Climate Change, Forests, Fire, Water, and Fish: A Synthesis for Land and Fire Managers

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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 *et al.*, 2005; Hamlet and Lettenmaier, 2007a; Luce and Holden, 2009; Isaak *et al.*, 2010). Some of the changes have been subtle, others more noticeable, and they are expected to shift distributions of fishes (Rieman *et al.*, 2007; Wenger *et al.*, 2011a; Wenger *et al.*, 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 *et al.*, 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 *et al.*, 2004; Breshears *et al.*, 2005; Westerling *et al.*, 2006; Morgan *et al.*, 2008; Westerling *et al.*, 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 U.S. 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 *et al.*, 2003; Shakesby and Doerr, 2006; Hessburg *et al.*, 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 *et al.*, 2009; Littell *et al.*, 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 *et al.*, 2001; Jentsch *et al.*, 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 which 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.

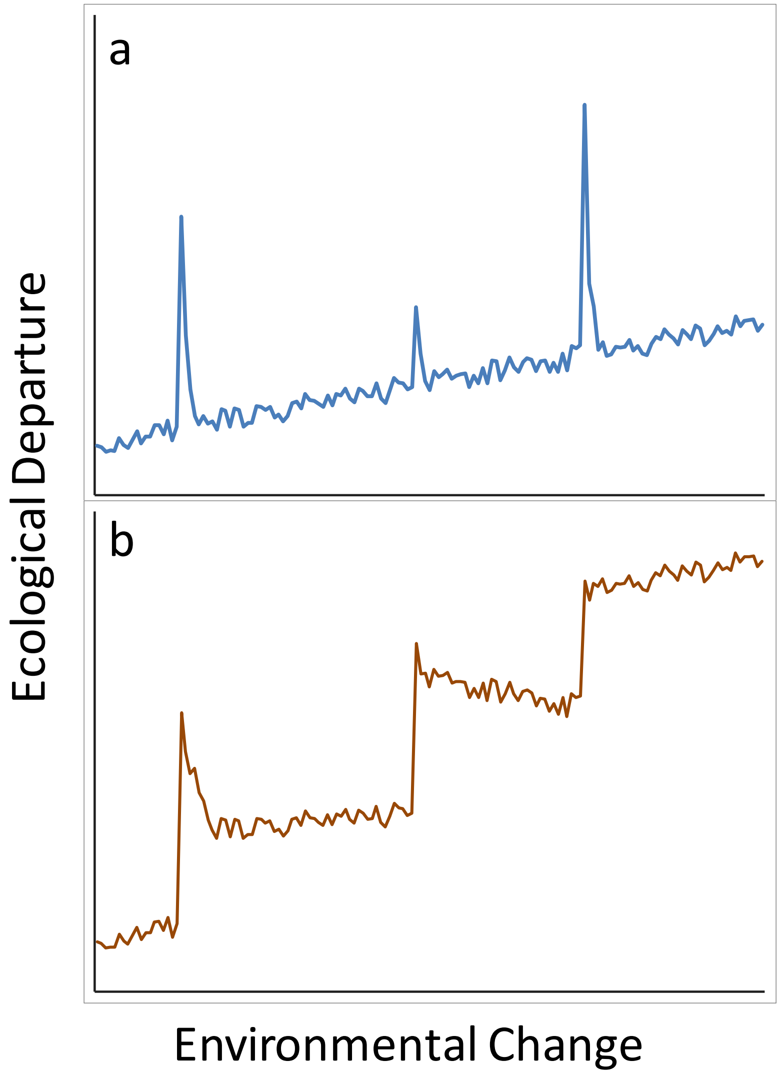


Figure 1: Conceptual roles for disturbance in a changing climate. Disturbance could continue to operate much as it always has, with unique disturbance/recovery patterns, or it could become the catalyst that allows ecosystems to shift in response to climate.

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 *et al.*, 2007 and see 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, with certainty, 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 *et al.*, 2000; Rieman *et al.*, 2010).

Earlier syntheses of fire effects on fish and streams (Bisson *et al.*, 2003; Rieman *et al.*, 2003a and papers therein) provided new ideas and new 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, suggesting 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. A strong cautionary note raised by many scientists and managers was that where connectivity and fragmentation of forests and streams was changed through historical land management, natural cycling is 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 *et al.*, 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, there was an opportunity to identify a fairly simple dichotomy between places where wildfire could operate freely without dramatically changing the natural dynamics of an environment and those places where some restoration of forests or streams would be necessary before that was true. A concept called "historical range of variability" (e.g. Keane *et al.*, 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 U.S. 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 *et al.*, 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 *et al.*, 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 *et al.*, 2003; Jentsch, 2007). The nature of interventions if any should draw from a deep understanding of which characteristics of the dynamics structure ecosystems of interest 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 *et al.*, 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 *et al.*, 2003; Turner *et al.*, 2003; Füssel and Klein, 2006; Millar *et al.*, 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 *et al.*, 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 *et al.*, 2010). Philosophical discussions of the goals of management intervention without the context of a specific physical landscape can be interesting, but a long literature would suggest that they might be unproductive absent a rooting in a real landscape (e.g. Cissel *et al.*, 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 *et al.*, 2010). Some details will be about how particular goals are achieved, probably reflecting a general bias towards 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 *et al.*, 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 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

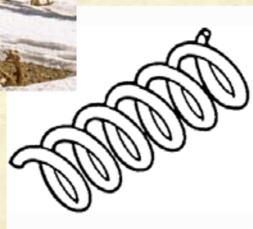
Resilience is one of the most commonly discussed ecological concepts with respect to fire and climate change. Based on such broad usage, one might suppose that it is a concretely defined term. There are, however, nuances that are sometimes unclear to new readers, and different connotations may appear 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 *et al.*, 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 US is explained, in part, by climate (Westerling *et al.*, 2006; Holden *et al.*, 2011b). The National Wildfire Coordinating Group (2009) has predicted that the annual area burned across the US 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 al., 2006) and increased tree mortality from bark beetles and drought (Breshears *et al.*, 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 *et al.*, 2007; Archer and Caldeira, 2008; Fu *et al.*, 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 *et al.*, 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).

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

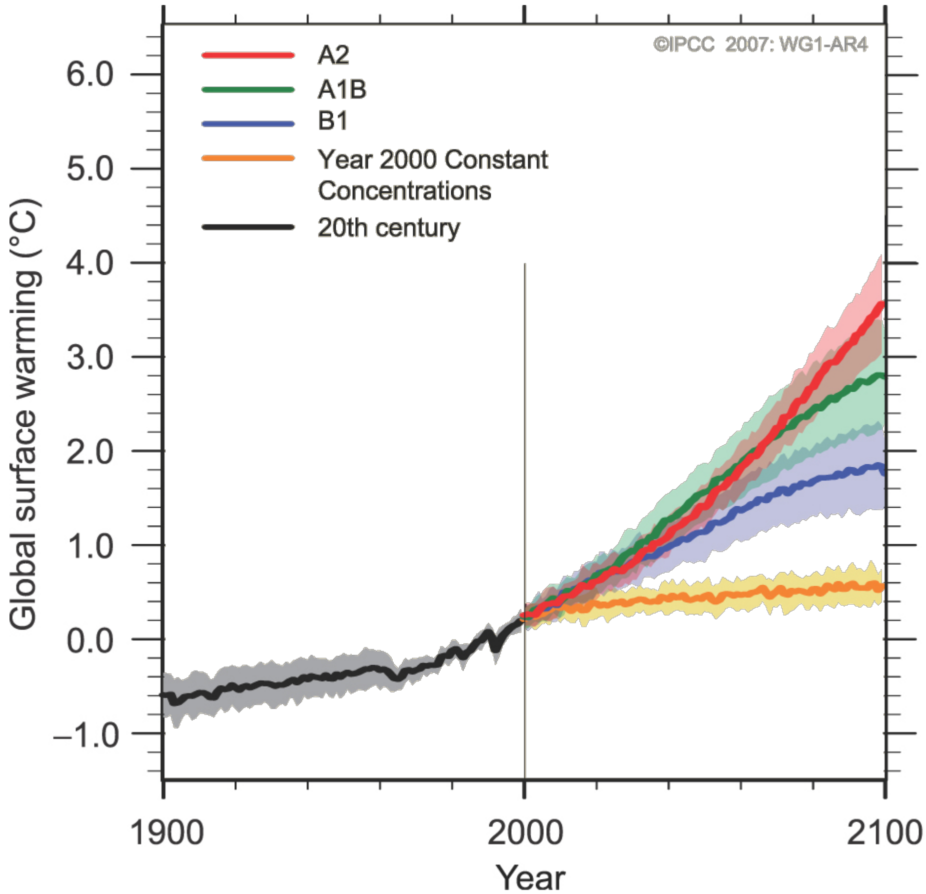


Figure 2: Global temperature trends showing the 20th century rise in temperature and projections for continued global average temperature increases depending on alternative carbon dioxide emission scenarios for the future (Solomon *et al.*, 2007)

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 *et al.*, 2007, Chapter 11).

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 a 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 CO_2 in the atmosphere, and it has been increasing over time. Although levels of CO_2 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).

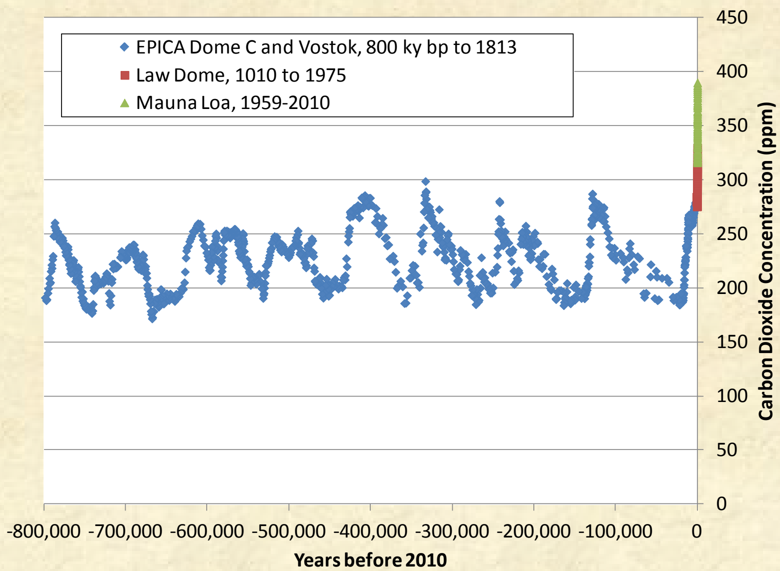


Figure T2-1: Composite of CO_2 data from 800 ky before present to 2010. Green: Mauna Loa data (Tans and Keeling, 2011), Rust: Law Dome ice core data (Etheridge *et al.*, 1996), Blue: composited Vostok and Dome C ice core data (Petit *et al.*, 1999; Monnin *et al.*, 2001; Siegenthaler *et al.*, 2005; Lüthi *et al.*, 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 CO_2 as of 2005 was about 1.5 W/m^2 (Solomon *et al.*, 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).

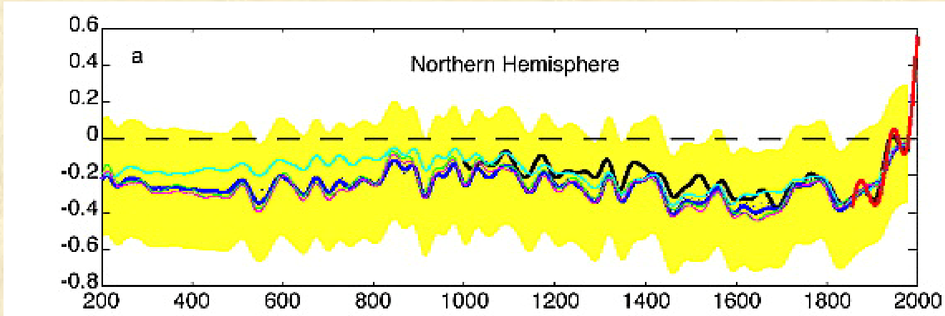


Figure T2-2: Northern hemisphere temperature over the last two millennia from several proxy reconstructions (different colors) with 95% confidence intervals (from Mann and Jones, 2003)

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 CO_2 '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 *et al.*, 1997; Braganza *et al.*, 2004).

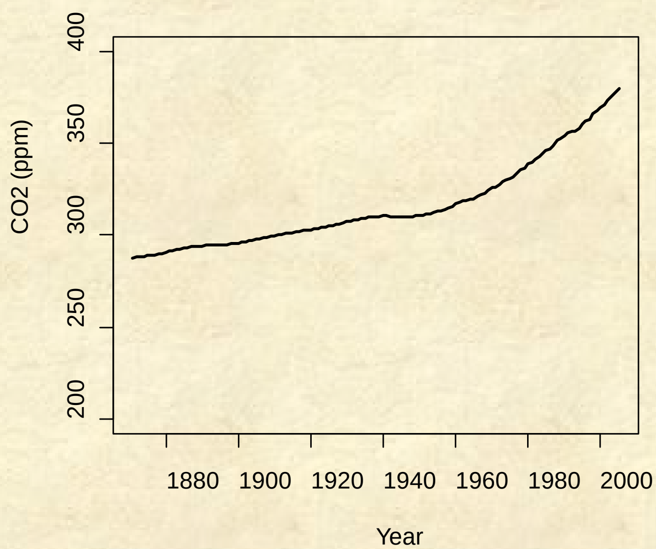


Figure T2-3: The meridional (equator to pole) temperature gradient over time (raw and smoothed) plotted with CO_2 concentrations. CO_2 data from Law Dome (Etheridge *et al.*, 1996), MTG data from (Karamperidou *et al.*, 2010). Note that the MTG axis is negative, and that the increasing trend represents 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 *et al.*, 1996; Guo *et al.*, 2008).

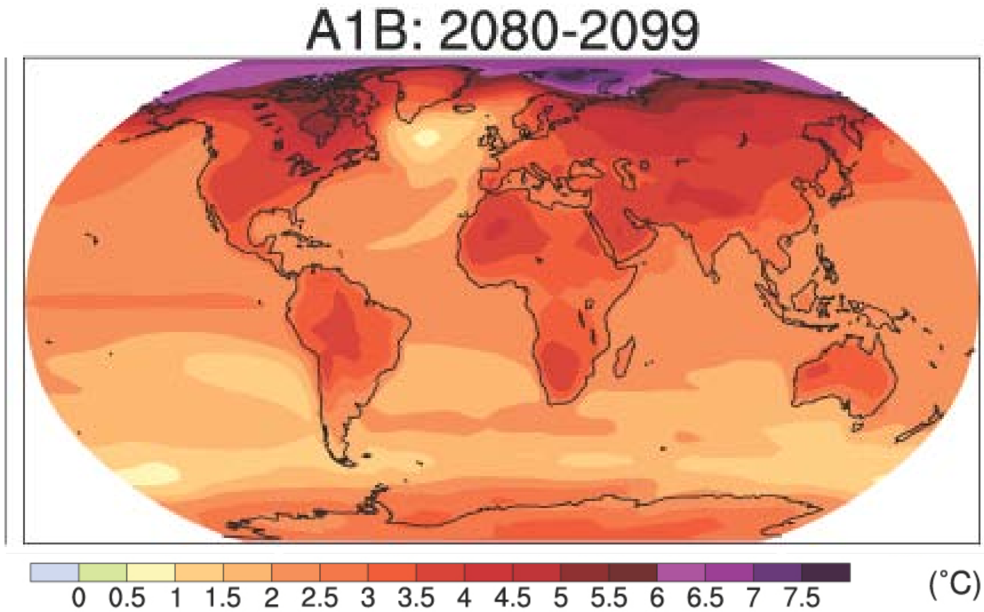


Figure 3: Multi-model mean surface temperature warming relative to 1980-1999 mean temperature (from Figure 10.8 in Solomon *et al.*, 2007)

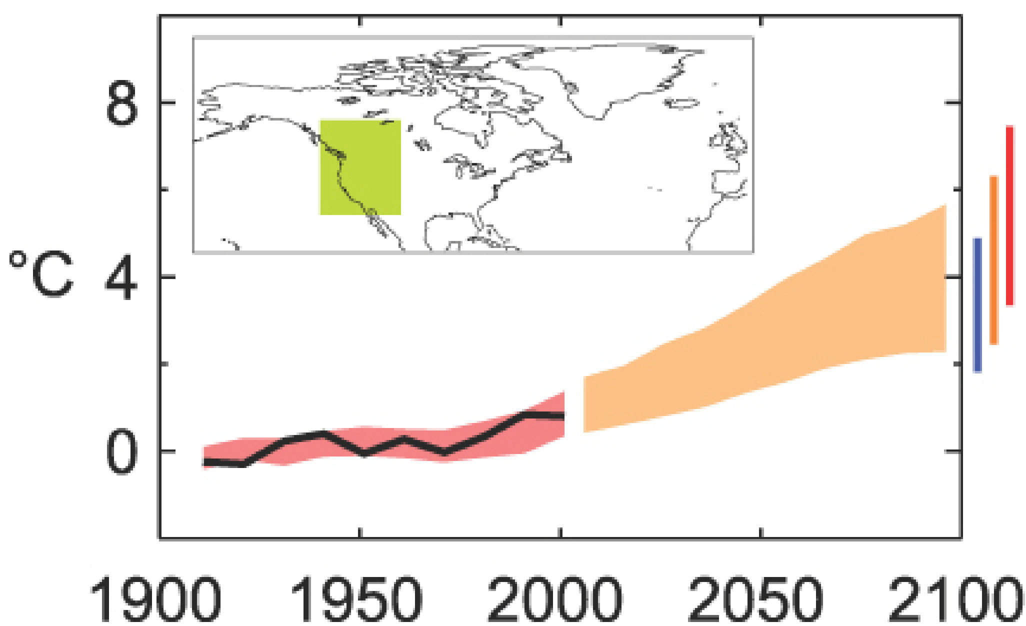


Figure 4: Temperature increases relative to 1901-1950 average temperature in western North America (from Figure 11.11 in Solomon et al., 2007). Bars on right show ranges for B1 blue, A1B orange, and A2 red.

