# Section 6—Water Resources and Aquatic Ecosystems

Water resources and aquatic ecosystems in the Sierra Nevada and southern Cascade Range support critical ecological and socioeconomic values both within and well beyond the region. This section contains four chapters on different types of water-based ecosystems in the synthesis area. Chapter 6.1, "Watershed and Stream Ecosystems," considers challenges and threats facing those systems, including climate change and wildfire, before turning to recent research on water quantity and quality, including how macroinvertebrates serve as indicators of water quality. Chapter 6.2, "Forested Riparian Areas," focuses on the ecologically important transition zones between upland forests and streams. It discusses current understanding of the role of fire in riparian ecosystems, as well as findings about opportunities for management to restore those areas. Wet meadows, the subject of chapter 6.3, have been a special focus of restoration efforts and research in the synthesis area and in other regions. Chapter 6.4, "Lakes," discusses recent research and restoration strategies for high-elevation lake ecosystems; it examines a

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Pope Marsh in South Lake Tahoe represents a rare wetland habitat adjoining the largest natural lake in the synthesis area.

Jim Frazier

Perennial streams such as Reynolds Creek on the Stanislaus National Forest and associated riparian areas are an important focus for promoting socioecological resilience in the synthesis area

multitude of stressors, including climate change, pollution, introduced fishes, and diseases. Although these different kinds of systems are related through the flow of water, they have distinct ecological issues and management challenges. Taken together, these chapters feature strategies to promote resilience that complement the broader themes of the synthesis, including an emphasis on promoting or emulating natural disturbance regimes, considering the larger spatial and temporal contexts of these systems, and understanding linkages between ecological processes and social values. As water travels, it integrates landscape influences, so that downstream waterbodies and their aquatic organisms reflect the condition of terrestrial and aerial environments. Accordingly, these chapters emphasize the connections between aquatic ecosystems and other forest components that are discussed in chapters 2.0, “Forest Ecology;” 4.1, “Fire and Fuels;” 4.2, “Fire and Tribal Cultural Resources;” 4.3, “Post-Wildfire Management;” 5.0, “Soils;” and 8.0, “Air Quality.”

# Chapter 6.1—Watershed and Stream Ecosystems

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## Summary

Water and aquatic ecosystems in the synthesis area have high social, cultural, and ecological values. National forests in the synthesis area are a major source of water supply, hydropower, and recreational activity for much of California. Recent research has provided more information on water and nutrient budgets; these data are fundamental to understanding the interaction of terrestrial and aquatic ecosystems, current forest conditions, and response to stressors. Rapid changes in climate pose a threat to water resources, as warming has produced a shift toward more precipitation falling as rain than snow, which reduces snowpack water storage, causes earlier runoff, increases the frequency of major floods through rain-on-snow events, and diminishes late-season flows and the stability of headwater habitats. Because climate change is expected to increase impacts from storms and wildfires, flooding and sediment movement may increase, which could in turn reduce channel stability and habitat quality. Hydrologic response to climate change is expected to be different for the northern, central, and southern Sierra Nevada. Recent research has noted that natural disturbances such as fires and floods and associated erosion can be important for maintaining stream functions and biodiversity; however, systems that are already degraded or have limited connectivity for aquatic life may be vulnerable to losses following disturbances. Significant increases in sedimentation rates may negatively affect sensitive aquatic organisms and reservoirs. Therefore, efforts to promote a fire regime that results in fewer uncharacteristically large and severe wildfires can help maintain resilience of aquatic systems.

Forest restoration treatments may promote resilience to drought, wildfire, insects, and disease, and they could increase water available to soils, groundwater, and streams owing to reduced transpiration and increased snowpack. However, forest treatments that are not designed primarily to increase water yield may not remove sufficient trees to result in an easily measured and sustained increase in water. Consequently, evaluations of the water benefits of general restoration treatments will depend on the combination of long-term experimental studies and modeling, which have been initiated in the past decade. Aquatic systems that have not

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been degraded by historical land management may demonstrate relatively greater declines in water quality because of warming and deposition of nitrogen, so it is important to include them in a long-term monitoring network. It is also important to have data on physical, chemical, and biological indicators of water quality and quantity to detect both change and the causes of change. Stream invertebrates are good indicators of small-stream conditions where many forest management activities take place. Additional management strategies to promote resilience of aquatic ecosystems to stressors include restoration of natural processes, including fire regimes, flow regimes, hyporheic exchange, lateral channel migration, and habitat connectivity.

## Introduction

All life depends on an adequate supply of water. The national forests play an important role in water protection, stemming from the Organic Administration Act of 1897, which asserted that one of the primary reasons for establishing forest reserves was to "secure favorable conditions of water flows." How much and where water occurs is a direct function of climate and weather patterns. Soils, topography, and vegetation affect how water is partitioned in the landscape, and these factors, along with human activities, air quality, and ecosystem disturbances, affect the quality of water. Measurements of physical, chemical, and biological characteristics serve to characterize the condition or health of water resources and aquatic ecosystems. Monitoring environmental attributes at different scales or doing research, especially with designed experiments, can help to gain knowledge about effects of land use practices. The Adaptive Management Plan, Appendix E, in the Sierra Nevada Framework (USDA FS 2001, 2004) described the need for status and trends monitoring and research; it also identified priority questions and knowledge gaps that required new information to improve Forest Service management of water resources and aquatic ecosystems. Since that time, most Forest Service efforts toward aquatic resources in the Sierra Nevada have been directed to studies of amphibians, grazing practices, and invertebrates. More recent attention has been given to meadow restoration. A long-term watershed research project in the Sierra Nevada was established by the Pacific Southwest Research Station in 2000 at the Kings River Experimental Watersheds (KREW), which includes a portion of the Teakettle Experimental Forest. This research site has attracted National Science Foundation funding for the establishment of the Southern Sierra Critical Zone Observatory (Lin et al. 2011),[4] which is starting to provide new information on hydrology and geosciences in the Sierra Nevada. Older watershed research sites

[4] Southern Sierra Critical Zone Observatory, http://criticalzone.org/sierra.

with long investments exist at University of California's Blodgett Forest Research Station and Sagehen Experimental Forest (see fig. 1 in chapter 1.5).

This chapter begins with a review of values and services associated with aquatic ecosystems. It then considers climate change and wildfire before turning to recent science on water quantity, water quality, and macroinvertebrates as indicators of water quality. This chapter concludes with a discussion of management strategies to promote resilience of aquatic ecosystems. A challenge for the future is to better integrate water and aquatic ecosystems into forest planning, which typically has focused on tree structure and composition, fire, and a few wildlife species in the synthesis region (e.g., North 2012, North et al. 2009).

## Values and Services of Aquatic Ecosystems

> Water, in all its forms, is indeed the crowning glory of the Sierra. Whether in motion or at rest, the waters of the Sierra are a constant joy to the beholder. …Above all, they are the Sierra's greatest contribution to human welfare.
>
> —*History of the Sierra Nevada*, F.P. Farquhar (1965: 1)

National forests supply 45 percent of California's water, and most of the state's surface water originates in the Sierra Nevada. In the Pacific Southwest Region, one of the most valuable ecosystem services that national forests provide is an adequate supply of good water for aquatic ecosystems and human needs. California's economy is highly dependent on agriculture, and much of the country relies on fresh fruits and vegetables produced in California. Water is therefore pivotal to California's economy and the Nation's food supply. Furthermore, human recreation is highly influenced by the presence of water bodies.

Streams, riparian areas, and wet meadows support a wide range of social, cultural, and ecological values, including plant and wildlife diversity, water quality, water quantity, cultural values, aesthetic values, sport fishing, and tourism. Native American cultural resources are often concentrated along perennial streams owing to availability of water and culturally important plants, travel corridors, and other patterns that facilitated settlement (Jackson 1988). Activities such as timber harvest, recreation, and livestock grazing (see chapter 9.5, "Managing Forest Products for Community Benefit") can affect the condition of riparian and meadow areas. Such areas can sustain a diverse array of ecosystem services and ameliorate effects of climate change (see also chapter 6.3, "Wet Meadows").

The value of riparian ecosystems (termed aquatic-terrestrial ecotones in the international literature) has been discussed extensively (Holland et al. 1991, Malanson 1993, Petts 1990, Pinay et al. 1990). Historically, riparian ecosystems were valued for their economic uses: transportation corridors, water supply and

electricity, construction materials and waste disposal, agriculture and livestock, and settlement. The more recently recognized economic, social, and biological values of riparian ecosystems are listed below. Luce et al. (2012) identified valued functions of riparian plant communities for native fishes: provision of shade for thermal modification of stream temperature, inputs of large wood for in-stream habitat complexity, organic matter inputs to aquatic food webs, and provision of streamside habitat and stabilization of streambanks.

**Values of riparian ecosystems from the referenced literature**

Economic:

- Reduce downstream flooding
- Recharge aquifers
- Surface water supply in arid regions
- Support secondary productivity, e.g., for fisheries
- High yields of timber

Social:

- Recycle nutrients
- Store heavy metals and toxins
- Filter of diffuse pollution from uplands
- Accumulate organic matter as a sink for carbon dioxide
- Intermediate storage for sediments
- Natural heritage
- Recreation
- Aesthetics
- Natural laboratories for teaching and research

Biological:

- Special habitat for some endangered or threatened species
- Habitat for aquatic species

Riparian areas are unique environments because of their position in the landscape; they are both ecotones between the terrestrial and aquatic zones, and corridors across regions (Malanson 1993). The term ecotone was first used in 1905 by Clements; with the development of the discipline of landscape ecology, there was a renewed interest in the ecotone concept in Europe around 1990, and it was explored by the Man and Biosphere program (Holland et al. 1991, Naiman and Decamps 1990). An ecotone was then defined as "a zone of transition between adjacent ecological systems, having a set of characteristics uniquely defined by space and time scales and by the strength of interactions between adjacent ecological systems" (Naiman and Decamps 1990: 3).

## Resilience and Degradation in Stream Ecosystems

Ecological resilience has been defined as the amount of disturbance an ecosystem can absorb without crossing a threshold to a different stable state, where a different range of variation of ecological processes and structures reigns (Gunderson 2000) (see also chapter 1.1). This general idea is also reflected in the concept of dynamic equilibrium, which Heede (1980) described as the capacity of streams to adjust to perturbations within a few years. These concepts depend on the timeframe being considered and the range of variation in processes and structures. For example, understanding thresholds of erosion beyond which long-term sustainability is jeopardized requires extensive monitoring and understanding of reference conditions and natural variability. Developing site-specific restoration and management strategies therefore requires consideration of the evolutionary history of particular sites (Miller et al. 2001).

Some scientists have challenged the concept of dynamic equilibrium by arguing that many fluvial systems are inherently unstable (Lave 2009). Fluvial systems in Mediterranean climates in particular have been characterized as highly variable and ever-changing (Kondolf et al. 2012). Reflecting this view, scientists in recent years have challenged efforts that emphasize promoting channel stability, and they cautioned that management and restoration approaches are often overprotective in seeking to avoid disturbances and erosion. They pointed out that channel instability may have desirable consequences; for example, erosion and deposition following major disturbances, such as wildfires, can be important for maintaining stream functions and biodiversity (Bisson et al. 2003). Florsheim et al. (2008) outlined the various benefits of streambank erosion for maintaining aquatic habitat diversity and reiterated that total elimination of bank erosion should not be a goal when restoring rivers.

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## Climate Change Effects on Watersheds and Stream Ecosystems

### Effects on Hydrology

Anticipating that a changing climate in California will substantially affect water resources and aquatic ecosystems, strategies for assessing the impacts of altered stream flows need to be developed. Changes in the Sierra Nevada, the primary source area of water in the state, are of particular concern. Warming has produced a shift toward more precipitation falling as rain than snow, and this reduces snowpack water storage, causes earlier runoff, increases the frequency of major floods through rain-on-snow events, and diminishes late-season flows and the stability of headwater habitats that are important for maintaining watershed hydrological and

ecological function (fig. 1). The California Department of Water Resources (DWR 2006) identified some other potential effects of climate change in California on water resources, including changes in vegetation, increased incidence of wildfires, increased water temperatures, and changes in human water demand.

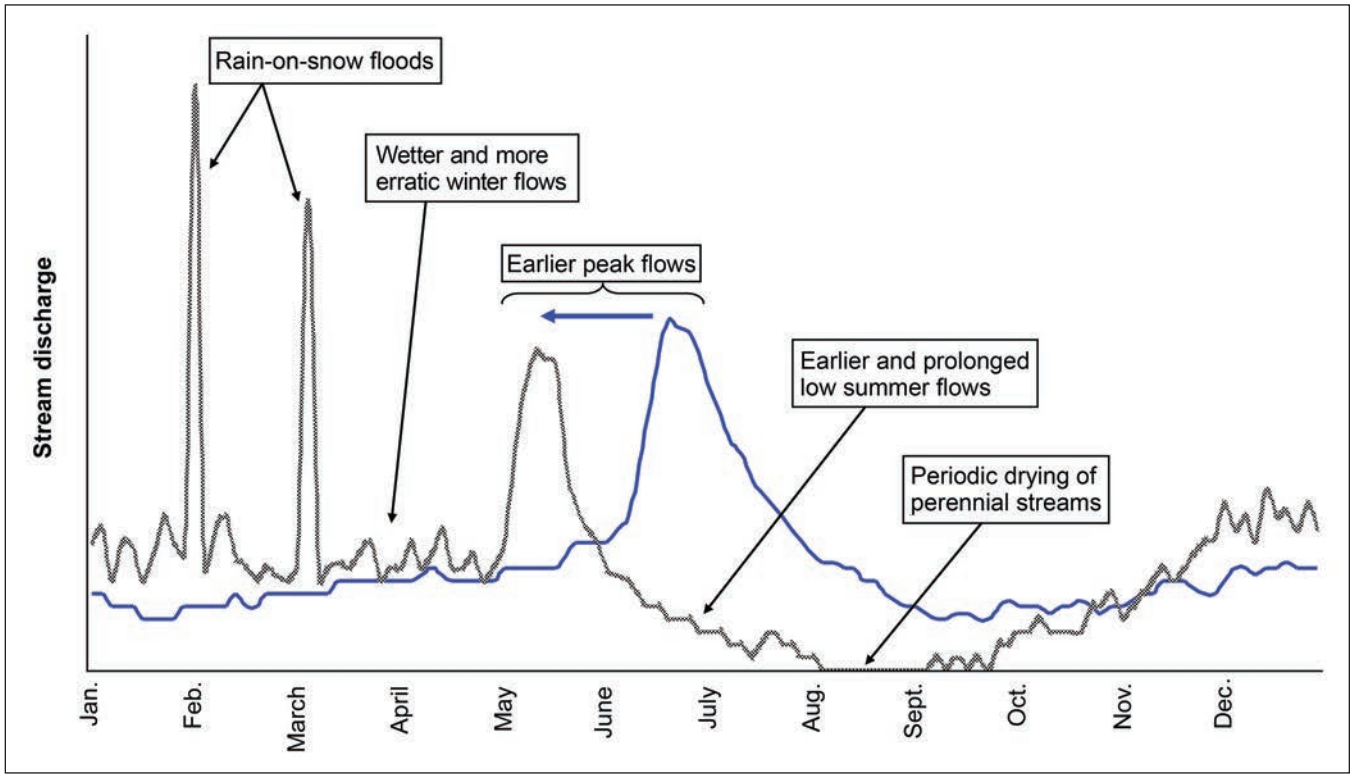


Figure 1—Conceptualization of the climate-driven changes (gray line) to the natural hydrograph (blue line) of a Sierra Nevada stream.

The water resources of the Western United States depend heavily on snowpack to store part of the winter precipitation into the drier summer months. A well-documented shift toward earlier runoff in recent decades has been attributed to more precipitation falling as rain instead of snow and earlier snowmelt (Knowles et al. 2006). The starting date of snowmelt is earlier now by about 15 days, based on data from 1960 to 2000 (fig. 2). A decline in the mountain snowpack of western North America has also been documented (Barnett et al. 2008, Mote et al. 2005). The California Department of Water Resources report (DWR 2006) showed how hydrologic patterns by river basin have already changed in California during the past 100 years. There is a slight decreasing trend in precipitation in central and southern California and increased variability in precipitation. There is also a difference between changes in northern and southern California. For example, the total annual water year runoff has been increasing for the Sacramento River basins (northern

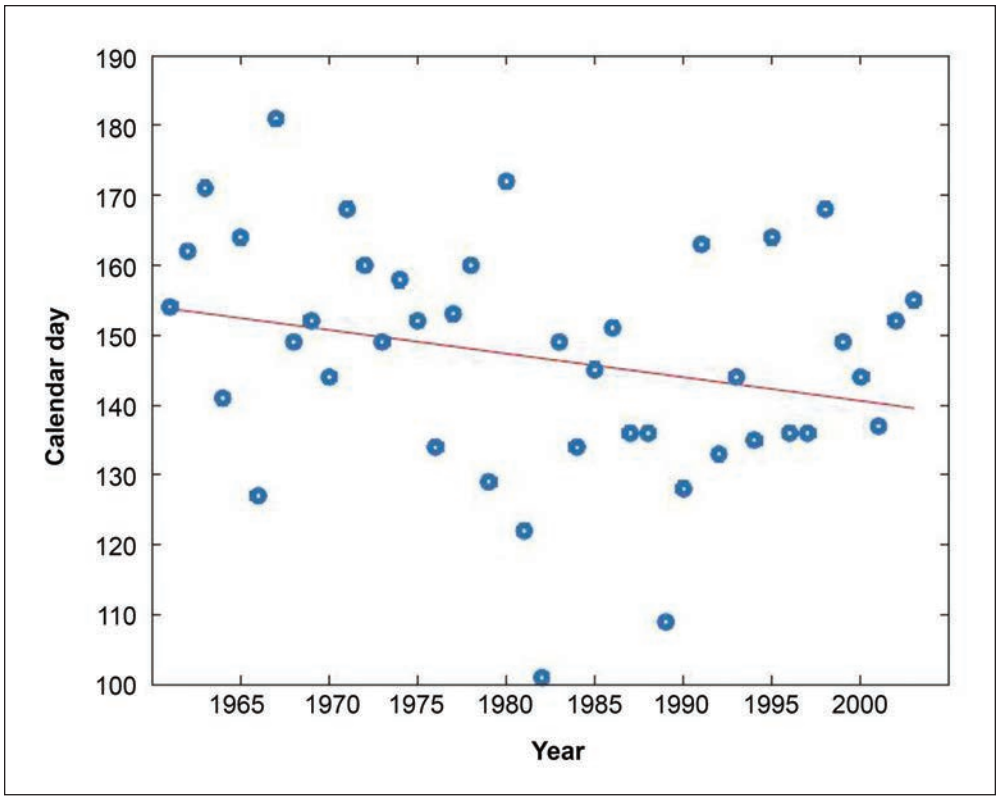


Figure 2—Trend in the timing of snowmelt discharge based on the day of maximum daily discharge, Kern River, California. Figure is from Peterson et al. 2008.

and central Sierra Nevada) and decreasing for the San Joaquin River basins (central and southern Sierra Nevada). However, both areas experienced decreases in spring runoff; runoff from April through July declined by 9 percent for the Sacramento River basins and declined by 7 percent for the San Joaquin River basins.

Snowpack provides 20 percent of California's total runoff and 35 percent of its usable surface water. Climate modeling predicts a loss of snowpack for the mountains in California (table 1), with a greater effect in the northern Sierra Nevada, where the mountains are lower in elevation than in the southern Sierra (Knowles and Cayan 2002). A change in surface water quantity of this magnitude will affect aquatic ecosystems and human uses.

**Table 1—Temperature increase and effect on Sierra Nevada snowpack**

| Temperature increase | Loss of snowpack | Predicted year of effect |
|---|---|---|
| *°C* | *Percent* | |
| 0.6 | 5 | 2030 |
| 1.6 | 33 | 2060 |
| 2.1 | 50 | 2090 |
| | (43 in south and 66 in north) | |

Source: Knowles 2002.

In the past 10 years, many publications and modeling efforts have focused on predicting climate change effects on critical ecological variables, including air temperature, amount and timing of precipitation and stream discharge, and soil moisture (Dettinger 2005, Hayhoe et al. 2004, Jeton et al. 1996, Knowles et al. 2006, Maurer 2007, Stewart et al. 2004, Vicuna et al. 2008). Some of these findings for the synthesis area are briefly discussed to highlight their importance for watershed and aquatic ecosystem condition.

Global climate models use a grid that is too coarse to adequately depict the complex structure of temperature and precipitation in California, especially within the rugged terrain of the synthesis area. A statistical technique allows coarse data to be "downscaled" to a finer level of detail, and a grid scale of 12 km (7 mi) was available by 2006 (Cayan et al. 2006). More recent work is downscaling data to even finer grids that allow predictions on possible changes to other attributes, such as stream discharge, water quality, and erosion. For example, Ficklin et al. (2012) developed and applied a hydroclimatological stream temperature model within the Soil and Water Assessment Tool (SWAT) to mountain areas of the Western United States. These scientists are also working on projections of (1) future hydrologic flow components for the major river basins of the Sierra Nevada using an ensemble of general circulation models, and (2) the effects of climate change on water quality (stream temperature, dissolved oxygen concentration, and sediment concentration) in the Sierra Nevada. The U.S. Geological Survey in Sacramento has an ongoing study of the effects of climate on snowmelt and water availability in the southern Sierra Nevada (Tuolumne, Merced, San Joaquin, King, and Kaweah River basins). These new downscaling efforts and predictions at the river-basin scale and for smaller watersheds will be useful to forest managers in considering climate change effects on water resources.

Null et al. (2010) used WEAP21 (Stockholm Environment Institute's Water Evaluation and Planning System), a weekly one-dimensional rainfall-runoff model, to compare the hydrologic responses of 15 watersheds on the west slope of the Sierra Nevada to air temperature increases of 2, 4, and 6 °C while keeping other climatic variables unchanged. Predicted changes in mean annual flow were largely driven by area and increased evapotranspiration from climate warming, while snowfall and snowmelt timing resulted in runoff timing changes. Predicted changes in low flow duration were a function of deep soil moisture capacity and infiltration. Null et al. (2010) found that vulnerabilities varied from north-to-south within the Sierra Nevada. Northern watersheds were sensitive to decreased mean annual flow owing to their extensive use for water storage; central watersheds were sensitive to length of low-flow conditions owing to their large areas of mountain meadows;

south-central watersheds were sensitive to changes in runoff timing owing to their importance for hydropower; and the southernmost watershed, the Kern, appeared to be the most resilient (table 2). For the 15 watersheds included in the study, total water storage is approximately 24 590 mcm[5] for all dams greater than 1.2 mcm, and total online hydropower capacity is approximately 8,751 MW. Null et al. (2010) state that their results are broadly consistent with other climate forecasts (Brekke at al. 2004, Lettenmaier and Gan 1990, Miller et al. 2003).

Bales et al. (2006) identified three pressing hydrologic information needs for the western mountains of the United States given climate change, population growth, and land use change:

- To better understand the processes controlling the partitioning of energy and water fluxes within and out from these systems.
- To better understand feedbacks between hydrological fluxes and biogeochemical and ecological processes.
- To enhance our physical and empirical understanding with integrated measurement strategies and information systems.

[5] Million cubic meters. 1 mcm = 810.7 acre feet or the volume of water necessary to cover 1 acre to a depth of 1 foot (also equivalent to 325,851 U.S. gallons).

**Table 2—Modeled mean annual flow by watershed and air temperature increase**

| | Annual average flow (MCM) | | | Percent decrease from basecase | |
|---|---|---|---|---|---|
| **Watershed** | **Basecase** | **2 °C increase** | **6 °C increase** | **2 °C increase** | **6 °C increase** |
| Feather | 5776 | 5649 | 5264 | 2.2 | 8.8 |
| Yuba | 3020 | 2960 | 2806 | 2.0 | 7.1 |
| Bear | 492 | 475 | 445 | 3.6 | 9.6 |
| American | 3556 | 3448 | 3218 | 3.1 | 9.5 |
| Cosumnes | 603 | 571 | 518 | 5.2 | 14.0 |
| Mokelumne | 979 | 946 | 887 | 3.4 | 9.4 |
| Calaveras | 330 | 319 | 301 | 3.3 | 8.9 |
| Stanislaus | 1561 | 1523 | 1435 | 2.4 | 8.1 |
| Tuolumne | 2445 | 2401 | 2304 | 1.8 | 5.8 |
| Merced | 1348 | 1308 | 1237 | 3.0 | 8.2 |
| San Joaquin | 2294 | 2265 | 2201 | 1.3 | 4.1 |
| Kings | 2117 | 2094 | 2041 | 1.1 | 3.6 |
| Kaweah | 586 | 564 | 519 | 3.8 | 11.5 |
| Tule | 199 | 190 | 171 | 4.6 | 14.3 |
| Kern | 926 | 887 | 813 | 4.2 | 12.2 |

Source: Null et al. 2010.
MCM = million cubic meters.

The KREW research (see box 6.1-5 on page 290) can help one to understand climate change effects for the southern Sierra Nevada, because five of the KREW streams are located in the rain-snow interface zone and five are in the snow-dominated zone. The current functioning of the lower elevation streams provides valuable insight into what can be expected for higher elevation streams with a 2 °C air temperature shift (Bales et al. 2011a, Hunsaker et al. 2012). New information is also being developed on these topics with field instrumentation and data analyses by the Sierra Nevada Adaptive Management Project (SNAMP) and the Southern Sierra Critical Zone Observatory (SSCZO).

Using data from four eddy covariance towers, Goulden et al. (2012) reported a large decline in evapotranspiration (ET) between 2015 and 2700 m that is associated with development of winter dormancy in trees. This elevation range marks the transition from a mixed rain-and-snow precipitation regime to one dominated by snow (Hunsaker et al. 2012). During 3 years of observations, the ET at 2015 m ranged from 70 to 80 cm/yr (Bales et al. [2011a] estimated 96 cm/yr at this location), whereas it was only 35 to 50 cm/yr at the 2700 m elevation. Goulden et al. (2012) interpreted their results as suggesting that winter transpiration could increase because of climate change, thus decreasing the amount of water available for streams.

**Because climate change is expected to increase the incidence of severe wildfire, high-intensity storms, and rain-on-snow events, the threat of post-wildfire debris flows is expected to increase and become more widespread.**

## Effects on Channel Processes

Because climate change is expected to increase rainfall and storm intensity (Moody and Martin 2009) and the occurrence of uncharacteristically severe wildfire (Miller et al. 2009), flooding and sediment movement may increase owing to the incidence of rain-on-snow events or post-wildfire floods, which could in turn reduce channel stability and habitat quality. Negative impacts of climate change may be especially pronounced in high-elevation, currently snow-dominated systems that shift toward more winter rainfall (Battin et al. 2007). Herbst and Cooper (2010) suggested that increased rain-on-snow floods might pose a particular threat to streams that are already degraded (see "Effects of floods" on page 300). Riebe et al. (2001) concluded that outside of glacial transition periods, climate change is unlikely to substantially affect watershed-scale erosion rates in the Sierra Nevada; however, they cautioned that climate change could alter sediment storage in floodplains, terraces, and colluvial hollows, which would affect short-term sediment delivery and channel stability. In reviewing effects of climate change on streams in the mountains of Idaho, Goode et al. (2012) contended that sediment yield could increase tenfold compared to recent historical levels owing to increases in postfire debris flows. Because climate change is expected to increase the incidence of severe wildfire,

high-intensity storms, and rain-on-snow events, the threat of post-wildfire debris flows is expected to increase and become more widespread (Cannon and DeGraff 2009). If postfire landforms persist beyond the wildfire recurrence interval, successive wildfires will have an important cumulative impact on watershed morphology (Moody and Martin 2001).

**Debris flows—**

Intense storms, in many cases following wildfires, can trigger debris flows.[6] Most debris-flow activity occurs within about 2 years following a fire, because revegetation tends to quickly stabilize hillslopes. Substantial hazards from flash flooding, however, could remain for many years after a fire (Cannon and Michael 2011). In studies of postfire debris-flow processes throughout the Western United States, the great majority of fire-related debris flows initiate through a process of progressive bulking of storm runoff with sediment eroded both from hillslopes and from channels, rather than from infiltration-triggered landsliding. Statistical-empirical models have been developed to estimate the probability and volume of debris flows that may be produced from burned drainage basins as a function of different measures of fire severity and extent, gradient, and soil physical properties in the basin (Cannon and Michael 2011). The probability model was developed using data from 388 basins in 15 recently burned areas of the Western United States, and the volume model was developed from 55 debris-flow-producing basins burned by eight different fires where the volume could be attributed to a single storm. This modeling work used a 30-minute-duration, 10-year-recurrence rainstorm of 0.73 in (18.5 mm) to trigger an event. Intense rainfall events, rain-on-snow storms, and rapid snowmelt are all associated with debris flow occurrence in the Sierra Nevada. Cannon et al. (2008) summarized research on runoff and sedimentation events from recently burned watersheds and found that a 30-minute peak rainfall intensity greater than 10 mm/h resulted in significant increases in runoff, and intensities greater than 20 mm/h resulted in significant sediment movement. The association between wildfire, debris flow, and floods is well established in the southern Sierra Nevada (DeGraff et al. 2011), and the modeling work by Cannon and Michael (2011) enables risk potential and volume of sediment from wildfires to be estimated for comparison to sediment from management activities.

[6] Land and rock slides are another geomorphic disturbance that can deliver sediment to stream networks in the synthesis area. However, compared to some mountain ranges, such as the European Alps or the Himalayas, the Sierra Nevada generates relatively infrequent massive rock slides. In the southern Sierra Nevada, nine slides have been documented from prehistoric times to 1997, ranging in size from 23 000 $m^3$ to 11 million $m^3$ (Harp et al. 2008). Such slides have had severe impacts on people, communities, and infrastructure and can create dams in steep river canyons.

Large debris flows down channels may be among the most persistent effects of wildfires (Benda et al. 2003, Goode et al. 2012, Moody and Martin 2001). Debris flows are a major concern in southern California and the Intermountain Region, and those flows tend to be larger than flows in more humid climates such as the Pacific Northwest (Santi and Morandi 2012). The wetter western slopes of the Sierra Nevada have experienced debris flows associated with landslides following high rates of rainfall, rapid snowmelt, or rain-on-snow events (DeGraff 1994). In addition, there have been instances of post-wildfire debris flows from burned watersheds upslope from El Portal, California, near Yosemite National Park (Cannon and DeGraff 2009). In the southern Sierra Nevada, monsoonal storms on July 12, 2008, produced intense rainfall that triggered large debris flows in the southern Sierra Nevada (fig. 3) (DeGraff et al. 2011). One flow traveled down the north and south forks of Oak Creek through the town of Independence on the east side of the Sierra Nevada; it resulted in substantial damage to homes, a Forest Service campground,

Jerome V. De Graff

Figure 3—Damage to residences along Oak Creek following the postfire debris flow incident of July 12, 2008, on the Inyo National Forest.

and other infrastructure. The other flow traveled down Erskine Creek through the town of Lake Isabella, California, and into the Kern River on the southern end of the Sierra Nevada. The Inyo Complex fire had burned 30 percent of the Oak Creek watershed at high or moderate severity in 2007, and the Piute Fire had burned 15 percent of the Erskine Creek watershed at high or moderate severity, but the two events shared relatively intense rainfall (16 to 30 mm/hour) (DeGraff et al. 2011). These incidents demonstrate that postfire debris flows are a significant concern in the southern and eastern parts of the Sierra Nevada, which experience high-intensity rainstorms. Further research would be needed to evaluate risks within the synthesis area, given high amounts of variability in these watershed processes. A comparison of rainfall regimes by Moody and Martin (2009) showed that the region that includes the Sierra Nevada experiences less intense rainfall than the mountains of Arizona, but more intense rainfall than in the Great Basin. However, they found very high variability within the Pacific region and poor correlation between post-wildfire sediment yields and rainfall intensity (measured as the average 2-year event over 30-minute periods).

