logistically difficult to implement (but see Dunham and others 2007). Overall, a major challenge to understanding resilience of salmonid populations in fire-prone environments is related to moving beyond localized case studies to those with broader applicability in wildfire management (Dunham and others 2003).

Genetic data can be useful for overcoming many of the limitations inherent in ecological studies (Neville and others 2006a; Schwartz and others 2006). Here

we review several case studies of western trout where population genetic data have provided insight about fish responses to fragmentation and disturbance more generally, and specifically in relation to fire. Results of these studies confirm the importance of movement and landscape connectivity for ensuring fish persistence in fire-prone landscapes, and highlight the usefulness of genetic approaches for broad-scale evaluation and monitoring of population responses to fire and related management actions.

General Effects of Fragmentation on Trout Populations

We begin with a brief review of several mechanisms by which habitat fragmentation may reduce the likelihood of persistence for salmonid populations. General consequences of isolation include increased risk of extinction from random demographic and genetic processes due to small population size (McElhany and others 2000), reduced potential for re-colonization or repopulation following disturbance (“metapopulation dynamics”; Gilpin and Hanski 1991), and decreased habitat diversity and “internal dynamics,” which can affect population persistence (Pickett and Thompson 1978). Isolated populations are typically smaller, and so are likely to have greater temporal fluctuations or other stochastic effects that increase variation in individual reproductive success and lead to smaller effective population sizes (Waples 1990, 2002) and frequent population bottlenecks (Neville and others 2006a). The consequent loss of genetic diversity from stronger genetic drift and increased frequency of inbreeding in these small isolated populations lowers adaptive capacity (Bijlsma and Loeschke 2011) and can increase the risk of extinction (Frankham 2005). Additionally, in smaller, simplified habitat patches fish cannot move to use a full complement of alternate local habitats that might bolster recruitment, provide refugia, or facilitate metapopulation dynamics (Dunning and others 1992; Schlosser 1995; Ebersole and others 2001; Ebersole and others 2003). Small populations are thus more vulnerable to physical disturbance because of the increased likelihood that a single event will affect the entire population. Finally, isolation of western trout populations has greatly reduced the occurrence of a migratory life history (Young 1995; Fausch and others 2009). Where present, migratory individuals using alternate habitats can boost reproductive capacity (Jonsson and others 2001; Morita and others 2009) and potentially buffer populations from environmental variability and disturbances, such as fire (Dunham and others 1997; Rieman and others 1997b; Dunham and others 2003; Neville and others 2006a).

Genetic Characterization and Monitoring of Populations

Because fragmentation and disturbances such as fire are expected to affect the genetic characteristics of populations, genetic data can be used as an indirect measurement of these processes. In fact, for many questions (e.g., measurements of abundance or dispersal rates), genetic assessments requires less time and less expense than traditional methods such as mark-recapture and radio telemetry (Schwartz and others 2006). Additionally, genetic approaches can provide information about temporal dynamics or cryptic influences that may not be captured by traditional methods. In such cases, evaluations of population characteristics are often more accurate than traditional methods (Neville and others 2006a). For instance, population fluctuations or founder events following fires may not be reflected in a

current census if a population has recovered demographically (Dunham and others 2007), but these events may have had significant long-term effects on effective population size and can be captured using genetic approaches (e.g., they might be expected to show evidence of population bottlenecks, particularly if they are isolated). As another example, there is growing concern that invasion of nonnative trout and hybridization with native species may become more prevalent following fire-related disturbances (Dunham and others 2003). Detecting hybridization in the field can be difficult, but hybridization can be assessed with confidence using genetic techniques (Boecklen and Howard 1997). Genetic approaches should be particularly valuable for capturing impacts from fire or other disturbances on natural populations, then, because samples can be collected across large landscapes (Manel and others 2003), and a sample collected at one point in time can provide a retrospective view of influences that may have affected a population over decades (a “snapshot,” but one that provides information about past influences). After an initial assessment, genetic data can then be collected periodically to monitor changes in population characteristics through time (Schwartz and others 2006).

Applications of Genetic Data for Evaluating Effects of Fragmentation and Disturbance

Using genetic diversity as an indicator, several recent studies of trout populations demonstrate how movement and life history variability are related to population resilience in fragmented and disturbance-prone landscapes. Neville and others (2006b) evaluated genetic characteristics of Lahontan cutthroat trout *Oncorhynchus clarkii henshawi* residing in high-desert stream habitats vulnerable to different types of disturbances including fire. The study area encompassed one of the last interconnected and relatively pristine stream networks in the current range of this fish. Because some streams in the network were degraded and isolated by barriers, this system provided a rare example of diverse movement and life history variation in a complex habitat that could be contrasted with population characteristics in isolated habitats within the same catchment. The mainstem river, a hypothesized migratory corridor connecting several tributaries, was found to maintain high genetic diversity and contain a mixture of genotypes representing fish from different streams that likely moved into mainstem river habitats when not spawning in the tributaries. In contrast, samples from physically connected but high-elevation tributaries where fish were thought to express a ‘resident’ life history showed moderate levels of differentiation from other samples and lower genetic variability within samples, confirming a certain degree of demographic and geographic segregation from migratory forms (Northcote and Hartman 1988; Varley and Gresswell 1988; Northcote 1992). Fish from habitats isolated by culverts or natural barriers had the lowest effective population sizes and levels of genetic diversity, often exhibited severe genetic bottlenecks, and were highly differentiated from other populations (patterns observed in other trout and charr populations isolated above barriers, Taylor and others 2003; Yamamoto and others 2004; Wofford and others 2005; Morita and others 2009). In one case, a population above a small natural waterfall that was thought to have been extirpated by drought (Dunham 1996) was later recolonized. Therefore, fish from interconnected larger habitats in this study seemed to maintain spatially segregated life history forms and functioning metapopulation dynamics, both of which are likely to contribute to the overall viability of this network of populations in a harsh desert environment (see