Western North America is predicted to warm at rates comparable to global averages (Solomon *et al.*, 2007). For the A1B emission scenario, this is on the 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 *et al.*, 2007). This seemingly modest amount of warming has been linked to many changes in the western U.S., including plant phenology (Cayan *et al.*, 2001), snowpack reduction (Mote *et al.*, 2005; Pierce *et al.*, 2008), and earlier streamflows (Stewart *et al.*, 2005).

Projected changes in precipitation suggest strong declines in the southwestern US but seem less certain in the northwestern US (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 *et al.*, 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 *et al.*, 2000). Both results imply an increased flood risk from precipitation events (Easterling *et al.*, 2000; Hamlet and Lettenmaier, 2007a). However, there is also recognition that increased dryness or length of dry spells could also have significant ecological consequences (Easterling *et al.*, 2000; Dale *et al.*, 2001; Westerling *et al.*, 2006; Holden *et al.*, 2011b).

Variations in precipitation are influenced by sea surface temperature anomalies, such as ENSO, PDO, PNA, NAO, and AMO (e.g. Dettinger *et al.*, 1998; Cayan *et al.*, 1999; Clark *et al.*, 2001; McCabe *et al.*, 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 *et al.*, 2000).

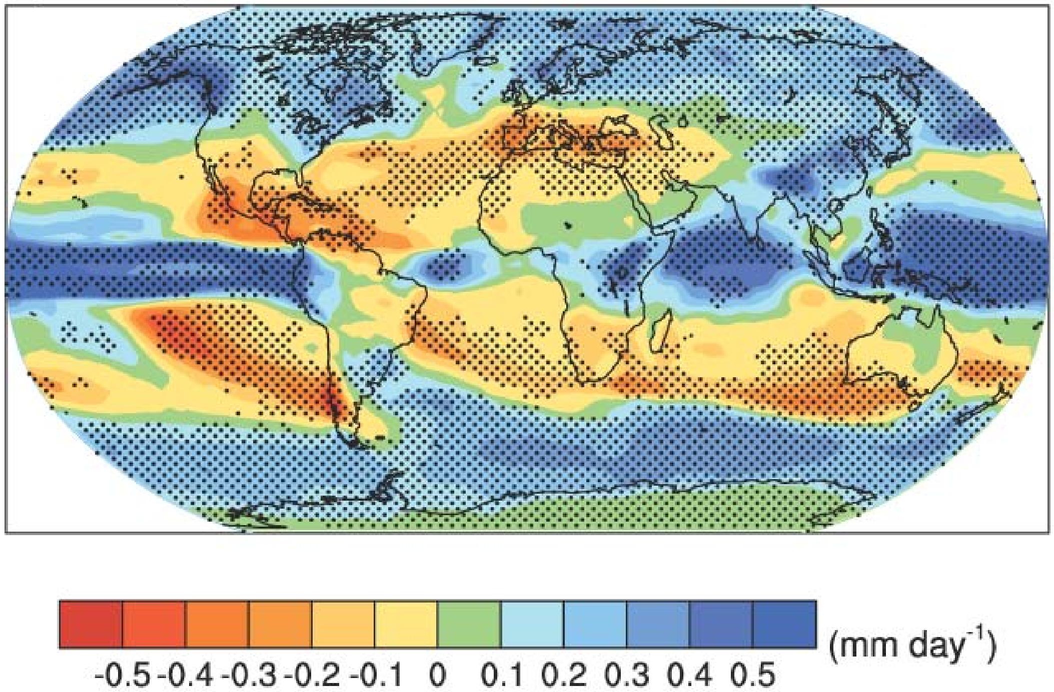


Figure 5: Precipitation anomalies (multi-model mean, A1B, 2080-2099 relative to 1980-1999). Stippling indicates areas where at least 80% of the models agree on the sign of the change.

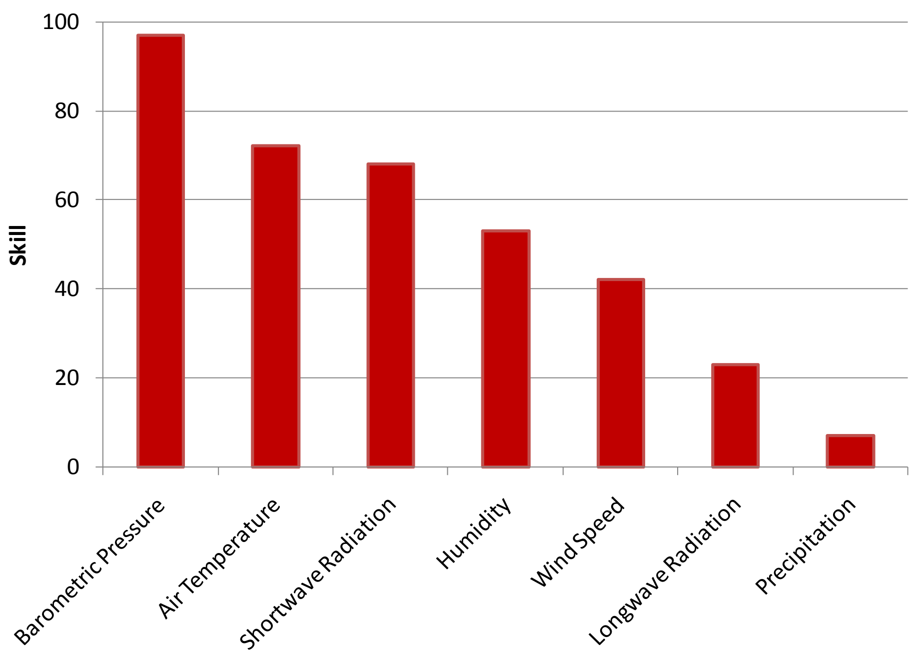


Figure 6: Relative skill of general circulation models in predicting different climate variables over Australia (Johnson and Sharma, 2009). Skill was indexed based on degree of agreement among models.

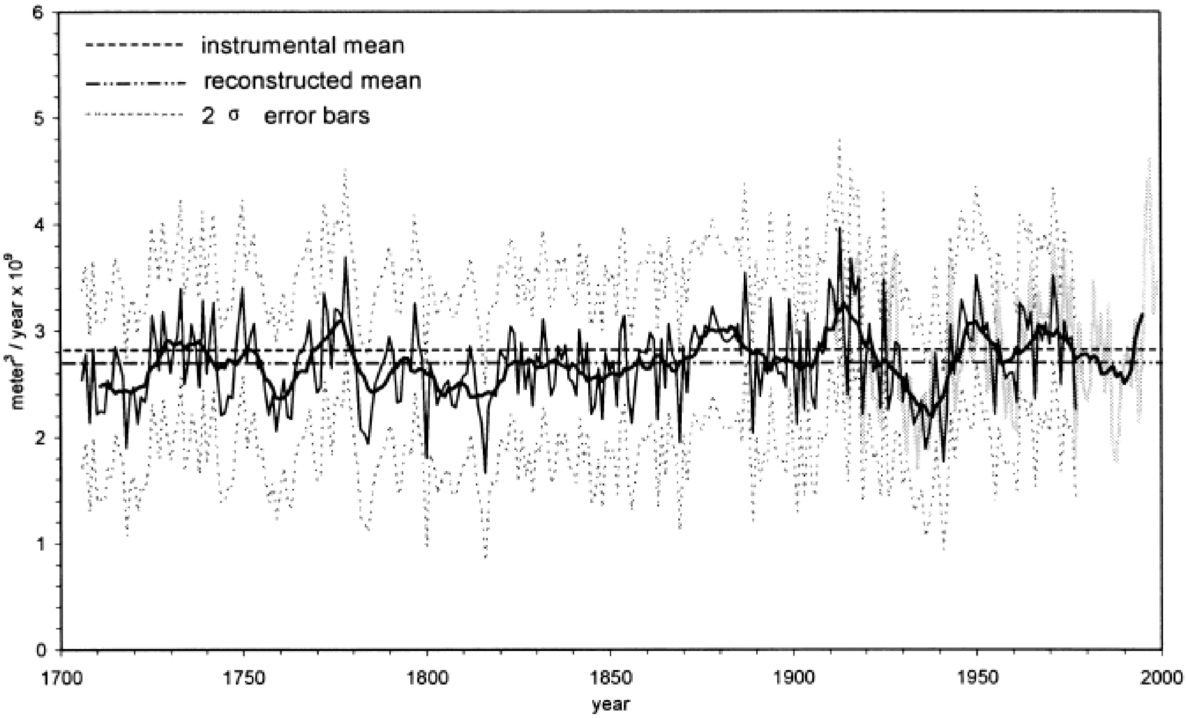


Figure 7: Streamflow reconstruction of annual flow volume of the Yellowstone River based on Tree Ring widths. Annual flows are in a solid black line bounded by 2 S.E. dashed lines and with a 10-yr moving average smooth in a heavier black line. From Graumlich et al. (2003).

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 U.S. is simply the availability of water. The interior west is a dry place, and even the wetter portions of the western U.S. 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 U.S. 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 *et al.*, 2008), and include reduced precipitation as snow compared to rain (SWE/P) (Knowles *et al.*, 2006), reduced snowpack on April 1st (Mote *et al.*, 2005; Regonda *et al.*, 2005), and earlier runoff timing (Stewart *et al.*, 2005). Changes to precipitation related to climate change are expected in the southwestern U.S. in the coming century caused by spreading Hadley cells (Seager and al., 2007; Johanson and Fu, 2009), but changes in the northwestern US are uncertain, leaving attribution difficult (Easterling *et al.*, 2000). Historical changes show increases in the southwestern streamflows (Regonda *et al.*, 2005) and declines in the northwestern US (Luce and Holden, 2009); such changes are partially consistent with general expectations for climate cycles (Dettinger *et al.*, 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 US falls in fall and winter, leaving a dry summer. Most of the precipitation in the western US also falls in mountains. It is not surprising then that about 75% of runoff in the western US 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 US (Barnett *et al.*, 2005).

Snowpacks in many parts of the western US are sensitive to variations in temperature (Mote *et al.*, 2005; Regonda *et al.*, 2005), and therefore to anthropogenic increases in greenhouse gases (Pierce *et al.*, 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 U.S. (Barnett *et al.*, 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 US 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 *et al.*, 2006). Conceptually we expect to see higher snowline elevations for individual storms (Casola *et al.*, 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 *et al.*, 2005). In the western U.S., 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 *et al.*, 2005; Pierce *et al.*, 2008; Nayak *et al.*, 2010a).

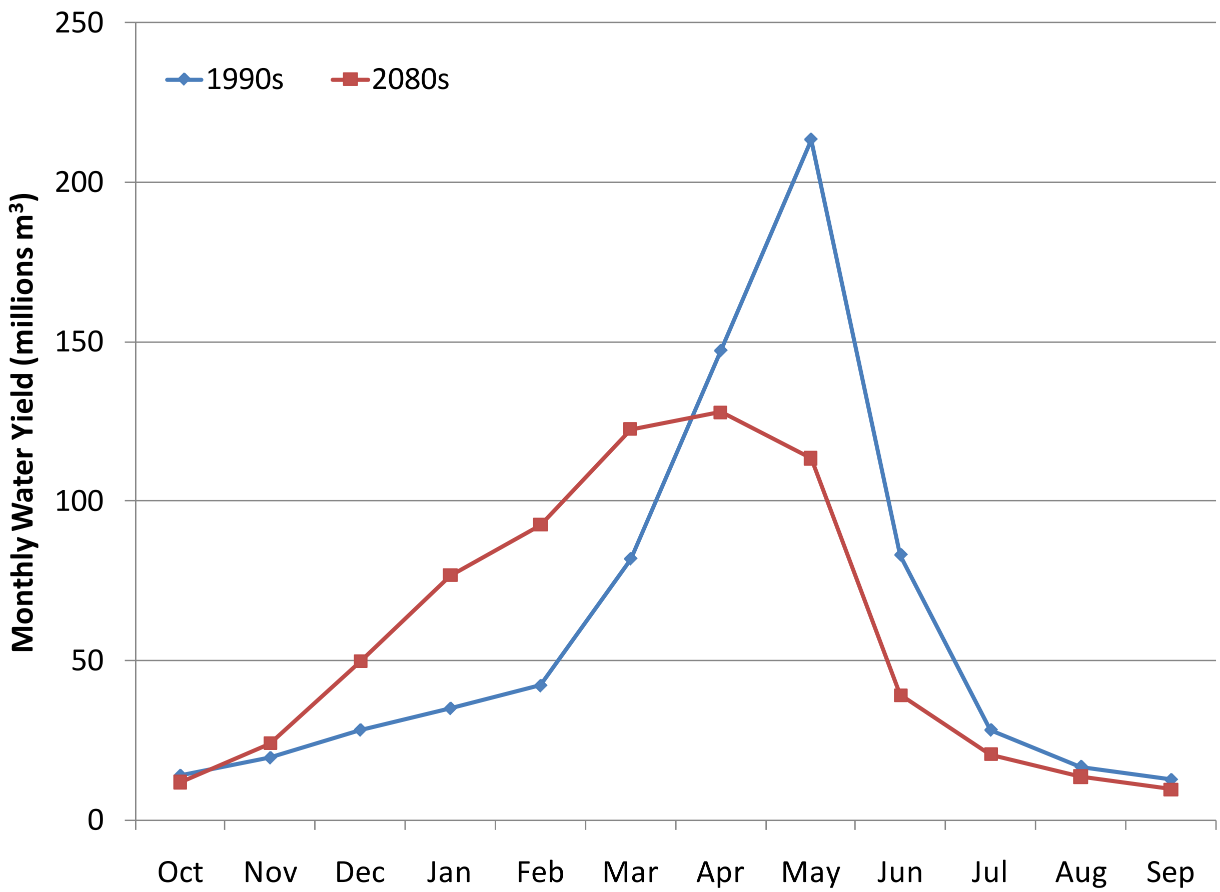


Figure 8: Average annual hydrographs simulated using the VIC model based on historical (1990s) climate compared to the projected climate of the 2080s under an A1B scenario. Data derived from VIC runs done by University of Washington Climate Impacts Group and US Forest Service Rocky Mountain Research Station (Elsner et al., 2010; Wenger et al., 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 *et al.*, 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, 2007b) in currently spring-snowmelt-dominated basins and because of increasing precipitation intensity in rain-dominated basins (Easterling *et al.*, 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 U.S. (e.g. Mote *et al.*, 2005; Regonda *et al.*, 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 *et al.*, 2007; Luce and Holden, 2009; Clark, 2010; Fu *et al.*, 2010a), particularly in the northwestern US 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 *et al.*, 2007).