## Effects on Aquatic Ecosystems

Projected effects of climate change on aquatic ecosystems include hydrologic effects discussed above (especially lower summer baseflows, earlier runoff, and higher summer water temperatures), as well as biological effects, such as increased isolation of native aquatic species and spread of invasive species (Viers and Rheinheimer 2011).

**Impacts on trout—**

The projected impacts of climate change on trout and salmon species are a particular concern because of the vulnerability of those species to increased stream temperatures (Moyle et al. 2011). Although the middle of the Sierra Nevada includes a large area that was historically fishless, the northern and southern Sierra Nevada support several endemic strains of native trout (fig. 4). Several varieties of redband or rainbow trout (*Oncorhynchus mykiss*) evolved in the Sacramento-San Joaquin drainages of the Sierra Nevada, while varieties of cutthroat trout (*Oncorhynchus clarkii*) evolved within the interior east-side drainages of the Lahontan basin (Behnke 2002). Because of stocking with nonnative trout species, translocations of trout outside of their native streams, and impacts to habitats, most of these native trout have become confined to relatively small streams, leaving them vulnerable to the effects of climate change and wildfire (see box 6.1-1). Wenger et al. (2011) forecasted significant declines in trout habitat and associated socioeconomic consequences across the interior Western United States over the next 60 years.

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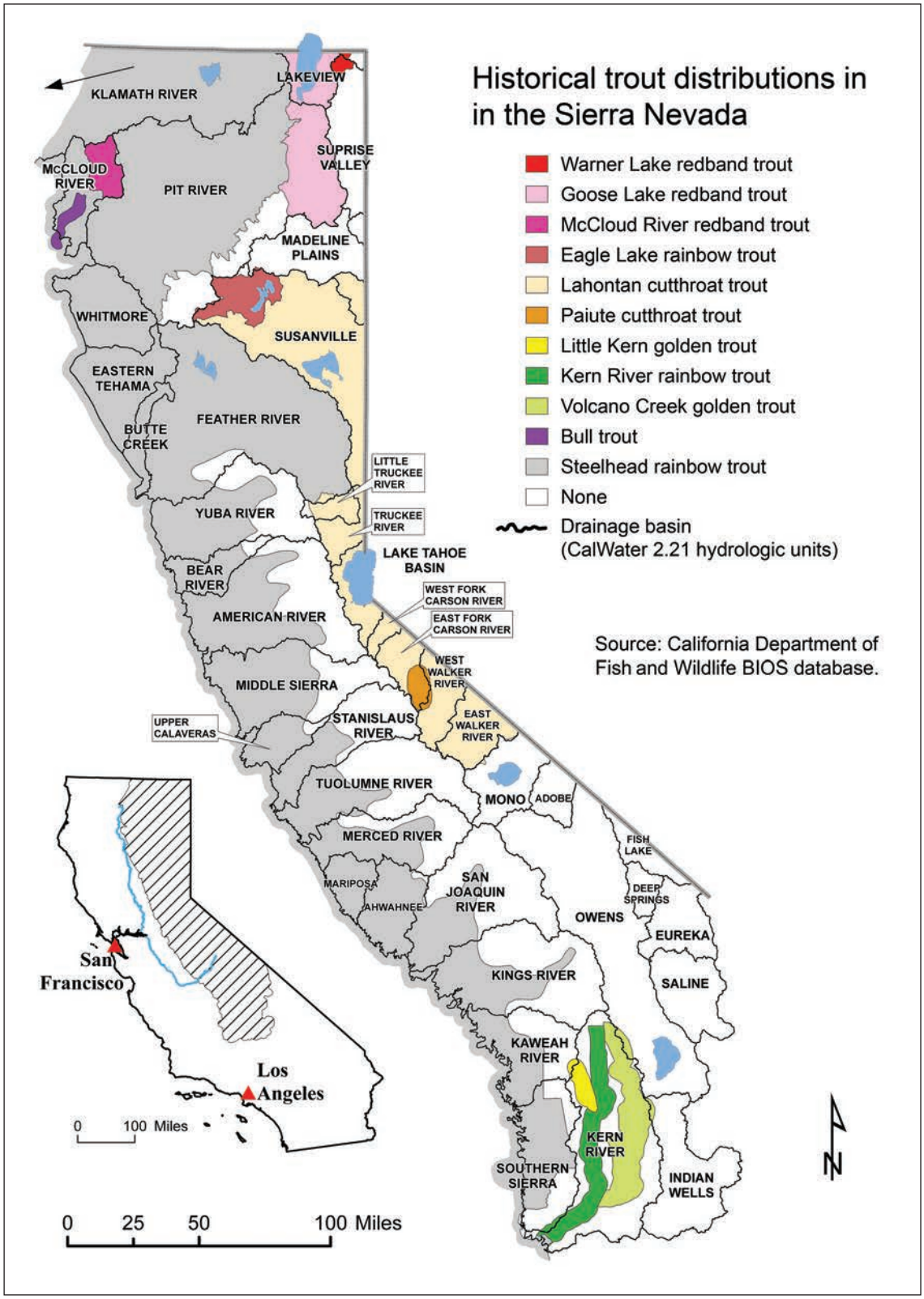


Figure 4—Historical distribution of native trout within the synthesis area. Map by Ross Gerrard.

Post-wildfire floods that reorganize channel habitats can have significant impacts on fish populations, including extirpation of isolated native trout populations in headwater streams in the Southwest (Brown et al. 2001). A study by Isaak et al. (2010) in Idaho demonstrated that severe wildfires followed by channel-reorganizing floods can increase rates of stream warming over long periods. These events can cause streams to warm by removing vegetation and widening channels; those effects may offset the potential of such events to lower temperatures by increasing base flow (a potential short-term effect resulting from reduced transpiration in the watershed) (Sugihara et al. 2006), or by increasing heat exchange with colder groundwater (Dunham et al. 2007). In contrast, native fish populations that inhabit relatively intact stream networks in the Northwest and the northern Rocky Mountains have demonstrated resilience following wildfires (Gresswell 1999, Neville et al. 2009). However, the responses of aquatic systems to wildfire and climate change observed in other regions may not transfer well across the synthesis area because of variations in climate, topography, extent of nonnative competitors, and connectivity of aquatic populations.

**Box 6.1-1**

### Reports on Threats From Climate Change and Wildfire on Aquatic Species

Trout Unlimited has generated a series of reports under its Conservation Success Index program (Williams et al. 2007) that characterize risks for native salmonids from changes in climate and fire regime within their ranges.

More recently, researchers at the University of California–Davis prepared a white paper report on the effects of future climates on freshwater fishes (Moyle et al. 2012).

In addition to the recently published study of postfire debris flows by DeGraff et al. (2011), the Forest Service has conducted monitoring of impacts to aquatic ecosystems following recent fires, including the Moonlight Fire (2007), Cub Fire (2008), and Lion Fire (2011). These observational efforts should afford opportunities to evaluate resilience of streams in the synthesis area to wildfires of different severities.

Tom Whitaker

Figure 5—Intense rain events or rain-on-snow events can increase suspended sediment and turbidity in streams, as shown at one of the flumes in the Kings River Experimental Watersheds, Sierra National Forest.

## Effects on Biological Indicators of Water Quality

Understanding particular effects of climate change on the biology of mountain streams would be valuable, because many management agencies use biological criteria to measure trends of ecological health, water quality, and the integrity of ecosystem function. In particular, streams in relatively undisturbed watersheds serve as references to evaluate condition. Against a background of climate-driven alteration to the ecology of streams across the Sierra Nevada, the biological integrity of reference streams may decline. Biological diversity in confined headwater and alpine streams may be especially sensitive to shifting hydrologic patterns. Even though all streams and lakes are affected by climate change, the reference habitats may have more to lose than disturbed streams that have already been affected by localized sources of pollution and other forms of degradation. If reference streams

lose a higher proportion of aquatic life to warming and hydrologic disruption, then the "signal," or difference relative to disturbed test sites, would be decreased. The reference condition for streams is typically developed based on many sites sampled over many years, so if these streams slowly degrade, the range of variability or "noise" in the cumulative reference condition will increase. The net effect of a declining reference condition is that it will be harder to detect degradation by nonclimate factors. Establishment of current conditions and quantification of climate-induced drift would help to monitor conditions within the Sierra Nevada and to recalibrate standards as climate conditions change.

**Box 6.1-2**

### Stream Monitoring Network for Climate Change

In 2010, the Management Indicator Species Program of the Pacific Southwest Region funded the establishment of a stream observatory network, designed by David Herbst of the Sierra Nevada Aquatic Research Laboratory, to detect the ecological impacts of climate-induced changes in hydrologic balance and temperature of Sierra Nevada streams and to provide a historical context for recovery of degraded ecological values. The network includes 12 sites (6 in the southern Sierra Nevada and 6 in the northern Sierra Nevada) that serve as undisturbed reference sites for streams that are expected to have high and low risk for climate-induced loss of snow cover and hydrologic stability, in combination with high and low resistance to climate change. This network also sets up a natural experiment within which differing hypothesized risks based on forecasted climate conditions and hydrographic susceptibility can be contrasted. The sites are broadly representative of Sierra Nevada streams across a range of elevations from 1220 to 3660 m (4,000 to 12,000 ft). Meas-urements at these sites include stream invertebrates, algae periphyton, water chemistry, geomorphic characteristics, stage height, riparian cover, and water and air temperature. Benthic invertebrate samples from this network partition into two community groupings—those sites north of Yosemite National Park and those south of the park. A possible difference is that southern Sierra Nevada streams have less groundwater recharge based on a comparison of their chemistry with that of northern streams, and they are thus more susceptible to low flows and drying, and support less biological invertebrate diversity. Southern Sierra Nevada streams with greater upstream length may be less prone to drying and therefore have a higher level of diversity.

## Water Resources

This section includes a discussion of ecosystem processes and issues that are important to water quantity and quality. Stream benthic macroinvertebrates (BMIs) are included here as important biological indicators of water quality.

### Water Quantity

Streamflow response to a change in forest density is strongly related to climate, species composition, and the percentage change in vegetation density. Troendle et al. (2010) provided a review of the many studies on this topic and made the following observation (Troendle et al. 2010: 126–127): "The data from 95 watershed experiments conducted in the United States show that, on average, annual runoff increases by nearly 2.5 mm for each 1 percent of watershed area harvested (Stednick 1996). Because runoff is quite variable from year to year, the general conclusion is that approximately 20 percent of the basal area of the vegetation must be removed before a statistically significant change in annual runoff can be detected (Bosch and Hewlett 1982, Hibbert 1967, Stednick 1996)." Because most of these paired watershed experiments imposed a partial or complete clearcutting of the mature trees, our current understanding of the hydrologic impacts of thinning and prescribed fire comes from inference supported by some plot and process studies.

**Forest treatments that remove less than 20 percent of tree basal area may result in a change in flow, although it likely will not be detectable.**

Many fuels management treatments or forest restoration efforts remove less than 20 percent of the basal area of trees; although this may result in a change in flow, it likely will not be detectable, especially in dry years. With best management practices (BMPs), which should not cause overland flow from skid trails or soil compaction, there should be little or no detectable effect on peak discharges. Any change will be short-lived because of vegetation regrowth, except in cold snow zones (Rocky Mountain region). Prescribed fire by itself is less likely to influence water yield than mechanical treatments because of the smaller reduction in basal area and lack of ground disturbance by heavy machinery (Troendle et al. 2010).

During the past decade, a better understanding of hydrologic processes has developed for the southern Sierra Nevada. Hunsaker et al. (2012) characterized the climate and hydrologic patterns for eight headwater catchments, including both the rain-snow transition zone and the snow-dominated zone of the southern Sierra Nevada. A water-balance instrument cluster at these rain-snow catchments enabled an estimate of total annual evapotranspiration at 76 cm in 2009, a value higher than previous estimates for the Sierra Nevada (Bales et al. 2011a). Water loss rates from soil were estimated to be 0.5 to 1.0 cm $d^{-1}$ during the winter and snowmelt seasons. Soon there will be data on the effect of both prescribed fire and mechanical removal of vegetation on streamflow (see box 6.1-5).

Engle et al. (2008) provided the only new experimental data on streamflow response to prescribed fire in the Sierra Nevada. The Tharps Creek watershed (13 ha) was burned after having no fire for at least 120 years; the preburn surface fuel load was 210 Mg $ha^{-1}$ and fuels were reduced by 85 percent as a result of the fire. After fire, runoff coefficients increased by 7 percent in dry years and 35 percent in wet years. (The runoff coefficient is the relationship between the amount of runoff to the amount of precipitation; the value is large for areas with low infiltration and high runoff.) Mean annual runoff in the 50-ha Log watershed (control) during the dry years was 29 percent of precipitation; mean annual runoff was 56 percent of precipitation when drought years are excluded. Runoff coefficients in the Tharps watershed were consistently lower than in the Log watershed, averaging 51 percent during wet years and 8 percent during dry years. Nine years after the burn, there was no evidence that runoff in the Tharps watershed was returning to prefire levels.

**Box 6.1-3**

### Water Yield Predictions

A recent report based on modeling suggests a somewhat different conclusion from Troendle et al. (2010) about the ability to increase water yield through forest harvesting. Bales et al. (2011b) suggested that reducing forest cover by 40 percent of maximum levels (based on leaf area index) across a watershed could increase water yields by about 9 percent. They estimated that proposed treatments at the Onion Creek Experimental Forest on the Tahoe National Forest could increase water yield by as much as 16 percent and extend snow storage (i.e., delay snowmelt) by days to weeks. They also cited recent studies in the Sierra Nevada that report potential increases in snow accumulation of 14 to 34 percent after forest harvest (see Bales et al. 2011b). However, a well-integrated approach to forest management considers many values in an area rather than focusing on opportunities to maximize any one value such as water yield.

## Water Quality

**General condition—**

Water from the Sierra Nevada accounts for 60 percent of the total dollar value of all natural products or services produced by the entire region—more than forest products, agricultural products, recreational services, or even residential development (SNEP Science Team 1996). A number of indicators can be used to

characterize water quality, including chemical indicators (nutrients, conductivity, pH, metals, pathogens, pesticides, and organics), physical indicators (temperature and sediment), biological indicators such as stream invertebrates, and human use criteria (water that is "swimmable, fishable, drinkable"). A general overview of water quality for major Sierra Nevada river basins is based on publicly available data about various indicators—some quantitative and some qualitative (Timmer et al. 2006).

The conditions of waters in the synthesis area may be affected not only by ongoing human activities but also by legacies associated with historical mining, logging, and construction for homes, roads, and railways. Because of enhanced planning, BMPs, and state and federal regulations, impacts from contemporary development activities are typically much less severe than during historical periods. Even though legacy impacts (e.g., mercury contamination from historical mining, high sediment loads from old and poorly constructed roads, or human health hazards from old septic systems) can be substantial in local areas, they are not a focus of this synthesis. Instead, this chapter focuses on water quality issues that tend to be common across the synthesis area and are a focus for monitoring to maintain or improve water quality. Some human health issues with water quality are briefly discussed in chapters 6.3, "Wet Meadows," and 9.3, "Sociocultural Perspectives on Threats, Risks, and Health").

The U.S. Environmental Protection Agency (EPA), state of California, and Forest Service all use macroinvertebrates as a biological indicator of water quality along with measures of stream physical habitat and water chemistry. An ecological condition assessment (2000 through 2006) of California's perennial wadeable streams provided an overview of water quality for forested lands (Ode 2007). This assessment reported that forests had 80 percent of the monitored stream segments in an **unimpaired** condition, compared with agriculture and urban land uses that had more than 80 percent being in the **impaired** and very impaired condition classes. When a forest stream segment was highly impaired, the following stressors were associated the most with that poor condition: total nitrogen (N) (30 percent), chloride (20 percent), total phosphorus (P) (10 percent), lack of habitat complexity (20 percent), and riparian disturbance and streambed stability (10 percent).

National forests are mostly in the headwaters of Sierra Nevada river basins; often the impaired portion of a river or stream is downstream of the forests or associated with reservoirs or other impoundments. As an example, Timmer et al. (2006) reported the following information for the Kings River watershed. The upper North Fork has been listed as impaired for wetland habitat and flow alterations by the EPA and listed as threatened for habitat, fishery, and freshwater by the California State

Amber Olsson

Figure 6—Sediment is removed, weighed, and sampled to quantify and characterize the annual load in a stream. A sediment basin is emptied at the Teakettle Creek, which is used as a control watershed in the Kings River Experimental Watersheds.

Water Resources Control Board. The Main Fork Kings River is listed by the EPA as impaired for flow alterations and threatened for habitat, fishery, and freshwater. Timmer et al. (2006) listed the probable sources for these detrimental impacts as construction, agriculture or nursery operations, and modification of the streambed. At Pine Flat Reservoir, EPA listed the Kings River as impaired for pathogens, habitat, and freshwater and as threatened for swimming, fishing, fish tissue concentrations, and recreation user days. The **threatened** designation means that the water currently supports designated uses, but may become impaired in the future if pollution control actions are not taken. **Impaired** means that a designated water use is not supported. This report also indicates if a water body is affected by a particular metal and if human exposure is a concern. Similar general condition information exists for all major watersheds in the Sierra Nevada (Timmer et al. 2006).

**Stream sediment and erosion—**
Undisturbed forests are an important source of the clean water that is necessary for ecosystem health as well as urban and agricultural uses. By altering infiltration rates and evapotranspiration rates and disturbing the soil, forest management activities (including road construction, timber harvesting, site preparation, fuels reduction, and prescribed fire) can increase overland flow rates and sediment yields. Sediment yields are dependent on many factors: climate, topography, soil type, vegetation, historical land use, and the dominant erosion processes (Stednick 2000). Robichaud et al. (2010) provided a review of fuels management effects on erosion. Reported sediment yields from undisturbed forests in the Western United States are 0.007 Mg $ha^{-1}$ (0.003 t $ac^{-1}$), but values up to 25 Mg $ha^{-1}$ (11 t $ac^{-1}$) have been reported (Stednick 2000). Hunsaker and Neary (2012) reported an average of 16 ± 21 kg $ha^{-1}$ (0.016 ± 0.021 Mg $ha^{-1}$) over 7 years of measurement at the undisturbed Teakettle Experimental Forest in the headwaters of the Kings River (Mg = $10^6$ grams or 1 metric tonne). Breazeale (1972) reported mean annual sediment rates of 260 Mg $km^{-2}$ (110 t $mi^{-2}$) for the North Fork of the Kings River, and Dunne and Reid (1985) reported 100 Mg $km^{-2}$ (43 t $mi^{-2}$) for the Teakettle Experimental Forest.

Research to better understand cumulative watershed effects in the Sierra Nevada has focused on sediment data from roads, timber harvests, and wildfires. On the Eldorado National Forest, MacDonald et al. (2004) found the median sediment production rate from roads was 0.2 kg $m^{-2}$, nearly an order of magnitude higher than any of the other sources they evaluated (including skid trails, off-road vehicle trails, prescribed fire, and wildfire). In a more recent study of 200 harvest units on the Eldorado, Lassen, Plumas, and Tahoe National Forests, Litschert and MacDonald (2009) found only 15 rills and four sediment plumes on 14 of the units. Only 7 percent of the units had sediment entering the streamside management zones, and the majority of these involved skid trails. They attributed these relatively limited impacts to the increased use and refinement of BMPs and a shift from clearcuts to thinning and group selection. Litschert and MacDonald (2009) suggested that attention should focus on proper construction and postharvest treatment of skid trails for additional reduction of sediment input to streams (e.g., use of proper water bars, ripping to maximize infiltration where soils are compacted, and adequate surface roughness at water bar outlets).

Any type of land use change that causes soil disturbance or vegetation removal (timber harvest, brush clearing for fuels reduction, fire, and road construction, use, and decommissioning) has the potential to cause erosion and subsequent sediment delivery to water bodies. Historically, roads have been considered the primary source of sediment and a significant problem in many landscapes. The potential in

the Sierra Nevada for erosion and sediment effects on water quality and aquatic habitat from forest management and wildfire needs to be evaluated with respect to current knowledge and practices. Coniferous forests across western North America are experiencing widespread mortality as a result of drought, insect outbreaks, and wildfire associated with climate change. In many of these landscapes, wildfires and subsequent storms commonly result in the delivery of large, infrequent pulses of sediment to water bodies. Goode et al. (2012) suggested that sediment yields may be roughly 10 times greater with climate-modulated processes than those observed during the 20$^{th}$ century. Although coarse sediment is important for forming geomorphology and aquatic habitat, an order of magnitude increase may have undesirable impacts to aquatic organisms and reservoir management and life expectancy.

Gucinski et al. (2001) synthesized scientific information on forest roads and noted that the Forest Service has a framework (USDA FS 1999) in place for evaluating benefits, problems, risks, and tradeoffs of roads. On November 9, 2005, the Forest Service published the Final Travel Management Rule (70 Federal Register [Fed. Reg.] 216, November 9, 2005; p. 68264–68291), which required designation of roads and trails for motor vehicle use. Implementing travel management plans to meet this requirement should help reduce sediment from roads. Despite the size of the forest road network, road effects have been examined in only a few places, especially in the Appalachians, Pacific Northwest, and Rocky Mountains. Given the wide variability in road history, age, construction methods, and use patterns in relation to topography, climate, and social setting, the narrow geographical scope of these studies limits their extrapolation to other regions or their usefulness in addressing more subtle effects. In forests along the west side of the Sierra Nevada, major roads were built along broad ridges, with secondary roads leading down into headwater areas. In general, Sierra Nevada roads create less erosion and landslides when compared to roads in western Oregon forests, which usually entered watersheds along narrow stream bottoms and then climbed the adjacent steep, unstable hillslopes to access timber extending from ridge to valley floor. Road placement in the landscape, combined with local geology and climate, resulted in different effects of roads on watershed, vegetation, and disturbance processes in the Western United States (Gucinski et al. 2001). A summary of points discussed by Gucinski et al. (2001) about road erosion effects is included here.

- Although mass erosion rates from roads typically are one to several orders of magnitude higher than from other land uses based on unit area, roads usually occupy a relatively small fraction of the landscape, so their combined effect on erosion may be more comparable to other activities, such as timber harvest.

- Roads interact directly with stream channels in several ways, depending on orientation to streams (parallel, orthogonal) and landscape position (valley bottom, midslope, ridge).
- The geomorphic consequences of these interactions, particularly during storms, are potentially significant for erosion rates, direct and offsite effects on channel morphology, and drainage network structure, but they are complex and often poorly understood.
- Encroachment of forest roads along the mainstem channel or floodplain may be the most direct effect of roads on channel morphology in many watersheds.
- Poorly designed channel crossings of roads and culverts designed to pass only waterflow also may affect the morphology of small tributary streams, as well as limit or eliminate fish passage.
- Indirect effects of roads on channel morphology include the contributions of sediment and altered streamflow that can alter channel width, depth, local gradients, and habitat features (pools, riffles) for aquatic organisms.
- Extensive research has demonstrated that improved design, building, and maintenance of roads can reduce road-related surface erosion at the scale of individual road segments.

Although poorly constructed roads in the Sierra Nevada can cause soil erosion and increase sedimentation to water bodies, their effects have not been studied very much in the synthesis area. Two studies do provide recent information for the Sierra Nevada (see box 6.1-4). Road impacts to water quality and aquatic habitat should be less in the future because very little new road construction is expected and knowledge exists about how to construct and maintain roads to lessen impacts. Larger sources of soil erosion may include increased wildfires (see "Debris flows" on page 275), as well as a lack of road maintenance that results in progressive degradation of road-drainage structures and functions (Furniss et al. 1991). A nationwide, multi-site study by Meadows et al. (2008) found that all-terrain vehicle (ATV) traffic on existing trails can increase runoff and sediment, but that proper trail design and maintenance can reduce those impacts.

Goode et al. (2012) projected that climate change would increase sediment yield in semiarid basins, primarily through changes in temperature and hydrology that promote vegetation disturbance (i.e., wildfire, insect/pathogen outbreak, and drought-related die-off). Although their case study took place in central Idaho, it is relevant for the Sierra Nevada because of similarities in conditions, such as coarse-textured, granitic soils and forests on steep mountain terrain. Istanbulluoglu et al. (2004) demonstrated that the mechanism driving higher long-term sediment yields

**Box 6.1-4**

## Master's Research: Roads and Measured Sediment Loads in the Sierra Nevada

In master's degree theses, Coe (2006) and Stafford (2011) provided data on sediment production and delivery from forest roads in mixed, rain-snow climate regimes in the Sierra Nevada. After studying roads for 3 years on the Eldorado National Forest, Coe (2006) found that sediment production rates from native surface roads (median value of 0.14 kg $m^{-2}$ $yr^{-1}$) were much greater (12 to 25 times) than those from rocked roads (median value of 0.009 kg $m^{-2}$ $yr^{-1}$), and that sediment from roads with slopes greater than 7 percent was 75 percent higher than for less steep roads. He also found that the annual amount of sediment delivered from episodic gully erosion below road segments was comparable to the amount of sediment being delivered from the native road surface. For comparison, sediment production from unpaved roads in the Sierra Nevada was approximately 67 percent of reported values for the Idaho batholith and only 4 percent of that for the Pacific Northwest. Fifty-nine percent of the road segments that connected to streams were at stream crossings, whereas 35 percent of the connected segments resulted from road-induced gullies. Sediment traveled less than 42 m below the drainage outlet for 95 percent of the road segments. In-sloped roads drained by relief culverts, longer road segments on steeper slopes, and drainage outlets discharging onto hillslopes with compacted soils were usually associated with road-induced gullies. Coe (2006) concluded that road sediment production can be mitigated by rocking native-surface roads, improving and maintaining drainage from roads, reducing the number of stream crossings, and rocking the approaches to stream crossings.

Stafford (2011) performed a study similar to Coe's within the southern Sierra Nevada. She measured an average sediment production rate of 1.8 kg $m^{-2}$ $yr^{-1}$ for native surface road segments in the rain zone; higher than was measured by Coe. At the rain-and-snow and snow-dominated zones, the average sediment production for native surface road segments is more than an order of magnitude lower (0.13 kg $m^{-2}$ $yr^{-1}$), which is similar to that measured by Coe (2006) in the same type of precipitation regime. There was no significant difference in sediment production from native and gravel-surfaced road segments in the lower elevation area, but in the higher elevation area, the sediment production was 22 percent less for graveled roads. Surprisingly, Stafford (2011) found no significant difference in sediment production and delivery between road segments in a highly erodible soil type (Holland) and road segment less erodible soil types. She suggested that sediment production from forest roads can be reduced by using more than 30 percent gravel cover on native surface roads, minimizing grading, and improving the construction of water bars.

**Road maintenance and decommissioning are generally effective and beneficial for water quality but will not mitigate an increase in sediment yields from increased wildfire frequency.**

in smaller catchments (<25 km) is rare, postfire erosional events that are typically two orders of magnitude larger than the long-term average yields of 146 t $km^{-2}$ $yr^{-1}$ (determined from cosmogenic dating). Sediment yields from experimental basins with roads are on the order of 101 t $km^{-2}$ $yr^{-1}$, whereas yields from individual fire-related events in this region are three orders of magnitude greater ($10^4$ t $km^{-2}$ $yr^{-1}$). An experiment (Ketcheson et al. 1999) showed that for 1 to 2 km/$km^2$ of new road, the amount of sediment yield increased by 7 to 12 t $km^{-2}$ $yr^{-1}$ compared with 2.5 t $km^{-2}$ $yr^{-1}$ for the control basin (more than a doubling during the four study years). Goode et al. (2012) concluded that road maintenance and decommissioning are generally effective and beneficial for water quality but will not mitigate an increase in sediment yields from increased wildfire frequency. They also highlighted the substantial uncertainty about the efficacy of postfire treatments for vegetation and hillslope erosion in forest mountain basins (Robichaud et al. 2000) and the growing body of literature discouraging further interference in natural landscape disturbance processes because the dynamic response to such disturbances may help maintain more diverse ecosystems that are more resilient to changed climates (DellaSala et al. 2004). Therefore, work to reduce the magnitude and frequency of wildfire is likely important to influence total sediment yields from forests in the Sierra Nevada drainage basins.

**Box 6.1-5**

## Current Research—Kings River Experimental Watersheds

The Kings River Experimental Watersheds (KREW) is a watershed-level, integrated ecosystem project for headwater streams in the Sierra Nevada (http://www.fs.fed.us/psw/topics/water/kingsriver). Eight watersheds at two study sites are fully instrumented to monitor ecosystem changes. The KREW project was designed to address several of the information gaps for water resources and aquatic ecosystems included in the Monitoring and Adaptive Management Plan for the Sierra Nevada (Appendix E, Sierra Nevada Forest Plan Amendment [USDA FS 2001, 2004]). A few examples of these questions are:

- What is the effect of fire and fuels reduction treatments (i.e., thinning of trees) on the physical, chemical, and biological conditions of riparian areas and streams?
- Does the use of prescribed fire increase or decrease the rate of soil erosion (long term versus short term) and affect soil health and productivity?
- How adequate and effective are current stream buffers (areas on both sides of a stream with restricted uses) at protecting aquatic ecosystems?

*Continued on next page*

**Box 6.1-5 (continued)**

Prior to 2000, when KREW was designed, there was no long-term experimental watershed study in the southern Sierra Nevada to guide future land management activities. KREW has a site in the rain-snow zone and a site in the snow-dominated zone of Sierra Nevada mixed conifer. Data have been gathered for a 9-year pretreatment period (Brown et al. 2008; Hunsaker and Eagan 2003; Hunsaker and Neary 2012; Hunsaker et al. 2007, 2012; Johnson et al. 2011a; Liu et al. 2012). Tree thinning was completed in 2012, some prescribed underburns were completed in 2013, and final burns are planned for 2014. The experimental design will allow partitioning of effects between thin only, underburn only, and the preferred treatment of thin and burn.

This research is evaluating the integrated condition of the streams and their associated watersheds (i.e., physical, chemical, and biological characteristics).

- Physical measurements include upland erosion, turbidity (suspended sediment), stream temperature, streamflow, channel characteristics, and weather conditions.
- Chemical measurements for stream water, shallow soil water, precipitation, and snowmelt include nitrate, ammonium, and phosphate (primary biological nutrients); chloride; sulfate; calcium; magnesium; potassium; sodium; pH; and electrical conductivity.
- Biological measurements include stream invertebrates (like dragonflies and mayflies), algae, and riparian and upland vegetation (herbs, shrubs, and trees). Yosemite toads (*Anaxyrus* [=*Bufo*] *canorus*) are also being studied at the Bull Creek site.

Unique aspects of KREW include the following:

- An integrated design of physical, chemical, and biological components being measured at the same locations and at several spatial scales.
- A control watershed that can provide the “natural range” of variability (no roads or timber harvesting).
- A designed comparison of fuels and vegetation for riparian and upland parts of the watersheds.
- Use of a long-term data set for stream invertebrates to evaluate water quality effects after management treatments at a designed before-after control-impact (BACI) experiment.
- A comparison of adult Yosemite toad movement before and during treatments.
- A comparison of rain-snow and snow-dominated watershed function before and after land disturbances.

**Nutrients and chemistry—**

Water chemistry is of interest for various reasons: human health, aquatic ecosystem condition, and agricultural and industrial uses downstream of the forests. Although the chemistry of water is usually very good within national forests, it is prudent to monitor some characteristics. The EPA and state of California use chloride, specific conductance, and total N and P as indicators of stress in perennial, wadeable streams (Ode 2007). Additional information on chemistry processes in forests can be found in chapters 5.1 and 8.1. This discussion focuses on nitrogen, which is a necessary nutrient for vegetation, but in high enough amounts can be a substantial stressor on both terrestrial and aquatic ecosystems. Sudduth et al. (2013) reported on the significant role that forests have in reducing concentrations of nitrate in stream water, as compared to watersheds that have been developed for urban use or agriculture. Soil solution and stream water nitrate were positively correlated across 40 undisturbed forest watersheds, with stream water values usually being half of the soil solution concentrations. A similar relationship was seen in 10 disturbed (usually clearcut tree harvest) forest watersheds.