Rieman and Dunham 2000). Populations in isolated habitats exhibited negative genetic effects of isolation, likely resulting from the small and demographically unstable nature of these populations. It is not surprising that many isolated populations across the historic range of the subspecies have disappeared in recent decades (Elliott and others 1997).

Other studies have also highlighted the importance of habitat connectivity and complexity in disturbance-prone environments. Disturbances such as landslides and debris flows are common throughout the range of coastal cutthroat trout *Oncorhynchus clarkii clarkii*. In western Oregon, however, these fish exist in several ecoregions differing significantly in landscape attributes such as stream gradients, topology (arrangement of tributary branching), and connectivity, all of which might influence the effect of disturbances on local populations (Guy and others 2008). For example, the interior Cascades ecoregion is characterized by steep streams with little branching structure, but the Coast Range ecoregion is lower gradient and has highly dendritic stream systems with few instream barriers to movement. Accordingly, Guy and others (2008) found patterns of genetic diversity to reflect these landscape characteristics in each ecoregion. The authors had hypothesized that Coast Range populations would show genetic resilience to disturbance, given that a disturbance in any single stream would affect only a portion of a dendritic system, where the capacity for dispersal from other parts of the network was high. As expected, they found the Coast Range populations exhibited greater genetic diversity than populations from the Cascades ecoregion, where landslides and debris flows in steep single-channel streams likely affected whole populations directly with little possibility for subsequent gene flow. Other patterns related to the physical distances separating habitats suggested that genetic structure in the Coast Range ecoregion were driven by within-watershed dispersal, whereas in the Cascades ecoregion genetic drift and disturbance-associated population bottlenecks seemed to be the dominant factor influencing genetic patterns. The study design could not differentiate between influences of disturbance regimes *versus* historical influences related to colonization of the two ecoregions; however, Guy and others (2008) suggest that the effect of disturbances may be greater in habitats with reduced connectivity and complexity where fish have little ability to respond to these events by movement (see also Gresswell 1999). These results should inspire further investigation into these connections.

Another recent genetic study of 55 stream populations of rainbow trout (*Oncorhynchus mykiss*) in the Boise and Payette river basins of Idaho suggested similar influences of isolation on population resilience in the face of disturbance (Neville and others 2009). This study was designed specifically to compare the effects of wildfire and related disturbance *and* habitat fragmentation, and incorporated the natural variability in fire history across these watersheds over the last several decades. The authors compared genetic diversity in samples without a history of recent wildfire to those with a history of stand-replacing wildfire, as well as those that had experienced both wildfire and a severe channel-reorganizing disturbance that may have extirpated or greatly reduced fish populations. Stream habitats also varied in size (catchment basin area) and isolation caused by human-constructed road culverts. Surprisingly, there was no evidence that fish from streams with different fire histories had different levels of genetic diversity; overall, fish in watersheds that had experienced wildfires and associated channel reorganizing events maintained similar levels of genetic variability as fish in streams with no known disturbance. Genetic diversity did increase, however, with increasing habitat size, and was lower in populations found above culvert barriers. Recent demographic assessment of a subset of these trout populations sheds detailed light

on the rapid recolonization (Rieman and others 1997a; Burton 2005; Dunham and others 2007) and flexible life history strategies that enable populations to recover quickly after fire in interconnected stream networks (Dunham and others, this volume). Furthermore, results of genetic analyses suggest that human influences such as barriers to dispersal may pose greater threats to populations of native trout than wildfire (Neville and others (2009).

In total, these recent studies yield substantial empirical evidence of a linkage between landscape structure and resilience as indicated by genetic diversity in salmonid fishes. Because genetic diversity can be used to characterize connectivity and population dynamics (Neville and others 2006a), these relationships underscore the importance of connectivity for both the persistence and long-term viability of salmonids (see McElhany and others 2000; Dunham and others, this volume). The genetic examples presented here support the growing consensus that salmonid fishes are able to withstand disturbances, including fire (Waples and others 2008), if given a habitat template for expression of various movement and life history strategies (Schlosser and Angermeier 1995; Hendry and Stearns 2004; Dunham and others this volume).

Management Implications

Given the link between stream network connectivity and population resilience, an obvious management strategy would be to reconnect trout habitats wherever physically possible to allow dispersal processes to occur naturally. But management decisions today are complicated by wide-spread invasions and hybridization with nonnative fishes which, it has been hypothesized, may be facilitated following fires (Dunham and others 2003). Although examples in this paper suggest that strategies promoting the isolation of populations above artificial barriers may not be successful, short-term isolation may be the only alternative for preserving the integrity of native populations in cases where invasion is certain (Fausch and others 2009).