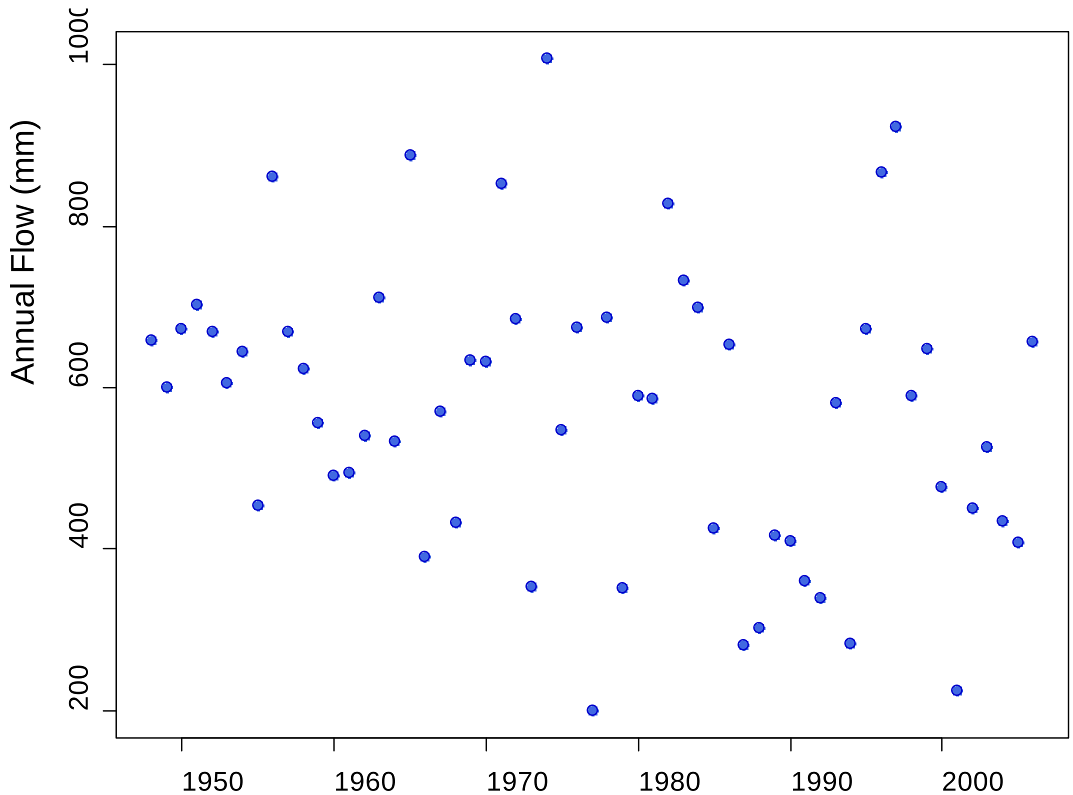


Figure 9: Trends in streamflow means and quantiles 1948-2006 for Johnson Creek at Yellow Pine, Idaho (USGS gage sta. 13313000). The dashed red line is the trend in the mean annual flow (24% decline, P=0.049), solid black line is the trend in the median (31% decline, P=0.025), the lower dashed black line 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 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 *et al.*, 2010a), others hypothesizing warmer temperatures are increasing evapotranspiration (Hoerling and Eischeid, 2007), and still others suggesting changes to land use and land cover (Wang and Hejazi, 2011). This raises the question, for example, if the 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 *et al.*, 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 *et al.*, 2005). 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 *et al.*, 2001; Ford *et al.*, 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 ferret 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 et al., 1997; Braganza et al., 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 *et al.*, 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 *et al.*, 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 *et al.*, 2005; Moore *et al.*, 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 is more rain and less snow, others are occurring simply because there is less snow (Figure 11).

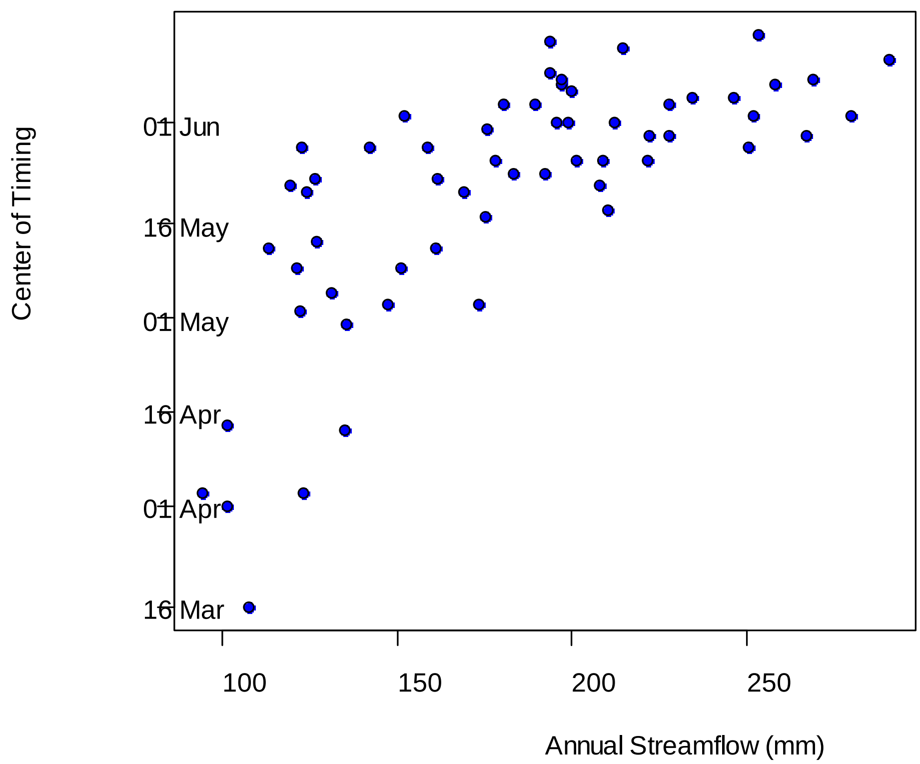


Figure 10: Relationship of snowmelt timing to total annual streamflow. Because sun angles are higher later in the spring, the last inch of snow can take weeks to melt if exposed by March but can melt in a few days if not exposed until June. The concave downward relationship has implications for causes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

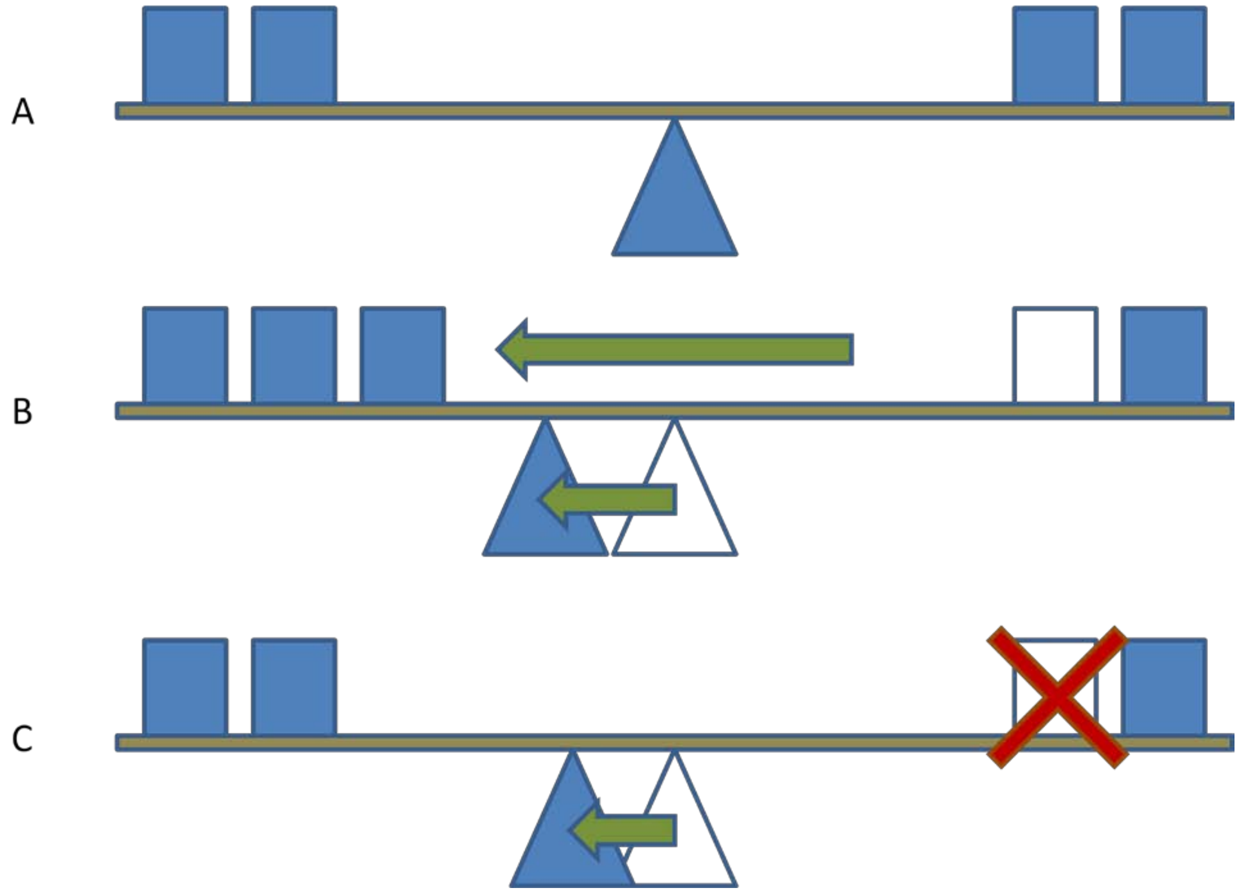


Figure 11: The flow timing seesaw. If the blue boxes represent buckets of runoff from a watershed, the center of timing for streamflow (blue triangle) is the balance point for when those buckets runoff over the year (A). If some of the flow starts coming off sooner because of earlier melt or falling 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.

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 *et al.*, 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 *et al.*, 2005; Stewart *et al.*, 2005; Westerling *et al.*, 2006; Barnett *et al.*, 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 *et al.*, 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 *et al.*, 2000; Cannon *et al.*, 2001; Istanbulluoglu *et al.*, 2002; Neary *et al.*, 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 *et al.*, 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 *et al.*, 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 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 *et al.*, 2000). Potentially the volatilization and recondensation of the chemicals has an annealing effect.

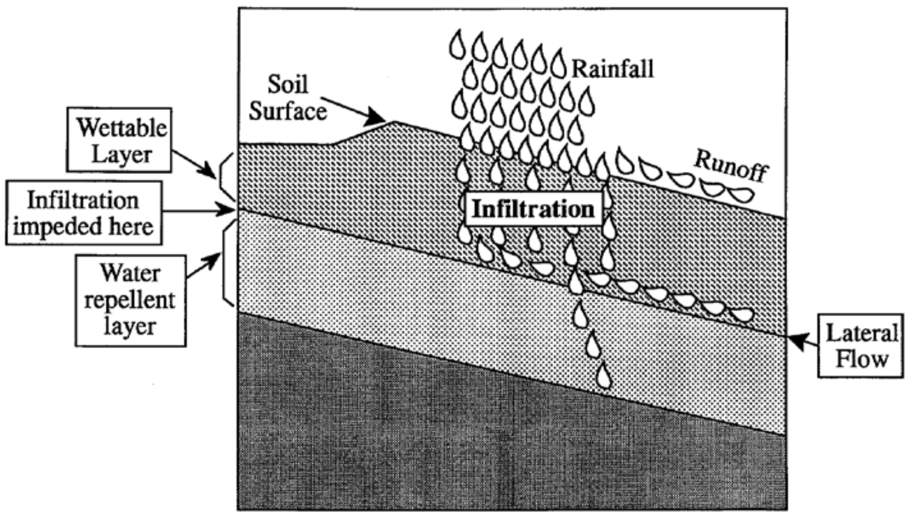


Figure 12: Schematic of water repellency effects on infiltration and runoff generation after fire (after DeBano, 1969).

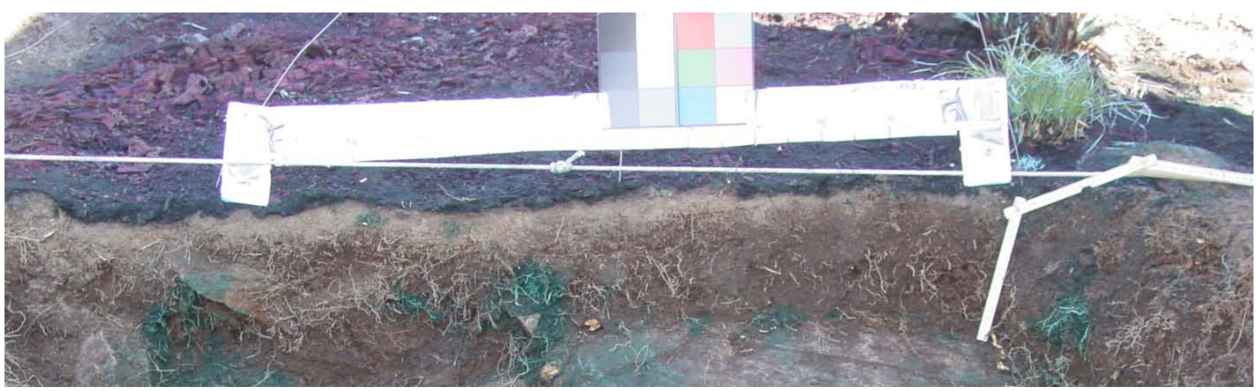


Figure 13: Photograph of wetted layer over a water repellent layer after rainfall simulation with blue dye showing substantial similarity to the schematic by Debano (1969) The scale bar is 1-m between the legs.

Water repellency is sensitive to the soil moisture state, and soils with water repellent substances generally repel water when dry (Doerr *et al.*, 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 U.S. 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).

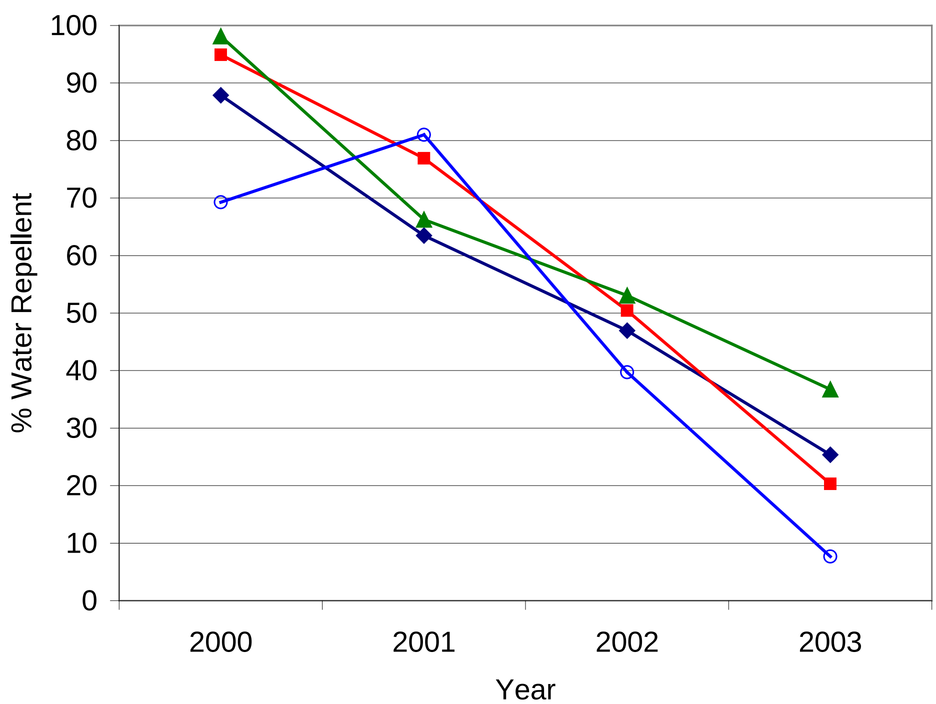


Figure 14: Declining fractional water repellent area over time following fires near Boise, Idaho and Bozeman, Montana. The four lines are four separate burned areas (unpublished data).

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), however similar behaviors have been seen on burned soils under intense rainfall (Larsen *et al.*, 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. Decreases in hydraulic conductivity related to surface sealing seem to be less severe than those associated with water repellency. For example Larsen et al. (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 U.S., the largest basin with a reported event was 122 km^2, and 99% of the basins were less than 25 km^2 (Gartner *et al.*, 2005). After the Tillamook burn in the Oregon Coast Range in 1933, (Anderson *et al.*, 1976) estimated a 45% increase in the peak flow of two basins close to 400 km^2 in size the first year after the fire. In the Boise River, no increase in peak flow was noted at the 2,000-km^2 scale despite measurable changes to water yield and some dramatic events in basins up to 20 km^2 in extent (see textboxes on debris flow scale and Boise River hydrology).

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 *et al.*, 1995; Troendle *et al.*, 2010; also see textbox on Boise R.). 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 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 *et al.*, 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 *et al.*, 2011; Guardiola-Claramonte *et al.*, 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 *et al.*, 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).

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 *et al.*, 2001; Andréassian, 2004; Brown *et al.*, 2005). There are, however, questions about whether water yield increases realized in small experimental basins (typically less than 10 km^2) translate into increases from large basins on the order of a few thousand km^2 (Troendle, 1983; Troendle *et al.*, 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.

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%

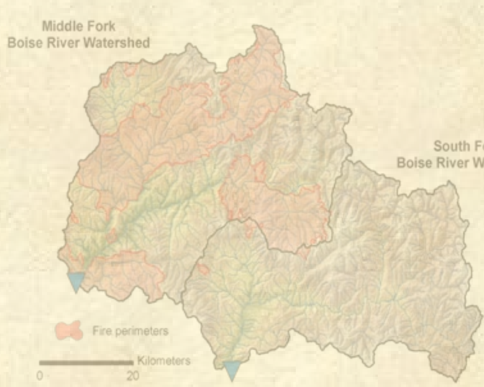


Figure T3-1: Middle and South Fork Boise watersheds with fire polygons. Blue triangles are locations of USGS gages.

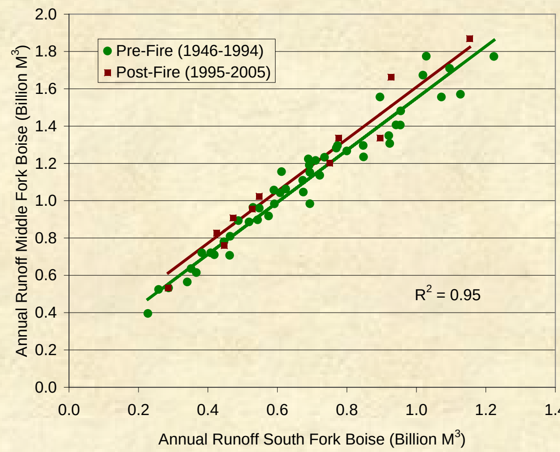


Figure T3-2: Paired watershed analysis regression for annual water yield.