Nitrogen is an important indicator of the overall health of a forest, and knowing its concentration over time in atmospheric deposition, vegetation, soils, and stream water provides a useful assessment tool (see chapter 5.1). Usually, Sierra Nevada stream water has very low N concentrations, almost at detection limits (Engle et al. 2008, Hunsaker et al. 2007). However, atmospheric deposition (as discussed in chapter 8.1) is high and moderately high in the southern and central Sierra Nevada, respectively. Nitrogen leaching from soils to water, which can lead to acid conditions and a fine root biomass loss of 26 percent, are expected at N deposition levels of 17 kg N $ha^{-1}$ $yr^{-1}$ (Fenn et al. 2008). High N deposition, coupled with recent findings of high N content in mineral soils and water flowing through forest floor litter in both the western and eastern Sierra Nevada (Johnson et al. 2010, 2011a, 2011b; Miller et al. 2005), are reasons to have long-term measurements on N partitioning in the forest ecosystem.

A recent study by Argerich et al. (2013) highlighted the importance of long-term, uninterrupted stream chemistry monitoring data. They examined long-term data (greater than 12 years) from seven Forest Service experimental forests to see whether stream N concentrations in undisturbed catchments have changed over long periods and if patterns were consistent across the United States. They found high variability both across the country and within individual sites, which suggests that local factors were important. The analysis did not include data from the synthesis area, but two Western sites were included: H.J. Andrews Experimental Forest in Oregon and Fraser Experimental Forest in Colorado. Stream nitrate and ammonium

mostly decreased or showed no trend in Oregon, whereas stream nitrate in Colorado showed an increasing trend and stream ammonium was neutral or decreasing. Nitrogen concentrations in wet deposition at both locations usually showed no trend. As mentioned earlier, N is an important indicator of forest health. Nine years of extensive N data will soon be available from the KREW project in the synthesis area; these data can be compared to the trends for these other western watershed studies. This evaluation of N is especially important where deposition levels are high, as they are in the southern Sierra Nevada (see chapter 8.1).

The long-term (decade-scale) effects of fire on watershed chemical balances relate to changes in vegetative cover and include N fixation and the accumulation of elements in aggrading plant biomass (Johnson et al. 2009). Tree mortality, crown scorch, and stand replacement can affect canopy-related processes that are important in watershed balances of water and nutrients, such as interception of precipitation and cloud water, scavenging of aerosols and gases, and transpiration. The changes in nutrient budgets expected soon after fire (first few years) are a result of many processes, as listed by Engle et al. (2008): wind- and water-driven sediment export, changes in the physical properties of soil, dissolution of ash, shifts in soil water pH, changes in microbial biomass and activity, increased decomposition, and changes in biological demand for water and nutrients.

Johnson et al. (1998) stated that fire and postfire N fixation are more important than atmospheric deposition and leaching for N fluxes in most semiarid forests of the Southwestern United States. Exceptions may occur in areas with high atmospheric inputs of N from local air pollution. They argued that existing literature shows that the nutrient cycling paradigm established for humid forest ecosystems, which emphasizes fluxes into and out of the ecosystem by water, needs modification for semiarid forests. Odum et al. (1994) proposed that a realistic paradigm for natural systems is one in which the system is subjected to regular pulses of N from processes such as fire rather than the concept of steady-state and one-dimensional, vertical nutrient cycling. Nitrogen cycling studies in semiarid forests require a long-term, landscape-scale perspective that encompasses episodic fire and periods of intensive postfire N fixation. Johnson et al. (1998) concluded that the frequency of fire and the occurrence and duration of postfire N fixation are crucial factors that determine the long-term productivity of semiarid forest ecosystems and require more study; datasets comparing N fluxes via fire and water at the same site are very rare.

Fire has both short- and long-term effects on nutrient availability and cycling in forest ecosystems. Because of its low volatilization temperature (200 °C), nearly all N in burned biomass is lost in gaseous form, and N losses are disproportionately

large compared to carbon (C) losses. Fire can also result in the loss of other nutrients, including sulfur and phosphorus, by volatilization, though to a lesser extent than for N (Raison et al. 1985, 1990). The mean and median values for N losses during wildfire and prescribed fire are 360 and 280 kg $ha^{-1}$, respectively, which equal approximately 500 to 12,000 years leaching loss of N from semiarid forests (measured rates 0.1 to 0.6 kg $ha^{-1}$ $yr^{-1}$). Data exist for the east side of the Sierra Nevada (Little Valley, Nevada) to compare fluxes of N via deposition and leaching versus wildfire (Johnson et al. 1997), and they show that at a 100-year interval, wildfire was the dominant factor in long-term N losses, exceeding leaching losses by more than two orders of magnitude (3 to 6 compared with 0.03 kg $ha^{-1}$ $yr^{-1}$).

Some combination of restoring natural fire frequency, vegetation conditions, or fuel loading to landscapes is usually the goal of prescription burning and forest restoration, thus it is important to understand how watershed balances respond to fire on time scales that match target fire return intervals (FRIs). Tree ring studies indicate that, from 1700 to 1900, natural FRIs in the region of Sequoia National Park averaged 10 to 20 years (Engle et al. 2008); this is an accepted FRI for mixed-conifer forests in the southern Sierra Nevada.

Engle et al. (2008) provided the only new long-term research on stream chemistry before and after prescribed fire in the Sierra Nevada. Their research is for a 16-year paired catchment study in sequoia/mixed-conifer forest in Sequoia National Park. Seven years of prefire chemistry data were compared to 9 years of postfire chemistry data for two adjacent headwater streams—the intermittent Tharps Creek (13 ha) and the perennial Log Creek (50 ha) in the rain-snow zone of the southern Sierra Nevada. This study provides an excellent opportunity for the comparison of water chemistry data collected before and after thinning and burning treatments at KREW; these two long-term research projects should provide the necessary information for resource managers to understand small-stream response and recovery processes to forest restoration practices in the southern Sierra Nevada. Measurement of ecosystem outputs after fire using gauged streamflow is rare (Engle et al. 2008).

Inorganic N was elevated in stream water for 3 years after fire in the Tharps Creek watershed. Increased export of water, $SO_4^{-2}$, $Cl^-$, $SiO_2$, and base cations continued through the end of the study. This loss was calculated to be less than 1 percent of the N, up to one-third of the calicum (Ca) and magnesium (Mg), and up to three-fourths of the potassium (K) contained in the forest floor prior to combustion. Changes in watershed balances indicated that low-end natural FRIs may prevent complete reaccumulation of several elements between fires. However, this result needs to be considered in the context of the high fuel loads that had built up over 120 years. Unfortunately, we do not have data that relate nutrient losses to more historical or natural fuel loads that would have built up under a historical FRI of 10 to 20 years.

Stephan et al. (2012) compared the effects of wildfire and spring prescribed burning on headwater streams in central Idaho. They found that stream nitrate concentrations were elevated by an order of magnitude for three growing seasons after wildfire and appeared to be related positively to the percent of watershed burned in wildfire. They found no significant effects of the much less severe prescribed burning on aquatic N dynamic. Terrestrial plants and in-stream mosses were found to sequester postfire available inorganic N, which Stephan et al. (2012) interpreted as evidence that the systems were adapted to pulses following fire. For that reason, they concluded that rehabilitation measures for N cycling are not generally required after wildfire, and prescribed burning treatments may need to sometimes result in higher severity to stimulate N cycling.

**Spatial variability of nutrients on the landscape—**
Understanding soil nutrient hot spots is important for water quality and plant nutrition. "Hotspots" are areas (or patches) that show disproportionately high reaction rates relative to the surrounding soil area (or matrix). In semiarid soils, these patches have long been recognized where "islands of fertility" occur near widely spaced shrubs or patches of vegetation (Johnson et al. 2011b). Schimel and Bennett (2004) highlighted the importance of hot spots as sources of nutrients to plant roots, which are otherwise outcompeted by microbes for nutrients in the rest of the soil matrix. A review of data sets for forests in the eastern Sierra Nevada mountains showed N hot spots in soils, resin lysimeters, and resin capsules; other measured nutrients (extractable P, $Mg^{2+}$, $K^{+}$, $SO_4^{2-}$, and $Ca^{2+}$) also showed positive skew and outliers, but less so than N (Johnson et al. 2010). A recent study at KREW on the western side of the Sierra Nevada showed that nutrient hot spots occur in mixed-conifer forests for nearly all measured nutrients; these nutrients were measured using soil cores, resin collectors, resin probes, and resin capsules in 6 by 6-m plots (Johnson et al. 2011b).

Recent literature shows that the lack of rooting in the O horizon of semiarid forests (during extreme summer drought) results in a spatial decoupling of the processes of decomposition/nutrient mineralization and vegetation uptake, and a lack of the intense competition for N between roots and decomposers (Johnson et al. 2011b). Because of this vertical decoupling, nutrients released during decomposition in O horizons are not immediately taken up and are solubilized by rain or snowmelt, creating solutions with very high inorganic N and P concentrations, which presumably infiltrate the soil at preferential flow paths and contribute to hot spots and possible leaching to surface water (Miller et al. 2005). Nutrient-enriched O horizon interflow has been quantified in both the eastern (Miller et al. 2005) and western (Johnson et al. 2011b) Sierra Nevada. At KREW, values for ammonium-N

ranged from less than 0.1 to 456 µmol $L^{-1}$ (<0.1 to 6.3 mg N $L^{-1}$), $NO_3$-N concentrations ranged from less than 0.1 to 622 µmol $L^{-1}$ (<0.1 to 8.8 mg N $L^{-1}$), and ortho-P concentrations ranged from less than 0.1 to 98 µmol $L^{-1}$ (<0.1 to 3.1 mg P $L^{-1}$); upper values exceeded those found in soil solutions and stream waters at these sites by ten to a hundredfold (Hunsaker et al. 2007).

**Stream benthic macroinvertebrates—**
This synthesis examines studies in which the ecology of aquatic invertebrates in streams (fig. 7) has been used to monitor or evaluate management actions, stressors from different kinds of disturbance, and natural processes in stream ecosystems. Informed decisionmaking in the Forest Service or other agencies is founded on reliable science, so the studies reported here emphasize practical applications for management planning and design. The studies cited in this section come exclusively from the Sierra Nevada, have not been covered in previous summaries (such as Erman 1996), and include only those with some invertebrate data component.

***Development of biomonitoring tools and bioassessment programs in California—*** Use of stream invertebrates as biological indicators has become one of the most common water quality tools of regulatory agencies (Allan 1995, Rosenberg and Resh 1993). Invertebrates are especially useful indicators in small, wadeable streams in the Sierra Nevada, and in headwaters and fishless or intermittent streams. These organisms can be used as sentinels to show how much the ecological integrity of watersheds is changing, and how effective management may be in protecting these natural resource values. Adaptive management requires monitoring tools for tracking the progress of desired outcomes. Bioassessment sampling by California's Surface Water Ambient Monitoring Program (SWAMP) provides extensive new data on community composition and indicators of environmental quality. The database, which is under construction, compiles surveys from portions of the Sierra Nevada, and these will be used to develop quantitative numeric biological objectives for use in water quality monitoring programs of the state of California (Ode and Schiff 2009). The Southwest Association of Freshwater Invertebrate Taxonomists has summarized various regional taxonomic updates and species descriptions from the broader region.[7] Online documents provide lists by state, taxonomy resources, compilation of tolerance values, and functional feeding groups as sources for bioassessment, but those documents cannot currently be compiled just for the Sierra Nevada (although the SWAMP database can). Benthic macroinvertebrates are being used in a probabilistic sampling design for monitoring

[7] Southwest Association of Freshwater Invertebrate Taxonomists, http://www.safit.org.

Carolyn Hunsaker

Figure 7—Benthic macroinvertebrates are collected from streams with nets, woody material is removed, and organisms are sorted in the field before being preserved and taken back to the laboratory for identification.

of status, trend, and health in response to management activities in the Pacific Southwest Region of the Forest Service.

Hawkins et al. (2000) provided the foundation for developing predictive models (RIVPACS) for assessing stream health in the Sierra Nevada and Klamath Mountains, and those models are now being used in SWAMP to establish biological objectives for streams across California. Results suggest that logged sites had subtle losses of diversity (10 percent) compared to reference areas (<5 percent of the basin logged), with losses related mostly to reduced riparian cover rather than amount of area logged or number of roads (logging intensity and type were not specified in the analysis).

Herbst (2004) provided an overview of stream survey work done in the Sierra Nevada through 2002, including initial steps in identifying reference stream standards, monitoring of grazing and mining practices, and gaps in understanding of stream invertebrate ecology. An eastern Sierra Nevada multimetric index of biotic

integrity (IBI) and predictive models (RIVPACS) show how differing methods and analytic tools are robust in giving the same assessments of loss of stream health related to channel modifications or livestock grazing (Herbst and Silldorff 2006).

***Natural patterns of variation in stream communities in space and time—***Carter and Fend (2001) found that differences in the richness of invertebrates in riffles and pools appear to depend on annual discharge regime and are more pronounced during low discharge years and disappear when flow is higher. This study, which took place on the Merced River in the Yosemite Valley, suggests that differences in erosional and depositional features between riffles and pools diminish when flows increase, so communities become more similar. An implication of this is that it is possible that flow regulation may influence the natural variations in habitat-based diversity and that channelization (eliminating riffle-pool geomorphology) may also produce less diverse assemblages of aquatic life and more limited ecosystem processes (such as nutrient recycling, productivity, organic matter transport, conversion, and decomposition). Beche and Resh (2007) found that traits related to environmental adaptations vary in response to gradients of flow between years, from dry to wet conditions. Traits that provide adaptation to drying (e.g., desiccation resistance, aerial respiration) were more common in drought years, whereas traits permitting survival during high flows (e.g., flat body shape, drift dispersal) were more common in wet years. Prolonged drought or wet conditions result in shifts in trait composition. Despite this relationship, traits among taxa are often sufficiently redundant that taxa can be replaced without losses in trait diversity. Consequently, it may be more difficult to conserve species diversity than trait diversity in the face of a changing climate regime.

Erman (2002) studied the invertebrates of spring and springbrook (outflow) communities over 20 years to describe the biota and physical/chemical properties of Sierra Nevada cold springs. Results showed the individualistic nature of springs even within the same stream basin. Spring invertebrate assemblages differed greatly from one spring to another, as did timing of insect emergence and abundance of species. Invertebrate species richness was greater in deeper, more permanent springs, which were distinguished by high concentrations of dissolved ions, especially calcium. Spring permanence was also determined by direct observations over time, measurement of discharge variability, correlation of discharge with ionic concentration, and water dating. This study demonstrated the high conservation value of spring habitats and the high levels of diversity that can serve as a biodiversity refuge in cold-water environments.

***Fire effects*—**Beche et al. (2005) found minimal effects from prescribed fire on stream invertebrates in a Sierra Nevada study. Prescribed fire altered BMI community composition only within the first weeks postfire, but there were no lasting (1 year) impacts on BMIs. Densities and percentage of sensitive taxa were significantly reduced after an intense wildfire on Angora Creek in the Lake Tahoe basin, but there were no consistent changes in taxonomic richness or diversity (Oliver et al. 2012). Canopy cover and bank stability declined dramatically following the wildfire and substrate also changed substantially, with fine sediment more abundant and cobble less abundant postfire. There were large reductions in relative abundances of shredder and scraper taxa, whereas collector-gatherer abundances increased. Community composition shifted away from prefire configurations, and continued to diverge in the second year following the fire. Scores from a regionally derived IBI were variable, but overall they were much lower in postfire samples and did not show recovery after 2 years. This study demonstrated substantial postfire effects to aquatic ecosystems even in the absence of large flooding or scouring events, and it showed that these effects can be transmitted downstream into unburned reaches. These findings, when compared to those from Beche et al. (2005), suggest that fire effects are strongly related to fire intensity.

***Forest management practices*—**Although stream invertebrates have been adopted as good water quality indicators for the Forest Service, little published information exists regarding effects of mechanical forest management practices (road building and maintenance, tree thinning and commercial harvesting, tractor piling of slash and burning) on stream invertebrates in the Sierra Nevada. A few publications exist on prescribed fire effects on stream invertebrates (see previous discussion). The usefulness of stream invertebrates for monitoring aquatic ecosystem condition and associated information gaps were recognized in the Adaptive Management Plan, Appendix E, of the Sierra Nevada Framework (USDA FS 2001, 2004), and one new research experiment exists (see box 6.1-5 on page 290). McGurk and Fong (1995) found that there was reduced diversity and increased dominance by common taxa in stream invertebrate communities in the Sierra Nevada where equivalent roaded area (ERA, a metric of cumulative land disturbance associated with timber harvest and roads) exceeded 5 percent of a watershed.

***Flow regulation and impoundments*—**Aquatic organisms have evolved life history strategies to take advantage of high flood predictability and associated seasonal processes. The timing of the spring snowmelt recession and the shape of the recession hydrograph contribute to reproductive cues for many riparian and aquatic species, such as cottonwoods, willows, mayflies, amphibians, and salmonids. Yarnell et al. (2010) developed a conceptual model about snowmelt recession that

provides some testable hypotheses about regulated flows in streams and climate change effects on the hydrograph. As flows gradually decrease through spring, the hydrograph passes through windows of biological opportunity at magnitudes that support habitat (i.e., availability) in sufficient condition (i.e., suitability) for species persistence. Shifts in the timing of the recession or changes to the shape of the recession hydrograph that preclude suitable habitat during a particular species' window of reproduction can lead to a lack of success. Shifts in the timing of the recession may push periods of reproduction out of phase with the availability of suitable habitat. Shifts in the rate of the recession affect both abiotic and biotic conditions, creating the largest observed changes to the stream ecosystem. The effects of climate warming on aquatic ecosystems in Mediterranean-montane climates will be profound, with shifts in each of the three primary components of the recession (magnitude, timing, and rate of change). Shifts in the timing at the start of the recession and decreases in the magnitude of the flow, coupled with a shorter duration resulting from a relatively small increase in the rate of change, will alter in-stream and riparian species compositions, forcing cold-water aquatic species to inhabit higher elevations, and leading to a higher abundance of nonnative species. Shifts in the spring recession as a result of flow regulation can create similar patterns. On the basis of this conceptual model, the authors found that managed hydrographs with a flashy, short-duration spring snowmelt recession overlying a steady base flow can create channel conditions reflective of the two observed extremes in discharge, flood and base flow. Aquatic and riparian species will be reflective of the homogeneous channel conditions and lack diversity. Rehn (2008) used a reference stream dataset to establish a multimetric IBI for Sierra Nevada west-slope streams to evaluate the effects of hydropower releases on BMIs. Degradation of the invertebrate community (quantified through comparison with upstream sites) was found within 3 km of dams and was mostly related to flow regulation and constancy below dams (i.e., loss of natural flow regime). Rehn's findings support the conceptual model and hypotheses in Yarnell et al. (2010).

***Effects of floods***—Herbst and Cooper (2010) evaluated conditions before and after the 1997 New Year's Day floods for 14 small, headwater streams in the eastern Sierra Nevada. The streams showed loss of bank stability and riparian cover resulting from scour. Densities of BMIs in previously disturbed habitats increased, with increases mainly consisting of small opportunistic species (rapid-growing colonizers) feeding on fine particulate organic matter. These results show a shift to common taxa with generalized food habits. Undisturbed reference streams changed little from 1996 to 1997, suggesting that these more diverse and stable communities persist even in the face of short-term extreme flows, and are important biodiversity

refugia for downstream habitats where flooding may be more severe. These results highlight the importance of protecting the integrity of less disturbed headwater stream habitats.

***Introduced invasive species***—A paired watershed study of fishless streams with adjacent trout-stocked streams in Yosemite National Park showed that trout reduce native grazers and permit more dense growth of algae on stream rocks (Herbst et al. 2008b). This study showed losses of 20 percent of BMI taxa richness, mostly as losses of endemics and native montane species, in the presence of trout. The higher algae cover in streams with nonnative trout corresponds to more collector-gatherers and fewer predators and grazers. Conserving biodiversity and restoring natural food webs likely depends on removal of introduced trout.

**Conserving biodiversity and restoring natural food webs likely depends on removal of introduced trout.**

The New Zealand mud snail (NZMS) has caused significant disruptions in stream food chains across many trout streams of the Western United States. Herbst et al. (2008a) suggest that specific conductance levels may control which streams NZMS can colonize. In streams with specific conductance below 50 µS, snails do not survive, and at levels below 200 µS, their growth and survival are inhibited. Invasive herbivores like NZMS can have strong top-down and bottom-up influences on invaded ecosystems, but these impacts can be extremely variable across time and space.

***Information gaps***—The Sierra Nevada Ecosystem Project (SNEP) provided a summary on the status of invertebrates, highlighting high endemism (among caddisflies and stoneflies in particular) and dependence of diversity on habitat quality, but it provided little information on ecological structure, function, and ecosystem processes in streams (Erman 1996). The distribution and abundance of aquatic invertebrates in the Sierra is still mostly unknown. Many studies are localized at research areas (see fig. 1 in chapter 1.5), such as the Sierra Nevada Aquatic Research Laboratory (SNARL) on Convict Creek, Sagehen Creek Experimental Forest, the Blodgett Forest Research Station, and KREW, or where restoration project monitoring has occurred. Recent sampling done under the California SWAMP provides more survey data, but important geographic gaps remain. High elevations, intermittent streams, springs, remote regions, and whole catchments remain poorly characterized (most surveys represent only 100- to 200-m reaches within larger basins). Data from some studies show an emerging pattern of north-south distinctions in biogeography.

Researchers still do not know a lot about the biodiversity of aquatic invertebrates in the Sierra Nevada. Erman (1996) reported that species-level information was lacking for many taxa and that inventories or lists were incomplete; this statement remains true. She reported approximately 400 taxa from streams

and lakes, approximately 20 percent of which were endemic to the Sierra Nevada (see box 6.1-6 for an update). Further genetics research is likely to reveal unknown biodiversity. For example, studies of the Cascades stonefly (*Doroneuria baumanni*), which is the most common large predatory insect in the high Sierra Nevada, show genetic variation in isolated Sierra Nevada and Great Basin populations (Schultheis et al. 2012).

**Box 6.1-6**

### Ongoing Aquatic Macroinvertebrates Research in Eastern Sierra Nevada Streams

About 400 surveys in streams of the eastern Sierra Nevada have contributed to a database developed at the Sierra Nevada Aquatic Research Laboratory (SNARL) with more than 500 distinct taxa from about 200 stream surveys (see http://vesr.ucnrs.org/pages/Herbst_Research.html for more on this database). More comprehensive listings of taxa from high-elevation streams in the western Sierra Nevada, and at species-level resolution (many are only at the genus level currently), will likely place this total closer to 1,000 species or more. Reference specimen collection cataloguing is underway at SNARL for hundreds of surveys in the Sierra Nevada and Great Basin.

**Because fuels reduction activities are expected to reduce the risk of wildfires, they can be considered as an opportunity to maintain or improve water quality because wildfires can have significant impacts through increased sediment loads and phosphorus concentrations and debris flows.**

## Strategies to Promote Resilience of Water and Aquatic Ecosystems

Recent science indicates that watershed management should be guided by an understanding of disturbance regimes that recognizes the nonequilibrium nature of ecosystems (Bisson et al. 2003, Rieman et al. 2010, Wallington et al. 2005, Welsh 2011, and chapter 1.2, “Integrative Approaches: Promoting Socioecological Resilience”).

### Promoting Favorable Waterflows

Given changing climate conditions, maintaining or improving water quantity and quality in low-order streams through fuels reduction activities (mechanical thinning and prescribed fire) can be considered as a management opportunity rather than a constraint. For water quality, this approach requires a balancing of short-term, low-intensity effects against high-intensity effects from more extreme events. For example, potential for extreme wildfires is increasing with climate change. Because fuels reduction activities are expected to reduce the risk of wildfires, they can be considered as an opportunity to maintain or improve water quality because wild-

fires can have significant impacts through increased sediment loads and phosphorus concentrations and debris flows. Also, with warming temperatures, trees are expected to decrease soil moisture and increase evapotranspiration, thus leaving less water for movement to streams; mechanical thinning of trees and low-intensity underburning of vegetation would reduce evapotranspiration and help maintain soil and stream water amounts. The strategic orientation of PSW-GTR-220 (North et al. 2009) and PSW-GTR-237 (North 2013), which focuses on restoring heterogeneity and landscape-scale ecological processes, can be extended to aquatic ecosystems. The incorporation of key hydrologic and nutrient processes as treatment objectives facilitates a more holistic forest restoration effort.

Because N concentrations are a good indicator of forest health (both productivity and stress), it would be beneficial to have a few locations where N is measured periodically in wet and dry atmospheric deposition, mineral soil, and soil and stream waters. Such measurements are most likely to be done at research locations (fig. 1 in chapter 1.5), and they could continue through collaborative activities between research and forest management (examples include studies such as those reviewed in Argerich et al. [2013], Sudduth et al. [2013], and ongoing at KREW [see box 6.1-5]).

The scientific literature indicates that increased wildfire impacts in the future are a concern in terms of soil erosion, sedimentation, and impacts to water resources (Goode et al. 2012). To mitigate potential impacts to water bodies, it is important to evaluate overall watershed potential for erosion and sedimentation resulting from wildfires, as well as fuels reduction activities; timber harvest; and road construction, use, improvement, and decommissioning. Some research exists and can be augmented with monitoring data from future forest management projects.

**Addressing roads—**

Although the ability to measure or predict the hydrologic consequence of building or modifying a specific road network might be limited, general principles and models are available to decrease the negative effects of roads. These principles can be useful during upgrading or decommissioning of roads to meet various objectives. A list of principles provided by Gucinski et al. (2001) includes the following:

- Locate roads to minimize effects by conducting careful geologic examination of all proposed road locations.
- Design roads to minimize interception, concentration, and diversion potential, including measures to reintroduce intercepted water back into slow (subsurface) pathways by using outsloping and drainage structures rather than attempting to concentrate and move water directly to channels.

- Evaluate and eliminate diversion potential at stream crossings.
- Design road-stream crossings to pass not just water but also woody debris, sediment, and fish.

**Assessing water quality using stream macroinvertebrates—**
Much work remains to be done to characterize the health of Sierra Nevada streams using stream invertebrates. Mapping invertebrate distributions would provide a basic understanding of biodiversity patterns, hot spots, and biogeographic regionalization of aquatic invertebrate fauna. Future planning efforts will benefit when forestwide analyses and multiforest syntheses can be done on stream invertebrate data and stream physical properties data (Stream Condition Inventory) that already exist for the synthesis area.

Spatial analysis of watersheds, from headwaters to major river systems, would provide information on how the ecological health of interconnected stream systems changes as a function of land use disturbances, habitat fragmentation, and reservoirs/dams. Combining data from SWAMP, the regionwide Management Indicator Species Program, the Forest Service Pacific Southwest Research Station and other organizations like SNARL could yield valuable information to improve assessments of cumulative watershed affects. Furthermore, distribution data on fish and amphibians could be combined with the accumulating data on aquatic invertebrates to assist in identifying areas of high biological value (richness, endemism, index of biological integrity) for conservation management to provide a means for identifying and protecting reference areas of biological integrity. Knowledge about forest stream condition (as shown by Ode 2007) can improve substantially if invertebrate monitoring occurs at selected management projects that have not yet proceeded to an implementation phase. It is possible to use bioassessment tools to gather data using a before-after control-impact (BACI) statistical design to evaluate project outcomes in terms of aquatic invertebrate indicators of desired ecological improvements. These case histories could provide a foundation for adaptive management (monitoring informs decisions on how to proceed with actions) and advance restoration as a prescriptive science (what works and where and how long it takes) (see box 6.1-5 about KREW). Opportunities exist at the Sierra Nevada Adaptive Management Project, the Dinkey Collaborative Forest Landscape Restoration Project on the Sierra National Forest, and for similar projects where planned management affords an opportunity to learn more about prescribed fire, selective logging practices, or fuels reduction in protecting aquatic ecosystems within the context of forest ecosystem health.

## Promoting Resilience of Aquatic Ecosystems

**Landscape-scale consideration of tradeoffs in managing forests for wildfires—** Although wildfire can have negative or neutral impacts on fish, wildfire-related disturbances can also help to maintain diverse and productive habitats (Rieman et al. 2003). Many of the processes associated with wildfire disturbance have potential to benefit aquatic habitats, including contributions of nutrients, wood, and coarse substrate; reorganization of in-channel habitat structure; increases in streamflow; and increases in temperature, light, and in-stream food production (in systems that are below optimum growth levels) (Gresswell 1999). The overall impact of a wildfire on aquatic organisms depends on the specific context of that event; however, scientists have come to a general conclusion that fishes in large habitat networks are more likely to benefit even after relatively severe wildfires, whereas fishes in small, isolated systems are more vulnerable to losses (Rieman et al. 2010). A recent synthesis of short-term effects of wildfire on amphibians yielded similar conclusions, specifically that (1) wildfire can provide important benefits for amphibian diversity overall; (2) wildfire can pose threats to small, isolated, or stressed populations, particularly in the Southwest; and (3) negative effects on populations or individuals are greater in fire-suppressed forests, and high-severity burns cause greater negative effects on populations or individuals (Hossack and Pilliod 2011). In addition, some research suggests that native fishes may be better adapted to fire-associated disturbances than nonnative competitors. This relationship, combined with the potential for wildfire to extirpate or greatly reduce nonnative trout species, suggests that wildfires, even uncharacteristically severe ones, could provide important opportunities to enhance native species.

**Scientists have come to a general conclusion that fishes in large habitat networks are more likely to benefit even after relatively severe wildfires, whereas fishes in small, isolated systems are more vulnerable to losses.**

Forests may be treated to reduce the threat of uncharacteristically severe wildfire or to emulate some of the desired effects of natural disturbances, consistent with the principle of disturbance-based management (see chapter 1.2). Burton (2005) and Rieman et al. (2003) emphasized the need to carefully weigh the costs and benefits of treatments, because road networks and stream crossings that may be used to implement the treatments have potential to perpetuate impacts to streams and aquatic populations. Collectively, these studies recommend spatially explicit analysis of risks for aquatic species in the synthesis area.

**Stream and riparian restoration to promote resilience to climate change—** Restoration of stream and riparian ecosystems is a core strategy for enhancing ecological resilience to postfire impacts and climate change (see chapter 1.1), because of the important roles of streams in providing linear habitat connectivity, laterally

connecting aquatic and terrestrial ecosystems, and creating thermal refugia for cold water species such as salmonids (Seavy et al. 2009).

***Promoting connectivity*—**Increasing longitudinally connected networks from mainstem rivers to headwater tributaries has been recommended to help native trout species cope with the threat of wildfire under projected climate change (Haak and Williams 2012). However, because many remaining native fish populations have been purposefully isolated from nonnative invaders, efforts to reconnect isolated populations could leave populations exposed to potential invaders (Fausch et al. 2009). For that reason, Williams et al. (2009) observed that a shift away from an isolation strategy would require increasing efforts to reduce nonnative fishes.

***Ameliorating high temperatures*—**An analysis by Wenger et al. (2011) suggested that proactive trout conservation strategies in the face of climate change should include ameliorating high temperatures along with reducing interactions with non-native species. Stream restoration has potential to ameliorate increases in stream temperatures, reductions in base flows, and other projected effects of climate change. Restoration efforts can promote vegetation growth and channel narrowing that reduce solar exposure, and they can also promote channel complexity and associated hyporheic exchange (where surface water mixes with shallow groundwater) by developing riffles, secondary channels, and floodplain sediments (Hester and Gooseff 2010, Kondolf 2012, Poole and Berman 2001). As an example of this strategy, researchers are investigating how wet meadow restoration could increase summer discharge and reduce water temperatures to help sustain California golden trout (*Oncorhynchus mykiss aguabonita*) (see chapter 6.3).