This issue emphasizes the usefulness of genetic data for monitoring native trout populations in the context of fire management. Where isolation is deemed necessary, genetic techniques have potential for evaluating the probability of persistence of isolated populations over time (see Yamamoto and others 2004, for an example with white-spotted charr *Salvelinus leucomaenis*) by assessing the effective population size prior to and following isolation, and evaluating the effects of any observed perturbations, such as fire. Genetic data are a powerful tool for monitoring threats from hybridization where native populations are at risk of invasion by congenators (e.g., Neville and Dunham, in press), and can be helpful in prioritizing management resources by evaluating the genetic integrity (i.e., whether or not they have already been hybridized) of above-barrier populations before barrier removals are planned and executed.

Wherever possible, therefore, information concerning the genetic characteristics of a population should be integral to assessments of trade-offs associated with isolation or connectivity (Kruse and others 2001; Peterson and others 2008; Fausch and others 2009). Undoubtedly, consideration of these trade-offs will become even more complex as suitable habitat becomes further fragmented and degraded in association with climate change, and the need for improving resiliency through restoring and reconnecting habitats increases (Rahel and others 1996; Rieman and others 2007; Williams and others 2009). Long-term monitoring of the genetic 'health' and purity of populations (Dunham and others 1999) will be

highly valuable in fire-prone habitats, especially when paired with insight from other ecological methods evaluating movement (e.g., telemetry or stable isotope analysis, see examples in Balkenhol and Waits 2009; Sepulveda and others 2009). This type of research will continue to improve our understanding of the complex mechanisms by which salmonids have adapted to the effects of fire (e.g., Dunham and others, this volume). Furthermore, genetic research on habitat connectivity and complexity underscores the need to develop a broad-based forest management strategy that focuses on protecting remaining fish populations and habitat from invasion and further anthropogenic degradation, while restoring degraded habitat and connectivity among habitats.

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Fish Life Histories, Wildfire, and Resilience—A Case Study of Rainbow Trout in the Boise River, Idaho

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Introduction

In this short piece we address the question of how aquatic ecosystems and species can change in response to disturbances, such as those related to the influence of wildfire on stream ecosystems. Our focal species is rainbow trout (*Oncorhynchus mykiss*) in the Boise River, Idaho. Rainbow trout in this system have persisted in the face of widespread and often severe wildfires occurring since the 1990s (Rieman and others 1997; Burton 2005; Dunham and others 2007).

Wildfire can lead to a variety of changes in stream environments (Minshall 2003). In the Boise River, recent wildfires ranged from light to severe burns (Dunham and others 2007), leading to variable changes in riparian and hillslope vegetation, and in some cases initiation of major channel reorganizing events (Benda and others 2003; Miller and others 2003). These events often involved massive erosion of stream channels, sometimes throughout an entire tributary. In the latter case, local populations of rainbow trout were reduced to undetectable levels (Burton 2005). In others, a patchwork of wildfire appeared to eliminate rainbow trout from some reaches of stream, but not others (Rieman and others 1997). Following these wildfires, water temperatures remained elevated for several years, particularly for streams influenced by channel reorganization (Dunham and others 2007; Figure 1). In spite of these dramatic environmental changes, rainbow trout have remained widespread in streams within the Boise River (Dunham and others 2007). What characteristics of this species' biology have conferred resilience in the face of these massive disturbances? What constraints may limit resilience?

One key to species resilience may be the expression of diverse and flexible life histories (Bisson and others 2009; Healey 2009; Waples and others 2009; Greene and others 2010). Our work on rainbow trout suggests the species is flexible in terms of how individuals and populations respond to variable environments, and that responses can be constrained by human influences that alter natural variability. The two major constraints we focus on here are invasions of nonnative trout (Neville and Dunham, in press) and loss of connectivity caused by stream culverts that block fish passage (Neville and others 2009).

Our work in the Boise River focused on smaller streams because both physical (Benda and others 2003; Miller and others 2003; Wondzell and King 2003) and biotic responses to wildfire may be more evident (Dunham and others 2003; Gresswell 1999; Minshall 2003). We recognized three broad classes of streams (Dunham and others 2007): 1) those lacking a recent history of wildfire; 2) streams with a recent history of moderate to high severity wildfire; and 3) streams in watersheds with a recent history of moderate-high severity wildfire that were followed by massive channel reorganization from a debris flow or severe flood. These events



Figure 1. Immediate aftermath of the 2003 Hot Creek Fire in the Boise National Forest. Pictured stream is a fish-bearing tributary to the upper Middle Fork Boise River (photo credit: Joseph Benjamin).

(fires, channel reorganization) occurred from 8-10 years prior to when we sampled fish, allowing for at least 2-3 generations of local reproduction.

We examined patterns of population density among these three classes of streams (for methods see Rosenberger and Dunham 2005; Dunham and others 2007), and found that density of age 1+ fish differed among the three classes of streams (Kruskal-Wallis Analysis of Variance, $X^2=5.96$, $df=2$, $p = 0.02$). In streams without a recent history of wildfire densities of age 1+ fish were highest (0.22 fish/m²), those in burned streams were intermediate (0.17 fish/m²), and densities in reorganized streams were lowest (0.05 fish/m²).