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 T2-1). The middle fork of the Boise River has been gauged since 1912, while the

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

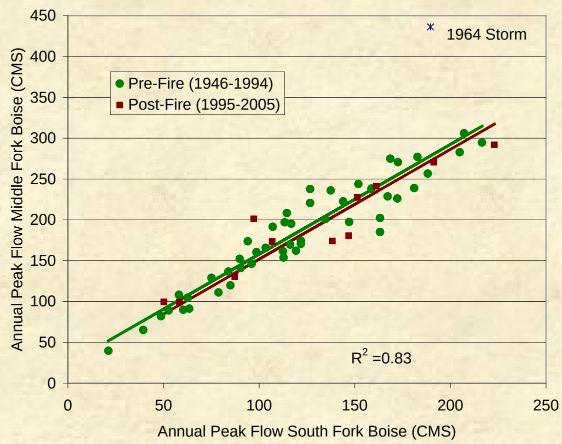


Figure T3-4: Paired watershed analysis regression for annual peak flows.

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 T2-2).

Despite severe changes in peak flows in some severely burned small basins (Figure T2-3) there was no increase in peak flows at the basin scale (Figure T2-4). The seasonal distribution of the flow increases was primarily in winter and spring, although late summer low flows also increased a little (Figure T2-5). 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.

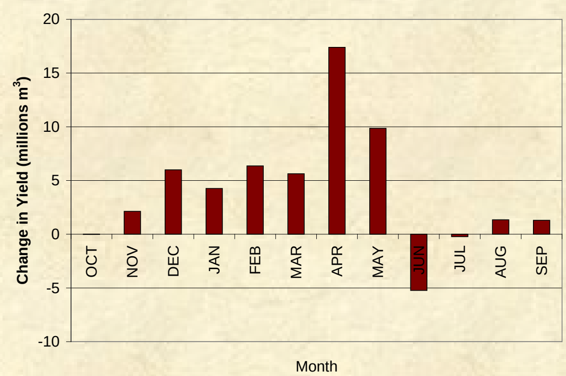


Figure T3-5: Monthly distribution of the water yield changes. Increases are shown through most of the year. The decreases in June and July are related to faster snowmelt in burned areas and relate to the much higher increases in April and May.

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 *et al.*, 2009). Likely the effects of climate change on fire, vegetation and streams will be synergistic. In the western US, more large fires (Westerling *et al.*, 2006) and more widespread fires are more likely when early, warm springs are followed by warm, dry summers in the forests of the US northern Rocky Mountains and elsewhere (Westerling *et al.*, 2006; Morgan *et al.*, 2008; Littell *et al.*, 2009) . Westerling et al. (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 et al. (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 US. Littel et al. (2010) predict increased area burned in many different regions across the western US 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 *et al.*, 2006), multiple decades (McKenzie *et al.*, 2004; Morgan *et al.*, 2008; Littell *et al.*, 2010), and multiple centuries as inferred from crossdated fire scars on trees (e.g. Kitzberger *et al.*, 2007; Heyerdahl *et al.*, 2008, and others) and from charcoal in lakes (Whitlock *et al.*, 2003) and debris flows (Pierce *et al.*, 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 *et al.*, 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 US (Dillon *et al.*, 2011), as north-facing slopes are more likely to burn severely than north-facing slopes at the same elevation and high elevation forests often burn more severely than lower elevation forests (Holden *et al.*, 2011b). North-facing slopes are often relatively moister than south-facing slopes, with soils with higher organic matter and with higher biomass, which once dry can burn severely (Dillon *et al.*, 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 et al. (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 et al. (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 US 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.

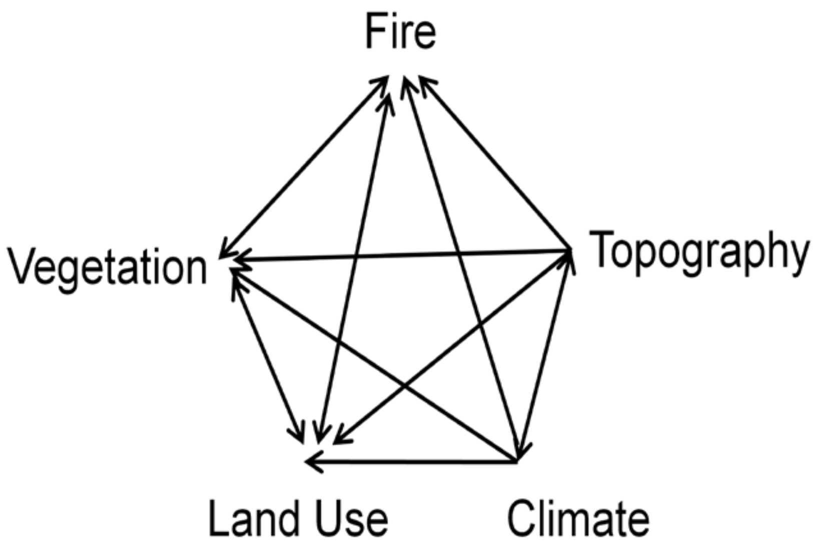


Figure 15. The interactions among factors influencing fire patterns are complex. Adapted from Canadian Forest Service (2001) Forest fire: context for the Canadian Forest Service's science program. <http://www.nrcan-rncan.gc.ca/cfs-scf/science/context_fire/index_e.html>

The relative role of climate/weather is likely different between forests and rangelands and among different rangeland and forest ecosystems (Collins *et al.*, 2006; Littell *et al.*, 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 *et al.*, 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 U.S. 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, 2007b), 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 *et al.*, 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 *et al.*, 1996; Tonina *et al.*, 2008) or sweep fry downstream (Fausch *et al.*, 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 *et al.*, 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 *et al.*, 2004; Pörtner and Farrell, 2008; Wenger *et al.*, 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 *et al.*, 2003; Rieman *et al.*, 2007; Isaak *et al.*, 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 *et al.*, 2003; Rieman *et al.*, 2007; Wenger *et al.*, 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 *et al.*, 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 *et al.*, 2002; Isaak *et al.*, 2010; Kelleher *et al.*, 2011)

Historical trends in stream temperature show increases in many places in recent decades even without land cover changes (Langan *et al.*, 2001; Petersen and Kitchell, 2001; Morrison *et al.*, 2002; Bartholow, 2005; Hari *et al.*, 2006; Isaak *et al.*, 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 *et al.*, 2010). For example, some streams in mountains, particularly with glaciers or snowfields, show a distinct buffering

The Stream Energy Balance

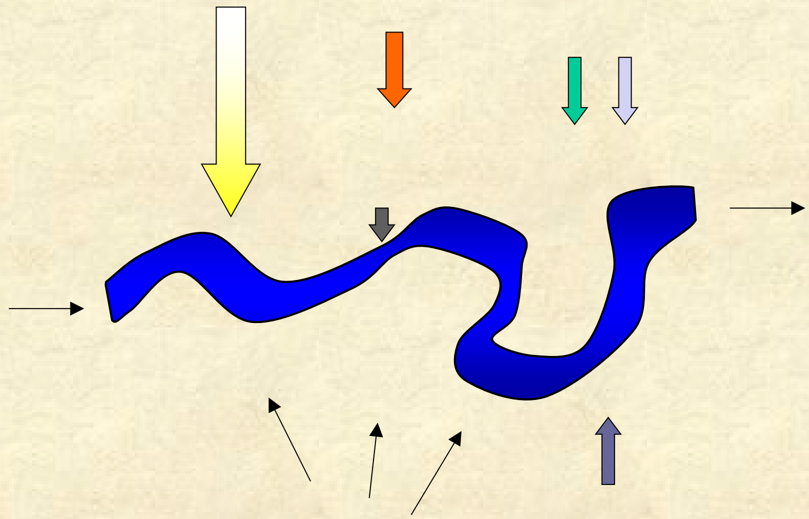


Figure T4-1: Components of the stream energy balance. Q's are water fluxes, and T's are temperatures of inflowing (in), ouflowing (out), and groundwater (g) respectively. Together these comprise the advective heat fluxes to and from a stream reach. Radiant fluxes, denoted by R, are separated as net shortwave (sn) from the sun, and net longwave (ln). The sensible (Fs) and latent (Fe) heat fluxes are together called turbulent heat fluxes and represent the energy carried by wind in terms cool or warm air convection on the stream (sensible) or evaporation or condensation on the stream (latent). The friction flux (Ff) 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 T3-1 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 *et al.*, 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 *et al.*, 2005a; Dunham *et al.*, 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 *et al.*, 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.

Although air temperature is commonly applied as an index of stream temperature, and strong correlations between air temperature and stream temperatures can be developed (Stefan and Preud'homme, 1993; Mohseni *et al.*, 2003) (Pilgrim *et al.*, 1998; Webb *et al.*, 2003), changes in air temperature are generally not the direct cause of major changes in stream temperature (Johnson, 2003; Johnson, 2004). Rather, both air temperature and stream temperature are mutually driven by external forcings, e.g. net radiation. Application of air temperature – stream temperature relationships to project future stream temperatures, therefore, has some uncertainty, because the empirical relationships between air temperature and stream temperature may themselves change. The relationships, for instance are affected by streamflow (Webb *et al.*, 2003), which may change in the future as well. Climate change will most directly and predictably increase the incoming longwave radiation, and increases in solar radiation may occur if forest vegetation is lost. Turbulent heat fluxes may change little if relative humidity is unchanged, and these comprise a small portion of the energy budget even if they were to change substantially.

Estimates of stream temperature sensitivity to air temperature are among the most readily applied tools for climate change estimation, but those estimates vary substantially from place to place (Webb and Nobilis, 1997; Kelleher *et al.*, 2011), and insights from energy balance studies will be helpful in interpreting results and accounting for the range of direct and indirect effects.

due to increased snowmelt inputs from increased summer melt rates (Hari *et al.*, 2006), and greater groundwater inputs can buffer warming as well (Kelleher *et al.*, 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 *et al.*, 1989; Hitt, 2003; Dunham *et al.*, 2007; Isaak *et al.*, 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 *et al.*, 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 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 et al. (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.

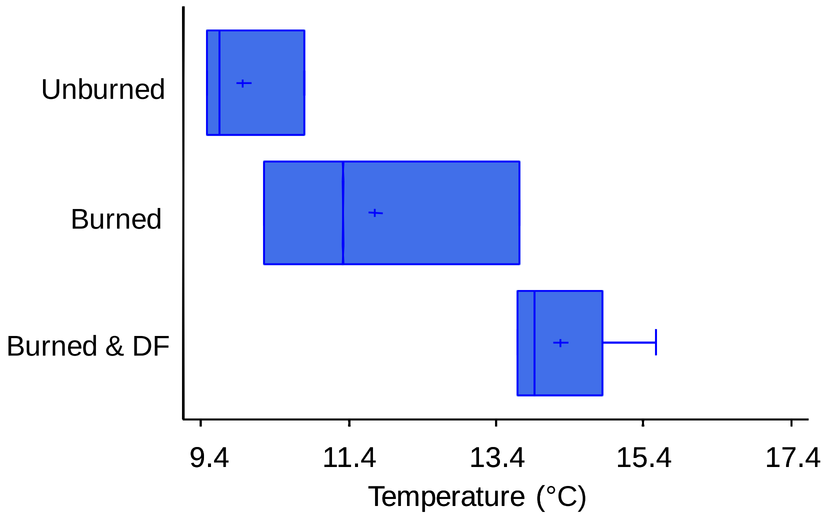


Figure 16: Maximum summer temperatures in streams after fire. Data from 10 data loggers placed along 9 streams (Dunham *et al.*, 2007).

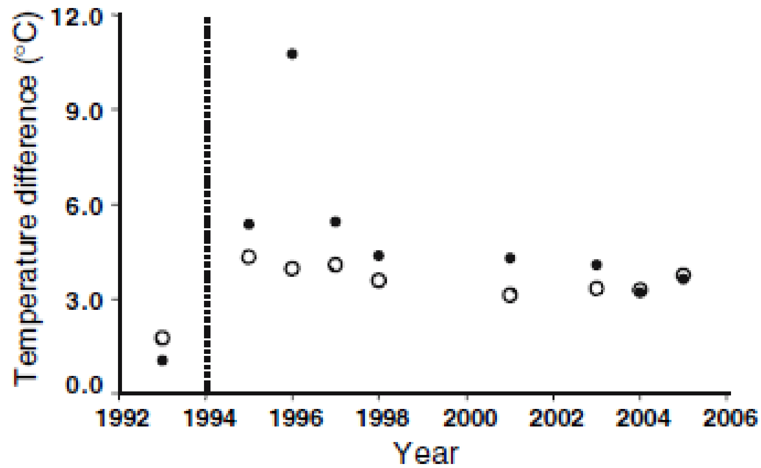


Figure 17: Temperature differences between Cottonwood Creek, which burned in 1994, and Roaring River in the Boise River basin. Open symbols are summer mean temperatures and filled symbols are the summer maximum (Dunham et al., 2007).

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 *et al.*, 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 *et al.*, 2006; McCullough *et al.*, 2009), which shows in the spatial and elevation distributions of fishes (e.g. Wenger *et al.*, 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 *et al.*, 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 further into headwater streams (Rieman *et al.*, 2007; Isaak *et al.*, 2010; Wenger *et al.*, 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 U.S. 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 yield may be less important than the potential of increased floods due to higher precipitation intensity (e.g. Easterling *et al.*, 2000; Hamlet and Lettenmaier, 2007b) or increased probability of occurrence of rain-on-snow floods (Lettenmaier and Gan., 1990; Nayak *et al.*, 2010b).

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; Istanbulluoglu and Bras, 2006; Collins and Bras, 2008). For example, Figure 18 (Goode *et al.*, 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.

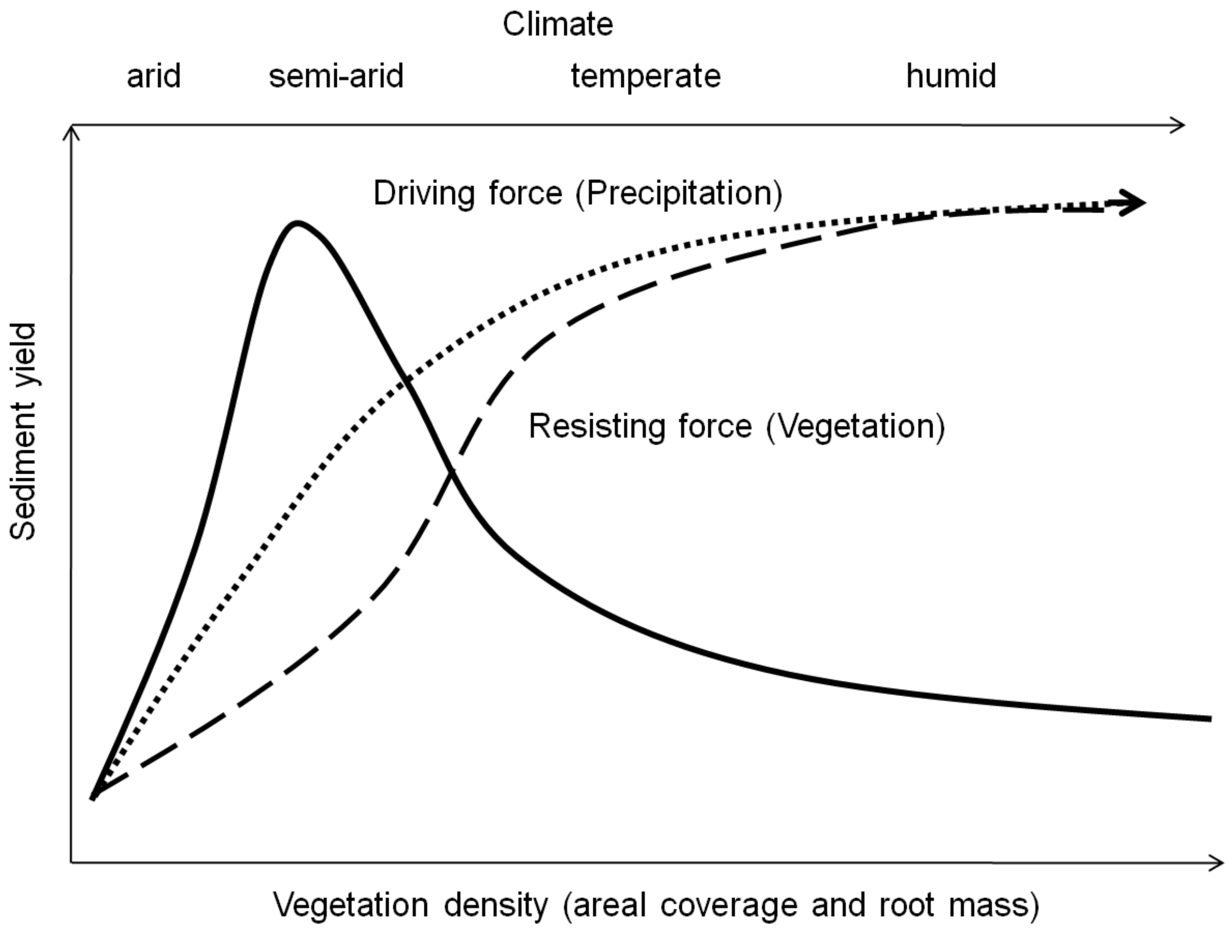


Figure 18: Conceptual relationship between sediment yield and climate. The sediment yield curve (solid) is based on the empirical relationship of Langbein and Schumm (1958). The dashed line represents the resistance to erosion by vegetation while the dotted line represents the relative driving force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