**Restoring flow regimes on regulated rivers—**
Management of reservoirs and regulated rivers was not a focus of this synthesis, although these systems are clearly important as a source of ecosystem services in terms of recreation opportunities, flood control, water supply, and power generation (Null et al. 2010). Recent research demonstrates that the cross-cutting theme of disturbance-based management relates strategies for river management to conservation of endemic species. A recent study examined time series data for the foothill yellow-legged frog (*Rana boylii*) and California red-legged frogs (*R. draytonii*); the findings suggest that flow management that emulates natural flow regimes is likely to promote resilience of populations of these native frogs (Kupferberg et al. 2012). Dams generally reduce overall heterogeneity in flow regime, while also permitting unnaturally rapid changes in flow, such as sharp decreases in flow following spring runoff (Kupferberg et al. 2012, Moyle and Mount 2007). Climate change may also alter hydrologic regimes in ways that are similarly detrimental to these aquatic

species by reducing snowpacks, inducing earlier and more rapid snowmelt in the spring, and by extending periods of low flow in the summer and fall (Null et al. 2010). Those changes may harm species that are adapted to gradual spring recession flows (Yarnell et al. 2010), and also reduce whitewater boating opportunities. Kupferberg et al. (2012) reported a negative association between hydrologic modification, as suggested by dam height, and persistence of foothill yellow-legged frog populations. In regulated rivers, a management strategy to protect native riverine species is to emulate natural flow patterns, especially by limiting rapid fluctuations in water levels; this strategy could benefit the species directly, by helping them avoid mortality, and perhaps indirectly, by promoting changes in channel morphology and in-stream habitat (Yarnell et al. 2012). Another component of a management strategy to benefit these species could be meadow restoration efforts, to the extent that they can extend base flows longer into the summer (see chapter 6.3).

**Restoring fluvial processes—**

Researchers have emphasized the fundamental importance of restoring fluvial processes in dynamic stream systems (e.g., by removing levees or other artificial structures) as a strategy to restore fluvial form and aquatic habitats (Kondolf 2012, Kondolf et al. 2012). This approach shares close parallels with the idea of restoring fire as an ecological process, and as such, its success will depend on careful articulation of goals and understanding of the ecological and social context of the system. In particular, the more passive approach of setting aside a "zone of liberty," where natural riverine processes of deposition and erosion can occur freely, is more likely to succeed in relatively large and powerful streams or rivers (Kondolf 2012).

**Researchers have emphasized the fundamental importance of restoring fluvial processes in dynamic stream systems as a strategy to restore fluvial form and aquatic habitats.**

## Literature Cited

**Allan, J.D. 1995.** Stream ecology—structure and function of running waters. New York: Chapman and Hall. 388 p.

**Argerich, A.; Johnson, S.L.; Sebestyen, S.D.; Rhoades, C.C.; Greathouse, E.; Knoepp, J.D.; Adams, M.B.; Likens, G.E.; Campbell, J.L.; McDowell, W.H.; Scatena, F.N.; Ice, G.G. 2013.** Trends in stream nitrogen concentrations for forested reference catchments across the USA. Environmental Research Letters. 8: 014039. doi:10.1088/1748-9326/8/1/-14039.

**Bales, R.C.; Battles, J.J.; Chen, Y.; Conklin, M.H.; Holst, E.; O'Hara, L.O.; Saksa, P.; Stewart, W. 2011b.** Forests and water in the Sierra Nevada: Sierra Nevada watershed ecosystem enhancement project. Report number 11.1. Merced, CA: Sierra Nevada Research Institute. 39 p.

**Bales, R.C.; Hopmans, J.W.; O'Geen, A.T.; Meadows, M.; Hartsough, P.C.; Kirchner, P.; Hunsaker, C.T.; Beaudette, D. 2011a.** Soil moisture response to snowmelt and rainfall in a Sierra Nevada mixed-conifer forest. Vadose Zone Journal. 10: 786–799.

**Bales, R.C.; Molotch, N.P.; Painter, T.H.; Dettinger, M.D.; Rice, R.; Dozier, J. 2006.** Mountain hydrology of the western United States. Water Resources Research. 42: W08432. doi:10.1029/2005WR004387.

**Barnett, T.P.; Pierce, D.W.; Hidalgo, H.G.; Bonfils, C.; Santer, B.D.; Das, T.; Bala, G.; Wood, A.W.; Nozawa, T.; Mirin, A.A.; Cayan, D.R.; Dettinger, M.D. 2008.** Human-induced changes in the hydrology of the western United States. Science. 319: 1080–1083.

**Battin, J.; Wiley, M.W.; Ruckelshaus, M.H.; Palmer, R.N.; Korb, E.; Bartz, K.K.; Imaki, H. 2007.** Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences of the United States of America. 104(16): 6720–6725.

**Beche, L.A.; Resh, V.H. 2007.** Biological traits of benthic macroinvertebrates in California Mediterranean-climate streams: long-term annual variability and trait diversity patterns. Archiv fur Hydrobiologie. 169: 1–23.

**Beche, L.A.; Stephens, S.L.; Resh, V.H. 2005.** Effects of prescribed fire on a Sierra Nevada (California, USA) stream and its riparian zone. Forest Ecology and Management. 218: 37–59.

**Behnke, R.J. 2002.** Trout and salmon of North America. New York: The Free Press. 359 p.

**Benda, L.; Miller, D.; Bigelow, P.; Andras, K. 2003.** Effects of post-wildfire erosion on channel environments, Boise River, Idaho. Forest Ecology and Management. 178(1–2): 105–119.

**Bisson, P.A.; Rieman, B.E.; Luce, C.; Hessburg, P.F.; Lee, D.C. 2003.** Fire and aquatic ecosystems of the western USA current knowledge and key questions. Forest Ecology and Management. 178: 213–229.

**Bosch, J.M.; Hewlett, J.D. 1982.** A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. Journal of Hydrology. 55: 2–23.

**Breazeale, R.E. 1972.** An estimation of the natural geologic sedimentation rate of the North Fork Kings River. Open-file report. Clovis, CA: U.S. Department of Agriculture, Forest Service, Sierra National Forest. 61 p.

**Brekke, L.D.; Miller, N.L.; Bashford, K.E.; Quinn, M.W.T.; Dracup, J.A. 2004.** Climate change impacts uncertainty for water resources in the San Joaquin River Basin, California. Journal of the American Water Resources Association. 40: 149–164.

**Brown, D.K.; Echelle, A.A.; Propst, D.L.; Brooks, J.E.; Fisher, W.L. 2001.** Catastrophic wildfire and number of populations as factors influencing risk of extinction for Gila trout (*Oncorhynchus gilae*). Western North American Naturalist. 61(2): 139–148.

**Brown, L.R.; May, J.T.; Hunsaker, C.T. 2008.** Species composition and habitat associations of benthic algal assemblages in headwater streams of the Sierra Nevada, California. Western North American Naturalist. 68(2): 194–209.

**Burton, T.A. 2005.** Fish and stream habitat risks from uncharacteristic wildfire: Observations from 17 years of fire-related disturbances on the Boise National Forest, Idaho. Forest Ecology and Management. 211(1–2): 140–149.

**California Department of Water Resources [DWR]. 2006.** Progress on incorporating climate change into management of California's water resources. Technical memorandum report. Sacramento, CA. 339 p.

**Cannon, S.H.; DeGraff, J. 2009.** The increasing wildfire and post-fire debris-flow threat in western USA, and implications for consequences of climate change. In: Sassa, K.; Canuti, P., eds. Landslides—disaster risk reduction. Berlin: Springer-Verlag: 177–190.

**Cannon, S.H.; Gartner, J.E.; Wilson, R.C.; Bowers, J.C.; Laber, J.L. 2008.** Storm rainfall conditions for floods and debris flows from recently burned areas in southwestern Colorado and southern California. Geomorphology. 96: 250–269.

**Cannon, S.H.; Michael, J.A. 2011.** Emergency assessment of postwildfire debris-flow hazards for the 2011 Motor Fire, Sierra and Stanislaus National Forests, California. Open-File Report 2011-1251. Reston, VA: U.S. Department of the Interior, Geological Survey. 10 p.

**Carter, J.L.; Fend, S.V. 2001.** Inter-annual changes in the benthic community structure of riffles and pools in reaches of contrasting gradient. Hydrobiologia. 459: 187–200.

**Cayan, D.; Luers, A.L.; Hanemann, M.; Franco, G.; Croes, B. 2006.** Scenarios of climate change in California: an overview. CEC-500-2005-186-SF. Sacramento, CA: California Climate Change Center. 47 p.

**Coe, D.B.R. 2006.** Sediment production and delivery from forest roads in the Sierra Nevada, California. Fort Collins, CO: Colorado State University. 107 p. M.S. thesis.

**DeGraff, J.V. 1994.** The geomorphology of some debris flows in the southern Sierra-Nevada, California. Geomorphology. 10(1–4): 231–252.

**DeGraff, J.; Wagner, D.; Gallegos, A.; DeRose, M.; Shannon, C.; Ellsworth, T. 2011.** The remarkable occurrence of large rainfall-induced debris flows at two different locations on July 12, 2008, Southern Sierra Nevada, CA, USA. Landslides. 8(3): 343–353.

**DellaSala, D.A.; Williams, J.; Deacon Williams, C.; Franklin, J.F. 2004.** Beyond smoke and mirrors: a synthesis of fire policy and science. Conservation Biology. 18: 976–986.

**Dettinger, M.D. 2005.** From climate-change spaghetti to climate-change distributions for 21st century California. San Francisco Estuary and Watershed Science. 3(1): 1–14.

**Dunham, J.B.; Rosenberger, A.E.; Luce, C.H.; Rieman, B.E. 2007.** Influences of wildfire and channel reorganization on spatial and temporal variation in stream temperature and the distribution of fish and amphibians. Ecosystems. 10(2): 335–346.

**Dunne, T.; Reid, L. 1985.** Effects of the HAAS-Kings River Project on sediment transport and sedimentation. Seattle, WA: R.W. Beck Engineering Co. 166 p.

**Engle, D.L.; Sickman, J.O.; Moore, C.M.; Esperanza, A.M.; Melack, J.M.; Keeley, J.E. 2008.** Biogeochemical legacy of prescribed fire in a giant sequoia-mixed conifer forest: a 16-year record of watershed balances. Journal of Geophysical Research. 113: G01014. doi:10.1029/2006JG000391.

**Erman, N.A. 1996.** Status of aquatic invertebrates. In: SNEP Science Team, eds. Sierra Nevada ecosystem project: final report to Congress. Vol. II: Assessments and scientific basis for management options. Report No. 37. Davis, CA: Centers for Water and Wildland Resources, University of California–Davis: 987–1008. Chapter 35.

**Erman, N.A. 2002.** Lessons from a long-term study of springs and spring invertebrates (Sierra Nevada, California, U.S.A.) and implications for conservation and management. Conference proceedings. Spring-fed wetlands: important scientific and cultural resources of the Intermountain Region, 2002. Las Vegas, NV: Desert Research Institute. http://www.dri.edu. (6 January 2014).

**Farquhar, F.P. 1965.** History of the Sierra Nevada. Berkeley, CA: University of California Press. 278 p.

**Fausch, K.D.; Rieman, B.E.; Dunham, J.B.; Young, M.K.; Peterson, D.P. 2009.** Invasion versus isolation: trade-offs in managing native salmonids with barriers to upstream movement. Conservation Biology. 23(4): 859–870.

**Fenn, M.E.; Jovan, S; Yuan, F.; Geiser, L; Meixner, T.; Gimeno, B.S. 2008.** Empirical and simulated critical loads for nitrogen deposition in California mixed conifer forests. Environmental Pollution. 155: 492–511.

**Ficklin, D.L.; Luo, Y.; Stewart, I.T.; Maurer, E.P. 2012.** Development and application of a hydroclimatological stream temperature model within the Soil and Water Assessment Tool. Water Resources Research. 48: W01511. doi:10.1029/2011WR011256.

**Florsheim, J.L.; Mount, J.F.; Chin, A. 2008.** Bank erosion as a desirable attribute of rivers. BioScience. 58(6): 519–529.

**Furniss, M.J.; Roelofs, T.D.; Yee, C.S. 1991.** Road construction and maintenance. In: Meehan, W.R., ed. Influences of forest and rangeland management on salmonid fishes and their habitats. Spec. Publ. 19. Bethesda, MD: American Fisheries Society: 297–323.

**Goode, J.R.; Luce, C.H.; Buffington, J.M. 2012.** Enhanced sediment delivery in a changing climate in semi-arid mountain basins: implications for water resource management and aquatic habitat in the northern Rocky Mountains. Geomorphology. 139: 1–15.

**Goulden, M.L.; Anderson, R.G.; Bales, R.C.; Kelly, A.E.; Meadows, M.; Winston, G.C. 2012.** Evapotranspiration along an elevation gradient in California's Sierra Nevada. Journal of Geophysical Research. 117: G03028. doi:10.1029/2012JG002027.

**Gresswell, R.E. 1999.** Fire and aquatic ecosystems in forested biomes of North America. Transactions of the American Fisheries Society. 128(2): 193–221.

**Gucinski, H.; Furniss, M.J.; Ziemer, R.R.; Brookes, M.H. 2001.** Forest roads: a synthesis of scientific information. Gen. Tech. Rep. PNW-GTR-509. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 103 p.

**Gunderson, L.H. 2000.** Ecological resilience—in theory and application. Annual Review of Ecology and Systematics. 31: 425–439.

**Haak, A.L.; Williams, J.E. 2012.** Spreading the risk: native trout management in a warmer and less-certain future. North American Journal of Fisheries Management. 32(2): 387–401.

**Harp, E.L.; Reid, M.E.; Godt, J.W.; DeGraff, J.V.; Gallegos, A. 2008.** Ferguson rock slide buries California State highway near Yosemite National Park. Landslides. doi:10.1007/s10346-008-0120-9.

**Hawkins, C.P.; Norris, R.H.; Hogue, J.N.; Feminella, J.W. 2000.** Development and evaluation of predictive models for measuring the biological integrity of streams. Ecological Applications. 10: 1456–1477.

**Hayhoe, K.; Cayan, D.; Field, C.; Frumhoff, P.; Maurer, E. 2004.** Emissions pathways, climate change, and impacts on California. Proceedings of the National Academy of Sciences of the United States of America. 101: 12422–12327.

**Heede, B.H. 1980.** Stream dynamics: an overview for land managers. Gen. Tech. Rep. RM-72. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 26 p.

**Herbst, D.B. 2004.** Establishing reference conditions for streams and measuring ecological responses to management actions using aquatic invertebrate biological assessments. In: Murphy, D.D.; Stine, P.A., eds. Proceedings of the Sierra Nevada Science Symposium. Gen. Tech. Rep. PSW-GTR-193. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 133–141.

**Herbst, D.B.; Bogan, M.T.; Lusardi, R.A. 2008a.** Low specific conductivity limits growth and survival of the New Zealand mud snail from the Upper Owens River, California. Western North American Naturalist. 68: 324–333.

**Herbst, D.B.; Cooper, S.D. 2010.** Before and after the deluge: rain-on-snow flooding effects on aquatic invertebrate communities of small streams in the Sierra Nevada, California. Journal of the North American Benthological Society. 29: 1354–1366.

**Herbst, D.B.; Silldorff, E.L. 2006.** Comparison of the performance of different bioassessment methods: similar evaluations of biotic integrity from separate programs and procedures. Journal of the North American Benthological Society. 25: 513–530.

**Herbst, D.B.; Silldorff, E.L.; Cooper, S.D. 2008b.** The influence of introduced trout on native aquatic invertebrate communities in a paired watershed study of streams in Yosemite National Park. Freshwater Biology. 54: 1324–1342.

**Hester, E.T.; Gooseff, M.N. 2010.** Moving beyond the banks: hyporheic restoration is fundamental to restoring ecological services and functions of streams. Environmental Science & Technology. 44(5): 1521–1525.

**Hibbert, A.R. 1967.** Forest treatment effects on water yield. In: Sopper, W.E.; Lull, H.W., eds. Forest Hydrology. New York: Pergamon: 527–543.

**Holland, M.M.; Risser, P.G.; Naiman, R.J. 1991.** Ecotones: the role of landscape boundaries in the management and restoration of changing environments. New York: Chapman and Hall. 142 p.

**Hossack, B.R.; Pilliod, D.S. 2011.** Amphibian responses to wildfire in the western United States: emerging patterns from short-term studies. Fire Ecology. 7(2): 129–144.

**Hunsaker, C.; Bytnerowicz, A.; Auman, J.; Cisneros, R. 2007.** Air pollution and watershed research in the central Sierra Nevada of California: nitrogen and ozone. The Scientific World. 7(S1): 206–221.

**Hunsaker, C.T.; Eagan, S.M. 2003.** Small stream ecosystem variability in the Sierra Nevada of California. In: Renard, K.G.; McElroy, S.A.; Gburek, W.J.; Canfield, H.E.; Scott, R.L., eds. First interagency conference on research in the watersheds. Washington, DC: U.S. Department of Agriculture, Agricultural Research Service: 716–721. http://www.tucson.ars.ag.gov/icrw/Proceedings/Hunsaker.pdf. (6 January 2014).

**Hunsaker, C.T.; Neary, D.G. 2012.** Sediment loads and erosion in forest headwater streams of the Sierra Nevada, California. In: Webb, A.A.; Bonell, M.; Bren, L.; Lane, P.N.J.; McGuire, D.; Neary, D.G.; Nettles, J.; Scott, D.F.; Stednick, J.; Wang, Y., eds. Revisiting experimental catchment studies in forest hydrology Proceedings of a workshop held during the XXV IUGG general assembly. Wallingford, United Kingdom: International Association of Hydrological Sciences Publications. 353: 195–203.

**Hunsaker, C.T.; Whitaker, T.W.; Bales, R.C. 2012.** Snowmelt runoff and water yield along elevation and temperature gradients in California's southern Sierra Nevada. Journal of the American Water Resources Association. 48: 667–678.

**Isaak, D.J.; Luce, C.H.; Rieman, B.E.; Nagel, D.E.; Peterson, E.E.; Horan, D.L.; Parkes, S.; Chandler, G.L. 2010.** Effects of climate change and wildfire on stream temperatures and salmonid thermal habitat in a mountain river network. Ecological Applications. 20(5): 1350–1371.

**Istanbulluoglu, E.; Tarboton, D.G.; Pack, R.T.; Luce, C.H. 2004.** Modeling of the interactions between forest vegetation, disturbances, and sediment yields. Journal of Geophysical Research—Earth Surface. 109: F01009.

**Jackson, T.L. 1988.** Geologic setting and prehistoric settlement patterning in the central Sierra Nevada, California. Journal of California and Great Basin Anthropology. 10(2): 227–236.

**Jeton, A.E.; Dettinger, M.D.; Smith, J.L. 1996.** Potential effects of climate change on streamflow, eastern and western slopes of the Sierra Nevada, California and Nevada. Water Resources Investigations Report 95-4260. Sacramento, CA: U.S. Department of the Interior, Geological Survey. 44 p.

**Johnson, D.W.; Fenn, M.E.; Miller, W.W.; Hunsaker, C.F. 2009.** Effects on carbon and nitrogen cycling in forests of the Sierra Nevada. In: Bytnerowicz, A.; Arbaugh, M.; Riebau, A.; Andersen, C., eds. Developments in environmental science, vol. 8: Wildland fires and air pollution. Amsterdam: Elsevier: 405–423.

**Johnson, D.W.; Glass, D.W.; Murphy, J.D.; Stein, C.M.; Miller, W.W. 2010.** Hot spots and hot moments: another look at nutrient variability in Sierra Nevada forest soils. Biogeochemistry. 101: 93–103.

**Johnson, D.W.; Hunsaker, C.T.; Glass, D.W.; Rau, B.M.; Roath, B.A. 2011a.** Carbon and nutrient contents in soils from the Kings River Experimental Watersheds, Sierra Nevada Mountains, California. Geoderma. 160: 490–502.

**Johnson, D.W.; Miller, W.W.; Rau, B.M.; Meadows, M.W. 2011b.** The nature and potential causes of nutrient hotspots in a Sierra Nevada forest soil. Soil Science. 176(11): 596–610.

**Johnson, D.W.; Susfalk, R.B.; Dahlgren, R.A. 1997.** Nutrient fluxes in forests of the eastern Sierra Nevada Mountains, USA. Global Biogeochemical Cycles. 11: 673–681.

**Johnson, D.W.; Susfalk, R.B.; Dahlgren, R.A.; Klopatek, J.M. 1998.** Fire is more important than water for nitrogen fluxes in semi-arid forests. Environmental Science and Policy. 1: 79–86.

**Ketcheson, G.L.; Megahan, W.F.; King, J.G. 1999.** "R1–R4" and "BOISED" sediment prediction model tests using forest roads in granitics. Journal of the American Water Resources Association. 35: 83–98.

**Knowles, N.; Cayan, D.R. 2002.** Potential effects of global warming on the Sacramento/San Joaquin watershed and the San Francisco estuary. Geophysical Research Letters. 29(18): 38-1–38-4.

**Knowles, N.; Dettinger, M.D.; Cayan, D.R. 2006.** Trends in snowfall versus rainfall in the western United States. Journal of Climate. 19(18): 4545–4559. doi:10.1175/JCL13850.1.

**Kondolf, G.M. 2012.** The espace de liberté and restoration of fluvial process: When can the river restore itself and when must we intervene? In: Boon, P.J.; Calow, P.; Petts, G.E., eds. River Conservation and Management. Hoboken, NJ: John Wiley & Sons: 223–241.

**Kondolf, G.M.; Podolak, K.; Grantham, T. 2012.** Restoring mediterranean-climate rivers. Hydrobiologia. 719: 527–545.

**Kupferberg, S.J.; Palen, W.; Lind, A.; Bobzien, S.; Catenazzi, A.; Drennan, J.; Power, M.E. 2012.** Effects of altered flow regimes by dams on survival, population declines, and range-wide losses of California river-breeding frogs. Conservation Biology. 26: 513–524.

**Lave, R. 2009.** The controversy over natural channel design: substantive explanations and potential avenues for resolution. Journal of the American Water Resources Association. 45(6): 1519–1532.

**Lettenmaier, D.P.; Gan, T.Y. 1990.** Hydrologic sensitivities of the Sacramento-San Joaquin River Basin, California, to global warming. Water Resources Research. 26: 69–86.

**Lin, H.; Hopmans, J.W.; Richter, D.B. 2011.** Interdisciplinary sciences in a global network of Critical Zone Observatories. Vadose Zone Journal. 10(3): 781–785. doi:10.2136/vzj2011.0084.

**Litschert, S.E.; MacDonald, L.H. 2009.** Frequency and characteristics of sediment delivery pathways from forest harvest units to streams. Forest Ecology and Management. 259: 143–150.

**Liu, F.; Hunsaker, C.T.; Bales, R.C. 2012.** Controls of streamflow generation in small catchments across the snow-rain transition in the Southern Sierra Nevada, California. Hydrological Processes. 27(14): 1959–1972. doi:10.1002/hyp.9304.

**Luce, C.; Morgan, P.; Dwire, K.; Isaak, D.; Holden, Z.; Rieman, B. 2012.** Climate change, forests, fire, water and fish: building resilient landscapes, streams, and managers. Gen. Tech. Rep. RMRS-GTR-290. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 207 p.

**MacDonald, L.H.; Coe, D.B.; Litschert, S.E. 2004.** Assessing cumulative watershed effects in the Central Sierra Nevada: hillslope measurements and catchment-scale modeling. In: Murphy, D.D.; Stine, P.A., eds. Proceedings of the Sierra Nevada science symposium. Gen. Tech. Rep. PSW-GTR-193. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 149–157.

**Malanson, G.P. 1993.** Riparian landscapes. New York: Cambridge University Press. 308 p.

**Maurer, E.P. 2007.** Uncertainty in hydrologic impacts of climate change in the Sierra Nevada, California, under two emission scenarios. Climatic Change. 82: 309–325.

**McGurk, B.J.; Fong, D. 1995.** Equivalent roaded area as a measure of cumulative effect of logging. Environmental Management. 19: 609–621.

**Meadows, D.; Foltz, R.B.; Geehan, N. 2008.** Effects of all-terrain vehicles on forested lands and grasslands. 0823 1811-SDTDC. San Dimas, CA: U.S. Department of Agriculture, Forest Service, San Dimas Technology and Development Center. 110 p.

**Miller, J.; Germanoski, D.; Waltman, K.; Tausch, R.; Chambers, J. 2001.** Influence of late Holocene processes and landforms on modern channel dynamics in upland watersheds of central Nevada. Geomorphology. 38: 373–391.

**Miller, J.D.; Safford, H.D.; Crimmins, M.; Thode, A.E. 2009.** Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems. 12(1): 16–32.

**Miller, N.I.; Bashford, K.E.; Strem, E. 2003.** Potential impacts of climate change on California hydrology. Journal of the American Water Resources Association. 39: 771–784.

**Miller, W.W.; Johnson, D.W.; Denton, C.; Verburg, P.S.J.; Dana, G.L.; Walker, R.F. 2005.** Inconspicuous nutrient laden surface runoff from mature forest Sierran watersheds. Water Air Soil Pollution. 163: 3–17.

**Moody, J.A.; Martin, D.A. 2001.** Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surface Processes and Landforms. 26(10): 1049–1070.

**Moody, J.A.; Martin, D.A. 2009.** Synthesis of sediment yields after wildland fire in different rainfall regimes in the western United States. International Journal of Wildland Fire. 18(1): 96–115.

**Mote, P.W.; Hamlet, A.F.; Clark, M.P.; Lettenmaier, D.P. 2005.** Declining mountain snowpack in western North America. Bulletin of the American Meteorological Society. 86: 39–49. doi:10.1175/BAMS-86-1-39.

**Moyle, P.B.; Katz, J.V.E.; Quiñones, R.M. 2011.** Rapid decline of California's native inland fishes: a status assessment. Biological Conservation. 144(10): 2414–2423.

**Moyle, P.B.; Kiernan, J.D.; Crain, P.K.; Quiñones, R.M. 2012.** Projected effects of future climates on freshwater fishes of California. CEC-500-2012-028. Sacramento, CA: California Energy Commission. 53 p.

**Moyle, P.B.; Mount, J.F. 2007.** Homogenous rivers, homogenous faunas. Proceedings of the National Academy of Sciences of the United States of America. 104(14): 5711–5712. doi:10.1073/pnas.0701457104.

**Naiman, R.J.; Decamps, H., eds. 1990.** The ecology and management of aquatic-terrestrial ecotones. Nashville, TN: The Parthenon Publishing Group. 316 p.

**Neville, H.; Dunham, J.; Rosenberger, A.; Umek, J.; Nelson, B. 2009.** Influences of wildfire, habitat size, and connectivity on trout in headwater streams revealed by patterns of genetic diversity. Transactions of the American Fisheries Society. 138(6): 1314–1327.

**North, M.; Stine, P.A.; O'Hara, K.L.; Zielinski, W.J.; Stephens, S.L. 2009.** An ecosystems management strategy for Sierra mixed-conifer forests, with addendum. Gen. Tech. Rep. PSW-GTR-220. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 49 p.

**North, M. 2012.** Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 184 p.

**Null, S.E.; Viers, J.H.; Mount, J.F. 2010.** Hydrologic response and watershed sensitivity to climate warming in California's Sierra Nevada. PLoS ONE, 5(4):e9932. doi:10.1371/journal.pone.0009932.

**Ode, P.R. 2007.** Ecological condition assessment of California's perennial wadeable streams. Report to the State Water Resources Control Board's Non-Point Source Program. Rancho Cordova, CA: California Department of Fish and Game Aquatic Bioassessment Laboratory.

**Ode, P.; Schiff, K. 2009.** Recommendations for the development and maintenance of a reference condition management program (RCMP) to support biological assessment of California's wadeable streams: report to the State Water Resources Control Board's Surface Water Ambient Monitoring Program (SWAMP). Tech. Report 581. Sacramento, CA: California Environmental Protection Agency, State Water Resources Control Board. 44 p.

**Odum, W.E.; Odum, E.P.; Odum, H.T. 1994.** Nature's pulsing paradigm. Estuaries. 18: 547–555.

**Oliver, A.A.; Bogan, M.T.; Herbst, D.B.; Dahlgren, R.A. 2012.** Short-term changes in stream macroinvertebrate communities following a severe fire in the Lake Tahoe basin, California. Hydrobiologia. 694(1): 117–130. doi:10.1007/s10750-012-1136-7.

**Organic Administration Act of 1897;** Act of June 4, 1897; 16 U.S.C 475.

**Peterson, D.H.; Stewart, I.; Murphy, F. 2008.** Principal hydrologic responses to climatic and geologic variability in the Sierra Nevada, California. San Francisco Estuary & Watershed Science. 6(1): Article 3.

**Petts, G.E. 1990.** The role of ecotones in aquatic landscape management. In: Naiman, R.J.; Decamps, H. eds. The ecology and management of aquatic-terrestrial ecotones. Man and the Biosphere series, vol. 4. Paris: UNESCO: 227–261.

**Pinay, G.; Decamps, H.; Chauvet, E.; Fustec, E. 1990.** Functions of ecotones in fluvial systems. In: Naiman, R.J.; Decamps, H. eds. The ecology and management of aquatic-terrestrial ecotones. Man and the Biosphere series, vol. 4. Paris: UNESCO: 114–169.

**Poole, G.C.; Berman, C.H. 2001.** An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental Management. 27(6): 787–802.

**Raison, R.J.; Keith, H.; Khanna, P.K. 1990.** Effects of fire on the nutrient-supplying capacity of forest soils. In: Dyck, W.J.; Mees, C.A., eds. Impact of intensive harvesting on forest site productivity. Forest Research Institute Bulletin No. 159, IEA/BE T6/A6 Report No. 2. Rotorua, New Zealand: 39–54.

**Raison, R.J.; Khanna, P.K.; Woods, P.V. 1985.** Mechanisms of element transfer to the atmosphere during vegetation fires. Canadian Journal of Forest Research. 15: 132–140.

**Rehn, A.C. 2008.** Benthic macroinvertebrates as indicators of biological condition below hydropower dams on west slope Sierra Nevada streams, California, USA. River Research and Applications. 25: 208–228.

**Rice, R.; Bales, R.C.; Painter, T.H.; Dozier, J. 2011.** Snow water equivalent along elevation gradients in the Merced and Tuolumne River basins of the Sierra Nevada. Water Resources Research. 47: W08515. 11 p.

**Riebe, C.S.; Kirchner, J.W.; Granger, D.E.; Finkel, R.C. 2001.** Minimal climatic control on erosion rates in the Sierra Nevada, California. Geology. 29(5): 447–450.

**Rieman, B.E.; Hessburg, P.F.; Luce, C.; Dare, M.R. 2010.** Wildfire and management of forests and native fishes: Conflict or opportunity for convergent solutions? BioScience. 60(6): 460–468.

**Rieman, B.; Lee, D.; Burns, D.; Gresswell, R.; Young, M.; Stowell, R.; Rinne, J.; Howell, P. 2003.** Status of native fishes in the western United States and issues for fire and fuels management. Forest Ecology and Management. 178(1–2): 197–211.

**Robichaud, P.R.; Beyers, J.L.; Neary, D.G. 2000.** Evaluating the effectiveness of postfire rehabilitation treatments. Gen. Tech. Rep. RMRS-GTR-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 85 p.

**Robichaud, P.R.; MacDonald, L.H.; Foltz, R.B. 2010.** Fuel management and erosion. In: Elliot, W.J.; Miller, I.S.; Audin, L. eds. Cumulative watershed effects of fuel management in the western United States. Gen. Tech. Rep. RMRS-GTR-231. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 79–100.

**Rosenberg, D.M.; Resh, V.H. 1993.** Freshwater biomonitoring and benthic macroinvertebrates. New York: Chapman and Hall. 504 p.