The sizes of fish at different ages were also different among streams. For the youngest three age classes (0+, 1+, 2+), fish were smallest in unburned streams and largest in reorganized streams (A. Rosenberger and J. Dunham, unpublished). This paralleled the pattern of water temperature in these streams, with unburned streams being coldest and reorganized streams being warmer (Dunham and others 2007). Warmer water temperatures could lead to faster metabolism and growth, if sufficient food is available (Hughes and Grand 2000). Additionally, fish in warmer streams may experience a longer growing season, and thus more opportunity to gain a larger size. Even though summer temperatures in the warmest streams exceeded levels that can be stressful to individuals, availability of suitable temperatures in spring and fall can still lead to faster growth (e.g., Tattam 2007). Faster growth may also be attributed to lower population densities and less intraspecific competition (Grant and others 1998; Ward and others 2006), especially in reorganized streams.

In parallel with growth, age at maturity was youngest in the warmer reorganized streams (A. Rosenberger and J. Dunham, unpublished data). A few individuals in

reorganized streams began maturing as early as their first summer (age 0+), with the frequency of mature individuals increasing in the 1+ and 2+ age classes. By age 1+, we observed that 27% of the total number of fish sampled were mature, with males predominating among mature individuals (83% of total mature individuals). For the age 1+ cohort, larger males and females were much more likely to be mature. By age 2+ the length differences were no longer significant, perhaps reflecting the influences of size-selective mortality or emigration. In other words, fish that grew faster and matured at age 1+ may have reproduced and died, thus leaving behind only slower growing fish in the 2+ age class, or older fish may have emigrated. The pattern of growth and maturity we observed has been observed repeatedly in other salmonids grown in captivity (Wysujack and others 2008) and in the field (Olsson and others 2006; McMillan and others, in press).

We also compared genetic responses of rainbow trout to changes in habitat associated with wildfire in the Boise and adjacent Payette River basins (Neville and others 2009). We hypothesized that disturbances associated with wildfire and channel reorganization should lead to losses of genetic variation within local populations. Given that populations can potentially be extirpated by disturbances associated with wildfires (Burton 2005), or reduced in distribution (Rieman and others 1997) or abundance (see above), it is reasonable to expect that genetic drift may lead to losses of genetic diversity.

We surveyed genetic variability in samples of rainbow trout collected from 55 small streams representing gradients of disturbance that paralleled our fish population surveys (Dunham and others 2007). We found that variability was similar among streams representing unburned, burned, and reorganized classes, as described above. In contrast, in streams that were isolated by human-caused fish passage barriers (impassable road culverts), we found losses of genetic variability were more likely, as has been found consistently in many similar studies (Neville and others 2006). In some locations, we found unexpectedly high levels of hybridization with nonnative cutthroat trout (*O. clarkii*) and hatchery rainbow trout introduced historically for sport fisheries, although hybridization was not related to fire history (Neville and others 2009; see also Neville and Dunham, in press). Overall, genetic variability within populations was most strongly associated with habitat size, with greater diversity observed in larger habitats. These results highlighted the importance of habitat size and connectivity, both critical factors influencing the resilience of fish in dynamic environments (Rieman and Dunham 2000; Dunham and others 2003; Bisson and others 2009).

In summary, we observed responses of individuals and populations in association with influences of wildfire (e.g., increased water temperatures), and in association with human influences, such as fragmentation of habitat and introductions of nonnative trout. Fish in streams most dramatically impacted by wildfires grew faster, but matured earlier in life with some evidence for shorter overall lifespan resulting from early reproduction. In non-scientific terms, these fish appeared to adopt a “live fast – die young” strategy, but the genetic basis of these responses is unknown. More recent changes, such as loss of genetic variability and changes in allele frequencies due to hybridization were linked to human influences.

Implications for Species Resilience In the Face of Disturbance

In the case of responses of rainbow trout to wildfire and other human influences in the Boise River, we find that wildfire changes both ecosystems and species. We also found that human influences have fragmented habitats (e.g., movement

barriers at road crossings) and constrained opportunities for rainbow trout to express its full range of life histories (e.g., because of hybridization with nonnative trout). These constraints appear to operate in ecological time, but ultimately may also constrain the evolutionary potential of rainbow trout. For example, if isolated populations lose substantial amounts of genetic variability or if genomes become compromised by hybridization with nonnative cutthroat trout these fish may have less capacity for evolutionary responses to their environment. Maintaining opportunities for natural expression of diverse phenotypes may be critical to long-term productivity, persistence, and continued evolutionary resilience of rainbow trout (Bisson and others 2009; Healey 2009; Waples and others 2009; Greene and others 2010).

We found human influences within local habitats to be important, but human impacts may stem from influences or constraints operating much further downstream. For example, major dams in place in the Boise River and downstream in the Snake River have prevented migration of rainbow trout to the sea to express a “steelhead” life history. Steelhead trout were common in the Boise River and other nearby streams prior to their extirpation by these large dams (Busby and others 1996). In locations where migratory fish are available to recolonize empty habitats, populations can be highly resilient to disturbance. Examples include migrants within freshwater systems (e.g., Rieman and others 1997; Dunham and others 1997), but also migrants from marine ecosystems, such as salmon and steelhead (Bisson and others 2005; Howell and others 2006; Bisson and others 2009).