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 *et al.*, 1992; Swetnam and Betancourt, 1998; Briffa, 2000; Meyer and Pierce, 2003; Whitlock *et al.*, 2003; Pierce *et al.*, 2004; Marlon *et al.*, 2006). At shorter time scales, it is clear that years of widespread fire are linked to severely dry and warm years (e.g. McKenzie *et al.*, 2004; Heyerdahl *et al.*, 2008; Morgan *et al.*, 2008; Littell *et al.*, 2009). As we shift toward a drier and warmer climate in the western US, there is an expectation of greater areas burned annually (e.g. Running, 2006; Littell *et al.*, 2009; Spracklen *et al.*, 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 *et al.*, 2011). Hillslope and steep channel processes, such as surface erosion and mass wasting receive the greatest attention (e.g. Benavides-Solorio and MacDonald, 2001; e.g. Cannon *et al.*, 2001; Miller *et al.*, 2003; Pierce *et al.*, 2004; Moody and Martin, 2009; Robichaud *et al.*, 2009b), while the disposition of channels with aquatic habitat is comparatively poorly discussed, despite more direct connections to aquatic ecology (Benda *et al.*, 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 *et al.*, 1995; Benda and Dunne, 1997a, b; Rieman and Clayton, 1997; May and Gresswell, 2003; Miller *et al.*, 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 *et al.*, 1995; Benda *et al.*, 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 *et al.*, 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 *et al.*, 2008; Moody and Martin, 2009), but 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 *et al.*, 2001; Istanbulluoglu *et al.*, 2002; Istanbulluoglu *et al.*, 2003; Miller *et al.*, 2003; Santi *et al.*, 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 (Istanbulluoglu *et al.*, 2003; Meyer and Pierce, 2003; Istanbulluoglu *et al.*, 2004; Moody and Martin, 2009; Cannon *et al.*, 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 *et al.*, 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 *et al.*, 2004). Debris flow *passage* and *deposition* occur in distinct areas of a stream network and have greatly differing effects on habitat and biota.

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.

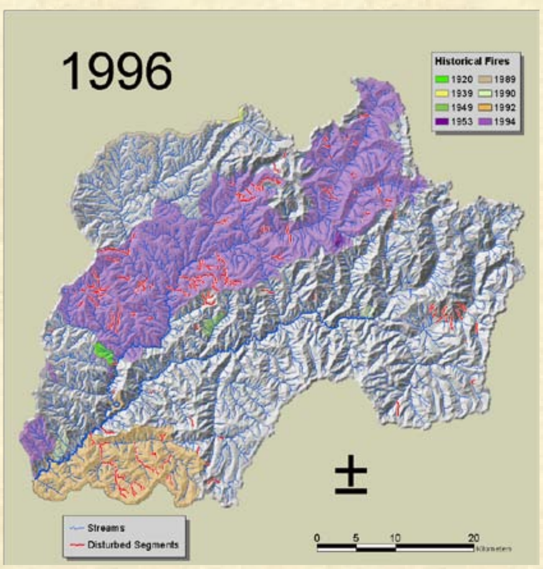


Figure T5-2: Map of debris flow affected segments from the 1996 photos.

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.

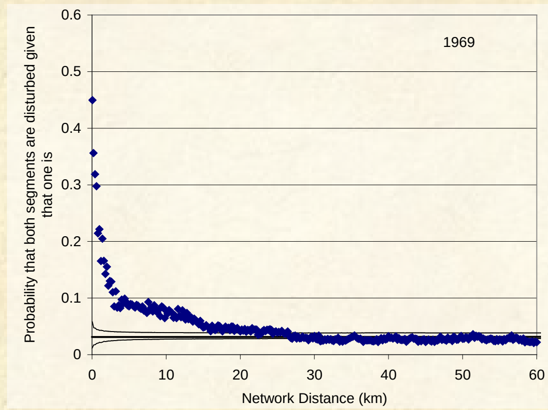


Figure T5-3: Probability that two segments separated by a given distance are simultaneously disturbed. Thin black lines show expected probability if disturbed segments were randomly distributed.

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 km^2, which corresponds to a stream network length of 20 km.

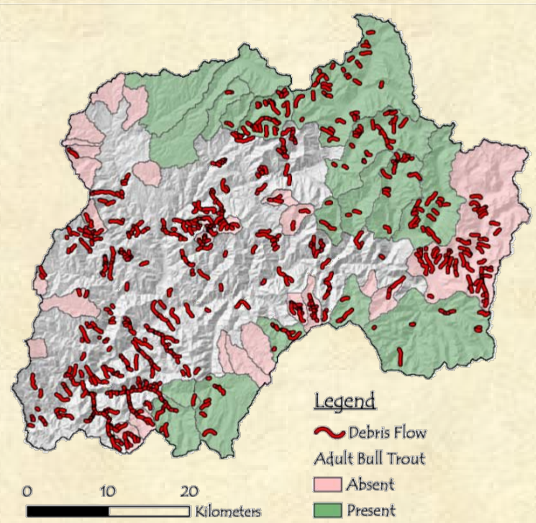


Figure T5-4: Map of all debris flows with bull trout patches from Dunham and Rieman (1999). Unoccupied patches tend to be small, on the order of debris flow affected basins.

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 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 *et al.*, 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 et al. (2005) looking at the western US, the 95th percentile basin size was about 15 km^2, while the median basin size was about 1.2 km^2. 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 *et al.*, 2010; Rieman *et al.*, 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 *et al.*, 1995; Benda *et al.*, 1998; Benda *et al.*, 2003a). Gravels, large boulders, woody debris, and soil entrained in debris flows bring in large cover and 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 *et al.*, 2003a), deposit fans that are rapidly reworked and transported (Lisle *et al.*, 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 *et al.*, 1989; Reeves *et al.*, 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 *et al.*, 1995; May and Gresswell, 2003; Scheidt, 2006; also see textbox on 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).

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 et al. (2001) compared long-term (~10,000 years) sediment yields measured using cosmogenic ^{10}Be 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. Istanbulluoglu et al. (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.

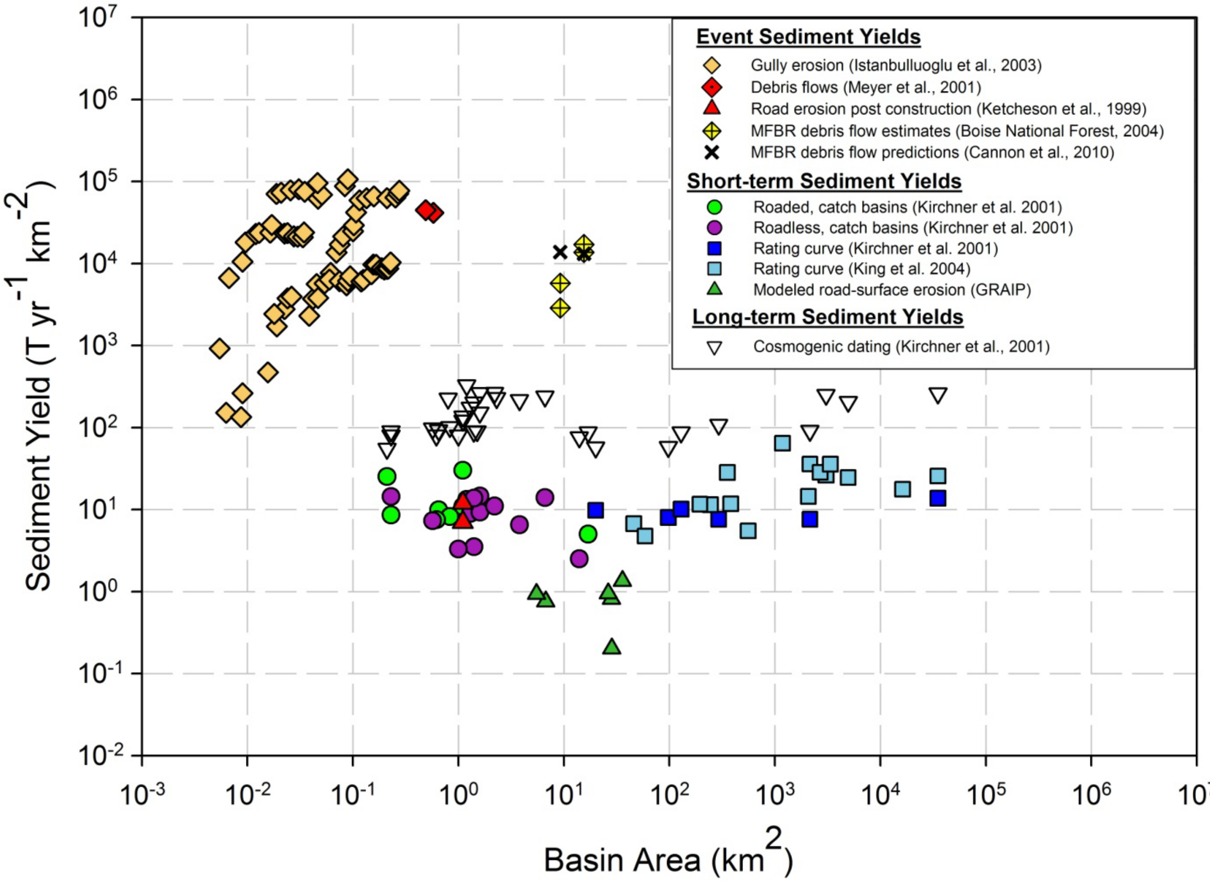


Figure 19: From Goode et al., (2011). Sediment yields for individual post-fire erosional events, long-term basin averages, short-term basin averages (~30 yrs.), and road-surface erosion. Individual post-fire erosional events include measurements of debris flow and gully tracks in the Boise and Payette River Basins (Meyer *et al.*, 2001; Istanbulluoglu *et al.*, 2003) and deposit estimates (Breidenbach *et al.*, 2004) and model predictions (Cannon *et al.*, 2010). Long-term basin averages are from analysis of cosmogenic nuclides in fluvial sediments (Kirchner *et al.*, 2001). Short-term averages for small basins (< 20 km^2) are from catchbasin dams (1950's-1980's, Kirchner *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 1999).

How important are roads in this context? Some of the watersheds studied by Kirchener et al. (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 *et al.*, 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 *et al.*, 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 *et al.*, 2006; Robichaud *et al.*, 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 *et al.*, 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 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 *et al.*, 2009; Spracklen *et al.*, 2009; Littell *et al.*, 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 *et al.*, 2004; Morgan *et al.*, 2008; Littell *et al.*, 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 *et al.*, 2004; Morgan *et al.*, 2008; Spracklen *et al.*, 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, 2004), changing climate (Solomon *et al.*, 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 *et al.*, 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 1900's, 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 *et al.*, 2004; McKenzie *et al.*, 2004; Morgan *et al.*, 2008; Littell *et al.*, 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. In what has been referred to as the grass-fire cycle; the more grass, the more fire, and the more fire the more grass (Vitousek *et al.*, 1996; Brooks, 2004). As a result, there are many ecosystems worldwide experiencing fires much more frequently than in the past (Vitousek *et al.*, 1996; Brooks, 2004; Shlisky *et al.*, 2007).

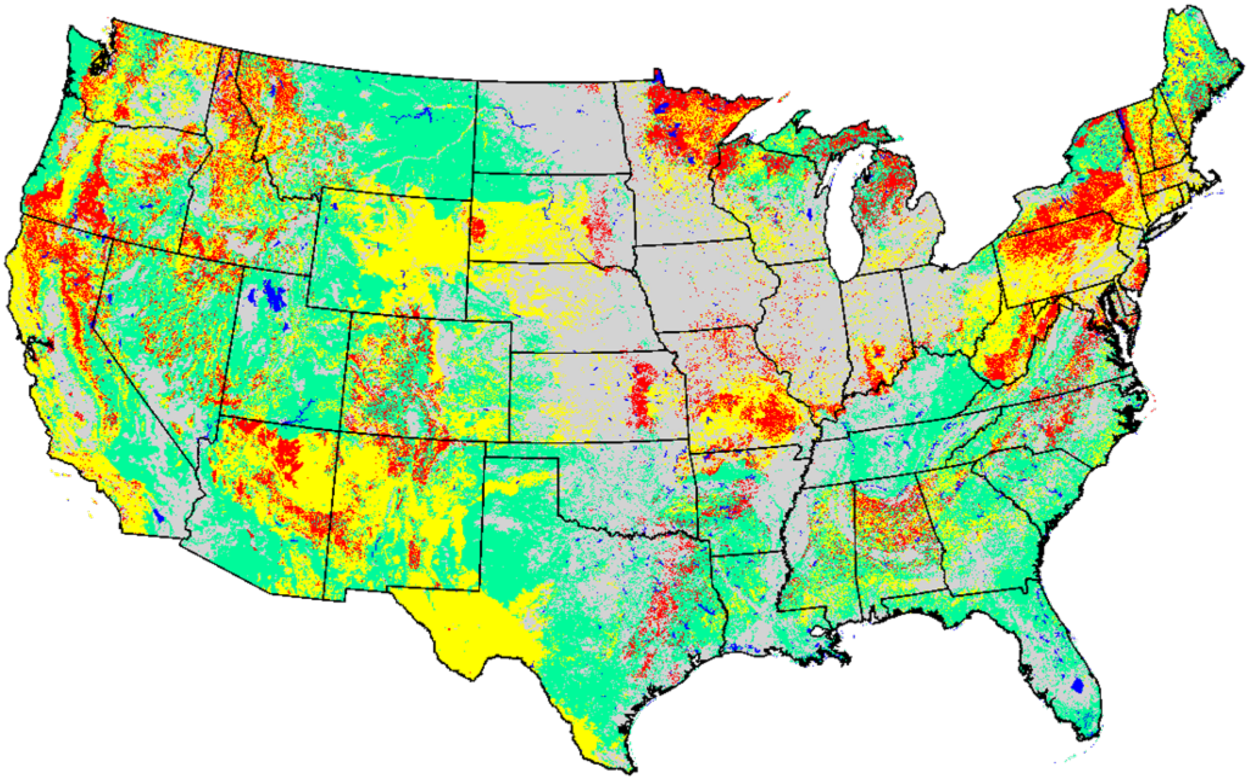


Figure 20: Fire Regime Current Condition (FRCC) class map version 2000 (from Schmidt et al., 2002). Red areas show the greatest departure from historical fire regimes and green show the least. Gray areas are non-forested. (www.frcc.gov).

Table 2: Fire regimes can be grouped by how often fires recur and the degree to which ecosystems change when they burn (from Barrett *et al.*, 2010), http://www.fire.org/niftt/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.frcc.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 al., 2005). Species may not be well adapted to the uncharacteristic fire regimes that develop when fire frequency and severity and vegetation composition are very different than historical range of variability (Keane *et al.*, 2008). More than 80% of the ecoregions examined worldwide have degraded or very degraded fire regimes (Shlisky and al., 2005). 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 *et al.*, 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 *et al.*, 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 *et al.*, 2008; Dillon *et al.*, 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 US northern Rockies (Running, 2006; Morgan *et al.*, 2008; Littell *et al.*, 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 *et al.*, 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).

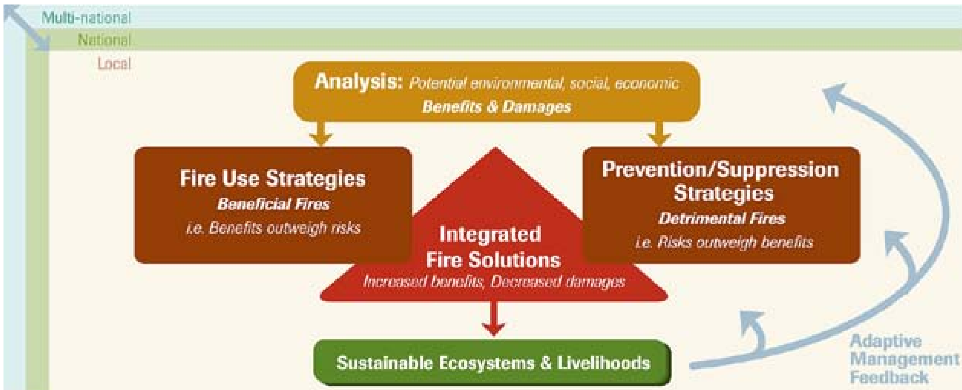


Figure 21. Integrated Fire Management encompass fire use, prevention, and suppression with an understanding of sustainable ecosystems and livelihoods (from Myers, 2006).