**Santi, P.; Morandi, L. 2012.** Comparison of debris-flow volumes from burned and unburned areas. Landslides. 10(6): 757–769.

**Schimel, J.P.; Bennett, J. 2004.** Nitrogen mineralization: challenges of a changing paradigm. Ecology. 85: 591–602.

**Schultheis, A.S.; Booth, J.Y.; Perlmutter, L.R.; Bond, J.E.; Sheldon, A.L. 2012.** Phylogeography and species biogeography of montane Great Basin stoneflies. Molecular Ecology. 13: 3325–3340. doi:10.1111/j.1365-294X.2012.05599.x.

**Seavy, N.E.; Gardali, T.; Golet, G.H.; Griggs, F.T.; Howell, C.A.; Kelsey, R.; Small, S.L.; Viers, J.H.; Weigand, J.F. 2009.** Why climate change makes riparian restoration more important than ever: recommendations for practice and research. Ecological Restoration. 27(3): 330–338.

**SNEP Science Team, eds. 1996.** Status of the Sierra Nevada, Sierra Nevada ecosystem project: final report to Congress. Vol. I: assessment summaries and management strategies. Report No. 36. Davis, CA: Centers for Water and Wildland Resources, University of California–Davis. 209 p.

**Stafford, A.K. 2011.** Sediment production and delivery from hillslopes and forest roads in the southern Sierra Nevada, California. Fort Collins, CO: Colorado State University. 190 p. M.S. thesis.

**Stednick, J.D. 1996.** Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology. 176(1–4): 79–95.

**Stednick, J.D. 2000.** Timber management. In: Dissmeyer, G.E., ed. Drinking water from forest and grasslands: a synthesis of the scientific literature. Gen. Tech Rep. GTR-SRS-39. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 103–119.

**Stephan, K.; Kavanagh, K.L.; Koyama, A. 2012.** Effects of spring prescribed burning and wildfires on watershed nitrogen dynamics of central Idaho headwater areas. Forest Ecology and Management. 263: 240–252.

**Stewart, I.T.; Cayan, D.R.; Dettinger, M.D. 2004.** Changes in snowmelt runoff timing in western North America under a “business as usual” climate change scenario. Climatic Change. 62: 217–232.

**Sudduth, E.B.; Perakis, S.S.; Bernhardt, E.S. 2013.** Nitrate in watersheds: Straight from soils to streams? Journal of Geophysical Research: Biogeosciences. 118: 1–12. doi:10.1002/jgrg.20030.

**Sugihara, N.G.; van Wagtendonk, J.W.; Fites-Kaufman, J.; Shaffer, K.E.; Thode, A.E. 2006.** Fire in California's ecosystems. Berkeley, CA: University of California Press. 596 p.

**Timmer, K.; Suarez-Brand, M.; Cohen, J.; Clayburgh, J. 2006.** State of Sierra waters: a Sierra Nevada watersheds index. South Lake Tahoe, CA: Sierra Nevada Alliance. 170 p.

**Troendle, C.A.; MacDonald, L.H.; Luce, C.H.; Larsen, I.J. 2010.** Fuel management and water yield. In: Elliot, W.J.; Miller, I.S.; Audin, L., eds. Cumulative watershed effects of fuel management in the western United States. Gen. Tech. Rep. RMRS-GTR-231. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 126–148.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 1999.** Roads analysis: informing decisions about managing the national forest transportation system. Misc. Rep. FS-643. Washington, DC: 222 p.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2001.** Sierra Nevada forest plan amendment: final environmental impact statement. Vallejo, CA: Pacific Southwest Region. http://www.sierraforestlegacy.org/Resources/Conservation/LawsPoliciesRegulation/KeyForestServicePolicy/SierraNevadaFramework/Framework-FSROD01.pdf. (December 13, 2013).

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2004.** Sierra Nevada forest plan amendment: final supplemental environmental impact statement. Vallejo, CA: Pacific Southwest Region. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsBdev3_046095.pdf. (December 13, 2013).

**Vicuna, S.; Leonardson, R.; Hanemann, M.W.; Dale, L.L.; Dracup, J.A. 2008.** Climate change impacts on high elevation hydropower generation in California's Sierra Nevada: a case study in the Upper American River. Climatic Change. 87: 123–137.

**Viers, J.H.; Rheinheimer, D.E. 2011.** Freshwater conservation options for a changing climate in California's Sierra Nevada. Marine and Freshwater Research. 62: 266–278.

**Wallington, T. J.; Hobbs, R.J.; Moore, S.A. 2005.** Implications of current ecological thinking for biodiversity conservation: a review of the salient issues. Ecology and Society. 10(1): article 15. http://www.ecologyandsociety.org/vol10/iss1/art15/. (December 13, 2013).

**Welsh, H.H. 2011.** Frogs, fish and forestry: an integrated watershed network paradigm conserves biodiversity and ecological services. Diversity. 3: 503–530. doi:10.3390/d3030503.

**Wenger, S.J.; Isaak, D.J.; Luce, C.H.; Neville, H.M.; Fausch, K.D.; Dunham, J.B.; Dauwalter, D.C.; Young, M.K.; Elsner, M.M.; Rieman, B.E.; Hamlet, A.F.; Williams, J.E. 2011.** Flow regime, temperature, and biotic interactions drive differential declines of trout species under climate change. Proceedings of the National Academy of Sciences of the United States of America. 108(34): 14175–14180.

**Williams, J.E.; Haak, A.L.; Gillespie, N.G.; Colyer, W.T. 2007.** The conservation success index: synthesizing and communicating salmonid condition and management needs. Fisheries. 32(10): 477–492.

**Williams, J.E.; Haak, A.L.; Neville, H.M.; Colyer, W.T. 2009.** Potential consequences of climate change to persistence of cutthroat trout populations. North American Journal of Fisheries Management. 29(3): 533–548.

**Yarnell, S.M.; Viers, J.H.; Mount, J.F. 2010.** Ecology and management of the spring snowmelt recession. BioScience. 60: 114–127.

**Yarnell, S.; Lind, A.; Mount, J. 2012.** Dynamic flow modeling of riverine amphibian habitat with application to regulated flow management. River Research and Applications. 28: 177–191.

# Chapter 6.2—Forested Riparian Areas

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## Summary

Riparian areas are typically highly productive areas that sustain important socioecological benefits, including the capacity to modulate effects of watershed disturbances on aquatic systems. Recent studies have shown that fire behavior in riparian areas varies with landscape attributes. Smaller, headwater riparian areas often burn similarly to adjacent uplands, whereas riparian areas next to larger streams (4$^{th}$ order and higher) often burn less frequently and less severely because of moister microclimates, and therefore can serve as fire breaks within a landscape. However, other riparian areas may accumulate fuels rapidly owing to their high productivity, and during dry fire seasons they can serve as wicks that carry high-intensity fire through a landscape. These localized relationships with fire suggest that treatment strategies for riparian areas should be customized and likely would differ. However, riparian areas that are vulnerable to uncharacteristically high-severity fire may benefit from being included in upland treatments to render them and their associated landscapes more resilient to wildfire. Furthermore, treatments that reduce tree density and increase light may have positive effects on understory plant diversity and aquatic productivity in some riparian areas, including those with aspen. Studies on prescribed fire in Sierra Nevada riparian areas have found relatively benign impacts. However, information about the effects of both mechanical treatments and fire treatments is still relatively limited, which suggests a need for experimental treatments. Overall, an adaptive management strategy based upon active management within some riparian areas may promote resilience better than a broad hands-off approach.

## Introduction

Riparian areas are important transition zones between terrestrial and aquatic ecosystems that can modulate effects from the watershed and provide valuable socioecological benefits. This chapter uses the term **riparian area** broadly to describe the "stream-riparian corridor," which consists of the stream channel,

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adjacent floodplains, and the transitional upland fringe, as defined by Dwire et al. (2010).[3] When Dwire et al. (2010) synthesized the state of knowledge about the potential impacts of streamside and upland fuels management on riparian areas, they found that most information was derived from studies on the effects of forest harvest or wildland fire. Although research about fire history strongly suggests a need for treatments within many riparian areas, limited information about the effects and effectiveness of mechanical treatments and prescribed fire treatments currently limits guidance for managing these valuable riparian ecosystems. As a consequence, these systems continue to present an important opportunity for research on riparian responses to treatments as well as to fires of different severities.

**Fire is an agent of renewal and redistribution in riparian and aquatic systems within large landscapes over long periods.**

Luce et al. (2012) provided a timely and relevant synthesis of information concerning strategies for promoting resilience in riparian and aquatic ecosystems in the face of wildfire and climate change. They emphasized several important functions of riparian areas, including provision of shade, inputs of large woody debris and allochthonous organic matter, streamside habitat, and bank stability. One of their central themes is the role of fire as an agent of renewal and redistribution in riparian and aquatic systems within large landscapes over long periods.

## Fire History and Behavior in Riparian Areas

Riparian plant communities evolved within the ecological context of regional fire regimes. A broader review of fire and fuels in the synthesis area is provided in chapter 4.1, "Fire and Fuels." Luce et al. (2012) presented four generalized scenarios of fire behavior and effects in riparian areas:

- Riparian areas burn like adjacent uplands because of similar vegetation and topography;
- Riparian areas burn less frequently or less severely than adjacent uplands because of soil and terrain that maintain moist microclimates;
- Riparian areas serve as fire breaks, particularly on large, perennial streams;
- Riparian areas burn more frequently or more severely than adjacent uplands where fuel loads are higher along low-order streams in steep terrain with south-facing aspects.

[3] Riparian areas have been defined in the planning rule by the Forest Service as "three-dimensional ecotones of interaction that include terrestrial and aquatic ecosystems that extend down into the groundwater, up above the canopy, outward across the floodplain, up the near-slopes that drain to the water, laterally into the terrestrial ecosystem, and along the water course at variable widths" (Office of the Federal Register 2012: 1411).

Research findings provide support for these different scenarios. Data from perennial streams in the Klamath Mountains suggest that fire return intervals (FRIs), and possibly fire behavior, are more variable within riparian zones than in adjacent uplands. Skinner (2003) found that median FRIs were generally twice as long on riparian sites than on neighboring uplands, but found no substantial differences in the range of FRIs between the two landscape types. Taylor and Skinner

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Figure 1— (A) An intermittent stream and (B) a perennial stream (Hull Creek), both on the Stanislaus National Forest, demonstrate some of the diversity in riparian habitats within the synthesis area.

(2003) found that areas with similar timing of fires were several hundred hectares in size and bounded by topographic features (e.g., ridgetops, aspect changes, riparian zones, and lithologic units) that affect fuel structure, fuel moisture, and fire spread. However in very dry years, fires would spread across such boundaries. Thus, by affecting fire spread, riparian areas contribute to the structure and dynamics of upland forest landscapes (Skinner 2003, Skinner et al. 2006, Taylor and Skinner 2003). This is an example of the linkage between aquatic and terrestrial ecosystems.

Using remotely sensed Burned Area Reflectance Classifications from four fires in the Intermountain Region, Fisk et al. (2004) found that riparian areas burned less severely than upland areas in general, but that lower order streams burned more like uplands, and that slope and aspect were more watershed-specific factors. Luce et al. (2012) summarized the average FRI for riparian areas across five studies in dry forests as 12 to 36 years. That value is very similar to the return interval of 10 to 31 years for uplands in the same studies. However, they found that the average FRI in mesic forest types, based upon two studies from the Klamath Mountains in California and the Cascade Range in Oregon, is much longer: 26 to 41 years in riparian versus 17 to 25 years in associated uplands. Two of the studies included in the synthesis by Luce et al. (2012) are discussed in greater detail below.

**Research in the Sierra Nevada has indicated that riparian forests have higher fuel loads than adjacent uplands, and that on smaller and more incised streams, forested riparian areas have fire histories similar to adjacent uplands.**

First, research in the Sierra Nevada has indicated that riparian forests have higher fuel loads than adjacent uplands, and that on smaller and more incised streams, forested riparian areas have fire histories similar to adjacent uplands (Van de Water and North 2010, 2011). Conducted at 36 sites in the northern Sierra Nevada (Lassen National Forest, Onion Creek Experimental Forest, and Lake Tahoe Basin), these studies developed dendrochronological fire records in adjacent riparian and upland areas across a variety of forest and stream conditions. They sampled first- through fourth-order streams, with a particular focus on first- and second-order streams. Riparian and upland FRIs were significantly different in only one quarter of the sites they sampled. They found that the historical seasonality of fire did not differ between riparian and upland areas; in both, fires typically occurred in late summer to early fall. Riparian FRIs ranged from 8.4 to 42.3 years. Fire return intervals were shorter in forests with a higher proportion (>23 percent) of pine species, sites east of the Sierra Nevada crest, lower elevation sites (<1944 m), and riparian zones bordering narrower, more incised streams (width/depth ratio <6.2).

Second, a recent study of two fires in southern Oregon similarly reported that smaller headwater streams had characteristics similar to adjacent uplands (such as low composition of riparian deciduous hardwoods) that were associated with high riparian fire severity (Halofsky and Hibbs 2008). Research in dry inland forests of Oregon also showed that historical fire frequencies in riparian areas

were comparable to those in adjacent uplands (the differences were not statistically significant), but high patchiness and mixed severity meant that many fires occurred only at a riparian plot or only in an upslope plot within a pair, but not at both (Olson and Agee 2005). In some areas, riparian fires may also be less frequent but more severe than those in uplands (Arno 1996, Everett et al. 2003). Aspect may be an important factor within landscape areas, as Everett et al. 2003 found that fire frequencies were more similar across site types on north-facing aspects (higher moisture and cooler temperatures) than on south-facing slopes. These studies demonstrate the wide variation in relationships between fire regimes across the riparian-upland interface.

### Wildfire Effects on Riparian Areas

Kobziar and McBride (2006) studied the relationships between wildfire burn patterns, stream channel topography, and the short-term response of riparian vegetation to the Lookout Fire along two creeks in mixed-conifer forest in the northern Sierra Nevada (Plumas National Forest). The study streams were perennial (3 m wide) with 7.4- to 9.9-m-wide riparian corridors on their southern aspects. One stream burned at lower severity, with 53 percent of transects at low to moderate severity and 47 percent at moderate to high severity. In the other stream, 86 percent of transects burned at low to moderate severity and 14 percent burned at moderate to high severity. The entire riparian corridor burned only 14 to 26 percent of the time, and one-third of the study transects were not burned. The authors noted that wider floodplain terraces supported mountain alder, which has been shown to slow backing wildfires moving toward streams. That study found that postfire seedling recruitment and sprouting allowed riparian vegetation to be resilient and maintain stream quality even following high-severity fire. Wildfire effects on streams and aquatic systems are discussed more in chapter 6.1, “Watershed and Stream Ecosystems.”

## Influences on Riparian Functions

### Stream Order

Distinctions between headwater streams and larger stream orders may be relevant for predicting fire effects and for disturbance-based management. The definition of headwater streams often differs, although first through third order may be a reasonable division for parts of the synthesis area. For example, streams of those orders often have very narrow riparian areas (1 to 3 m on a side in the Kings River Experimental Watersheds [KREW]) that have a unique plant community from the adjacent uplands (Dolanc and Hunsaker 2007). These distinctions may have an influence on management plans, because first- to third-order streams represent

approximately 90 percent of all streams in the continental United States (Leopold et al. 1995). Agreement on delineation rules and verification of stream order and flow regime in the field is necessary to determine the extent of different stream types and to direct management to protect water quality and aquatic habitats (Hansen 2001). Streams at the fourth-order size up to large rivers usually support wider riparian areas and create a larger, moister microclimate; these downstream riparian areas likely impede some fires from burning all or some of their vegetation or crossing their stream channels.

## Large Wood

Scientific literature has described the hydrological, ecological, and geomorphic effects of in-stream large wood and reported on the important role that large wood plays in linking upland, riparian, and aquatic portions of watershed ecosystems. Wohl and Jaeger (2009) provide a conceptual model of large wood loads and spatial distribution in streams of the Colorado Front Range that is summarized here as the findings may inform issues in the synthesis area. They note that in-stream large wood (LW) loads are generally highest in the headwater reaches, where trees are large and small channel size and stream power limit transport. Intermediate reaches often exhibit a dynamic equilibrium where LW pieces are moved out at approximately the same rate that they enter the reach. In headwaters and intermediate reaches, landscape disturbances like fire, windthrow, landslides, and debris flows are responsible for delivering large pulses of wood to streams. However, large, low-gradient streams and rivers are supply limited because of the larger proportion of open water compared to riparian contact area. Wohl and Jaeger (2009) surmise that stream reaches at lower elevations in the Colorado Front Range may have a deficit in LW resulting from historical reductions in supply and active removal of wood, whereas loads in higher elevation streams may be closer to a historical reference condition. If similar patterns occur for streams in the synthesis area, then their conceptual model might suggest that short-term reductions in LW from active management would pose little risk in small, low-order streams. However, Wohl and Jaeger (2009) also noted that local recruitment of wood is limited in reaches along large meadows and bedrock outcrops, which do occur in parts of the synthesis area.

**Short-term reductions in large wood from active management would pose little risk in small, low-order streams.**

## Microclimate Effects

Riparian areas are supported by a moister, three-dimensional air and soil microclimate as compared with adjacent uplands. Rambo and North (2009) compared microclimate (air temperature and humidity) gradients in trees from near the forest floor up through the canopy for both upland and riparian-influenced forest trees

(three trees for each landscape type). The study area was in the Teakettle Experimental Forest in old-growth mixed-conifer forest that received one of three treatments: none, understory thinning, or overstory thinning. Measurements were made at 5, 15, 25, 35, and 45 m above the forest floor. Riparian microclimate had significantly lower minimums and means, and greater daily ranges of temperatures and humidity. The largest temperature and humidity ranges were near the stream and forest floor. In summer, steep slopes cause drainages to be warmer than ridge tops from upslope winds in daytime, and cooler at night because of the downslope flow of cold air from surrounding higher terrain. Accumulation of cold air at night can result in a local temperature inversion in drainages. This phenomenon acts in conjunction with stream influence, which directly cools air temperature and indirectly supplies water for daytime evaporative cooling via plant transpiration. In another study assessing changes in microclimate conditions both vertically and horizontally from Teakettle Creek, Rambo and North (2008) found a very narrow area around the stream (< 5.0 m vertically and < 7.5 m horizontally) in which microclimate conditions differed from upland.

## Recent Research

### Prescribed Burning

Beche et al. (2005) published one of the few studies that focused on effects of prescribed fire in a Sierra Nevada riparian area. They examined prescribed fire effects in a mixed-conifer forest of the northern Sierra Nevada by comparing characteristics of the stream and its riparian zone in the burned watershed with those of five unburned watersheds (first- and second-order streams of low gradient). Effects were measured immediately and up to 1 year after the fire and compared with conditions 1 to 7 years prefire. They concluded that the prescribed fire either had no or short-lasting (≤1 year) impacts on the stream and its riparian zone. The prescribed fire in the riparian zone was patchy in terms of intensity, consumption, and severity; it consumed 79 percent of prefire fuels, 34 percent of total surface fuels, and 90 percent of total ground fuels. The prescribed fire significantly reduced percentage of cover of surface vegetation and plant taxa richness in comparison with unburned sites, but not plant diversity (Simpson's D). Community composition of understory riparian vegetation changed postfire, most likely as a result of the reduction in taxa richness and cover. Postfire riparian tree mortality (>11.5 in diameter at breast height) was only 4.4 percent. No postfire change occurred in large woody debris volume and recruitment or in the amount of fine sediment in pools. Some water chemistry parameters increased ($SO^{4-}$, total P, $Ca^{2+}$, and $Mg^{2+}$), and periphyton biomass decreased; however, these changes were short term (≤1 year).

**Prescribed fire either had no or short-lasting (≤1 year) impacts on the stream and its riparian zone.**

Macroinvertebrate community composition was affected 10 to 19 days postfire, but density, richness, and diversity were unaffected; furthermore, composition recovered within 1 year. These effects are discussed in more detail in chapter 6.1. Beche et al. (2005) explained that the limited observed impacts may be a result of the small portion (<20 percent) of the watershed area that burned, moderate topography, the low to moderate severity of the fire, and the below-average precipitation year that followed the fire.

In a study from the Lake Tahoe basin, prescribed burning in areas that included some ephemeral channels showed short-term (3-month) increases in calcium and pH but not a significant increase in the amount of soluble reactive phosphorus in stream waters (Stephens et al. 2004: 258).

## Aspen Management

Aspen stands provide important ecological services, including habitat for diverse wildlife and distinctive understory plants (Kuhn et al. 2011). Recent studies have demonstrated the benefits of selective conifer removal in restoring aspen stands; studies have taken place on the Eagle Lake Ranger District (ELRD) and the Lassen National Forest (Jones et al. 2005, 2011). At the sites in Lassen, the removal treatments were conducted in concert with control of heavy grazing pressure, and harvest was selected over the use of fire to avoid damage to the aspen trees. Jones et al. (2005) reported that hand pile burning within the treated stands killed aspen roots and appeared to inhibit regeneration. Another recent study on the east side of the Sierra Nevada (Krasnow et al. 2012) found that conifer thinning was effective in stimulating aspen release, although one stand that lost old aspen trees may have been past a threshold for restoration. The study also found that prescribed burning could be an effective restoration tool for aspen, although it noted that wildland fire use that resulted in higher intensity might be more effective. A detailed and comprehensive review of aspen ecology management in the Sierra Nevada is found in Shepperd et al. (2006); with regard to water resources, it is important to note that they suggested a possible benefit to water yield of restoring aspen stands, although studies suggest that more research is needed on that topic.

**Prescribed burning could be an effective restoration tool for aspen, although wildland fire use that resulted in higher intensity might be more effective.**

## Terrestrial Amphibians

Effects of fires of varying severities and timber harvest, and their interactions, on terrestrial amphibians have been proposed as an important topic for research (Hossack and Pilliod 2011). Appendix E of the Sierra Nevada Forest Management Plan amendment identified effects of fuels treatments as an important research topic, with a focus on site occupancy by the foothill yellow-legged frog. Other species

that may be important to consider are slender salamanders in the genus *Batrachoseps*, within which several new species have been recently described in the Southern Sierra Nevada (Jockusch et al. 2012). A synthesis of wildfire effects on amphibians noted that four studies found negative effects on populations or individuals on lungless salamanders, especially in uncharacteristically severe burns (Hossack and Pilliod 2011). One of the four studies was from southern California, where large fires in 2003 reduced occupancy by the slender garden salamander (*B. major*) in burned chaparral; the authors suggested that the effect may have resulted from a reduction in moist litter rather than be a direct effect of the fire, because the amphibians tend to move underground during the summer wildfire season (Rochester et al. 2010). Within the synthesis area, Bagne and Purcell (2009) examined effects of spring prescribed burning in ponderosa pine forest on two species of terrestrial salamanders (*Ensatina eschscholtzi platensis* and *B. gregarius*). They found no strong adverse effects of the treatment, which resulted in patchy burn effects, although they cautioned that sample sizes in the study were small. A review of studies suggests that heterogeneity of burn patches could be important in maintaining resilient populations of these amphibians (Hossack and Pilliod 2011).

**Box 6.2-1**

### Pending Research on Pile Burning

Recent research in the Lake Tahoe basin that examined effects of pile burning in riparian areas suggested that in most management settings, potential soil effects did not appear to be an overriding concern (see chapter 5.1, "Soils," for details on soil heating, although water quality results of that study are still in review).

Researchers from Humboldt State University have conducted research on pile burning in aspen stands in the Tahoe basin; although published studies are pending, the authors have released a report (Dagley et al. 2012).

## Research Gaps and Management Implications

Dwire et al. (2010) concluded that there is little information about specific and cumulative impacts of different fuels reduction treatments on riparian and aquatic ecosystems. Stone et al. (2010) similarly concluded that additional experimental studies of fuels treatment effects on aquatic and riparian ecosystems are needed before generalizations can be made across different forest types and local conditions. Study results are often quite variable and confounded by local effects of other past and current management activities (Wondzell 2001).

Luce et al. (2012: 52) enumerated a number of challenges for riparian management: "(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." They placed significant emphasis on the need for basic inventory and monitoring data about aquatic ecosystems, including stream temperature data and detailed mapping of riparian and aquatic habitats.

The 1996 Sierra Nevada Ecosystem Project (SNEP) report included a chapter with recommendations for riparian management that included a prohibition on vegetation removal and ground disturbance within riparian zones, which was intended to benefit both riparian and aquatic habitats (Kondolf et al. 1996). That section emphasized the importance of riparian tree canopies within first- and second-order streams in blocking summer sun and moderating water temperatures, as well as stream loading of large wood and other organic matter from riparian trees. It also suggested a fixed buffer width of 150 ft based on typical tree heights in the Sierra Nevada, and it recommended adopting wider, variable buffer widths that could be increased to account for variation in the riparian community and hillslope and soil erodibility. They asserted that "even the natural role of disturbance…does not require, in most situations, active restoration of the landscape in order to secure the habitat conditions necessary for the area" (Kondolf et al. 1996: 1026). The SNEP recommendations are similar to those established by FEMAT (1993) in the Pacific Northwest around the same time. The Sierra Nevada Framework (USDA FS 2001, 2004) established similar buffers with restricted activities; however, this planning effort also called for research on management in riparian areas. However, recent science has shown that higher stem densities and fuel loads in riparian forests can serve as a wick for high-intensity fire to move within treated upland forests under some conditions, such as the Angora Fire in the Lake Tahoe basin (Murphy et al. 2007, Pettit and Naiman 2007, Van de Water and North 2011). More studies of variation across riparian areas are needed, but limited evidence does suggest that some of these forests are vulnerable to uncharacteristically high-severity fires under severe weather conditions; as a result, scientists have noted the importance of considering treatments in riparian areas as part of landscape-scale restoration strategies (Messier et al. 2012, Van de Water and North 2011).

**Limited evidence suggests that some riparian forests are vulnerable to extreme fires and may benefit from treatment as part of landscape-scale restoration strategies.**

Broad principles based upon recent science discussed in this synthesis suggest that more active management within riparian areas, including mechanical harvest, could promote resilience to uncharacteristically severe wildfire. The principles

of restoring upland forests described in chapter 1.2, "Integrative Approaches: Promoting Socioecological Resilience," can extend to riparian areas. For example, it may be appropriate to design treatments to increase heterogeneity where it has been reduced. Customization to local conditions and consistency with principles designed to promote resilient soils (see chapter 5.1) would help to develop specific treatment approaches. Luce et al. (2012) suggested scenarios in which short-term risk may need to occur to promote a long-term benefit. For example, they considered that extending fuels reduction treatments into riparian areas may reduce effective shade for several years, while reducing the potential for severe wildfire and ultimately sustaining shade benefits over several decades.

Effects of fire suppression and lack of active treatment have contributed to high fuel loads, increased tree density, and shifted vegetation composition to less fire-resistant species in riparian areas as well as in uplands. Treatments should reduce the likelihood of high-severity wildfires where they are not characteristic of the landscape. Riparian areas support important resource values, they are well adapted to recovery from disturbance, and even uncharacteristically high-severity fires may not necessarily impair long-term recovery of key functions. Outcomes may depend on the extent and severity of fire in the surrounding landscape and the vulnerability of downstream aquatic resources (see chapter 6.1). Better information is needed to understand how uncharacteristically severe fire may alter trajectories in riparian areas over a range of time scales relevant to understanding particular ecological processes (such as aquatic life cycles, channel organization, recruitment of woody debris, etc.).

Rieman et al. (2003) stated that objectives for fuels reduction treatments should include the return to fuel loads that support ecosystem processes and natural disturbance regimes and incorporate short- and long-term targets for the vegetation condition of uplands and riparian areas. Fuel loads in many riparian forests are so high that mechanical treatments may be needed to reduce fuels to levels that facilitate safer reintroduction of fire. Studies in uplands show that mechanical fuels reduction treatments, if conducted properly (i.e., reducing surface and ladder fuels), can effectively reduce fire severity under most weather conditions (Safford et al. 2012). These treatments should work just as effectively in riparian areas, although higher productivity in riparian areas may necessitate more frequent maintenance.

Treating densely stocked riparian areas may not only offer benefits in terms of wildfire risk reduction and promotion of shade-intolerant riparian vegetation, but it may also yield benefits to aquatic systems. This idea has gained traction in the Pacific Northwest, where the use of riparian management areas has been widely adopted. For example, evidence suggests that light limitation of primary production

often overrides nutrient limitation in small, forested streams (Wilzbach et al. 2005). Bisson et al. (2005) reported that following the 1980 eruption of Mount St. Helens, fish populations thrived in what would otherwise be considered undesirable stream temperatures owing to the presence of abundant food supplies. An early study by Murphy et al. (1981) had reported that small, steep streams traveling through clearcuts in the Oregon Cascade Range had greater aquatic productivity than streams in shaded forest reaches, although they cautioned that such treatments might have imposed effects further downstream. In a similar vein, Newton and Cole (2005) reported that stream buffers on the south side of streams in western Oregon appeared to support increased production of benthic insects while avoiding creation of temperature hot spots. Although these studies do not necessarily test proposed management strategies for the synthesis area, they point to the potential benefits of reducing riparian forest canopies in western forests, which should be considered as part of an adaptive management framework.

Riparian treatments would need to be evaluated and monitored to assess impacts and guide approaches in the future. There may be valuable opportunities to better link management and research. For example, Stone et al. (2010) interviewed Forest Service fire management officers in 11 Western States and found that 43 percent were conducting fuels reduction treatments in riparian areas (California had 7 of 12 districts with riparian treatments). Although 88 percent of the districts reported monitoring activities to evaluate the effectiveness or ecological effects of the fuels reduction treatments in riparian areas, most monitoring was qualitative or not collected with sufficient spatial and temporal replication for quantitative summaries.

The special nature of these systems warrants developing localized prescriptions based in part upon historical fire regimes. For instance, approaches should differentiate riparian areas that function similar to upland landscapes in terms of fire frequency and spread; as discussed earlier, stream order may be a useful distinguishing characteristic. Van de Water and North (2010) suggest that the following riparian types could probably be treated similarly to upland areas, including:

- Lower elevation riparian areas;
- Riparian areas adjacent to small, incised headwater streams that historically experienced fire at frequencies similar to those of upland areas; and
- Riparian areas surrounded by forests with a high proportion (about one-third of the basal area or greater) of fire-tolerant pines, especially those on the east side of the Sierra Nevada.

For other kinds of forested riparian areas, including those at higher elevations and those bordering wider streams, they recommended considering less intensive treatments, such as hand thinning and pile burning small trees (Van de Water and North 2010). Understanding the life history requirements of aquatic taxa will help to inform prescriptions in different contexts. For example, the Cascades frog (*Rana cascadae*) is an example of a species that appears to need open-canopy basking sites along the riparian corridor (Pope et al. 2014).

An ongoing research experiment in eight Sierra Nevada watersheds in the mixed-conifer zone will provide new insight into restoration treatments in headwater riparian areas for both mechanical thinning and prescribed fire (see the box on KREW in chapter 6.1). However, the research gap is so large that more adaptive management research is needed to develop guidelines for mechanical and prescribed fire treatments in riparian areas within the synthesis area. Consequently, the approach of large experimental areas outlined in chapter 1.2 might incorporate adaptive management experiments within riparian areas to help fill this gap.

**Adaptive management research is needed to develop guidelines for mechanical and prescribed fire treatments in riparian areas within the synthesis area.**

## Literature Cited

**Arno, M.K. 1996.** Reestablishing fire-adapted communities to riparian forests in the ponderosa pine zone. In: Arno, S.F.; Hardy, C.C., eds. The use of fire in forest restoration. INT-GTR-341. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 42–43.

**Bagne, K.E.; Purcell, K.L. 2009.** Response of two terrestrial salamander species to spring burning in the Sierra Nevada, California. Res. Note RMRS-RN-41. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 10 p.