Even without a steelhead life history present, freshwater resident rainbow trout appear resilient in the face of wildfire, except in the smallest and most isolated habitats, as would be expected in theory and from the few empirical data available for salmonid fishes (Dunham and others 2003). Resilience is shown in the ability of rainbow trout to rapidly recolonize habitats following disturbance, variable responses to changing environments (e.g., growth, age at maturity), and maintenance of large enough numbers of adults to avoid genetic drift in the face of disturbance. Lower resilience is evident when other constraints imposed by humans (e.g., movement barriers, nonnative trout) limit the species’ ability to respond to disturbance (Bisson and others 2009; Healey 2009; Waples and others 2009; Greene and others 2010).

Finally, results of this work point to the importance of evaluating multiple responses of a species to wildfire. If, for example, we had relied only on ecological responses, we would have missed critical threats posed by intra and inter-specific hybridization. Furthermore, evaluating only distribution or abundance of a species provides only limited insights into possible mechanisms contributing to resilience. This has important implications for evaluating potential effects of wildfire or fire management alternatives on aquatic species. Whereas it is obvious that we need to understand more about processes that contribute to species responses to wildfire, studies of species responses typically address only the net results of such processes: presence or abundance. Because both of these responses can be influenced by multiple underlying processes, presence or abundance can offer only limited insight into the complex interplay of the underlying genetic, environmental, and demographic drivers. In conclusion, to better understand species responses in the face of changes in aquatic ecosystems caused by wildfire (or any other disturbance) we need to better understand the processes that constrain or contribute to resilience.

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Aquatic Species Invasions in the Context of Fire and Climate Change

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Introduction

This paper focuses on the nexus among native and nonnative fishes with respect to fire and climate change in the western United States. Although many taxa are involved, I emphasize native and nonnative salmonids because these are obligate coldwater species that might be expected to respond strongly to fire and because most research has been conducted on these fishes. Also, forested lands in federal ownership in the western United States represent the stronghold for many native coldwater taxa and are the subject of fire, fuels, and water management. I caution, however, that climate change, coupled with a growing human population and increasing demands for certain ecosystem services (e.g., water for domestic, agricultural, and industrial use) may alter the species of concern, their interaction, and the geographic focus of this issue.

Overview: Nonnative Aquatic Species

In waters of the western United States, invasions of nonnative aquatic species are both exceptionally frequent and frequently detrimental to native aquatic species. For example, in Oregon, Washington, and Idaho, 120 nonnative forms of fish, crustaceans, mollusks, and amphibians have become established, and nonnative species constitute about half of the taxonomic diversity of fishes in these three states (ISAB 2008; Sanderson and others 2009). Declines in native species have coincided with the establishment of these nonnative taxa. All inland forms of trout and charr in the western United States—the subspecies of cutthroat trout (*Oncorhynchus clarkii*), Gila trout (*O. gilae gilae*), Apache trout (*O. g. apache*), and bull trout (*Salvelinus confluentus*)—have been proposed for federal listing under the Endangered Species Act or are currently listed, and nonnative species invasions have been identified as a cause of decline in all petitions and recovery plans. Moreover, introductions of nonnative species are expected to continue because of the growing globalization in trade that intentionally or inadvertently includes many new species (Cox 2004). Vehicle-based aquatic recreation also facilitates their spread; nonnative mussels, aquatic plants, and pathogens are being distributed throughout the country amid debris on boats, boat trailers, and perhaps even on wading boots (Gates 2007; Idaho Invasive Species Council 2007; Benson and Raikow 2009). In the late 19th and 20th centuries, state and federal agencies introduced many nonnative fishes to promote fishing, but more recently illegal angler-assisted transport of sportfish or baitfish is the leading cause of the spread of nonnative fishes (Rahel 2004). Many of these species form the basis of economically valuable and socially prized fisheries, even in remote wilderness settings

(Dunham and others 2004). Consequently, the enhancement, monitoring, or control of nonnative aquatic species is often a management priority.

What Facilitates Invasions?

Although invasions of nonnative species are common, they are not necessarily inevitable (e.g., Fausch and others 2001). Invasion success is in part attributable to propagule pressure, because the number of introductions and number of individuals introduced is positively correlated with establishment (Lockwood and others 2005). A second characteristic leading to establishment is habitat matching, in which the receiving environment is suitable for an introduced species throughout its life history. For example, many high mountain lakes provide excellent conditions for the growth and maturation of introduced salmonids stocked for angling, but because they lack inlets or outlets with fluvial spawning habitat, these populations decline relatively rapidly unless they are refounded by stocking. Alternatively, where such spawning habitats are present, nonnative trout readily establish in many instances and these locations serve as springboards for invasions elsewhere in a watershed (Adams and others 2001). This points out another aspect of successful invasions: spread. Watershed connectivity permits nonnative fishes to move throughout a basin, sometimes with surprising speed. Brook trout (*Salvelinus fontinalis*) that were passed over a migration barrier in a small Wyoming stream were found up to 4 km upstream within a month (M. K. Young, unpublished data), and summer movements of 2-3 km by this species are common (Gowan and Fausch 1996). Iowa darters (*Etheostoma exile*), native to the Mississippi River basin, were first detected in the Yampa River in the Colorado River basin in 2003; by 2007, they were observed 229 km downstream in the Green River in Utah (Walford and Bestgen 2008).