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 *et al.*, 2006; Lentile *et al.*, 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 and tree mortality, soil heating and soil effects. The term burn severity is imprecise and carries inherently negative connotations (Lentile *et al.*, 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 charactertistics 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 *et al.*, 2006; Lentile *et al.*, 2007; Keane *et al.*, 2008; Keeley, 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.

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.

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 – those 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, measureable 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 *et al.*, 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 *et al.*, 2007; Parsons *et al.*, 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 *et al.*, 2005; Smith *et al.*, 2005; Lentile *et al.*, 2006; Hudak *et al.*, 2007). These continuous measures are often broken into classes, often based on field assessments made with the Composite Burn Index (CBI)(van Wagtendonk *et al.*, 2004; Brewer *et al.*, 2005; Cocke *et al.*, 2005). Unfortunately, there are few studies evaluating dNBR or CBI against quantitative biological or ecological measures of post-fire effects (Hudak *et al.*, 2007; Robichaud *et al.*, 2007; Santis and Chuvieco, 2007; Smith *et al.*, 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 US in the national Monitoring Trends in Burn Severity (MTBS, www.mtbs.gov) program (Cocke *et al.*, 2005; Epting *et al.*, 2005; Miller and Thode, 2007). There are limitations to these approaches (Roy *et al.*, 2006; Smith *et al.*, 2007; Lentile *et al.*, 2009). Briefly, limitations include the subjective and qualitative nature of breaks between classes and field assessments using CBI; 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 also want to know soil heating and fuel consumption (Roy *et al.*, 1987; Hudak *et al.*, 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 et al. (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 *et al.*, 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 *et al.*, 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 *et al.*, 2004; Keane *et al.*, 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 *et al.*, 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 *et al.*, 2009a; Parsons *et al.*, 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, and 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 *et al.*, 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 et al. (2009), Littel et al. (2010), and Westerling et al. (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 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 et al. (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 et al. (2011) found that annual proportion burned severely increased in just one of the three US

Southwest ecoregions and two of three ecoregions in the US 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 et al. (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 *et al.*, 2010).

Fuels 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 *et al.*, 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 *et al.*, 2002, 2006). Fuel treatments are designed to reduce fire hazard with the goal of altering fire behavior, thus easing fire suppression efforts (Graham *et al.*, 1999; Graham *et al.*, 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 surface fuels are treated (e.g. before hand piles are 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 *et al.* (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 *et al.*, 2007; Harbert *et al.*, 2007; Murphy *et al.*, 2007b; Martinson *et al.*, 2008; Hudak *et al.*, 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 et al. (2009) found similar results. Continued effectiveness will depend on maintenance and retreatment (Graham *et al.*, 2009). Hartsough et al. (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 *et al.*, 1999; Graham *et al.*, 2004; Graham *et al.*, 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 US 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 *et al.*, 2004; Graham *et al.*, 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 *et al.*, 2004). Further, what is ecologically appropriate, sustainable, and socially acceptable will vary from place to place (Graham *et al.*, 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 US 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 *et al.*, 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 *et al.*, 2004; Jenkins *et al.*, 2008; Hoffman *et al.*, 2011; Simard *et al.*, 2011), but outcomes are less certain for burn severity. Many people think trees killed by insect outbreaks are more likely to burn and burn severely (Geiszler *et al.*, 1980; Knight, 1987) because there are more dead needles in the trees or on the surface and eventually more large wood (Page and Jenkins, 2007; Jenkins *et al.*, 2008). Others argue the opposite (Bigler *et al.*, 2005; Simard *et al.*, 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 *et al.*, 1994; Bebi *et al.*, 2003; Kulakowski *et al.*, 2003). Likely, crown fire hazard is high in where the proportion of trees killed by bark beetles is high and the red needles are still in the trees, but then decreases as red needles fall (sometimes called the gray stage) (Hoffman *et al.*, 2011). Tree mortality could also increase fire hazard by increasing amount of solar radiation in the subcanopy, drying surface fuels (Hoffman *et al.*, 2011). These effects 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 *et al.*, 1998), and this was evident in some studies (Bradley and Tueller, 2001; McHugh *et al.*, 2003; Wallin *et al.*, 2003; Cunningham *et al.*, 2005), but not others (Elkin and Reid, 2004), or no relation was found (Sanchez-Martinez and Wagner, 2002).

Lynch et al. (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 et al. (2005) found that prior stand structure that resulted from multiple disturbances, including bark beetles, affected burn severity in spruce forests in Colorado. Kulakowski et al. (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 *et al.*, 2008). Hoffman et al.

(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.

Many efforts are focused on homes to reduce the likelihood 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.

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.

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 strategies. 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 *et al.*, 1993; Naiman *et al.*, 1998; Pollock *et al.*, 1998; Tabacchi *et al.*, 1998; Naiman *et al.*, 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 *et al.*, 2005; Wohl *et al.*, 2007). Streamside vegetation reflects these local physical features (Baker, 1989; Friedman *et al.*, 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 *et al.*, 1998). High levels of biodiversity in riparian areas are maintained by spatial habitat heterogeneity (Pabst and Spies, 1999; Sarr *et al.*, 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 *et al.*, 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 *et al.*, 1997; Corenbilt *et al.*, 2009; Merritt *et al.*, 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 *et al.*, 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 *et al.*, 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 US (e.g. Hansen *et al.*, 1995; Manning and Padgett, 1995; Crowe and Clausnitzer, 1997; Carsey *et al.*, 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.

Riparian areas cover a relatively small area in any given watershed, yet they provide critical ecological functions (Brinson *et al.*, 2002; Naiman *et al.*, 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 *et al.*, 1991; Prichard *et al.*, 1993; revised 1995, 1998; Naiman and Decamps, 1997; Brinson *et al.*, 2002; Naiman *et al.*, 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; (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.

<u>Stream temperature:</u> 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 *et al.*, 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 *et al.*, 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 *et al.*, 1999; Dunham *et al.*, 2007; Isaak *et al.*, 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 *et al.*, 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 et al. (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 describe 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 temperatures averaged 2-3 times greater than basin averages within the burned portions of watersheds, 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 *et al.*, 1987; Belt *et al.*, 1992). With predictions of rising stream temperatures in response to changing climate and increased incidence of fire, more focus will be directed towards manipulation and restoration of riparian vegetation to increase shade (Davies, 2010; Furniss *et al.*, 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 *et al.*, 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.

<u>Inputs of large wood for instream habitat complexity</u>: Over the last three decades, an 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 *et al.*, 2003a). Large wood strongly influences channel form in small streams, creating pools and waterfalls and affecting channel width and depth (Montgomery et al. 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, the creation and growth of gravel bars and channel and floodplain sedimentation (Montgomery *et al.*, 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 *et al.*, 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 the 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 *et al.*, 1990; Bragg, 2000; Benda *et al.*, 2003b; Reeves *et al.*, 2003). Large pulses of wood may originate from near channel sources following fire (Figure 24), windthrow, or insect infestations, or be transported from other portions of a watershed by debris torrents, avalanches, or landslides (Bilby and Bisson, 1998; Bragg, 2000; Benda *et al.*, 2003b). The relative importance of chronic LW inputs vs. episodic, disturbance-related inputs varies in time and space (Benda *et al.*, 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).

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.

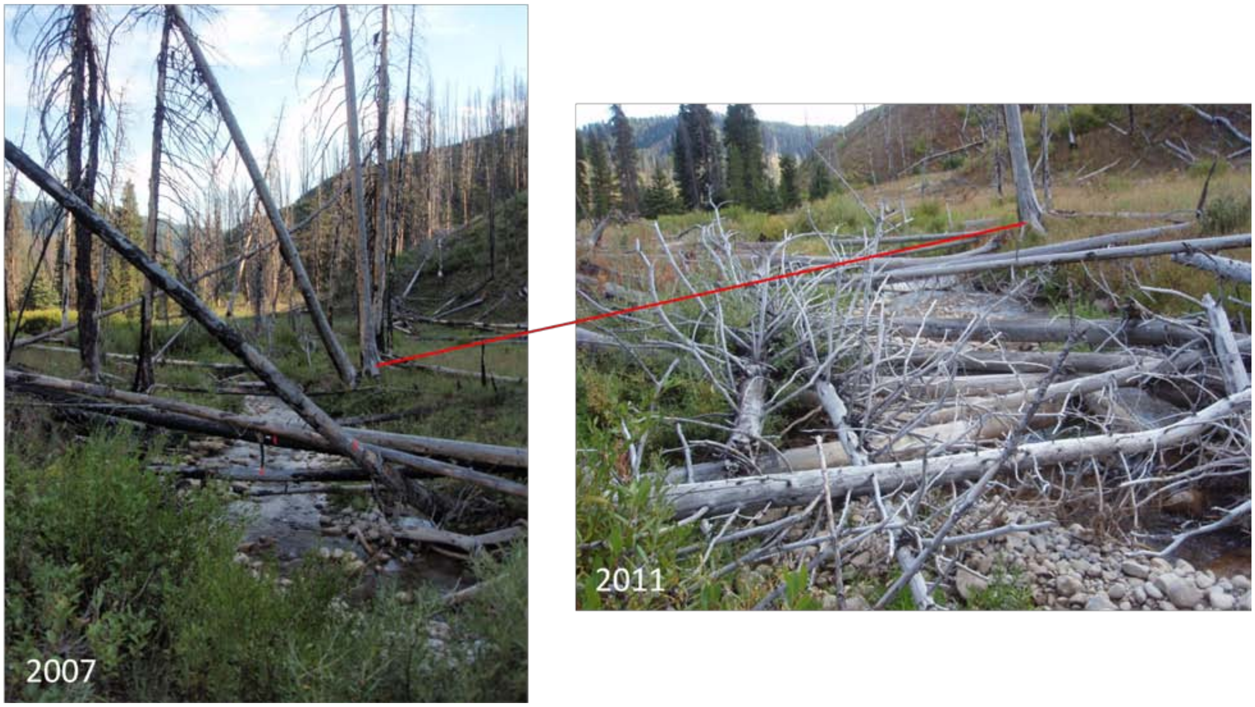


Figure 24b: Post-fire inputs of large wood (source = riparian) to Boulder Creek, Bridger-Teton National Forest, Wyoming. Photos were taken in 2007 (left) and 2011 (right), 7 and 11 years following the Boulder Fire (2000). In 2011, approximately 90% of the riparian 'recruitable wood' had either entered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

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 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 *et al.*, 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 *et al.*, 2000; Benda *et al.*, 2003b; Gregory *et al.*, 2003a; Gregory *et al.*, 2003b; Meleason *et al.*, 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 *et al.*, 2000; Gregory *et al.*, 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 *et al.*, 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 *et al.*, 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 instream wood.

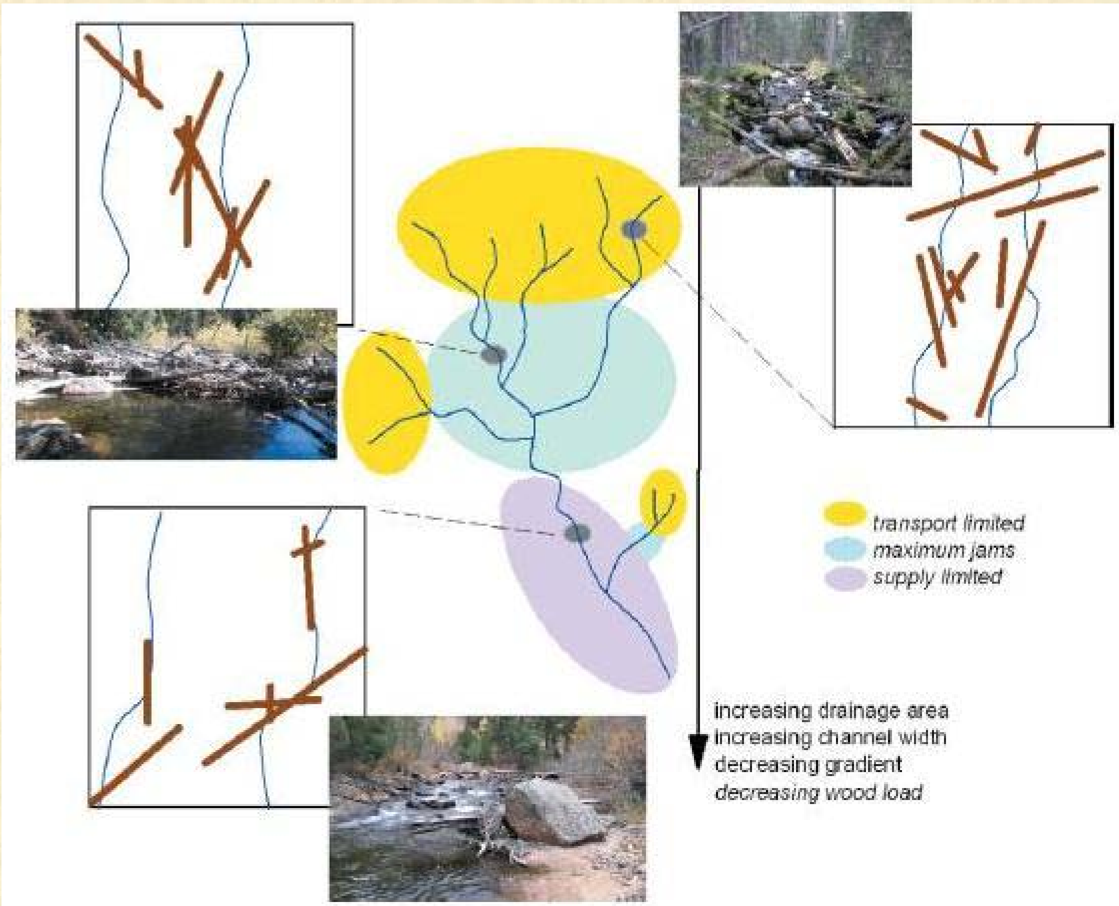


Figure T6-1. Conceptual model of large wood loads and spatial distribution along mountain streams. Although the model was generated from data collected in streams of the Colorado Front Range, the basic framework applies to most streams throughout the forested, mountainous West (reproduced without permission, Wohl and Jaeger, 2009).

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 *et al.*, 1980; Newbold *et al.*, 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 *et al.*, 1997); such inputs are highest from closed canopy riparian areas dominated by deciduous plant species (Edwards and Huryn, 1996; Nakano *et al.*, 1999; Baxter *et al.*, 2004; Baxter *et al.*, 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 *et al.*, 2003).

Streambank stability: Riparian vegetation can directly affect stream channel characteristics, particularly streambank habitat and stability (Gregory and A.M., 1988; Davies-Colley, 1997; Simon and Collison, 2002; Pollen *et al.*, 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 *et al.*, 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 *et al.*, 2000; Simon and Collison, 2002; Wynn *et al.*, 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, 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 1998). Floods can erode streambanks and undercut, topple and remove standing riparian vegetation. Entrained wood and debris can batter riparian trees (Johnson *et al.*, 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 *et al.*, 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 *et al.*, 1997;

Naiman *et al.*, 2005; Poff *et al.*, 2007; Merritt *et al.*, 2009). Many riparian plants are specifically adapted to flooding, as well as sediment deposition, physical abrasion, and stem breakage associated with flooding (Karrenberg *et al.*, 2002; Naiman *et al.*, 2005; Merritt *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 1998). Similar examples are common throughout the Western US and elsewhere worldwide. As the US 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 include a flood component and also address flow requirements for channel maintenance, in-channel habitat, and maintenance of water quality (Richter and Richter, 2000; Rathburn *et al.*, 2009).

Fire

Wildfire has played a critical role in shaping ecological heterogeneity across landscapes of the western USA (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 *et al.*, 2003; Skinner, 2003; Reeves *et al.*, 2006; Petitt 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 *et al.*, 2003; Dwire and Kauffman, 2003; Petitt and Naiman, 2007); recent work has advanced understanding of post-fire recovery in different settings (Mellon *et al.*, 2008; Jackson and Sullivan, 2009; Malison and Baxter, 2010).

Different scenarios of generalized fire behavior and effects in riparian areas have been proposed. Pettit 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 *et al.*, 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.

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 *et al.*, 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	

[1]including 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 et al. (2010)

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.

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 *et al.*, 2003; Skinner, 2003).

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 which 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 'wicks'. This fire behavior was observed during the Angora Fire, Tahoe National Forest, CA in late June 2007 (Murphy *et al.*, 2007a). 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 *et al.*, 2007a; 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 *et al.*, 2007a; Safford *et al.*, 2009).