**Beche, L.A.; Stephens, S.L.; Resh, V.H. 2005.** Effects of prescribed fire on a Sierra Nevada (California, USA) stream and its riparian zone. Forest Ecology and Management. 218: 37–59.

**Bisson, P.A.; Crisafulli, C.M.; Fransen, B.R.; Lucas, R.E.; Hawkins, C.P. 2005.** Response of fish to the 1980 eruption of Mount St. Helens. In: Dale, V.H.; Swanson, F.J.; Crisafulli, C.M., eds. Ecological responses to the 1980 eruption of Mount St. Helens. New York: Springer: 164–181.

**Dolanc, C.R.; Hunsaker C.T. 2007.** Riparian and upland vegetation on the Kings River Experimental Watershed, Sierra Nevada, California. In: Powers, R.F., ed. Restoring fire-adapted ecosystems: proceedings of the 2005 national silviculture workshop. Gen. Tech. Rep. PSW-GTR-203. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 299.

**Dagley, C.M.; Berrill, J.; Coppeto, S.; Jacobson, K. 2012.** Effects of slash pile burning after restoring conifer-encroached aspen. South Lake Tahoe, CA: U.S. Department of Agriculture, Forest Service, Lake Tahoe Basin Management Unit. 10 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5406067.pdf. (13 December 2013).

**Dwire, K.A.; Rhoades, C.C.; Young, M.K. 2010.** Potential effects of fuel management activities on riparian areas. In: Elliot, W.J.; Miller, I.S.; Audin, L., eds. Cumulative watershed effects of fuel management in the western United States. Gen. Tech. Rep. RMRS-GTR-231. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 175–205.

**Everett, R.; Schellhaas, R.; Ohlson, P.; Spurbeck, D.; Keenum, D. 2003.** Continuity in fire disturbance between riparian and adjacent sideslope Douglas-fir forests. Forest Ecology and Management. 175: 31–47.

**Forest Ecosystem Management Assessment Team [FEMAT]. 1993.** Forest ecosystem management: an ecological, economic, and social assessment: report of the forest ecosystem management team, July 1993. USDA Forest Service; USDC National Marine Fisheries Service; USDI Bureau of Land Management, Fish and Wildlife Service, and National Park Service; Environmental Protection Agency. Portland, OR: U.S. Government Printing Office.

**Fisk, H.; Megown, K.; Decker, L.M. 2004.** Riparian area burn analysis: process and applications. RSAC-57-TIP1. Salt Lake City, UT: U.S. Department of Agriculture, Forest Service, Remote Sensing Application Center. 4 p.

**Halofsky, J.E.; Hibbs, D.E. 2008.** Determinants of riparian fire severity in two Oregon fires, USA. Canadian Journal of Forest Research. 38: 1959–1973.

**Hansen, W.F. 2001.** Identifying stream types and management implications. Forest Ecology and Management. 143(1-3): 39–46.

**Hossack, B.R.; Pilliod, D.S. 2011.** Amphibian responses to wildfire in the western United States: emerging patterns from short-term studies. Fire Ecology. 7(2): 129–144.

**Jockusch, E.L.; Martinez-Solano, I.; Hansen, R.W.; Wake, D.B. 2012.** Morphological and molecular diversification of slender salamanders (Caudata: Plethodontidae: Batrachoseps) in the southern Sierra Nevada of California with descriptions of two new species. Zootaxa. 3190: 1–30.

**Jones, B.E.; Lile, D.F.; Tate, K.W. 2011.** Cattle selection for aspen and meadow vegetation: implications for restoration. Rangeland Ecology and Management. 64(6): 625–632.

**Jones, B.E.; Rickman, T.H.; Vazquez, A.; Sado, Y.; Tate, K.W. 2005.** Removal of encroaching conifers to regenerate degraded aspen stands in the Sierra Nevada. Restoration Ecology. 13(2): 373–379.

**Kobziar, L.N.; McBride, J.R. 2006.** Wildfire burn patterns and riparian vegetation response along two northern Sierra Nevada streams. Forest Ecology and Management. 222: 254–265.

**Kondolf, G.M.; Kattelman, R.; Embury, M.; Erman, D.C. 1996.** Status of riparian habitat. In: SNEP Science Team and Special Consultants, eds. Sierra Nevada ecosystem project: final report to Congress. Vol. II: Assessments and scientific basis for management options. Report No. 37. Davis, CA: Centers for Water and Wildland Resources, University of California: 1009–1030.

**Krasnow, K.D.; Halford, A.S.; Stephens, S.L. 2012.** Aspen restoration in the eastern Sierra Nevada: effectiveness of prescribed fire and conifer removal. Fire Ecology. 8(3): 104–118.

**Kuhn, T.J.; Safford, H.D.; Jones, B.E.; Tate, K.W. 2011.** Aspen (*Populus tremuloides*) stands and their contribution to plant diversity in a semiarid coniferous landscape. Plant Ecology. 212(9): 1451–1463.

**Leopold, L.B.; Wolman, M.G.; Miller, John P. 1995.** Fluvial processes in geomorphology. Mineola, NY: Dover Publications. 544 p.

**Luce, C.; Morgan, P.; Dwire, K.; Isaak, D.; Holden, Z.; Rieman, B. 2012.** Climate change, forests, fire, water and fish: building resilient landscapes, streams, and managers. Gen. Tech. Rep. RMRS-GTR-290. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 207 p.

**Messier, M.S.; Shatford, P.A.; Hibbs, D.E. 2012.** Fire exclusion effects on riparian forest dynamics in southwestern Oregon. Forest Ecology and Management. 264: 60–71.

**Murphy, K.; Rich, T.; Sexton, T. 2007.** An assessment of fuel treatment effects on fire behavior, suppression effectiveness, and structure ignition on the Angora Fire. Tech. Pap. R5-TP-025. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region. 32 p.

**Murphy, M.L.; Hawkins, C.P.; Anderson, N.H. 1981.** Effects of canopy modification and accumulated sediment on stream communities. Transactions of the American Fisheries Society. 110(4): 469–478.

**Newton, M.; Cole, E.C. 2005.** Linkage between riparian buffer features and regeneration, benthic communities and water temperature in headwater streams, western Oregon. In: Harrington, C.; Schoenholtz, S., eds. Productivity of western forests: a forest products focus. Gen. Tech. Rep. PNW-GTR-642. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 81–101.

**Office of the Federal Register. 2012.** Subpart A: National Forest System Land Management Planning. In: USDA, ed. 36 CFR Part 219. U.S. Government Printing Office, Federal Register: 48–71.

**Olson, D.L; Agee, J.K. 2005.** Historical fires in Douglas-fir dominated riparian forests of the southern Cascades, Oregon. Fire Ecology. 1(1): 50–74.

**Pettit, N.E.; Naiman, R.J. 2007.** Fire in the riparian zone: characteristics and ecological consequnces. Ecosystems. 10: 673–687.

**Pope, K.L.; Brown, C.; Hayes, M.J.; Green, G.; Macfarlane, D. 2014.** Cascades frog conservation assessment. Gen. Tech. Rep. PSW-GTR-244. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 116 p.

**Rambo, T.R.; North, M.P. 2008.** Spatial and temporal variability of canopy microclimate in a Sierra Nevada riparian forest. Northwest Science. 82: 259–268.

**Rambo, T.R.; North, M.P. 2009.** Canopy microclimate response to pattern and density of thinning in a Sierra Nevada forest. Forest Ecology and Management. 257: 435–442.

**Rieman, B.; Lee, D.; Burns, D.; Gresswell, R.; Young, M.; Stowell, R.; Rinne, J.; Howell, P. 2003.** Status of native fishes in the western United States and issues for fire and fuels management. Forest Ecology and Management. 178(1–2): 197–211.

**Rochester, C.J.; Brehme, C.S.; Clark, D.R.; Stokes, D.C.; Hathaway, S.A.; Fisher, R.N. 2010.** Reptile and amphibian responses to large-scale wildfires in Southern California. Journal of Herpetology. 44(3): 333–351.

**Safford, H.D.; Stevens, J.T.; Merriam, K.; Meyer, M.D.; Latimer, A.M. 2012.** Fuel treatment effectiveness in California yellow pine and mixed conifer forests. Forest Ecology and Management. 274: 17–28.

**Shepperd, W.D.; P. Rogers, P.; Burton, D.; Bartos, D.L. 2006.** Ecology, biodiversity, management, and restoration of aspen in the Sierra Nevada. Gen. Tech. Rep. RMRS-GTR-178. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 122 p.

**Skinner, C.N. 2003.** A tree-ring based fire history of riparian reserves in the Klamath Mountains. In: Farber, P.M., ed. California riparian systems: processes and floodplains management, ecology, and restoration. Mill Valley, CA: Pickleweed Press: 116–119.

**Skinner, C.N.; Taylor, A.H.; Agee, J.K. 2006.** Klamath Mountains bioregion. In: Sugihara, N.S.; van Wagtendonk, J.W.; Fites-Kaufmann, J.; Shaffer, K.; Thode, A., eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 170–194.

**Stephens, S.L.; Meixner, T.; Poth, M.; McGurk, B.; Payne, D. 2004.** Prescribed fire, soils, and stream water chemistry in a watershed in the Lake Tahoe Basin, California. International Journal of Wildland Fire. 13: 27–35.

**Stone, K.R.; Pilliod, D.S.; Dwire, K.A.; Rhoades, C.C.; Wollrab, S.P.; Young, M.K. 2010.** Fuel reduction management practices in riparian areas of the western USA. Environmental Management. 46(1): 91–100.

**Taylor, A.H.; Skinner, C.N. 2003.** Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications. 13(3): 704–719.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2001.** Sierra Nevada forest plan amendment: final environmental impact statement. Vallejo, CA: Pacific Southwest Region. http://www.sierraforestlegacy.org/Resources/Conservation/LawsPoliciesRegulation/KeyForestServicePolicy/SierraNevadaFramework/Framework-FSROD01.pdf. (13 December 2013).

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2004.** Sierra Nevada forest plan amendment: final supplemental environmental impact statement. Vallejo, CA: Pacific Southwest Region. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsBdev3_046095.pdf. (13 December 2013).

**Van de Water, K.M.; North, M. 2010.** Fire history of coniferous riparian forests in the Sierra Nevada. Forest Ecology and Management. 260(3): 384–395.

**Van de Water, K.; North, M. 2011.** Stand structure, fuel loads, and fire behavior in riparian and upland forests, Sierra Nevada Mountains, USA; a comparison of current and reconstructed conditions. Forest Ecology and Management. 262(2): 215–228.

**Wilzbach, M.A.; Harvey, B.C.; White, J.L.; Nakamoto, R.J. 2005.** Effects of riparian canopy opening and salmon carcass addition on the abundance and growth of resident salmonids. Canadian Journal of Fisheries and Aquatic Sciences. 62: 58–67.

**Wohl, E.; Jaeger, K. 2009.** A conceptual model for the longitudinal distribution of wood in mountain streams. Earth Surface Processes and Landforms. 34: 329–344.

**Wondzell, S.M. 2001.** The influence of forest health and protection treatments on erosion and stream sedimentation in forested watersheds of eastern Oregon and Washington. Northwest Science. 75: 128–140.

# Chapter 6.3—Wet Meadows

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## Summary

Wet meadows help to sustain favorable water flows, biological diversity, and other values; consequently, restoration of degraded wet meadows is an important part of a strategy for promoting socioecological resilience. This chapter focuses on high-elevation wet meadows that are associated with streams; thus restoration of such meadows may be considered a subset of stream restoration. However, it is important to recognize that degradation of high-elevation meadows often reflects site-level impacts rather than watershed-scale impacts that degrade lower-elevation streams and rivers. For that reason, and because of the cascade of impacts associated with incision of wet meadows, restoration of wet meadows is often expected to deliver a wide range of benefits. Published evaluations of wet meadow restoration efforts within the synthesis area have demonstrated gains at specific sites in certain functions, including water quality, water quantity, and macroinvertebrate diversity. Broader reviews of stream and river restoration in the past decade indicate that restoration efforts have often fallen short in demonstrating anticipated benefits, especially in terms of wildlife and fishes. These shortcomings may reflect a variety of causes, including incomplete documentation of projects, inability of treatments to address limiting factors, and limitations on monitoring resources, experimental designs, and timeframes for evaluating responses, which may require over a decade to gauge. Because many of these challenges are also relevant to meadow restoration projects, these findings reinforce the need for continued and increased monitoring of treatment outcomes, use of rigorous experimental designs, and use of conceptual models when evaluating the potential for improving site conditions, designing treatments, setting restoration objectives, and evaluating outcomes. An important theme in the synthesis that applies to wet meadow restoration is an emphasis on restoring ecological processes, such as overbank flooding, sediment transport, and establishment of native wetland vegetation. Active site-specific restorations may be warranted where local factors have caused degradation to a point where natural recovery is likely to be extremely slow. Monitoring the rate and extent of channel incision is important to avoid losses of socioecological values in stream and meadow ecosystems associated with erosion and lowering of water tables. However,

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considering broader landscape influences on meadows may be more important over the next several decades, given that changes in hydrology, wildfire regime, and spread of nonnative species may affect key ecological processes in meadows. Meadow restoration offers productive ground for understanding interactions among a wide range of ecological, social, cultural, and economic values. Designing, conducting, and evaluating restoration strategies in an adaptive management framework will likely benefit from broad participation by resource managers, researchers, and community members to facilitate this integrated understanding.

## Introduction

This chapter addresses wet meadows, and in particular, high-elevation wetlands that have fine-textured soils and shallow groundwater tables in the summer. These conditions support wetland vegetation, predominantly herbaceous plants, including sedges, other graminoids, and forbs, but also woody plants such as willows that can tolerate anaerobic conditions (Ramstead et al. 2012, Weixelman et al. 2011). This chapter focuses on meadows that are associated with defined stream channels (fig. 1). It does not focus on headwater fens and other peatlands, which are valuable but relatively uncommon in the synthesis area; recent publications provide guidance for assessing their conditions (Weixelman et al. 2011).

Jonathan Long

Figure 1—Trout Creek and adjacent wet meadow in the Lake Tahoe Basin Management Unit.

Stream restoration in general has been identified as an important part of the Forest Service's overall restoration strategy for the synthesis area (USDA FS 2011b). Restoration of wet meadows provides important opportunities to promote ecological resilience and benefit social values (Weixelman et al. 2011). Wet meadow restoration is expected to have an important role in securing favorable flows of high-quality water (Viers and Rheinheimer 2011) and mitigating emissions of carbon and nitrogen (Norton et al. 2011). By enhancing meadow wetness and enhancing diversity of meadow conditions, restoration could also promote biodiversity including pollinators (Hatfield and LeBuhn 2007) and nesting passerine birds (Cocimano et al. 2011). The final section of this chapter, "Integrated Socioecological Approaches to Stream and Meadow Restoration," considers further opportunities to link socioeconomic and ecological values to guide restoration of wet meadows.

**Restoration of wet meadows provides important opportunities to promote ecological resilience and benefit social values.**

Restoration of streams has been a focus of research across the Sierra Nevada region, the state of California, and the United States within the past decade. Earlier synthesis reports for the region, including the Sierra Nevada Ecosystem Project report (SNEP Science Team 1996b), the 1998 Science Review (Sierra Nevada Science Review Team 1998), and the 1999 report Sierra Nevada Ecosystems in the Presence of Livestock (Allen-Diaz et al. 1999), remain relevant and useful because they address a broader range of meadow ecosystems and related topics, including conservation of aquatic biodiversity, sustaining streamflows for wildlife and human uses, and grazing management on public lands. A more recent synthesis for Great Basin ecosystems by Chambers and Miller (2011) explained that strategies to restore streams and meadows should consider a wide range of watershed impacts, including roads, trails, grazing, and water diversions. Restoration strategies may be most effective if they consider where and when addressing these watershed influences is necessary to promote restoration, and in which cases active interventions are warranted and cost effective (Hobbs and Cramer 2008, Kauffman et al. 1997).

## Promoting Resilience in Wet Meadow Ecosystems

Chapter 6.1, "Watershed and Stream Ecosystems," provides a definition of resilience and emphasizes the importance of restoring natural fluvial processes. Although these concepts apply generally to montane wet meadow systems, a strategy of relying on natural disturbance processes may be less effective in these less physically dynamic systems because they have relatively small watershed areas and reduced stream power. There is widespread recognition that channel headcutting in headwater systems can be indicative of a more persistent disequilibrium, in part because the process is difficult to reverse through natural deposition in such small systems. In the Great Basin, many streams have exhibited a tendency toward incision during

the past two millennia (Germanoski and Miller 2004). However, most meadow systems on the western slope of the Sierra Nevada appear to have been stable within the past three millennia, and in a few cases, as many as 10,000 years, apart from more recent incision associated with anthropogenic disturbances such as livestock grazing (Benedict 1982, Ratliff 1985). Reference erosion rates appear very low in many small headwater streams and in wet meadow systems that have intact streambank vegetation, as reported by studies in the Sierra Nevada (Micheli and Kirchner 2002, Simon 2008). As a consequence, high rates of incision and bank erosion generally appear outside the range of historical variation in these systems and strengthen the rationale for active intervention.

**High rates of incision and bank erosion generally appear outside the range of historical variation in these systems and strengthen the rationale for active intervention.**

To evaluate what kinds of interventions, if any, are warranted in a particular ecological system requires analysis to determine whether abiotic or biotic thresholds have been passed (Hobbs and Cramer 2008). In considering these questions, Sarr (2002) described how systems with intact soils and geomorphology demonstrated capacity to recover quickly and predictably, while others had shifted to alternative states marked by channel incision (discussed in more detail below), lowering of local water tables, reduced connectivity of channels to broad floodplains, and encroachment of nonhydrophytic woody plants (particularly conifer trees and sagebrush) in formerly wet riparian areas. Revegetation measures such as transplanting sedges tend to be more effective where groundwater tables are sufficiently high (Steed and DeWald 2003). In more degraded sites, much more active restoration efforts to remove woody vegetation, raise groundwater levels, and reestablish burning regimes may be needed to restore native herbaceous communities (Berlow et al. 2003). Sagebrush encroachment is addressed in detail by Chambers and Miller (2011). A report by Eagan et al. (2000) noted that a trail used by hikers and stock in a subalpine meadow required recontouring and restoration of topsoil after demonstrating little recovery following 30 years of closure. Because restoration potential appears to be very site-specific, studies of geology, hydrology, and soils attributes, as well as assessment tools discussed in the section on monitoring and evaluation near the end of this chapter, are important to determine site potential and to select appropriate treatments (Ramstead et al. 2012).

## Channel Incision

Channel incision can cause a profound loss of productivity in wet meadow ecosystems. Shields et al. (2010) described incision as a syndrome that threatens many ecosystem services by triggering a cascade of geomorphic, hydrologic, and biological effects, including bank instability, channel erosion, perturbated

hydrology, non-point source pollution, conversions from wet to dry meadow vegetation, degradation of aquatic habitat, and reduced fish species richness. Causes of incision may be natural, such as geological uplift, lowering of channel base levels associated with changing climate, or extreme runoff events associated with wildfires or storms; or resulting from more specific human actions, such as blocked culverts that impair sediment movement or overgrazing that removes protective vegetation or substrates from stream channels. These influences result in excess capacity of a stream to transport sediment relative to the supply of sediment from upstream reaches (Simon and Rinaldi 2006). Reduced sediment supply resulting from the alteration of fire regimes, as well as loss of beaver activity, are additional potential influences that might have rendered meadows more vulnerable to incision (see "Research Gaps and Pending Research" at the end of this chapter).

Studies within the synthesis area have reinforced the importance of addressing channel incision. The potential for channel incision to pierce low-permeability layers and alter stream hydrology is a particular concern; such layers may be associated with peat or with compacted soils that are a legacy of historical heavy grazing (Hill and Mitchell-Bruker 2010). Local studies have also quantified some impacts from incision; for instance, a study of Monache Meadow in the southern Sierra Nevada found that banks without wet meadow vegetation are approximately 10 times more susceptible to erosion than banks with herbaceous wet meadow vegetation, such as sedges and rushes (Micheli and Kirchner 2002). Where channels have active headcuts, herbaceous vegetation may not be effective in preventing bank erosion (Zonge et al. 1996). Moreover, physical changes associated with channel widening or incision, including increased temperatures, are not readily changed through restoration of riparian vegetation alone (Poole and Berman 2001). Because of the profound losses in ecosystem functions that can occur as a result of incision (Sarr 2002), management strategies and monitoring would benefit from focusing on this process. Preventing incision and restoring incised meadows could be important components of a landscape strategy to promote system resilience to climate change (Seavy et al. 2009) and sequester carbon and nitrogen (Norton et al. 2011). Although incision is likely to be a predominant problem of concern, other processes in meadows are important, such as channel widening (Loheide et al. 2009).

Chambers and Miller (2011) proposed a general framework for addressing incised meadows based on the degree of incision (table 1). Their synthesis is particularly useful for considering issues that are important for east-side systems, such as sagebrush encroachment and management of the fire-adapted, invasive cheatgrass.

**Table 1—Framework for addressing incision within wet meadow systems by Chambers and Miller (2010)**

| Condition | Indicators | Treatment approach |
|---|---|---|
| Low to moderately incised | Channel has incised to 0 to 2 times bankfull channel depth | In-stream structures and bank stabilization measures to prevent knickpoint migration and maintain meadow vegetation |
| Highly incised | Channel has incised to >2 times bankfull channel depth | Careful design of in-stream structures to minimize further incision of the main channel and to maintain springs |
| Fully incised | Channel has previously incised but is no longer actively incising | Actively manage area to maintain meadow vegetation based upon knowledge of groundwater tables and riparian vegetation |

## In-Stream Structural Approaches

Because waiting for streams to stop incising may result in extensive erosion, structural interventions are often proposed for incising systems, and as shown in table 1, may be particularly warranted for early stages of incision. Such treatments include various kinds of grade control features and streambank or channel armoring (see Ratliff [1985] and Chambers and Miller [2011] for more extensive descriptions). In-stream structures have long been used to protect or enhance aquatic habitat, but there remain serious concerns about their potential for failure. For example, Stewart et al. (2009) concluded that resource managers should be circumspect in using in-stream engineered devices because evidence does not support their effectiveness. Failures, such as erosion around or under the structures, tend to be more common in larger, more dynamic streams but are still a concern for smaller meadow systems; consequently, structural treatments generally require careful design and installation as well as long-term monitoring and maintenance (Chambers and Miller 2011).

**An important trend in large river restoration strategies is to move from permanent structures toward temporary protection and enhancement that allows natural vegetation, sedimentation, and erosional processes to reestablish.**

An important trend in large river restoration strategies is to move from permanent structures toward temporary protection and enhancement that allows natural vegetation, sedimentation, and erosional processes to reestablish (Miller and Kochel 2010), as well as allowing streams access to a wider corridor to migrate (Kondolf 2011) (see chapter 6.1). Softer or “deformable” treatments may be somewhat more challenging to evaluate than harder in-stream structures because by their nature, they are intended to be overtaken by natural recovery processes. Although montane wet meadow systems are less dynamic than low-elevation riverine systems, the general principles of reducing constraints on historical floodplains (such as roads and culverts) and designing deformable treatments can extend to meadow treatments. For example, this principle is reflected in the use of natural rock and plant materials (rather than unnaturally large rip-rap or metal gabions) to construct grade control structures (e.g., riffle formations) and stabilize streambanks (see Chambers and Miller [2011] and Ramstead et al. [2012] for examples of such applications).

**Channel filling and plugging—**
Channel reconfigurations in the synthesis area have involved diversion, filling, or plugging of existing channels as well as excavation of new channels. Filling of incised channels has been conducted in a number of sites in the synthesis area (Loheide et al. 2009, Ramstead et al. 2012). For sites that are thought to have historically lacked defined channels, flows may be directed over the meadow surface (an example includes Halstead Meadow in Sequoia National Park, cited in Ramstead et al. [2012]). In other cases, they may be diverted into one or more remnant channels that have a more desirable geomorphic configuration and vegetation. Either approach requires careful attention to protecting or restoring native vegetation and hydrology to prevent re-incision. Where remnant conditions are not suitable for reintroducing flows, practitioners have often constructed new channels.

To reduce the volume of material needed to refill incised channels, practitioners have developed the "pond and plug method," wherein materials are excavated within the meadow, creating ponds, and then the channel is plugged at various locations using the excavated materials. This method has been the subject of studies in the Feather River watershed, which provide evidence that this method is effective in restoring many attributes of these systems (Loheide et al. 2009), as described further in the next section. However, researchers have also noted concerns about these treatments:

- Pond and plug creates novel conditions of deep ponds, which can become habitats for invasive aquatic species such as bullfrogs (*Rana catesbeiana*) and green sunfish (*Lepomis cyanellus*) (Adams and Pearl 2007, Ramstead et al. 2012).
- Channel reconstruction or pond and plug methods may be inappropriate in systems with fine-grained confining units, because the process of excavating alluvial materials could disrupt the meadow hydrology (Chambers and Miller 2011). Identifying low-permeability layers in meadows is an important component of a broader strategy for protection and restoration (see Hill and Mitchell-Bruker [2010] and the "Research Gaps and Pending Research" section).

## Evaluating Benefits of Meadow Restoration

Water quantity and quality effects of meadow restoration have been undertaken at a relatively small number of sites in the Sierra Nevada within the past decade, with considerable emphasis on large, low-gradient meadows along tributaries of the Feather River and streams in the Lake Tahoe basin. Published studies suggest that active meadow restoration designed to remedy incised channels has increased

groundwater levels and subsurface storage, which in turn promotes wetland vegetation (Hammersmark et al. 2010); increased frequency and duration of floodplain inundation, which in turn may filter sediment and nutrients; attenuated peak flows and increased mid-summer baseflows (Hammersmark et al. 2008, Tague et al. 2008); and reduced maximum water temperatures (Loheide and Gorelick 2006). Streamflow below restored meadows may be affected by higher evapotranspiration rates in the rewetted meadows (and any created ponds) and increased subsurface storage (Hammersmark et al. 2008, Loheide and Gorelick 2005). Research has improved understanding of how site qualities influence response, including the presence of impermeable layers that can maintain high water tables but also inhibit groundwater from upwelling to the meadow surface (Booth and Loheide 2012). The water quality and water quantity benefits of wet meadow restoration are an important topic for which the National Fish and Wildlife Federation has initiated a major research initiative in Forest Service Region 5 (Viers and Rheinheimer 2011).

Restoration of meadow hydrology and vegetation is often expected to result in a cascade of higher order functions, including increases in soil carbon and improvements in fish and wildlife habitat (Ramstead et al. 2012). Nevertheless, a number of reviews have cautioned not to oversell higher order benefits without further monitoring and research to quantify them. This concern has emerged in light of the popularization and commercial expansion of stream restoration in parts of the United States (Lave et al. 2010). Several reviews have recommended a rigorous application of ecological theory and greater emphasis on monitoring outcomes (Palmer 2009, Ramstead et al. 2012). Bernhardt and Palmer (2011) noted that research on river restoration in recent years has progressed from asking “Why don’t we know more about river restoration success?” to asking “Why aren’t river restoration projects more effective?” This general trend also appeared to unfold in California, where Kondolf et al. (2007) highlighted a lack of information needed to evaluate projects. In a meta-analysis of effects of stream restoration projects on macroinvertebrates, Miller et al. (2010) did not include any studies from the Sierra Nevada, presumably because they did not find ones that met their criteria for a controlled research design. A recent evaluation of wet meadow restoration in the Southwest, which included studies of about a dozen projects from the synthesis area, concluded that although there has been significant progress in restoring morphology and vegetation, there remains a need for long-term and better designed monitoring programs (Ramstead et al. 2012). These reviews noted lack of controls and confounded treatments as a common problem in evaluating project effects. For example, changes in grazing management are often confounded with structural restoration treatments, and sites that have not been treated recently may have an older history of treatments. Others

have noted the potential for a publication bias in favor of reporting more successful projects (Ramstead et al. 2012, Stewart et al. 2009).

Demonstrating benefits of stream and meadow restoration becomes more challenging when evaluating benefits to higher order ecosystem services, including biodiversity. Researchers have criticized shortcomings of some restoration projects as overly relying upon the "field of dreams hypothesis" that "if you build it, they will come," in which "it" refers to physical structure, hydrology, or vegetation, and "they" refers to the desired biological community, usually wildlife (Hobbs and Cramer 2008, Palmer et al. 1997). Defenders of that approach may counter that projects that fell short may have lacked restoration of critical ecosystem processes, such as overbank flooding and fire, so in effect, they did not "rebuild it." Nevertheless, researchers contend that this hypothesis needs to be rigorously tested for different habitats and different species (Palmer et al. 1997). In recent years, researchers have reviewed stream restoration efforts nationwide to evaluate this hypothesis. Bernhardt and Palmer (2011) cautioned that channel reconfiguration efforts may reduce bank erosion and increase sinuosity, but that evaluations have found little evidence for benefits to sensitive taxa and water quality (in particular, reduction of nutrients). They noted that many projects in the United States are undertaken at sites where watershed degradation is a key factor, so reach-specific channel restoration treatments do not treat the underlying causes of degradation. However, their review included many urbanized streams and other sites in heavily altered watersheds. Sites on national forests in the Sierra Nevada are less likely to have experienced severe watershed-scale impacts (although dams may have significantly altered hydrologic processes in some watersheds), and many restoration projects have targeted streams and meadows that have been significantly affected by localized road, channelization, or grazing impacts. Therefore, meadow restoration projects in the synthesis area should generally be less vulnerable to those potential shortcomings. Nevertheless, this research firmly underscores the importance of long-term monitoring and research to evaluate more complex, higher order outcomes of restoration.

**Long-term monitoring and research are needed to evaluate the more complex, higher order outcomes of wet meadow restoration, including effects on wildlife.**

Researchers have emphasized the importance of conceptual models to explicitly state and test the strength of linkages between various fundamental changes, such as modifying channels to reduce entrenchment and increase the areas flooded during frequent floods, to vegetative effects and higher order effects on fish, amphibians, and terrestrial wildlife. Through a national meta-analysis of two dozen studies, Miller et al. (2010) concluded that although habitat restoration may promote biodiversity and ecosystem resilience, its ability to increase biomass of macroinvertebrates for the benefit of higher tropic levels (e.g., fish, amphibians, and birds) was still uncertain. That study noted that channel reconfigurations yielded highly

variable invertebrate community responses. On the other hand, in a study from the Sierra Nevada, Herbst and Kane (2009) reported that active channel restoration yielded a rapid shift in macroinvertebrate communities toward reference conditions. Conceptually, restoration of wet meadow hydrology should yield benefits for a variety of wildlife species, including willow flycatcher (*Empidonax traillii*) (Cocimano et al. 2011). However, many of these higher order biological objectives may prove hard to achieve (or to demonstrate) in short timeframes because of confounding or limiting factors, including legacy effects of past management, including historical overgrazing, soil compaction, mining, and stocking of nonnative trout. For example, the stocking of trout into fishless systems has affected amphibians, reptiles, and birds by altering food webs in lakes and streams (Eby et al. 2006, Epanchin et al. 2010). A Sierra Nevada study by Purdy et al. (2011) found that fundamental indicators of vegetation and physical habitat tend to classify meadows as being in better condition than do aquatic indices, especially the native fish and amphibian index. This finding could reflect a variety of causes, including legacy effects, time lags in these indicators, and controlling influences that are beyond the site.