Nevertheless, invasions of nonnative species sometimes stall or remain relatively contained even when abiotic conditions appear favorable for their advance. Adams and others (2002) noted that the upstream extent of brook trout in several Idaho watersheds had remained unchanged over two decades despite the absence of migration barriers. In some cases this has been ascribed to introduced populations requiring extended periods of time to overcome small initial populations, as well as a delayed increase in fitness as natural selection operates on the genome of the founding individuals or new arrivals contribute genomic diversity (Carroll and others 2007; Roman and Darling 2007). It has been posited that strong populations of native fishes may prevent or delay nonnative invasions, but presently there appears to be little empirical support for this hypothesis (Moyle and Light 1996).

Some Examples of Invasive Species and Their Effects

The responses of native aquatic species to nonnative invasions are mixed. In some locations, native fishes have persisted in sympatry with nonnative fishes for extended periods, and in others nonnatives rapidly replaced the indigenous fauna. These responses have sometimes involved the same native-nonnative species pairs. Because the effects of introductions are specific to the native and the introduced taxa, I focus this discussion on the effects of three nonnative trout—brook trout, brown trout (*Salmo trutta*), and rainbow trout (*O. mykiss*)—on cutthroat trout native to the inland western United States.

Brook Trout and Brown Trout

Brook trout have been linked to the loss of headwater populations of cutthroat trout in many portions of the inland West (Behnke 1992; Young 1995; Dunham and others 1999). Often, the mortality of young cutthroat trout exposed to brook trout is high enough to produce recruitment failure (Dunham and others 2002; Shepard and others 2002; McGrath and Lewis 2007). Peterson and others (2004) demonstrated that age-0 cutthroat trout survival was near zero in the presence of brook trout, and that the suppression of brook trout led to large increases in survival of age-0 and age-1 fish. Yet how brook trout replace cutthroat trout remains unknown. Both competition and predation have been suggested as mechanisms that may work in concert with the tendency for brook trout to mature at younger ages and have greater size-specific fecundity than do cutthroat trout (Kennedy and others 2003).

Brook trout, however, are not universally successful at displacing cutthroat trout from small streams. Whereas cutthroat trout populations east of the Continental Divide in Montana appear particularly vulnerable to displacement, similar populations to the west have persisted in sympatry for decades. This difference may be attributable to differences in winter climate (Shepard 2004; Fausch and others 2006). This geographic difference also coincides with the prevalence of large cutthroat trout with fluvial life histories that may provide a demographic buffer to the presence of brook trout. And at smaller scales, the distribution of habitats serving as demographic sources—such as warmer water or reaches in unconfined river valleys (Benjamin and others 2007)—and sinks could be controlling brook trout establishment and spread (Adams 1999; Castric and others 2001; Petty and others 2005).

The effects of brown trout on inland cutthroat trout are less studied, but appear to be similar to those of brook trout. Brown trout have been shown to be competitively superior to cutthroat trout in laboratory trials (Wang and White 1994) and field enclosures (McHugh and Budy 2005). In addition, de la Hoz Franco and Budy (2005) felt that the presence of brown trout truncated the downstream distribution of Bonneville cutthroat trout (*O. c. utah*) in the Logan River, and McHugh and Budy (2006) observed an array of subtle but significant changes in the growth and behavior of cutthroat trout in the presence of brown trout. The influence of brown trout on cutthroat trout appears to have a geographic signature; the most pronounced effects are on Rio Grande cutthroat trout (*O. c. virginalis*) in New Mexico (Paroz 2005) whereas there are many examples of long-term sympatry between more northerly subspecies of cutthroat trout and brown trout. An important element that may be facilitating coexistence at higher latitudes is that brown trout tend to be found in larger streams at lower elevations and do not occupy many tributaries used by cutthroat trout for spawning, reducing interactions between brown trout and juvenile cutthroat trout. This downstream-upstream partitioning of stream habitats by brown trout and other salmonids (Weigel and Sorensen 2001) and their absence from small, high-elevation streams in the central Rocky Mountains (Bozek and Hubert 1992; Rahel and Nibbelink 1999) is thought to be related to their poor recruitment at low water temperatures (Jensen and Johnsen 1999; McHugh and Budy 2005). Because brown trout eggs and fry are also vulnerable to high flows during the latter part of incubation through the early stages of emergence (Lobón-Cerviá and Mortensen 2005), the timing of peak discharge in these environments may also present an obstacle to more-upstream invasions.

Rainbow Trout

Rainbow trout from coastal streams in the western United States have been widely introduced into inland waters originally containing cutthroat trout (Behnke 1992). Despite that these species tend to spawn at different times—cutthroat trout usually spawn after peak flows and rainbow trout before them—and in different places—cutthroat trout more often use small tributaries (De Rito 2004; Muhlfeld 2008)—hybrids frequently appear where the species co-occur (Shepard and others 2005). These hybrids are fertile and are capable of backcrossing with genetically pure fish of either source and with other hybrids, leading to introgression of nonnative genes into the cutthroat trout population. The spread of nonnative genes appears to rely primarily on straying by hybrid cutthroat trout rather than pure nonnative trout (Hitt and others 2003; Rubidge and Taylor 2004), although feral populations of rainbow trout in river main stems or a reservoir have also been implicated (Weigel and others 2003; Rubidge and Taylor 2005). First-generation hybrids do not appear to be at a selective disadvantage relative to parental stocks, but the fitness of later-generation hybrids is greatly reduced (Muhlfeld 2008). Despite this, introgression appears to progress via occasional reproductive bonanzas from hybrids and from rainbow trout parental stock (Muhlfeld 2008), and eventually rainbow trout genes may pervade virtually all remaining individuals to produce a hybrid swarm within a particular area (Allendorf and others 2001).