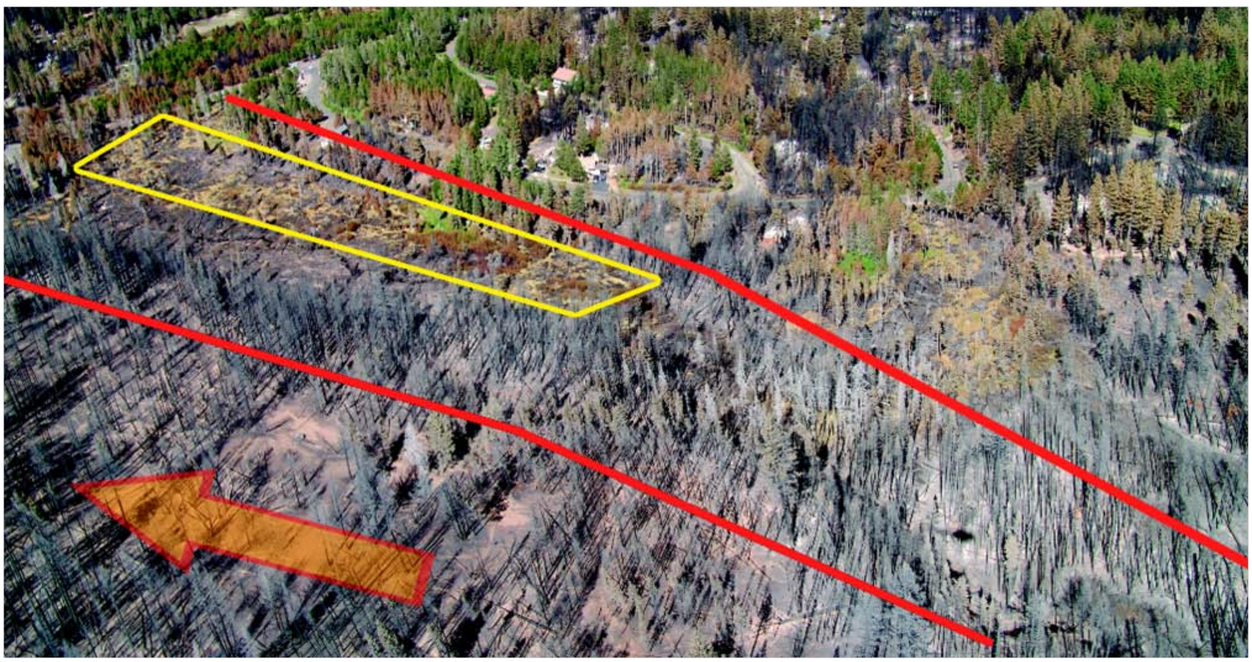


Figure 25: Stream Environment Zone (SEZ) along Angora Creek following the Angora Fire, Tahoe NF, California (2007). Dense, continuous stands of trees contributed to rapid spread rates (to the NNE) down this stream corridor. Arrow points in direction of wind and fastest fire spread (NNE). Note greater 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).

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 USA, where woody encroachment into riparian areas has increased streamside fuel loads.

<u>Riparian Areas Serve as Fuel Breaks</u>: 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 (CA), 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 *et al.*, 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 *et al.*, 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; Istanbulluoglu *et al.*, 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 *et al.*, 2003a). Although considered a major hazard in mountain regions worldwide (Coe *et al.*, 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 *et al.*, 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 *et al.*, 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 US, particularly in steep terrain. Numerous studies have documented increased frequency of debris flows following large-scale, severe fires (Swanson, 1981; Meyer *et al.*, 1992; Cannon *et al.*, 2001; Istanbulluoglu *et al.*, 2002; Istanbulluoglu *et al.*, 2003; Pierce *et al.*, 2004; Gabet and Bookter, 2008; Santi *et al.*, 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 *et al.*, 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 US, with a greater frequency and duration of droughts (Seager and al., 2007; Solomon *et al.*, 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 *et al.*, 2008; Bentz *et al.*, 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 USA 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 *et al.*, 1996).

Warming temperatures are predicted to dramatically affect insect outbreaks in forested areas (Bale *et al.*, 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 *et al.*, 2008; Bentz *et al.*, 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 al., 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 colleagues (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 *et al.*, 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 *et al.*, 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 US 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 *et al.*, 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 US. 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 *et al.*, 1998). The beaver is a considered a keystone riparian species due to its extensive influence on fluvial corridors (Pollock *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 1995; Naiman *et al.*, 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 attenuated water table decline in the drier summer months (Westbrook *et al.*, 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 *et al.*, 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 *et al.*, 1995; Naiman *et al.*, 1998). Beaver also affect plant community composition and the spatial-temporal dynamics of the vegetation through selective herbivory and foraging practices (Pollock *et al.*, 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 *et al.*, 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; 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 *et al.*, 2002; Naiman *et al.*, 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 *et al.*, 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 *et al.*, 1997; Merritt *et al.*, 2009). Land use practices, such as agriculture and urbanization, have added to the disruption of natural hydrologic regimes within stream networks (Poff *et al.*, 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 *et al.*, 2000; Rood *et al.*, 2007; Rieman *et al.*, 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 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).

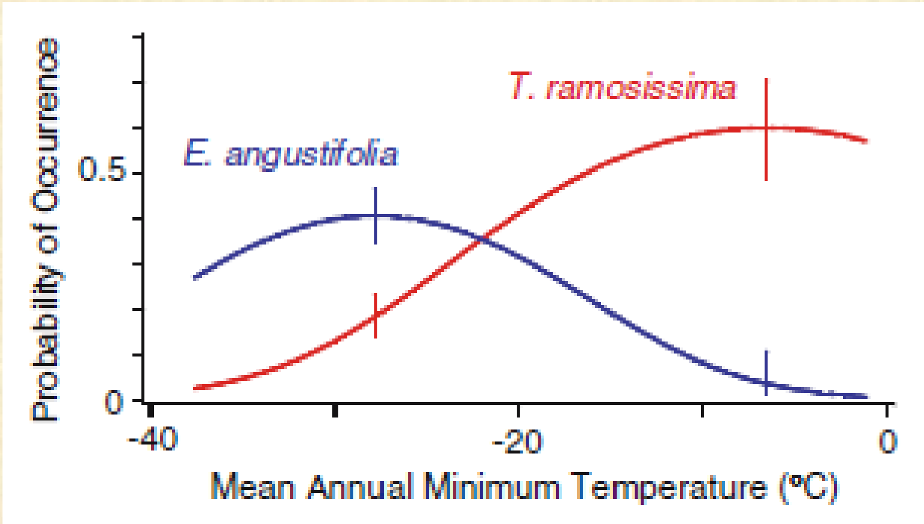


Figure T8-1. Gaussian logistic regression of occurrence of salt-cedar, *Tamarix ramosissima* and Russian olive, *Elaeagnus angustifolia* as a function of mean annual minimum temperature. Vegetation data were collected at 475 randomly selected stream gaging stations in 17 western states; temperatures were derived from weather station data recorded from 1961-1990 (reproduced without permission, Friedman *et al.*, 2005).

Throughout the western US, 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 *et al.*, 2006; Wolf *et al.*, 2007; Westbrook *et al.*, 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 *et al.*, 2007; Wilding and Poff, 2008; Poff *et al.*, 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 *et al.*, 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 *et al.*, 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 U.S. 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 attached compendium of "Advanced Topics on Fish Populations and Fire" for a richer and more thorough review of the general topic and several important aspects.

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 weter 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 western U.S. landscapes 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 *et al.*, 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 *et al.*, 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 *et al.*, 2008) with trends in streamflow timing (Stewart *et al.*, 2005), increasing variability in streamflows (Pagano and Garen, 2005; Luce and Holden, 2009), and warming streams (Isaak *et al.*, 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 *et al.*, 1989; Earl and Blinn, 2003; Spencer *et al.*, 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 *et al.*, 2003) almost certainly remove the fish that are present at the time.

The response of fish populations to these impacts are varied (see Rieman et al., 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 *et al.*, 2007). Although fine sediment increases are documented to interfere with life stages that use gravel interstices (Everest *et al.*, 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 *et al.*, 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 *et al.*, 1995; Benda *et al.*, 1998)(Figure 26). Inputs of nutrients released by fires may also provide at least a temporary boost in productivity (Spencer *et al.*, 2003; Malison and Baxter, 2010). Fires are one of many disturbances that regulate sunlight coming to streams, so 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.

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).

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 *et al.*, 2003; Moore *et al.*, 2010; Schindler *et al.*, 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 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 *et al.*, 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 et al. in Advanced Topics).

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 *et al.*, 2003; Peterson *et al.*, 2008a; Fausch *et al.*, 2009; Cook *et al.*, 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 km^2 (Dunham and Rieman, 1999) which equates to roughly 40 km of suitable habitat (Isaak *et al.*, 2010). A cartoon 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 River supported the notion that gene flow was much stronger at shorter distances especially under 20 km (Whiteley *et al.*, 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.

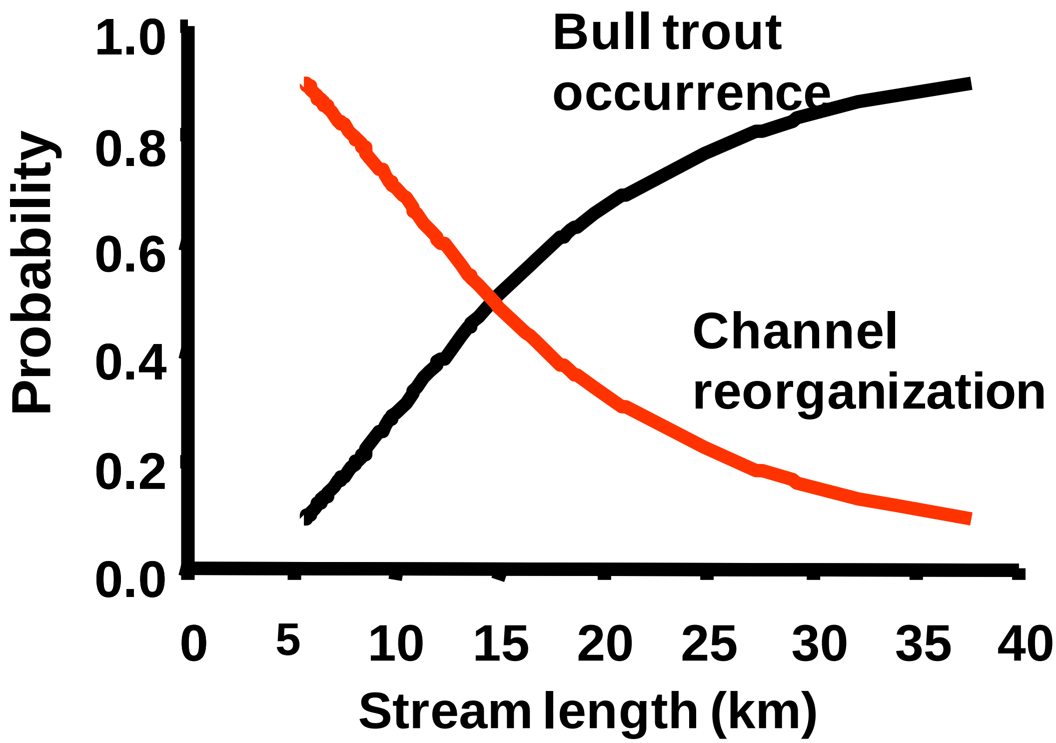


Figure 27: Probability of bull trout presence within a patch based on total stream length in the habitat (based on Dunham and Rieman, 1999) versus the probability that channel segments within a given distance of stream length might experience debris flows at the same time. Neither study directly measured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

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 *et al.*, 1997; Baxter *et al.*, 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 et al. in Advanced Topics). Barriers constructed to protect against non-native fish invasions can have similar consequences to road fragmentation (Peterson *et al.*, 2008b; Fausch *et al.*, 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. Istanbulluoglu *et al.*, 2004; O'Laughlin, 2005; Roloff *et al.*, 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 (Istanbulluoglu *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 2003; Giménez *et al.*, 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 *et al.*, 2003; Hessburg and Agee, 2003; Rieman *et al.*, 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 *et al.*, 2009). Many of the places with strongly altered fuels or fire regimes were directly affected by forest harvest and attendant road construction, thus seldom coincide with current habitats of sensitive, threatened, or endangered species (Rieman *et al.*, 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 *et al.*, 2010).

Discussions of fuel change issues and fish have focused largely on forests (e.g. Bisson *et al.*, 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 (*Artemesia tridentata*) communities by cheat grass (*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 *et al.*, 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 *et al.*, 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 *et al.*, 2011) or high temperatures.

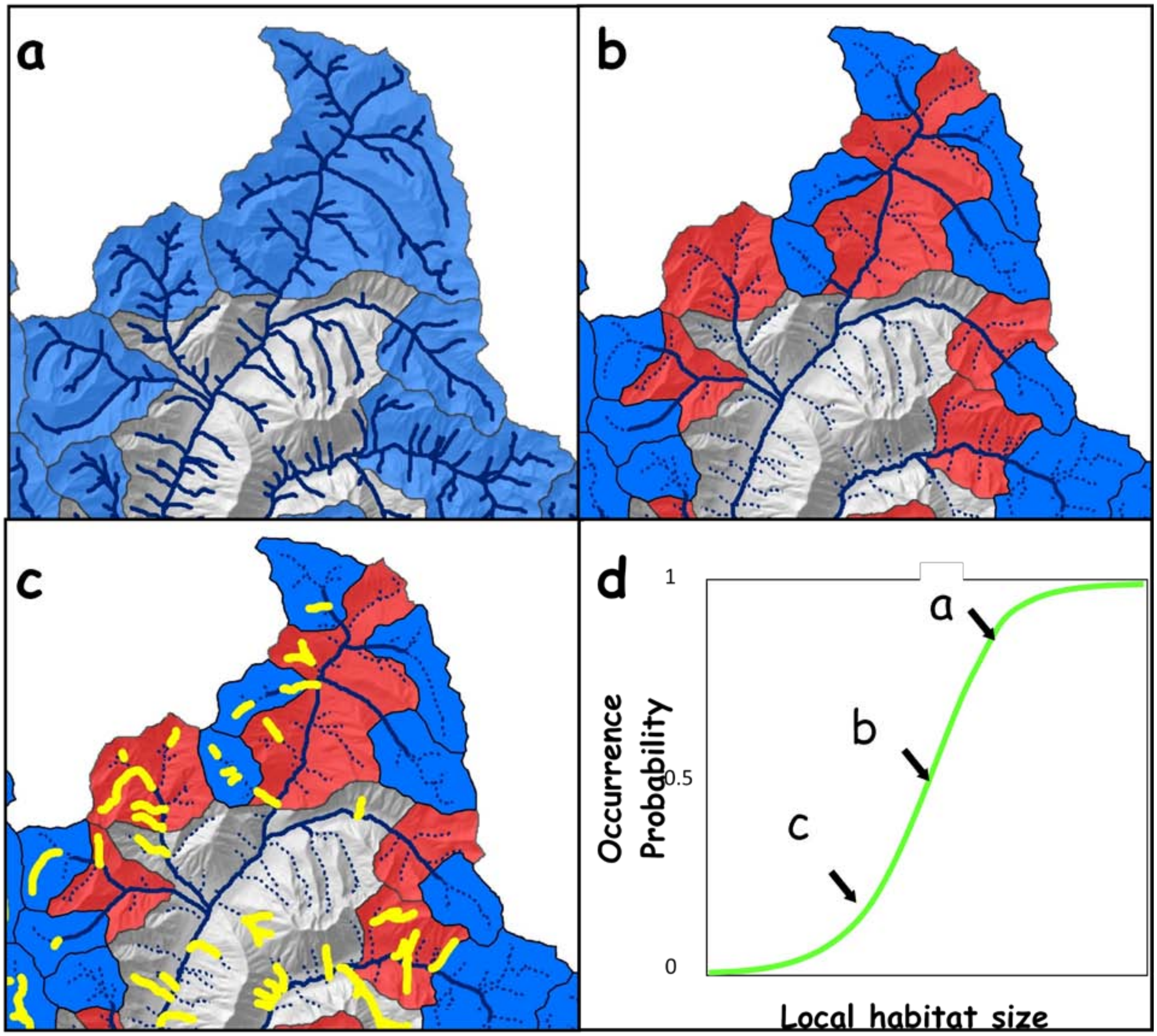


Figure 28: Decline in patch size and consequent probability of being occupied starting from (a) current conditions with (b) decreases due to warming temperatures [red areas] and (c) low flow decreases [dashed lines] and debris flow risks [yellow lines].

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 *et al.*, 2007; Wenger *et al.*, 2011a; Wenger *et al.*, 2011b)(Figures 29 & 30). Contrasting and complementary 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.