## Grazing Management and Wet Meadow Restoration

Livestock grazing involves a complex interplay of social and ecological factors (see chapter 9.5, "Managing Forest Products for Community Benefit"). Although grazing is only one of many land uses that affect streams and wet meadows, grazing management and hydrogeomorphic condition appear to be critical determinants of meadow restoration outcomes (Ramstead et al. 2012). In a recently published review of rotational grazing from a broad socioecological perspective, Briske et al. (2011b) offered frameworks to promote effective management of grazed systems, including adaptive management with an emphasis on stakeholder participation (Fernandez-Gimenez et al. 2008) as well as targeted grazing that explicitly emphasizes management outcomes, such as weed control, fire hazard reduction, and wildlife habitat improvement. The latter approach suggests that grazing management could be an important tool for promoting socioecological resilience in systems that evolved with grazing animals. This approach embodies the logic of disturbance-based management as described in North and Keeton (2008), and recognizes that grazing, like fire, can be a tool for rejuvenating areas by reducing accumulated vegetation. It is important to recognize that because different kinds of domesticated livestock (i.e., cattle, horses, and sheep) have different grazing behaviors and influences, they are not interchangeable with each other or with the prehistorical assemblage that may have grazed particular landscapes. Researchers have discussed the utility of grazing in "novel systems," where grazing has a long history and nonnative species

have become dominant (Hobbs et al. 2009). In such systems, carefully managed livestock grazing may be a useful, albeit often controversial, tool for maintaining biodiversity and ecological services (Hobbs and Cramer 2008). For example, studying spring systems in Sierra Nevada foothills, Allen-Diaz et al. (2004) found that removing livestock grazing may allow dead plant material to accumulate, which in turn can increase levels of nitrate in wetland waters and decrease plant diversity. Similar findings have come from research in vernal pool systems (Marty 2005). The ecological benefits of grazing-based management approaches to less invaded, high-elevation wet meadows of the Sierra Nevada are less clear. A report by the Sierra Nevada Ecosystem Project indicated that it was unknown whether grassland ecosystems in the Sierra Nevada were adapted to disturbance by prehistoric megafauna, and suggested that more intensive grazing practices, such as active herding, could avoid many undesirable impacts (SNEP Science Team 1996a).

A recent comprehensive report on riparian management practices provides an overview of prescribed grazing effects on a wide range of resource values, including wildlife habitat, water quantity and quality, streambank and soil stability, carbon storage, plant and animal diversity, composition and vigor of plant communities, forage for grazing and browsing animals' health and productivity, riparian and watershed function, soil condition, and fine fuel loads (Briske et al. 2011a). In a companion chapter on riparian management practices, George et al. (2011) found that grazing practices that result in heavy use of riparian vegetation, are too long in duration, or are poorly timed can be detrimental to aquatic values such as fisheries and streambank stability. They found support for grazing exclusion as a restoration strategy for degraded riparian systems because it promotes recovery of riparian plant community composition. However, they noted that other techniques for manipulating livestock distribution, including herding, supplement placement, water development, and fences, are effective in reducing livestock residence time and utilization in the riparian zone. Both direct effects (such as trampling) and indirect effects (reducing vegetation) of grazing on streambanks and channels within the riparian zone remain critical considerations for meadow resilience because of their potential to induce stream incision and other threshold changes (Ramstead et al. 2012, Trimble and Mendel 1995).

Studies have also examined effects beyond the streamside zone, including impacts on water quality, soils, nutrients, and vegetative composition. For example, Blank et al. (2006) reported that cattle grazing under short-term, high-density stocking conditions did not affect composition and root length density, but that the treatment deposited nutrients and altered soil pH at the edge of a wet meadow in the synthesis area. Also from the synthesis area, a recently completed 5-year study

addressed the effectiveness of excluding cattle from breeding areas of Yosemite toads (*Bufo canorus*). The researchers found no detectable differences in toad occupancy, toad density, or water quality between grazed and nongrazed meadows when livestock grazing met current standards, including 30 to 40 percent use (Roche et al. 2012b). In addition, they found that meadow hydrology there influences occupancy by toads, and cattle grazing intensity does not (Roche et al. 2012a). Recent studies on national forests of the Sierra Nevada have reported exceedances in levels of fecal coliform bacteria (*Escherichia coli*, specifically) in several meadow streams with cattle grazing as well as recreational use in some cases (Derlet et al. 2012, Myers and Kane 2011, Myers and Whited 2012). Additional research on a number of sites in the synthesis area should help to put these studies within a broader context of national forest management (see box 6.3-1).

Much of the research on grazing has had limitations on experimental design that constrain the range of inference to contexts that may not necessarily match conditions on national forest lands. Many studies of grazing impacts are difficult to translate to grazing management strategies when they lack details such as stocking rates or utilization levels (Briske et al. 2008). Many studies of grazing in the Western United States, including the Sierra Nevada, have provided a dichotomous view of grazing by comparing differences or trajectories of vegetation and channel morphology inside and outside of exclosures (examples from the Sierra Nevada include Kondolf [1993] and Knapp and Matthews [1996]). Studies have commonly reported that where physical thresholds had not been exceeded (e.g., channel incision that had lowered water tables below the rooting zone), long-term grazing exclusion or reduction has facilitated substantial growth of native wetland herbaceous and woody vegetation such as willows (Ramstead et al. 2012). A review of exclosure studies on the Kern Plateau by Sarr (2002) noted that particular vegetative and channel responses to exclusion differ owing to a host of factors, including watershed stability, climate, subsurface moisture availability, soil organic content, proximity of willow propagule sources, and degree of channel incision. A study of bumblebees at 20 meadow sites on the Tahoe National Forest (Hatfield and LeBuhn 2007) reported variable impacts associated with cattle and sheep grazing that suggested an important interaction between grazing and flower availability, but it did not quantify the level of grazing use.

Variation in responses has also been reported in some studies on higher order responses within the past decade. Studies assessing the impacts of cattle on amphibians have often been correlative and have yielded mixed results; for example, Bull and Hayes (2000) found no evidence of negative effects of grazing on the Columbia spotted frog (*Rana luteiventris*), but they noted their inability to control for wide

variation in grazing intensity and other landscape variables. More experimental studies using cattle exclosures have also reported mixed results with specific implications for particular taxa. For example, reporting from a study in Tennessee, Burton et al. (2009) suggested that fencing cattle from wetlands may benefit ranid frogs, and controlled grazing may benefit toads in the genus *Bufo*.

Quantifying the influence of livestock grazing in stream and meadow ecosystems has been difficult because experimental designs may not sufficiently address ecological variation. Sarr (2002) identified common problems in evaluating responses to livestock grazing and exclusion, including lack of proper controls and the small size of exclosures. Research experiments are often conducted at too small a scale to properly evaluate effects (Briske et al. 2008). In response to these challenges, Sarr (2002) suggested that resource managers evaluate treatments at watershed scales on experimental rangelands, and in modest-sized exclosures as part of an ongoing adaptive management process. This strategy is reflected in the recent study by Herbst et al. (2012), which suggested that treatments at broader scales may yield different outcomes than smaller riparian exclosures. Specifically, they reported greater macroinvertebrate diversity and measures of habitat quality after 4 years of rest from grazing at the allotment scale, but more limited differences within local riparian exclosures that had been in place over a decade.

## Monitoring and Evaluation

Recent stream restoration research indicates that monitoring remains very important in evaluating whether restoration is achieving desired objectives and in improving practice. The Forest Service Watershed Condition Framework set a goal of establishing a comprehensive monitoring effort by 2016 (USDA FS 2011a). Effort-based performance metrics, such as number of stream miles restored or enhanced, may create an unintended incentive for intervention, particularly in reaches that may have lower per-unit treatment costs. It is important to measure performance in terms of changes in ecological condition and associated benefits. Furthermore, metrics based primarily on physical structure (such as high sinuosity and low width-depth ratios) may underemphasize ecological functions (Bernhardt and Palmer 2011). Consequently, measurements of ecological processes (such as overbank flooding) and services (e.g., improved water quality, changes in seasonal water tables, dampened floods, and improvements in the diversity or abundance of target taxa) may be more appropriate for tracking progress (Bernhardt and Palmer 2011). The emphasis on process-based indicators is an important theme of this synthesis (see chapter 1.2, “Integrative Approaches: Promoting Socioecological Resilience”). In addition, monitoring is important in evaluating project outcomes

over a long period during which floods and vegetative development are expected to alter conditions. Finally, adopting a landscape perspective may be important in promoting socioecological resilience by giving increased weight toward meadows with greater potential to yield desirable outcomes. For example, stream reaches that are located close to less disturbed areas are more likely to be successful in reestablishing aquatic organisms (Bernhardt and Palmer 2011).

The Watershed Condition Framework considers channel incision within criteria for channel shape and function. Developing more specific quantitative criteria for this critical threshold of ecological function may be possible in different regions within the Sierra Nevada; for example, Micheli and Kirchner (2002) suggested that a bank height of 1 m in a southern Sierra Nevada meadow represents a threshold for shifting to dry meadow species and less stable streambanks, and Chambers and Miller (2011) suggested a threshold of incision occurs when channel depths exceed twice the bankfull depth.

A recent study in the Sierra Nevada concluded that soil properties correspond with rapid assessments of meadow condition using the Proper Functioning Condition methodology; specifically, they found that meadows categorized as "properly functioning" have greater nitrogen and dissolved organic nitrogen than "nonfunctioning" or "at-risk" meadows, and greater carbon than "nonfunctioning" meadows (Norton et al. 2011). To evaluate grazing management in wet meadows, Blank et al. (2006) suggested using root-length density as an indicator of ecological function. Root depth is another useful indicator of functional condition, and these types of qualities can be related to vegetation cover and composition data that is collected more routinely (Weixelman et al. 1999). Collectively, these studies provide a basis for using rapid assessments that focus on channel incision and shifts in vegetation away from native wet meadow graminoids; however, they also point to more quantitative metrics and possible threshold values for monitoring.

**Post-project evaluation, monitoring, and adaptive management beyond 3 years is important for advancing stream restoration.**

Stewart et al. (2009) contended that more research-based information about in-stream structural treatments is needed before widespread use can be recommended. Kondolf et al. (2007) pointed to the importance of post-project evaluation, monitoring, and adaptive management for advancing stream restoration and learning from both successes and failures. They further noted that restoration projects in California rarely provide for monitoring beyond 3 years, which is likely inadequate to observe effects of large, infrequent events. They also argued that projects do not always meet the standards needed to evaluate the restoration outcomes articulated by Bernhardt et al. (2007), including a clearly defined goal, objective success criteria, and use of controlled monitoring designs. Ruiz-Jaen and Aide (2005) recommended including measures of diversity, vegetation structure, and ecological

processes, and monitoring more than one reference site, to account for the temporal and spatial dynamics of ecosystems. Identifying suites of streams and watersheds that are in reference condition or could be brought into reference condition would facilitate evaluation of restoration potential and success in the face of anticipated stressors.

Monitoring frameworks need to consider temporal scale to account for effects of disturbances on key indicators (Berkes and Folke 2002, Bryant 1995). Because recovery of stream channels and floodplains may be limited by the episodic nature of flood disturbances (Sarr 2002), pulsed monitoring to coincide with climate and flood dynamics has been proposed as an efficient way to evaluate stream condition (Bryant 1995). Furthermore, interpreting indicators of stream condition relative to flood dynamics may be more appropriate than evaluating annual trends in systems where less frequent floods drive key ecological processes. The idea of quantifying a threshold to aid determination of when geomorphic monitoring is warranted was proposed by Florsheim et al. (2006) for dynamic lowland rivers. Even in relatively stable wet meadow ecosystems, environmental conditions can vary significantly from one year to the next, so reliable evaluations need a relatively long timeframe for monitoring, including pre- and posttreatment data, to demonstrate trends (Kiernan and Moyle 2012, Ramstead et al. 2012). These temporal dynamics further complicate efforts to evaluate impacts of grazing and rest, so one strategy is to consider long rest periods that provide opportunities to evaluate influences of grazing from multiple perspectives (Briske et al. 2011b).

**Because recovery of stream channels and floodplains may be limited by the episodic nature of flood disturbances, pulsed monitoring to coincide with climate and flood dynamics has been proposed as an efficient way to evaluate stream condition.**

Standardized monitoring and classification protocols could facilitate collection and comparison of data at larger spatial scales. Katz et al. (2007) provided a number of recommendations to facilitate evaluation of project effectiveness, including standardized metrics and a common reporting system for tracking restoration projects that includes common semantics for project type, location, timing, and magnitude. Stein et al. 2009 validated the utility of a rapid assessment tool, the California Rapid Assessment Methodology (CRAM) for lower elevation systems, although testing of a CRAM module for high-elevation meadow systems has not yet been published. Validation is important to ensure that the methods are effectively capturing important information about condition and trend. However, as Purdy et al. (2011) explained, variation across different biological indices within sites pose challenges for rapid assessments to concisely summarize the various dimensions of stream and meadow condition.

Classification systems are helpful in addressing heterogeneity within the large areas. A field key by Weixelman et al. (2011) provides a tool for classifying meadow types based on several hydrogeomorphic factors, including soils, water source, and gradient. Loheide et al. (2009) developed a framework for predicting potential

benefits of restoration based on several factors, including elevation, soil texture, and the degree of stream incision. Use of these classifications in project reporting and evaluation could help evaluate restoration treatment effectiveness across the synthesis area by identifying geomorphic settings and hydrologic characteristics that appear particularly sensitive to threats or responsive to treatments.

## Research Gaps and Pending Research

A host of pending research projects will help to fill some of the important gaps in knowledge regarding wet meadow restoration (see box 6.3-1 below). Many of the key cause-and-effect questions and information gaps identified in the Sierra Nevada Forest Plan Amendment Appendix E (USDA FS 2004) focused on impacts of livestock grazing practices. These questions will likely remain at the center of meadow management, although thorough review of long-term monitoring data on rangelands, streams, and meadows should provide better information on which to base decisions and to revisit the questions in Appendix E.

**Long-term studies of effects of meadow restoration on higher order values, such as favorable water flows and aquatic life, remain a topic for further research, particularly in light of anticipated effects of climate change.**

Many topics that influence the outcomes of meadow restoration warrant further research, especially groundwater interactions between meadow aquifers and surrounding systems and the effects of meadow properties and various treatments on hydrologic responses (Hill and Mitchell-Bruker 2010, Loheide et al. 2010). Long-term studies of effects of meadow restoration on higher order values, such as favorable water flows (Loheide et al. 2010) and aquatic life, remain a topic for further research, particularly in light of anticipated effects of climate change. The Sierra Nevada watersheds with the largest amount of mountain meadows are the Mokelumne, Tuolumne, and Stanislaus; these three basins, along with the American and Merced, have been projected to have longer periods of low flows, which threaten mountain meadows and the services they provide (Null et al. 2010).

The introduction of beaver (*Castor canadensis*) has been suggested as a strategy for restoring wet meadows through their potential to induce sediment deposition, raise water tables, and alter relatively large habitat patches (Johnston and Naiman 1990). Beaver introductions in Yellowstone have shown potential to promote increased water surface area, wetland herbaceous vegetation, and riparian shrubs (McColley et al. 2012). However, the complex interactions between beaver activity, wetland hydrology and vegetation, and human infrastructure have to be considered, especially given the potential impacts of beaver dam failures (Beier and Barrett 1987, Butler and Malanson 2005). Recently published evidence that beaver were native to at least some watersheds in the Sierra Nevada suggests that other areas warrant more indepth investigation (James and Lanman 2012, Lanman et al. 2012). These findings heighten the importance of research to determine the conditions under which beaver reintroductions may promote meadow restoration.

Large-scale restoration efforts are increasingly seeking to better incorporate understanding of disturbance regimes and multiple successional states (Lake et al. 2007). As noted in chapter 1.5, "Research Gaps: Adaptive Management to Cross-Cutting Issues," proposals to study landscape-scale effects of forest treatments on multiple resource values would benefit from including aquatic resources, especially to evaluate effects of managing riparian areas and impacts from wildfires. Ratliff (1985) noted a wide range of potential wildfire impacts to meadows, including reduced encroachment by conifer trees; increased flows of water, sediment, charcoal, and woody debris; consumption of peat layers in meadow soils by intense fires; and consumption of wood structures that serve as check dams. However, a more formal examination of outcomes would help to evaluate whether there are thresholds of existing meadow instability or wildfire severity that leads to undesirable outcomes, as well as under what conditions and timeframes wildfires promote desirable developments in wet meadows.

**Box 6.3-1**

## Current and Pending Research

These studies are included for reference although they may have not been published in peer-reviewed outlets.

**Golden trout and climate change adaptation—**

- The U.S. Forest Service Pacific Southwest Research Station (PSW) has a current project funded by the National Fish and Wildlife Foundation to examine the resiliency of stream habitats in the Golden Trout Wilderness to future climate warming. This project will spatially analyze stream temperatures and shading in restored and degraded sections of Mulkey and Ramshaw meadows to estimate what proportion of the habitat will be resilient to climate change and what proportion should undergo restoration treatments. The project is using peak temperature threshold values of 23 °C to trigger management responses, with a long-term goal of keeping streams below 20 °C so they will be resilient to future climate warming. Monitoring of degraded and recovering stream sections will enable comparisons of temperature, dissolved oxygen, stream depth and width, and shading, and it will be used to guide restorative management actions.

**Grazing management in national forests—**

- Researchers at the University of California–Davis (UC Davis), led by Kenneth Tate, are completing a study across several national forests to determine whether grazing best management practices avoid exceedance of water quality standards.

*(continued on next page)*

(*from page 357*)

- A joint Forest Service Region 5 and UC Davis project, also led by Kenneth Tate, will publish long-term (1999 to present) rangeland condition and trend monitoring data on more than 800 permanent plots from throughout the state.

**Hydrologic effects of meadow restoration—**

- Barry Hill, Region 5 Hydrologist, is preparing an assessment of meadow restoration and meadow hydrology for the Sierra Nevada.

**Meadow restoration guide and economic effects—**

- "A guide for restoring functionality to mountain meadows of the Sierra Nevada" (Stillwater Sciences 2012) provides an overview of restoration strategies in the synthesis area.
- "An Economic Analysis of Sierra Meadow Restoration" (Aylward and Merrill 2012) synthesized both peer-reviewed studies and unpublished reports in evaluating potential economic impacts of meadow restoration in the synthesis area.

**Evaluation of meadow restoration—**

- A recent PSW meadow restoration study included sites from the southern Cascade Range to the Stanislaus National Forest (fig. 2). Response variables of interest included basic physical and vegetative indicators of wetland condition, including soil moisture, soil carbon, vegetation cover and biomass, channel depth, and presence of headcuts.

**Meadow restoration and native trout reports—**

- Researchers at UC Davis published a report titled "Meadow Restoration to Sustain Stream Flows and Native Trout: a Novel Approach to Quantifying the Effects of Meadow Restorations to Native Trout" (Henery et al. 2011).
- American Rivers produced a report titled "Evaluating and Prioritizing Meadow Restoration in the Sierra" that documented a rapid assessment methodology based on six attributes (bank height, gullies, bank stability, ratio of graminoid to forb vegetation, bare ground, and encroachment) and a framework for prioritizing restoration activities (Hunt and Nylen 2012).
- UC Davis researchers have released a study that examined the number and size of meadows in the Sierra Nevada: "Sierra Nevada Meadow Hydrology Assessment" (Fryjoff-Hung and Viers 2013).
- UC Davis researchers have prepared reports on effects of climate change on fishes and amphibians associated with meadows: "Projected Effects of Future Climates on Freshwater Fishes of California" (Moyle et al. 2012) and "Montane Meadows in the Sierra Nevada: Changing Hydroclimatic Conditions and Concepts for Vulnerability Assessment" (Viers et al. 2013).

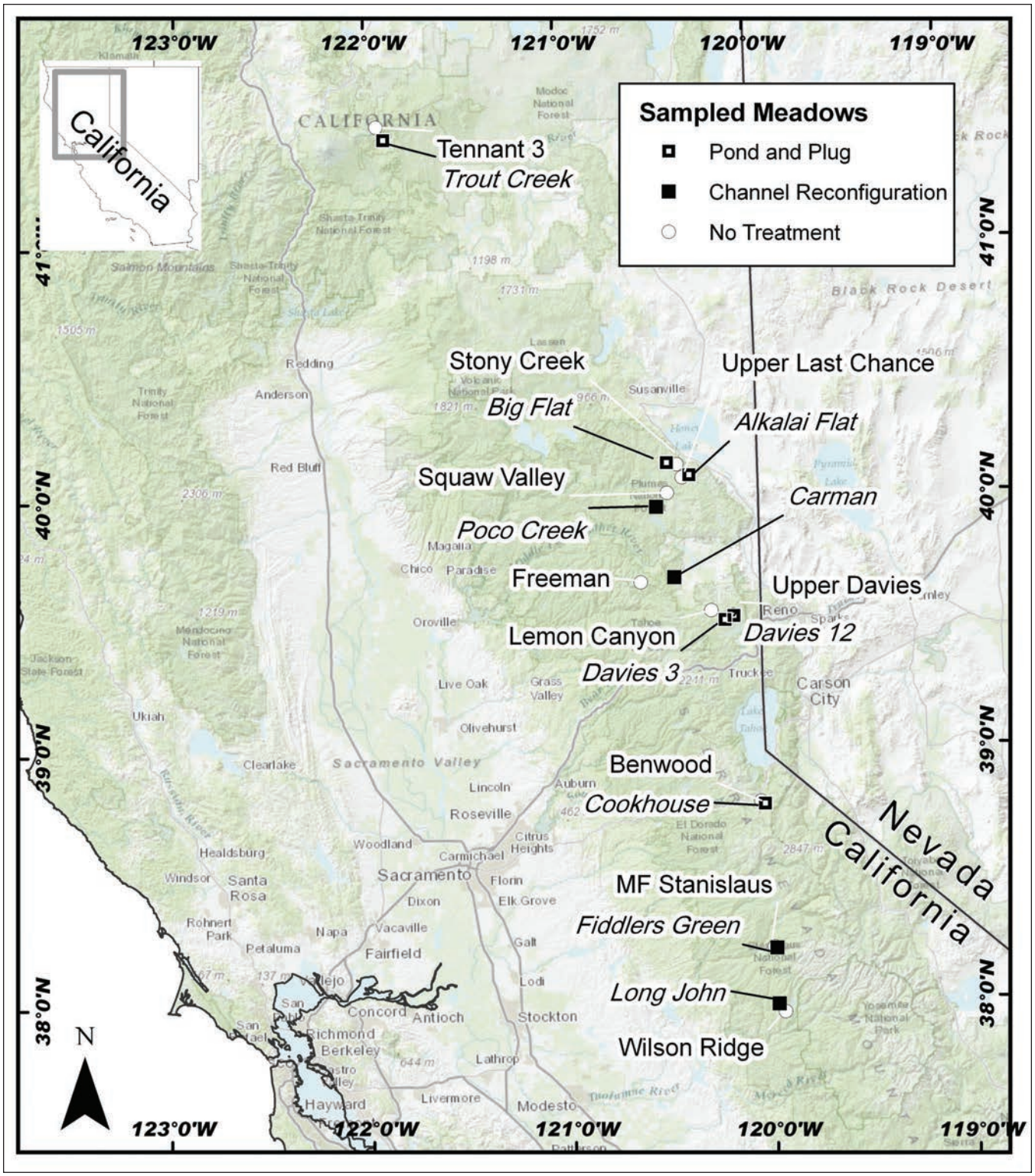


Figure 2—Map of sites in the meadow restoration study led by the Forest Service Pacific Southwest Research Station. Map by Diane Sutherland.

## Integrated Socioecological Approaches to Stream and Meadow Restoration

**Because streams and meadows provide important ecological services and other sociocultural values, they present opportunities to promote ecologically and socially integrated restoration consistent with the broader approaches in this synthesis.**

Because streams and meadows provide important ecological services and other sociocultural values (fig. 3), they present opportunities to promote ecologically and socially integrated restoration consistent with the broader approaches in this synthesis (see chapters 1.2, "Integrative Approaches: Promoting Socioecological Resilience"] and 9.1, "Broader Context for Social, Economic, and Cultural Components"). Advancing management may increasingly depend on a participatory or collaborative adaptive management program that accounts for both social and ecological objectives and promotes learning to meet those objectives (see chapter 9.6, "Collaboration in National Forest Management," as well as Briske et al. [2011b] for a discussion in the context of grazing management). Bernhardt et al. (2007) observed that the most effective stream restoration projects tend to involve local community members and advisory committees throughout all stages, and they hypothesized that those interactions promote accountability in ways that improve outcomes. Golet et al. (2006) also found that community participation in planning may also help to increase the success and social benefits of stream restoration. Chapter 9.1 explains that participation in restoration activities can be empowering for individuals and enhance social capital. Recognizing the importance of social benefits builds upon the distinction drawn by Higgs (1997) between "effective restoration" that is focused on meeting technical performance criteria and "good restoration," which addresses the value of restoration in sociocultural contexts.

Increasing interest in the idea of payments for ecosystem services through restoring montane meadows of the Sierra Nevada has generated excitement about the potential to accelerate the pace of restoration (Viers and Rheinheimer 2011). Rising interests and concerns about these approaches are discussed more generally in chapter 9.2, "Ecosystem Services." Market incentives have fostered interest in treating degraded streams and developing stream restoration as an applied science; however, the increasing importance of stream restoration as a private industry and potential market for ecosystem services could have unintended consequences as complex ecosystem functions are translated into specific credits (Lave et al. 2010). Kondolf et al. (2007) pointed out that although quantitative criteria are important measures of success, projects may also have broader goals regarding stakeholder involvement. Many meadow sites have strong cultural value to tribes and may have been managed by Native American tribes (Ramstead et al. 2012). A strategy discussed in chapter 4.2, "Fire and Tribal Cultural Resources," focuses on partnering with tribes to reestablish more frequent fire regimes and enhance growth of

Jonathan Long

Figure 3—Fly-fishing along Trout Creek, Lake Tahoe Basin Management Unit, represents one of the socioeconomic opportunities afforded by well-functioning stream/wet meadow ecosystems.

culturally desirable plants, such as sedges, willows, and various geophytes, including beargrass, that are commonly associated with riparian or wet meadow habitats.

Relationships between meadow condition and broader social values are a topic that invites further study, particularly to reconcile potential tensions in meadows between scenic views and more complex ecological aesthetics based upon ecological integrity (Gobster 1999, Gobster et al. 2007). For example, wildflower viewing is an important and burgeoning recreational activity (see chapter 9.1). However, Ratliff (1985) reported that maintaining wet meadows in good hydrologic and vegetative condition favors graminoid species that are less showy than many forbs. On the other hand, maintaining wet meadow hydrology may promote production of late-season wildflowers for the benefit of pollinators (Hatfield and LeBuhn 2007) as well as human visitors. Furthermore, because meadow degradation can stimulate dramatic shifts from vegetative reproduction by sedges to production of pollen and seeds by grasses and forbs (Klimkowska et al. 2009), evaluating production of allergens could present another opportunity to relate meadow integrity to social considerations.

## Management Implications

- Wet meadows can be vulnerable to transformations that result in diminished socioecological value. The flip side of that coin is that restoration of these systems holds great potential to provide multiple ecological and social benefits, despite their small share of the landscape. Research to date suggests that projects can promote important benefits; however, additional long-term monitoring and research would help to evaluate those benefits and prioritize investments in restoration.
- In particular, long-term studies of effects of stream and meadow restoration on higher order values such as water flows and aquatic life remain a topic for further research, particularly in light of anticipated effects of climate change.
- Assessments of the number, size, location, current condition (especially extent of incision), and recovery potential of degraded wet meadows in the synthesis area will help target and prioritize structural interventions, changes in grazing practices, or other restoration treatments.
- In addition to site-specific assessments and treatments, examination of disturbances (e.g., wildfire) and management practices (e.g., prescribed grazing practices) on a larger, watershed scale, could aid the design of more effective strategies to promote long-term resilience of these valuable systems.

## Literature Cited

**Adams, M.J.; Pearl, C.A. 2007.** Problems and opportunities managing invasive bullfrogs—Is there any hope? In: Gherardi, F., ed. Biological invaders in inland waters— profiles, distribution, and threats. Dordrecht, The Netherlands: Springer: 679–693. Chapter 38.

**Allen-Diaz, B.; Barrett, R.; Frost, W.; Huntsinger, L.; Tate, K. 1999.** Sierra Nevada ecosystems in the presence of livestock. Davis, CA: University of California–Davis, Rangeland Science Team. 149 p.

**Allen-Diaz, B.; Jackson, R.D.; Bartolome, J.; Tate, K.; Oates, L.G. 2004.** Long-term grazing study in spring-fed wetlands reveals management tradeoffs. California Agriculture. 58(3): 144–148.

**Aylward, B.; Merrill, A.G. 2012.** An economic analysis of Sierra Meadow restoration. Bend, OR: Ecosystem Economics, and Berkeley, CA: Stillwater Sciences. 76 p.

**Beier, P.; Barrett, R.H. 1987.** Beaver habitat use and impact in Truckee River Basin, California. Journal of Wildlife Management. 51(4): 794–799.

**Benedict, N.B. 1982.** Mountain meadows: stability and change. Madrono. 29(3): 148–153.

**Berkes, F.; Folke, C. 2002.** Ecosystem dynamics and local knowledge. In: Gunderson, L.; Holling, C.S., eds. Panarchy: understanding transformations in human and natural systems. Washington DC: Island Press: 121–146. Chapter 5.

**Bernhardt, E.S.; Palmer, M.A. 2011.** River restoration: the fuzzy logic of repairing reaches to reverse catchment scale degradation. Ecological Applications. 21(6): 1926–1931.

**Bernhardt, E.S.; Sudduth, E.B.; Palmer, M.A.; Allan, J.D.; Meyer, J.L.; Alexander, G.; Follastad-Shah, J.; Hassett, B.; Jenkinson, R.; Lave, R.; Rumps, J.; Pagano, L. 2007.** Restoring rivers one reach at a time: results from a survey of U.S. river restoration practitioners. Restoration Ecology. 15(3): 482–493.

**Blank, R.R.; Svejcar, T.; Riegel, G. 2006.** Soil attributes in a Sierra Nevada riparian meadow as influenced by grazing. Rangeland Ecology and Management. 59(3): 321–329.

**Booth, E.G.; Loheide, S.P. 2012.** Hydroecological model predictions indicate wetter and more diverse soil water regimes and vegetation types following floodplain restoration. Journal of Geophysical Research—Biogeosciences. 117, G02011. doi:10.1029/2011JG001831.

**Briske, D.D.; Derner, J.D.; Brown, J.R.; Fuhlendorf, S.D.; Teague, W.R.; Havstad, K.M.; Gillen, R.L.; Ash, A.J.; Willms, W.D. 2008.** Rotational grazing on rangelands: reconciliation of perception and experimental evidence. Rangeland Ecology and Management. 61(1): 3–17.

**Briske, D.D.; Derner, J.D.; Milchunas, D.G.; Tate, K. 2011a.** An evidence-based assessment of prescribed grazing practices. In: Briske, D.D., ed. Conservation benefits of rangeland practices: assessment, recommendations, and knowledge gaps. Lawrence, KS: Allen Press: 21–74.

**Briske, D.D.; Sayre, N.F.; Huntsinger, L.; Fernandez-Gimenez, M.; Budd, B.; Derner, J.D. 2011b.** Origin, persistence, and resolution of the rotational grazing debate: integrating human dimensions into rangeland research. Rangeland Ecology and Management. 64(4): 325–334.

**Bryant, M.D. 1995.** Pulsed monitoring for watershed and stream restoration. Fisheries. 20(11): 6–13.

**Bull, E.L.; Hayes, M.P. 2000.** Livestock effects on reproduction of the Columbia spotted frog. Journal of Range Management. 53(3): 291–294.

**Burton, E.C.; Gray, M.J.; Schmutzer, A.C.; Miller, D.L. 2009.** Differential responses of postmetamorphic amphibians to cattle grazing in wetlands. The Journal of Wildlife Management. 73(2): 269–277.

**Butler, D.R.; Malanson, G.P. 2005.** The geomorphic influences of beaver dams and failures of beaver dams. Geomorphology. 71(1–2): 48–60.

**Chambers, J.C.; Miller, J.R. 2011.** Geomorphology, hydrology, and ecology of Great Basin meadow complexes—implications for management and restoration. Gen. Tech. Rep. RMRS-GTR-258. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 125 p.