Although the probability of hybridization tends to rise with the number of introductions (Colautti 2005), spatial patterns in the distribution of nonnative genes are complex. Weigel and others (2003) reported that proximity to a source of stocked fish was only weakly related to hybridization between cutthroat trout and rainbow trout. Instead, larger streams at lower elevations were more likely to contain hybridized populations regardless of stocking location. They argued that isolating mechanisms, such as different spawning and emergence times or habitat preferences between adults of each species, were preventing hybridization from developing in headwater populations of cutthroat trout (Henderson and others 2000; Ostberg and Rodriguez 2006). In contrast, Hitt and others (2003), Rubidge and Taylor (2005), and Muhlfeld (2008) detected the upstream progression of hybrids between rainbow trout and cutthroat trout over time, and concluded that hybrid swarms would eventually occupy most of these waters except those with migration barriers.

Do Fires Favor Nonnative Fishes?

The immediate and long-term effects of wildfire on aquatic ecosystems and fish populations are the subject of this volume (Rieman and others, this volume) and of previous syntheses (Gresswell 1999; Young and others 2003), thus are reviewed only briefly here. The thermal or water chemistry changes associated with severe wildfire have led to direct mortality of trout populations (Rinne 1996; Howell 2006), although many populations have endured such fires with few or no ill effects or showed only temporary declines (Rieman and Clayton 1997; Burton 2005; Sestrich 2005). More problematic may be post-fire floods, blackwater events, and debris torrents triggered by summer thunderstorms that have reduced or eliminated salmonid populations (Bozek and Young 1994; Brown and others 2001). Nevertheless, the majority of events are in small (<2.6 km²), steep (>20%) basins (Parrett and others 2003), and these would usually not threaten entire populations (Rieman and Clayton 1997; but see Brown and others 2001). The probability of such events often declines rapidly in subsequent years (Cannon 1999).

Secondary effects of fire may depress or boost populations. Channel stability often declines following fire because of the loss of riparian vegetation, increases in stream flow produced by reductions in evapotranspiration, and increases in sediment, but regrowth and resprouting of vegetation eventually attenuate off-channel contributions of sediment and water. Infall of fire-killed snags will spike in the first few decades following fire (Lyon 1984; Bragg 2000), which may lead to extremely complex channels with an array of complementary habitats. Growth rates of salmonids in burned streams have been observed to increase (Dunham and Rosenberger, this volume). Although this may in part be attributable to temporary changes in water chemistry, it is more likely that decreased shading increases primary productivity that leads to greater macroinvertebrate abundance and food availability (Wilzbach and others 2005). Because water temperatures are directly related to the amount of solar radiation reaching the water surface (Johnson 2004), post-fire temperatures tend to increase, sometimes by several degrees (Dunham and others 2007; Mahlum and others 2011). Warming may harm or bolster population growth rates depending on the pre-fire temperature regime (Dunham and others 1999; Harig and Fausch 2002; Schrank and others 2003). Streams previously too cold for consistent reproduction or rapid growth may support larger, more stable populations, whereas those that are already relatively warm may become seasonally uninhabitable. Both conditions will gradually revert to their original state as canopy cover and stream shading increase (Luce and others, this volume), given that climatic patterns remain relatively constant.

It is sometimes thought that habitat degradation and disturbance in fluvial systems favors nonnative species either by creating conditions more suitable for them or by depressing the abundance of native species (ISAB 2008). Consequently, it has been hypothesized that post-fire habitats with increased water temperature, light, sediment transport and deposition, and channel instability may favor nonnative salmonids (Dunham and others 2003). At present, there is little evidence to refute or support this hypothesis. Similar to native salmonids, nonnative brook, brown, and rainbow trout have occasionally been eliminated from headwater streams during or after severe fires (Rinne 1996; Burton 2005). Furthermore, connectivity with unaffected portions of a watershed has facilitated the rapid recolonization of post-fire habitats (Novak and White 1990; Howell 2006), as has been observed for native species (Rieman and Clayton 1997).

Evaluations of the post-fire responses of sympatric populations of native and nonnative salmonids are more informative, but few such studies have been done (but see Howell 2006). One example involved monitoring the response of several native and nonnative species of salmonids in small watersheds throughout the Bitterroot River basin following fires in 2000 (Sestrich 2005; Sestrich and others 2011). Declines in or the extirpation of all salmonid species were observed in some but not all of the watersheds that experienced high-severity fire. Where such declines were observed, westslope cutthroat trout (*O. c. lewisi*) and bull trout tended to recover rapidly and sometimes exceeded pre-fire abundances within 3 years. In contrast, brook trout were slower to respond and in some cases their abundance remains depressed (M. Jakober, Bitterroot National Forest, unpublished data). Brown trout, however, appeared in several monitoring reaches for the first time after the fires. Whether this was attributable to suitable post-fire environments or their ongoing population growth in river main stems (C. Clancy, Montana Fish, Wildlife and Parks, unpublished data) is uncertain.