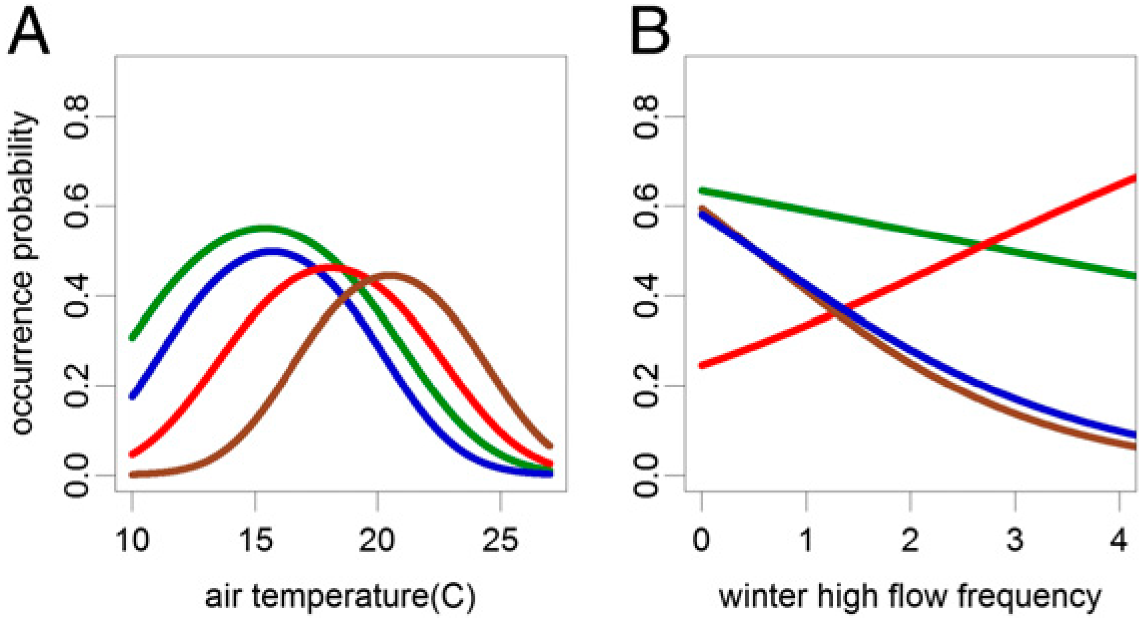


Figure 29: Comparative sensitivity of four trout species to stream temperature A and the frequency of winter high flows B. Green-cuthroat trout; blue-brook trout; red-rainbow trout; brown-brown trout. (from Wenger *et al.*, 2011b)

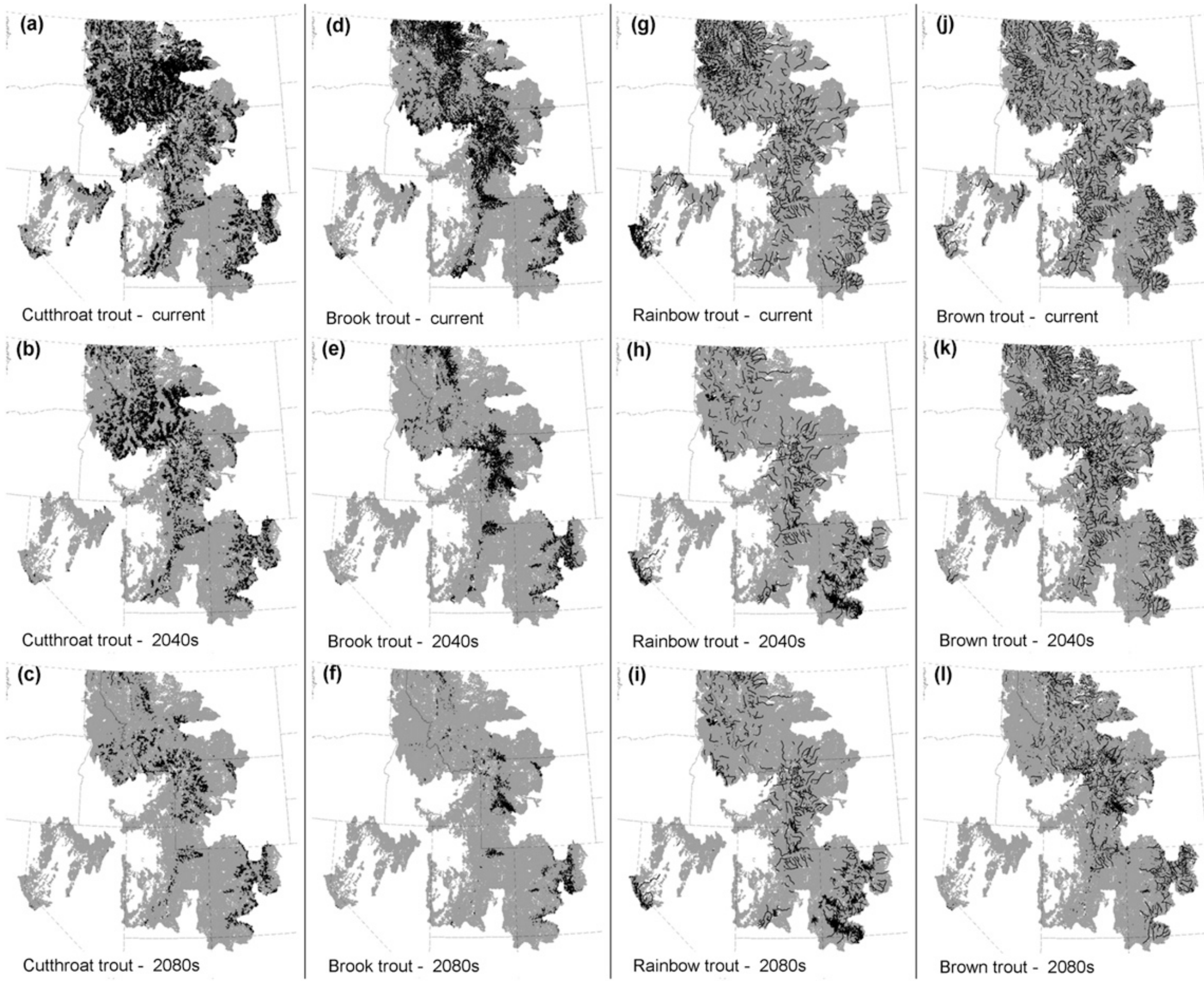


Figure 30: Modeled trout distributions under present and future (A1B) climate change scenarios. Gray streams are relatively unsuitable and black streams are relatively suitable (from Wenger *et al.*, 2011b)

Hydrologic changes will likely precipitate changes in water management (Barnett *et al.*, 2008), which in turn will have consequences for many aquatic systems. In the southwest, where water withdrawals are a common challenge for fish (Rieman *et al.*, 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 *et al.*, 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 US.

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 resilience of fish populations to fires is strongly influenced by diversity in life histories and the extent of habitat networks needed to support persistent populations can help to focus the discussion. 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.

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 *et al.*, 2008b; Fausch *et al.*, 2009). Under certain circumstances climate change may reduce threats of invasion by some species (Wenger *et al.*, 2011a; Wenger *et al.*, 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 *et al.*, 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 wildfire. 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 *et al.*, 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 *et al.*, 2010; Williams *et al.*, 2011), and See Rosenberger et al. in Advanced Topics.

Part III: Management Actions and Decisions

Land, wildlife, and water managers have heard the call for increasing resilience of ecosystems (Walker and Salt, 2006; Millar *et al.*, 2007; Baron *et al.*, 2009; Heller and Zavaleta, 2009; Joyce *et al.*, 2009; Keane *et al.*, 2009; Palmer *et al.*, 2009; West *et al.*, 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, however advances in understanding of aquatic ecosystem response to fire supports new ideas in framing key decisions (Bisson *et al.*, 2003; Rieman *et al.*, 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 *et al.*, 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 *et al.*, 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. However, in a broader sense there are related activities applied to mined, overgrazed, or logged sites. We distinguish these activities from post-fire stabilization (see below), however, which is focused on preventing (resisting) losses to populations whereas 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 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 *et al.*, 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 *et al.*, 2005).

Many forest and stream ecosystems in the western U.S. 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 *et al.*, 2003; Keane *et al.*, 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 et al., 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 nor 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 *et al.*, 2005). Road decommissioning assessments generally show substantial improvement, however remediation of steep slopes is problematic, and it is difficult to entirely erase road impacts (Luce, 1997; Madej, 2001; Switalski *et al.*, 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 *et al.*, 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 *et al.*, 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 before-hand, 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 *et al.*, 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 western US. 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 2008b).

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 (Monsen and Shaw, 2001; e.g. Shaw *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 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 *et al.*, 1999; Dellasalla, 2004; Rieman *et al.*, 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 *et al.*, 2000; Dellasalla, 2004; Rieman *et al.*, 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 *et al.*, 2001; Meyer and Pierce, 2003; Whitlock *et al.*, 2003; Pierce *et al.*, 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 (US 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 not only the high technical complexity of designing landscape-scale solutions to persistence in a dynamic environment, but huge challenges in convincing a diverse public that it is all in the 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 *et al.*, 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 because, a strategy relying on fire suppression and post-fire stabilization alone conceptually violates the first, third, and fourth principles because it allows fuel buildups that reduce forest resilience, it imposes a static conceptualization for the forest and aquatic habitats, and it requires substantial resources. We repeat it here for emphasis. We also note 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 [National Wildfire Coordinating Group], 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 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 an 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 et al. (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 *et al.*, 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, or upgraded to be resilient to storms if needed in the future, and values warrant the cost. 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.

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, 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.

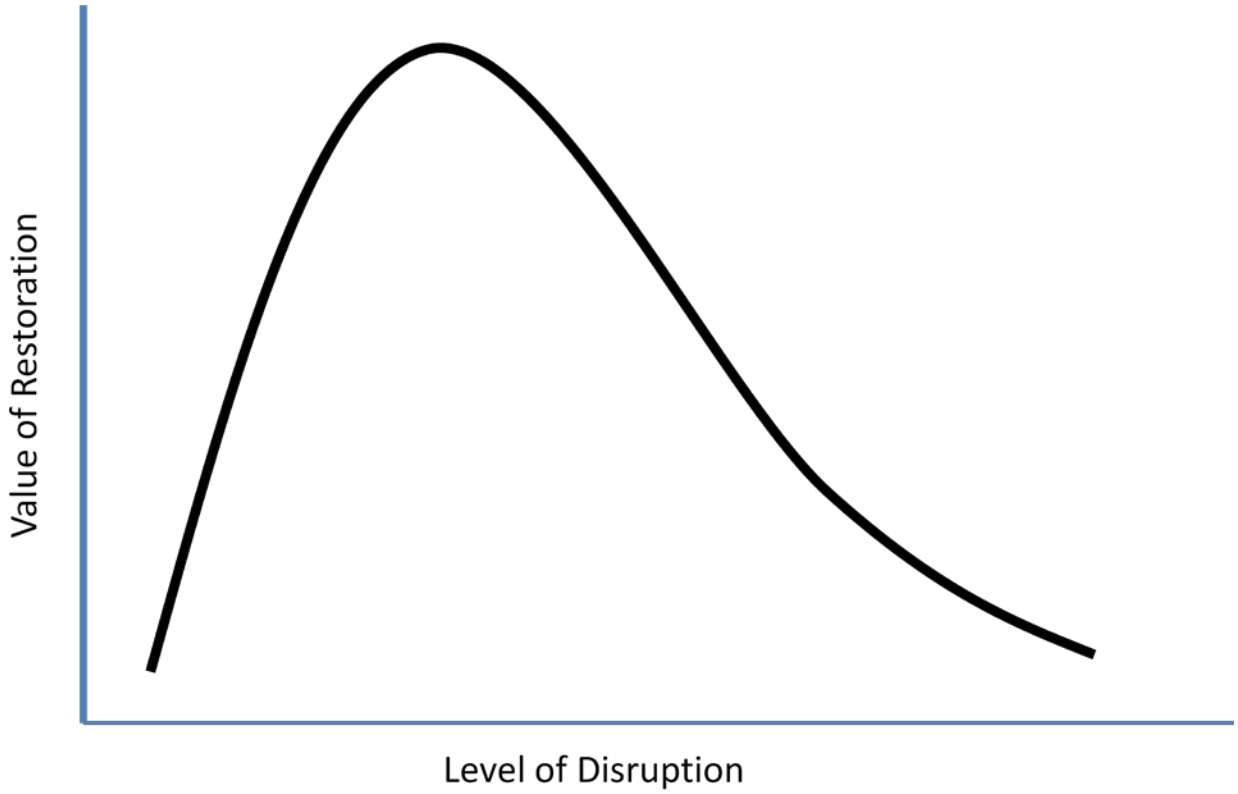


Figure 31: Most benefits from restoration are likely to be found from efforts in places only mildly or moderately degraded.

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 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 *et al.*, 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 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 groundwater 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 *et al.*, 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 *et al.*, 2004; O’Laughlin, 2005; Roloff *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 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, sediment effects within a given habitat patch along with 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 *et al.*, 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 lacking information is 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 *et al.*, 2003). One key uncertainty is the scale of habitat patches required for persistence in time (Rieman *et al.*, 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 *et al.*, 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 *et al.*, 2007; van Wagtendonk 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 *et al.*, 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 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 U.S. for example, there is substantial uncertainty with respect to changes in orographic precipitation from mountains (Dettinger *et al.*, 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 *et al.*, 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 *et al.*, 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 U.S. (e.g. Westerling *et al.*, 2006; Morgan *et al.*, 2008; Dillon *et al.*, 2011; Holden *et al.*, 2011b). 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 *et al.*, 2003) or homogenization of forests (Hessburg *et al.*, 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 et al., (2007) and others 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 et al. (2007) the nominal ideas presented were applied to helping make transitions to new habitats successful, however all of these steps could be 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 ecosystem.

Climatic changes can be viewed simplistically as shifting suitable habitat ranges higher in elevation and further 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 *et al.*, 2008; Luce and Holden, 2009; Leppi *et al.*, 2011). Even species that are less temperature sensitive may have reduced net productivity and fewer migratory individuals if temperatures warm (Dunham *et al.*, 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 *et al.*, 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 *et al.*, 1998; Baxter and Hauer, 2000; Hari *et al.*, 2006; Lundquist and Cayan, 2007; Tague and Grant, 2009; Millar and Westfall, 2010; Holden *et al.*, 2011a). 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 et al. in Advanced Topics). 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 *et al.*, 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 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 further 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. serrotinous 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 *et al.*, 1995a; Mann *et al.*, 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 *et al.*, 2009; Wenger *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 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, 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 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 *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 2005; Ford *et al.*, 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 km^2, 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 *et al.*, 1986). Within such optimization techniques however, are assumptions about certainty of outcomes. Some have promoted **robust** decision making (Regan *et al.*, 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, 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 apriori, and decisions must be made as events unfold, a task described as **dynamic decision making** (Brehmer, 1990).

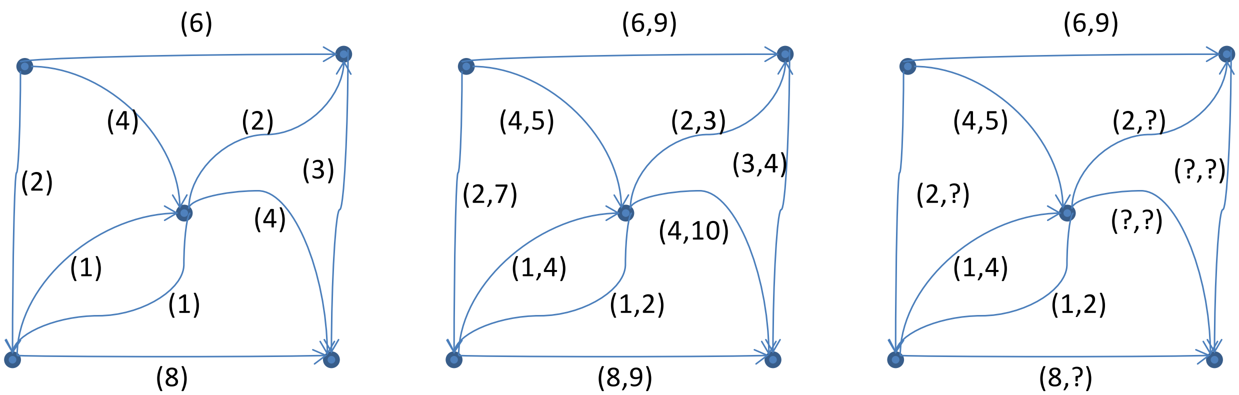


Figure 32: Conceptual path length diagrams for a) classic optimization, b) robust optimization, and c) dynamic decision making. In a) path lengths are given with uncertainty, in b) and c) path lengths are given across a range, however some of the path characteristics are unknown in c).

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 *et al.*, 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 *et al.*, 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 *et al.*, 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 *et al.*, 2003; Rieman *et al.*, 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 *et al.*, 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 *et al.*, 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, would ultimately be of benefit. Spring time information about fine spatial scale snowpack and soil moisture information can also help make planned fuel treatments more successful. 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 *et al.*, 1999; Hansen *et al.*, 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 *et al.*, 2003; Hessburg and Agee, 2003; Rieman *et al.*, 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 *et al.*, 2009). These uncertainties also have substantial impact on future fire occurrence and size (Holden *et al.*, 2011b). 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 Nation 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; that 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.

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 fire fighting community may provide a good example of how to improve other land management approaches in a dynamic and partially unpredictable future.

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, 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 U.S., 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 *et al.*, 2011a; Wenger *et al.*, 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). 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 may benefit from an increasing capacity for remote sensing of stream and riparian characteristics (see e.g. McKean *et al.*, 2008; McKean *et al.*, 2009 for geomorphic information; Isaak *et al.*, 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 *et al.*, 2010; Rieman and Isaak, 2010). Unfortunately water, riparian, and aquatic biota are not 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 such information for more punctual AND better decisions.

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. 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 their bottom line. Some examples may include:

- 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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