In summary, there is little evidence that nonnative fishes show greater resistance to short-term habitat changes wrought by severe fire than do native species. Moreover, some native species may exhibit greater resilience to fire-related

disturbance, in part because this represents a portion of the disturbance regime with which they evolved (Lytle and Poff 2004; Waples and others 2008). Nonetheless, I caution against generalizing the results of the limited research on this issue because the severity of habitat alteration, the environmental context of affected streams (e.g., their position in the watershed and arrangement of habitats), and the aquatic community (e.g., the abundance, diversity, and life histories of native and nonnative species) will influence the outcome. Because post-fire habitat changes—particularly warmer water temperatures—may persist for decades or centuries, long-term shifts to nonnative species remain possible.

Climate Change, Fire, and Nonnative Fishes

Climate change is expected to alter global patterns in the distribution of flora and fauna (Wilson and others 2005). Based on recent modeling (IPCC 2007), climate in the western United States is projected to warm substantially before the end of this century. In montane environments, this warming is expected to cause more winter precipitation to fall as rain rather than snow (Knowles and others 2006), leading to smaller winter snowpacks (Mote and others 2005). In many streams in this region, earlier runoff and a longer low flow period are already evident (Stewart and others 2005), and greater flow intermittency in summer is anticipated (Boughton and others 2009). Besides discharge, the primary change in physical habitat is expected to be an increase in water temperature. As a consequence, populations of obligate coldwater species such as salmonids are predicted to retreat upstream to smaller waters to track suitable thermal environments (Rieman and others 2007; Wenger and others 2011a,b) and more temperature tolerant species will likely spread upstream as well (Rahel and Olden 2008). Initially, these may include nonnative coldwater species that prefer slightly warmer temperatures e.g., brown trout (McHugh and Budy 2005) and rainbow trout (Sloat and others 2002; Bear and others 2007), but additional warming may favor coolwater species such as smallmouth bass (*Micropterus dolomieu*; Sharma and Jackson 2008), which are already widely distributed and expanding in rivers in the inland West (LaVigne and others 2008; Walford and Bestgen 2008). Mobile life history forms of native species that migrate downstream will encounter a gauntlet of potentially hostile nonnative species as well increasing areas of thermally unsuitable habitat and larger numbers of barriers to movement; selection against such forms may strongly favor resident life history strategies. Thus, the collective consequences of these changes will be to reduce overall occupied habitat and exacerbate the current patterns of population isolation for native coldwater fishes (Neville and others, this volume).

Climate change is also expected to increase the frequency, severity, and extent of fires (Westerling and others 2006). Combined effects of climate change and fire would place fish populations at even greater risk of extirpation during or shortly after severe wildfire because of low or no connectivity, fewer refuges, and smaller occupied habitats (Isaak and others, this volume). Thereafter, the loss of shading from stand-replacing fire in riparian zones and from associated debris torrents may further increase water temperatures to the extent that some sites may become uninhabitable for coldwater species. Moreover, climate warming coupled with severe fire may cross an ecological threshold that results in state changes in some forested basins in the western United States. This might include a shift from forest types adapted to infrequent severe fire to those tolerant of more frequent fire (Keane and others 2008) or from forested areas to grasslands (Schoennagel and others 2008). Such wholesale changes in the terrestrial vegetation and disturbance dynamics would alter stream conditions, probably leading to additional upstream

incursions of nonnative species and the further diminution of habitats occupied by native coldwater fauna.

Climate change, fire, and nonnative species invasions are likely to redirect future management efforts. Whereas the present emphasis is on in situ habitat and population protection and restoration, in the future population salvage (Brooks 2006), assisted migration (cf. Millar and others 2007), anthropogenic refounding, thermal mitigation (e.g., by increasing and retaining shading), and refuge designation (e.g., finding those areas most likely to provide suitable habitats for native species despite climate change) may assume importance as management tools in pre- and post-fire environments. The changing management of other resources may also provide some unforeseen opportunities for aquatic species management. The predicted decline in surface water available for human use under many climate change scenarios has led to proposals to build additional high-elevation reservoirs to store water for late summer release (Bates and others 2008). Such waters could become de facto refuges of native species that tolerate or require lentic habitats, and reservoir releases could increase (and sometimes cool) streamflow adequately to maintain coldwater species downstream. This should not be regarded as a panacea for native fish; reservoirs are often the target for nonnative species introductions that then spread elsewhere (Havel and others 2005).

There is substantial uncertainty associated with the rate of change and locations most likely to experience the greatest effects from climate change, in part because of the low predictability of whether, when, and how much major drivers of global climate, such as the North Atlantic circulation and El Niño-Southern Oscillation, will contribute (Kriegler and others 2009). Nevertheless, all trends and forecasts suggest a substantially altered and warmer climate in the western United States in this century. Retaining ecosystem services and providing habitats suitable for the persistence and evolution of the native aquatic fauna (Dunham and Rosenberger, this volume) under the constraints imposed by a changing climate, altered disturbance regime, and array of exotic newcomers represents a critical challenge in the not-so-distant future.

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