**Cocimano, M.C.; Morrison, M.L.; Mathewson, H.A.; Vormwald, L.M. 2011.** The influence of meadow moisture levels on activity of small mammal nest predators in the Sierra Nevada, California. Northwestern Naturalist. 92(1): 50–56.

**Derlet, R.W.; Richards, J.R.; Tanaka, L.L.; Hayden, C.; Ger, K.A.; Goldman, C.R. 2012.** Impact of summer cattle grazing on the Sierra Nevada watershed: aquatic algae and bacteria. Journal of Environmental and Public Health. 2012(Article ID 760108): 7 p.

**Eagan, S.; Newman, P.; Fritzke, S.; Johnson, L. 2000.** Restoration of multiple-rut trails in the Tuolumne Meadows of Yosemite National Park. In: Cole, D.N.; McColl, S.F.; Borrie, W.T.; O'Loughlin, J., eds. Wilderness science in a time of change conference. Vol. 5: Wilderness ecosystems, threats, and management. RMRS-P-15-Vol-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 188–192.

**Eby, L.A.; Roach, W.J.; Crowder, L.B.; Stanford, J.A. 2006.** Effects of stocking-up freshwater food webs. Trends in Ecology and Evolution. 21(10): 576–584.

**Epanchin, P.N.; Knapp, R.A.; Lawler, S.P. 2010.** Nonnative trout impact an alpine-nesting bird by altering aquatic-insect subsidies. Ecology. 91(8): 2406–2415.

**Florsheim, J.L.; Mount, J.F.; Constantine, C.R. 2006.** A geomorphic monitoring and adaptive assessment framework to assess the effect of lowland floodplain river restoration on channel-floodplain sediment continuity. River Research and Applications. 22(3): 353–375.

**Fryjoff-Hung, A.; Viers, J.H. 2013.** Sierra Nevada Meadow Hydrology Assessment: final project report to the USDA Forest Service Pacific Southwest Region. Davis, CA: University of California–Davis, Department of Environmental Science and Policy. 1114 p.

**George, M.R.; Jackson, R.D.; Boyd, C.S.; Tate, K.W. 2011.** A scientific assessment of the effectiveness of riparian management practices. In: Briske, D.D., ed. Conservation benefits of rangeland practices: assessment, recommendations, and knowledge gaps. Lawrence, KS: Allen Press: 213–252.

**Germanoski, D.; Miller, J.R. 2004.** Basin sensitivity to channel incision in response to natural and anthropogenic disturbance. In: Chambers, J.C.; Miller, J.R., eds. Great Basin riparian ecosystems—ecology, management, and restoration. Covelo, CA: Island Press. 88–123.

**Gobster, P.H. 1999.** An ecological aesthetic for forest landscape management. Landscape Journal. 18(1): 54–64.

**Gobster, P.H.; Nassauer, J.I.; Daniel, T.C.; Fry, G. 2007.** The shared landscape: What does aesthetics have to do with ecology? Landscape Ecology. 22: 959–972.

**Golet, G.H.; Roberts, M.D.; Luster, R.A.; Werner, G.; Larsen, E.W.; Unger, R.; White, G.G. 2006.** Assessing societal impacts when planning restoration of large alluvial rivers: a case study of the Sacramento River project, California. Environmental management. 37(6): 862–879.

**Hammersmark, C.T.; Dobrowski, S.Z.; Rains, M.C.; Mount, J.F. 2010.** Simulated effects of stream restoration on the distribution of wet-meadow vegetation. Restoration Ecology. 18(6): 882–893.

**Hammersmark, C.T.; Rains, M.C.; Mount, J.F. 2008.** Quantifying the hydrological effects of stream restoration in a montane meadow, northern California, USA. River Research and Applications. 24(6): 735–753.

**Hatfield, R.G.; LeBuhn, G. 2007.** Patch and landscape factors shape community assemblage of bumble bees, *Bombus* spp. (Hymenoptera: Apidae), in montane meadows. Biological Conservation. 139(1–2): 150–158.

**Henery, R.; Purdy, S.; Williams, J.; Hatch, J.; Fesenmyer, K.; Drew, M.; Lass, D.; Knight, C. 2011.** Meadow restoration to sustain stream flows and native trout: a novel approach to quantifying the effects of meadow restorations to native trout. Davis, CA: Trout Unlimited, California Trout, University of Nevada, Reno, and University of California–Davis. 110 p.

**Herbst, D.B.; Bogan, M.T.; Roll, S.K.; Safford, H.D. 2012.** Effects of livestock exclusion on in-stream habitat and benthic invertebrate assemblages in montane streams. Freshwater Biology. 57(1): 204–217.

**Herbst, D.B.; Kane, J.M. 2009.** Responses of aquatic macroinvertebrates to stream channel reconstruction in a degraded rangeland creek in the Sierra Nevada. Ecological Restoration. 27(1): 76–88.

**Higgs, E.S. 1997.** What is good ecological restoration? Conservation Biology. 11(2): 338–348.

**Hill, B.; Mitchell-Bruker, S. 2010.** Comment on "A framework for understanding the hydroecology of impacted wet meadows in the Sierra Nevada and Cascade Ranges, California, USA." Hydrogeology Journal. 18(7): 1741–1743.

**Hobbs, R.J.; Cramer, V.A. 2008.** Restoration ecology: interventionist approaches for restoring and maintaining ecosystem function in the face of rapid environmental change. Annual Review of Environment and Resources. 33: 39–61.

**Hobbs, R.J.; Higgs, E.; Harris, J.A. 2009.** Novel ecosystems: implications for conservation and restoration. Trends in Ecology and Evolution. 24(11): 599–605.

**Hunt, L.; Nylen, B.D. 2012.** Evaluating and prioritizing meadow restoration in the Sierra. Nevada City, CA: American Rivers. 22 p.

**James, C.D.; Lanman, R.B. 2012.** Novel physical evidence that beaver historically were native to the Sierra Nevada. California Fish and Game. 98(2): 129–132.

**Johnston, C.A.; Naiman, R.J. 1990.** Aquatic patch creation in relation to beaver population trends. Ecology. 71(4): 1617–1621.

**Katz, S.L.; Barnas, K.; Hicks, R.; Cowen, J.; Jenkinson, R. 2007.** Freshwater habitat restoration actions in the Pacific Northwest: a decade's investment in habitat improvement. Restoration Ecology. 15(3): 494–505.

**Kauffman, J.B.; Beschta, R.L.; Otting, N.; Lytjen, D. 1997.** An ecological perspective of riparian and stream restoration in the western United States. Fisheries. 22(5): 12–24.

**Kiernan, J.D.; Moyle, P.B. 2012.** Flows, droughts, and aliens: factors affecting the fish assemblage in a Sierra Nevada, California, stream. Ecological Applications. 22(4): 1146–1161.

**Klimkowska, A.; van Diggelen, R.; den Held, S.; Brienen, R.; Verbeek, S.; Vegelin, K. 2009.** Seed production in fens and fen meadows along a disturbance gradient. Applied Vegetation Science. 12(3): 304–315.

**Knapp, R.A.; Matthews, K.R. 1996.** Livestock grazing, golden trout, and streams in the Golden Trout Wilderness, California: impacts and management implications. North American Journal of Fisheries Management. 16(4): 805–820.

**Kondolf, G.M. 1993.** Lag in stream channel adjustment to livestock exclosure in the White Mountains of California. Restoration Ecology. 1(4): 226–230.

**Kondolf, G.M. 2011.** Setting goals in river restoration: When and where can the river "heal itself"? In: Simon, A.; Bennett, S.J.; Castro, J.M., eds. Stream restoration in dynamic fluvial systems: scientific approaches, analyses, and tools. Book 194. Washington, DC: American Geophysical Union: 29–43.

**Kondolf, G.M.; Anderson, S.; Lave, R.; Pagano, L.; Merenlender, A.; Bernhardt, E.S. 2007.** Two decades of river restoration in California: What can we learn? Restoration Ecology. 15(3): 516–523.

**Lake, P.S.; Bond, N.; Reich, P. 2007.** Linking ecological theory with stream restoration. Freshwater Biology. 52(4): 597–615.

**Lanman, R.B.; Perryman, H.; Dolman, B.; James, C.D. 2012.** The historical range of beaver in the Sierra Nevada: a review of the evidence. California Fish and Game. 98(2): 65–80.

**Lave, R.; Doyle, M.; Robertson, M. 2010.** Privatizing stream restoration in the US. Social Studies of Science. 40(5): 677–703.

**Loheide, S.P.; Deitchman, R.S.; Cooper, D.J.; Wolf, E.C.; Hammersmark, C.T.; Lundquist, J.D. 2009.** A framework for understanding the hydroecology of impacted wet meadows in the Sierra Nevada and Cascade Ranges, California, USA. Hydrogeology Journal. 17(1): 229–246.

**Loheide, S.P.; Deitchman, R.S.; Cooper, D.J.; Wolf, E.C.; Hammersmark, C.T.; Lundquist, J.D. 2010.** Reply to comment on "A framework for understanding the hydroecology of impacted wet meadows in the Sierra Nevada and Cascade Ranges, California, USA." Hydrogeology Journal. 18(7): 1745–1746.

**Loheide, S.P.; Gorelick, S.M. 2005.** A local-scale, high-resolution evapotranspiration mapping algorithm (ETMA) with hydroecological applications at riparian meadow restoration sites. Remote Sensing of Environment. 98(2–3): 182–200.

**Loheide, S.P.; Gorelick, S.M. 2006.** Quantifying stream-aquifer interactions through the analysis of remotely sensed thermographic profiles and in situ temperature histories. Environmental Science Technology. 40(10): 3336–3341.

**Marty, J.T. 2005.** Effects of cattle grazing on diversity in ephemeral wetlands. Conservation Biology. 19(5): 1626–1632.

**McColley, S.D.; Tyers, D.B.; Sowell, B.F. 2012.** Aspen and willow restoration using beaver on the northern Yellowstone winter range. Restoration Ecology. 20(4): 450–455.

**Micheli, E.R.; Kirchner, J.W. 2002.** Effects of wet meadow riparian vegetation on streambank erosion. 1. Remote sensing measurements of streambank migration and erodibility. Earth Surface Processes and Landforms. 27(6): 627–639.

**Miller, J.R.; Kochel, R.C. 2010.** Assessment of channel dynamics, in-stream structures and post-project channel adjustments in North Carolina and its implications to effective stream restoration. Environmental Earth Sciences. 59(8): 1681–1692.

**Miller, S.W.; Budy, P.; Schmidt, J.C. 2010.** Quantifying macroinvertebrate responses to in-stream habitat restoration: applications of meta-analysis to river restoration. Restoration Ecology. 18(1): 8–19.

**Moyle, P.B.; Kiernan, J.D.; Crain, P.K.; Quiñones, R.M. 2012.** Projected effects of future climates on rreshwater fishes of California. CEC-500-2012-028. Sacramento, CA: California Energy Commission. 53 p.

**Myers, L.; Kane, J. 2011.** The impact of summer cattle grazing on surface water quality in high elevation mountain meadows. Water Quality, Exposure and Health. 3(1): 51–62.

**Myers, L.; Whited, B. 2012.** The impact of cattle grazing in high elevation Sierra Nevada mountain meadows over widely variable annual climatic conditions. Journal of Environmental Protection. 3(28A): 823–837.

**North, M.P.; Keeton, W.S. 2008.** Emulating natural disturbance regimes: an emerging approach for sustainable forest management. In: Lafortezza, R.; Chen, J.Q.; Sanesi, G.; Crow, T., eds. Patterns and processes in forest landscapes: sustainable management of forest landscapes. New York: Springer-Verlag: 341–372. Chapter 17.

**Norton, J.B.; Jungst, L.J.; Norton, U.; Olsen, H.R.; Tate, K.W.; Horwath, W.R. 2011.** Soil carbon and nitrogen storage in upper montane riparian meadows. Ecosystems. 14(8): 1217–1231.

**Null, S.E.; Viers, J.H.; Mount, J.F. 2010.** Hydrologic response and watershed sensitivity to climate warming in California's Sierra Nevada. Plos One. 5(3).

**Palmer, M.A. 2009.** Reforming watershed restoration: science in need of application and applications in need of science. Estuaries and Coasts. 32(1): 1–17.

**Palmer, M.A.; Ambrose, R.F.; Poff, N.L. 1997.** Ecological theory and community restoration ecology. Restoration Ecology. 5(4): 291–300.

**Poole, G.C.; Berman, C.H. 2001.** An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental Management. 27(6): 787–802.

**Purdy, S.E.; Moyle, P.B.; Tate, K.W. 2011.** Montane meadows in the Sierra Nevada: comparing terrestrial and aquatic assessment methods. Environmental Monitoring and Assessment. 184(11): 6967–6986.

**Ramstead, K.M.; Allen, J.A.; Springer, A.E. 2012.** Have wet meadow restoration projects in the Southwestern U.S. been effective in restoring geomorphology, hydrology, soils, and plant species composition? Environmental Evidence. 1(11): 16.

**Ratliff, R.D. 1985.** Meadows in the Sierra Nevada of California: state of knowledge. Gen. Tech. Rep. GTR-PSW-84. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 59 p.

**Roche, L.M.; Allen-Diaz, B.; Eastburn, D.J.; Tate, K.W. 2012b.** Cattle grazing and Yosemite toad (*Bufo canorus* Camp) breeding habitat in Sierra Nevada meadows. Rangeland Ecology and Management. 65(1): 56–65.

**Roche, L.M.; Latimer, A.M.; Eastburn, D.J.; Tate, K.W. 2012a.** Cattle grazing and conservation of a meadow-dependent amphibian species in the Sierra Nevada. Plos One. 7(4): e35734.

**Ruiz-Jaen, M.C.; Aide, T.M. 2005.** Restoration success: How is it being measured? Restoration Ecology. 13(3): 569–577.

**Sarr, D.A. 2002.** Riparian livestock exclosure research in the western United States: a critique and some recommendations. Environmental Management. 30(4): 516–526.

**Seavy, N.E.; Gardali, T.; Golet, G.H.; Griggs, F.T.; Howell, C.A.; Kelsey, R.; Small, S.L.; Viers, J.H.; Weigand, J.F. 2009.** Why climate change makes riparian restoration more important than ever: recommendations for practice and research. Ecological Restoration. 27(3): 330–338.

**Shields, F.D.; Lizotte, R.E.; Knight, S.S.; Cooper, C.M.; Wilcox, D. 2010.** The stream channel incision syndrome and water quality. Ecological Engineering. 36(1): 78–90.

**Sierra Nevada Science Review Team. 1998.** Sierra Nevada science review: report of the Science Review Team charged to synthesize new information of rangewide urgency to the national forests of the Sierra Nevada. Vallejo, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 149 p.

**Simon, A. 2008.** Fine-sediment loadings to Lake Tahoe. Journal of the American Water Resources Association. 44(3): 618–639.

**Simon, A.; Rinaldi, M. 2006.** Disturbance, stream incision, and channel evolution: the roles of excess transport capacity and boundary materials in controlling channel response. Geomorphology. 79(3–4): 361–383.

**SNEP Science Team. 1996a.** Rangelands. In: Status of the Sierra Nevada, Sierra Nevada ecosystem project: final report to Congress. Vol. I: Assessment summaries and management strategies. Report No. 36. Davis, CA: Centers for Water and Wildland Resources, University of California: 113–122. Chapter 7.

**SNEP Science Team. 1996b.** Status of the Sierra Nevada, Sierra Nevada ecosystem project: final report to Congress. Vol. I: Assessment summaries and management strategies. Report No. 36. Davis, CA: Centers for Water and Wildland Resources, University of California–Davis. 209 p.

**Steed, J.E.; DeWald, L.E. 2003.** Transplanting sedges (*Carex* spp.) in southwestern riparian meadows. Restoration Ecology. 11(2): 247–256.

**Stewart, G.B.; Bayliss, H.R.; Showler, D.A.; Sutherland, W.J.; Pullin, A.S. 2009.** Effectiveness of engineered in-stream structure mitigation measures to increase salmonid abundance: a systematic review. Ecological Applications. 19(4): 931–941.

**Stillwater Sciences. 2012.** A guide for restoring functionality to mountain meadows of the Sierra Nevada. Technical memorandum. Berkeley, CA. 74 p. http://www.stillwatersci.com/resources/2012meadowrestguide.pdf. (9 September 2014).

**Tague, C.; Valentine, S.; Kotchen, M. 2008.** Effect of geomorphic channel restoration on streamflow and groundwater in a snowmelt-dominated watershed. Water Resources Research. 44(10): 1–10.

**Trimble, S.W.; Mendel, A.C. 1995.** The cow as a geomorphic agent—a critical review. Geomorphology. 13(1–4): 233–253.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2004.** Sierra Nevada Forest Plan Amendment, Final Supplemental Environmental Impact Statement Volume 4, Appendix E– Adaptive Management Strategy. Vallejo, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Region. 134 p.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2011a.** Watershed condition framework: a framework for assessing and tracking changes to watershed condition. FS-977. Washington, DC. 34 p.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2011b.** Region Five ecological restoration: leadership intent. Vallejo, CA: Pacific Southwest Region. 4 p.

**Viers, J.H.; Purdy, S.E.; Peek, R.; Fryjoff-Hung, A.; Santos, N.R.; Katz, J.; Emmons, J.D.; Dolan, D.V.; Yarnell, S.M. 2013.** Montane meadows in the Sierra Nevada: changing hydroclimatic conditions and concepts for vulnerability assessment. Tech. Rep. Davis, CA: University of California, Davis, Center for Watershed Sciences. 63 p.

**Viers, J.H.; Rheinheimer, D.E. 2011.** Freshwater conservation options for a changing climate in California's Sierra Nevada. Marine and Freshwater Research. 62: 266–278.

**Weixelman, D.A.; Hill, B.; Cooper, D.J.; Berlow, E.L.; Viers, J.H.; Purdy, S.E.; Merrill, A.G.; Gross, S.E. 2011.** A field key to meadow hydrogeomorphic types for the Sierra Nevada and southern Cascade Ranges in California. Gen. Tech. Rep. R5-TP-034. Vallejo, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Region. 33 p.

**Weixelman, D.A.; Zamudio, D.C.; Zamudio, K.A. 1999.** Eastern Sierra Nevada riparian field guide. R4-ECOL-99-01. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. [Irregular pagination].

**Zonge, K.L.; Swanson, S.; Myers, T. 1996.** Drought year changes in streambank profiles on incised streams in the Sierra Nevada Mountains. Geomorphology. 15(1): 47–56.

# Chapter 6.4—Lakes: Recent Research and Restoration Strategies

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## Summary

The Sierra Nevada and southern Cascade Range support thousands of montane lakes, from small, remote tarns to iconic destinations such as Lake Tahoe. Their beauty and recreational opportunities instill high social value, in particular by serving as destinations for hiking, camping, swimming, and fishing. Lakes also have high ecological value because they support a diverse aquatic fauna, including rare species such as the mountain yellow-legged frog (*Rana muscosa*/*R. sierrae*), and they provide food for terrestrial and aquatic predators. A number of stressors interact to affect lake ecosystems in the synthesis area. Climate change is expected to affect lakes by altering physical processes and reducing water levels. In shallow lakes and ponds, reduced hydroperiods could directly reduce the amount of available habitat for lentic amphibians and increase the instances of stranding mortality of eggs and tadpoles. Introductions of fish into lakes have altered food webs and particularly affected native amphibians. Chytridiomycosis, an amphibian-specific fungal disease caused by *Batrachochytrium dendrobatidis* (*Bd*), has caused significant declines and extirpations in populations of amphibians native to the Sierra Nevada and southern Cascade Range. Research is ongoing to determine ways to reduce impacts of *Bd* on native amphibian populations. Air pollution has potential to negatively affect lake-dwelling amphibians, especially owing to interactions with other stressors; however, recent studies from the synthesis area did not find associations between frog population status and measured pesticide concentrations. A metric using the ratio of the number of taxa observed at a site to that expected can be an effective tool for assessing resistance and resilience of expected native taxa to threats. Successful restorations will likely depend on the control of introduced fish, the presence and virulence of *Bd*, and habitat conditions that help frogs to withstand these and other stressors.

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## Socioecological Significance of Lakes

The Sierra Nevada and southern Cascade Range contain thousands of natural lakes that provide both ecological and recreational values. In high-elevation environments characterized by low productivity, lakes provide food and energy to terrestrial consumers, such as birds (Epanchin et al. 2010) and snakes (Matthews et al. 2002, Pope et al. 2008). In addition, native endemic species, such as the Sierra Nevada yellow-legged frog (*Rana sierrae*) (fig. 1) and Yosemite toad (*Anaxyrus* [=*Bufo*] *canorus*), are found in California's mountain lakes. These species, in addition to the Sierra Nevada population of mountain yellow-legged frog (*R. muscosa*), were recently added to the federal list of threatened and endangered species.

Because of their great beauty, lakes also serve as important destinations for recreational activities, including hiking, camping, swimming, and fishing. Historically, nearly all lakes in the high Sierra Nevada (above 1800 m) were fishless, but the introduction of nonnative trout has been a common practice since the early 1900s (Knapp 1996). Currently, the majority of large, deep lakes support introduced populations of fish, including brook trout (*Salvelinus fontinalis*), rainbow trout (*Oncorhynchus mykiss*), and brown trout (*Salmo trutta*) (CDFG 2011, Knapp and Matthews 2000). This fishery supports a multimillion dollar recreational industry by bringing anglers to the region.

*Kathleen Matthews*

Figure 1—Recent metamorph and adult forms of the Sierra Nevada yellow-legged frog.

## Conservation Issues and Restoration Options

### Climate Change

Lakes serve as sentinels of climate change because they provide easily detectable signals that may reflect the influence of the change on the broader catchment area (Williamson et al. 2009). Changes in lake temperatures and water levels are easy to measure. Available records can span extended periods; for example, Schneider et al. (2009) found that six large lakes in California (including Lake Tahoe) have been warming an average of 0.11 ± 0.02 °C per year since 1992. These rates of change are about twice as high as regional trends in air temperature (Schneider et al. 2009). As a consequence of climate change, increased surface water warming rates will likely affect lake ecosystems of the Sierra Nevada by potentially decreasing lake levels, but also by altering critical physical processes, such as stratification and deep mixing (Sahoo et al. 2012). In addition, predicted treeline advancement (Harsch et al. 2009) and increases in terrestrial vegetation surrounding alpine lakes (Luckman and Kavanagh 2000) may result in increased inputs of terrestrially derived dissolved organic matter into montane lakes (Sommaruga et al. 1999). These coupled changes in climate variability and physical and chemical processes have been found to dramatically alter the distribution and abundance of important lake zooplankton such as the grazer, *Daphnia* (Fischer et al. 2011). In addition, lake warming and increased nutrient inputs were found to enhance the exchange of energy and organisms between lakes and neighboring terrestrial ecosystems by increasing emergence of aquatic insects and increasing decomposition rates (Greig et al. 2012).

Climate change may interact with other stressors to alter lake community dynamics. Warmer air temperatures are predicted to result in less annual snowpack in the Sierra Nevada (Cayan et al. 2008, Young et al. 2009), which in turn will affect the hydroperiod of small lakes and ponds. Sierra Nevada yellow-legged frogs and mountain yellow-legged frogs (hereafter referred to as mountain yellow-legged frogs) have a 2- to 3-year larval stage and may experience an interactive effect of lower water levels and introduced fishes, with fish preventing breeding in the majority of deep lakes (Vredenburg 2004) and climate change increasing the likelihood of drying of shallow lakes and ponds that do not support fish (Lacan et al. 2008). Based on this expected interaction, mitigations for climate change effects on native lentic species include removing fish from some larger lakes to provide additional fish-free, permanent, cool water refuge habitat (see discussion below).

**Threatened amphibians face threats from introduced fishes in deep lakes and drying out in shallow ponds that do not support fish.**

## Introduced Species

Invasive species are a major threat to freshwater ecosystems and can cause a range of impacts, including behavioral shifts of native species and restructuring of food webs (e.g., Cucherouset and Olden 2011, Simon and Townsend 2003). Climate change is expected to accelerate the spread of aquatic invasive species (Rahel and Olden 2008). Although many invasive species have not entered the high Sierra Nevada's natural mountain lakes, some, such as the American bullfrog (*Lithobates catesbeiana*) and signal crayfish (*Pacifastacus leniusculus*), do occur in many of the reservoirs and constructed ponds in the range. These impoundments are much more likely to harbor invasive species than are natural lakes (Johnson et al. 2008), and they can allow invasive species to establish self-sustaining populations that can then spread to the adjacent natural waters (e.g., Ding et al. 2008). Additional detrimental invaders, such as the New Zealand mudsnail (*Potamopyrgus antipodarum*) and Quagga mussel (*Dreissena rostiformis bugensis*), are spreading rapidly in California's freshwater habitats (Karatayev et al. 2012, Richards 2002). The Sierra Nevada's natural lakes are relatively discrete ecosystems separated by dry land that is inhospitable to most aquatic species. In addition, limited road access to many lakes, as well as inspection programs, may inhibit unintentional spread of aquatic invasive species via recreational boats (Johnson et al. 2001). As a result, dispersal of invasive species may be expected to be slow, and many suitable mountain lake ecosystems have remained uninvaded for long periods (Johnson et al. 2008).

Nonetheless, aquatic invasive species pose a serious threat to native biodiversity and associated ecological services (Johnson et al. 2008). Once established, invasive species such as crayfish, sunfish, bass, and aquatic weeds have been shown to affect native biodiversity, reduce water quality, and create food and other habitat conditions that benefit other nonnative species (Caires et al. 2013, McCarthy et al. 2006, Nilsson et al. 2012). Losses of ecosystem services associated with introductions of species such as crayfish appear to have outweighed gains in some areas. Aquatic invertebrates are major components not only of aquatic communities but also adjacent terrestrial habitats. Larval insects and zooplankton serve as prey for larger aquatic insects and amphibians, and the winged adult stages of insects feed terrestrial predators, such as birds, bats, and spiders (Nakano and Murakami 2001, Sanzone et al. 2003). In high-elevation lakes, introduced trout can produce strong top-down effects on aquatic invertebrates (Knapp et al. 2005, Pope et al. 2009, Vadeboncoeur et al. 2003). This is important because changes in invertebrate abundance and composition can have cascading consequences for nutrient cycling (Schindler et al. 2001) and terrestrial communities (Epanchin et al. 2010, Finlay and Vredenburg 2007, Knight et al. 2005). For example, Knight et al. (2005) found that

fish reduce dragonfly emergence, with subsequent consequences on native pollinators and terrestrial plant reproduction. In the Sierra Nevada, Epanchin et al. (2010) linked introduced fish to dramatic decreases in mayfly emergence, with indirect consequences for feeding gray-crowned rosy-finches (*Leucosticte tephrocotis dawsoni*). These studies show that the effects of introduced trout permeate beyond the lake boundary and have ramifications for neighboring terrestrial ecosystems. These complex interactions are an important topic for further investigation to better understand potential implications of managing nonnative species.

Preventing establishment of aquatic invaders is recognized as a core management strategy because eradication and control are costly and difficult following establishment (Simberloff et al. 2005). When the basic biology of an invasive species is known, recent studies have been successful in identifying vulnerable sites so that prevention efforts can be targeted to where they are likely to produce the greatest ecological benefit (Olden et al. 2011, Vander Zanden and Olden 2008). Development and implementation of comprehensive invasive species plans for lake habitats of the synthesis area could help prevent or slow the secondary spread of California's freshwater invaders into the Sierra Nevada's backcountry lakes (e.g., Vander Zanden and Olden 2008).

Some species are stocked and actively maintained because they have social and economic value, especially for recreational fishing, and are therefore not considered nuisances. However, some of these species, such as nonnative trout, may exert negative effects on native species that historically inhabited these ecosystems. For example, deep, permanent waters are critical as overwintering and breeding habitat for mountain yellow-legged frogs (Bradford 1989, Knapp and Matthews 2000). Introduced trout are significant predators of these frogs (Grinnell and Storer 1924, Vredenburg 2004), and many studies have found that breeding populations of mountain yellow-legged frogs rarely co-occur with nonnative trout (Knapp 2005, Knapp and Matthews 2000).

California is one of several states that initiated reviews of hatchery and stocking programs in response to concerns over effects on wild populations of species listed under the Endangered Species Act (Kostow 2009). Recent policy changes by the California Department of Fish and Wildlife (CDFW, the agency responsible for fish stocking) have reduced fish stocking in water bodies where sensitive native fish and amphibians may occur. Introduced fish are likely to remain ubiquitous in Sierra Nevada lakes even where they are no longer stocked. For example, Armstrong and Knapp (2004) found that lakes in the John Muir Wilderness with >2.1 m of spawning gravels that were <3520 m in elevation nearly always showed signs of supporting self-sustaining populations of rainbow and golden trout.

Nevertheless, the stocking of fish in aquatic water bodies represents a management action over which participating agencies have the ability to exert both direct and indirect controls. Because stocking has occurred throughout the Sierra Nevada and southern Cascade Range, actions taken on this issue have the potential to be far reaching. Further, the rapid recovery of native frog populations and other native species following fish removal in lentic systems (Knapp et al. 2001, Knapp et al. 2007, Pope 2008, Vredenburg 2004) indicates that fish removals have the potential to be effective restoration tools. Both CDFW and the National Park Service have integrated strategic fish removal projects into their resource management plans, and the Forest Service is implementing fish removal projects in the Sierra Nevada (e.g., Desolation Wilderness, El Dorado National Forest). These projects focus on headwater lake basins with high ecological value and they make up a small fraction of fish-containing lakes in the Sierra Nevada. Projects to remove fish from Sierra Nevada lakes have primarily used non-chemical methods, such as setting gill nets and electrofishing in inlets and outlets, to reduce impacts to nontarget organisms.

**Determining lake basins to focus on native amphibian restoration involves science and management working together to maximize the positive outcomes of restoration actions while maintaining recreational fisheries.**

Research to understand the mechanisms of recovery following fish removals can help managers determine characteristics of lakes and amphibian populations best suited for recovery. For example, Pope (2008) found that the high reproductive output of Cascades frogs (*Rana cascadae*) may allow rapid population growth, even with only a small number of breeding-aged frogs onsite. Knapp et al. (2007) found that following rapid population increases, mountain yellow-legged frogs disperse to neighboring suitable habitat if it is available. Determining lake basins to focus on native amphibian restoration involves science and management working together to maximize the positive outcomes of restoration actions while maintaining recreational fisheries. Important basin characteristics include the presence of target amphibians and additional nearby suitable habitat, the ability to successfully eliminate introduced fish, and the level of use by anglers. Because it is extremely difficult to remove fish from streams without the use of toxicants (e.g., rotenone), lakes with natural fish barriers near their inlets and outlets are better targets than lakes without natural fish barriers (e.g., Knapp et al. 2007, Pope 2008).

## Amphibian Disease

Although amphibians are susceptible to a wide array of diseases, one disease has emerged as the greatest conservation concern for amphibians in the Sierra Nevada (and the world). Chytridiomycosis, an amphibian-specific fungal disease caused by *Batrachochytrium dendrobatidis* (*Bd*), has been implicated in declines and extinctions of amphibian populations worldwide (e.g., Bosch et al. 2001, Lips et al. 2004, Muths et al. 2003), and has contributed to widespread declines of mountain

yellow-legged frogs throughout the Sierra Nevada (Briggs et al. 2010, Rachowicz et al. 2006, Vredenburg et al. 2010). Mass die-offs of frogs and population extirpations have been observed soon after the arrival of *Bd* in mountain yellow-legged frog populations (Vredenburg et al. 2010). Although many populations have been driven to extinction by *Bd*, some populations have survived the population crash and persist with the disease (Briggs et al. 2010, Knapp et al. 2011). Research is ongoing to determine factors that allow some populations to persist while others are extirpated (Knapp et al. 2011). One important finding is that large populations have a higher likelihood of persisting with the disease (Knapp et al. 2011). This finding suggests that although *Bd* can be devastating to populations even where fish have been removed, these "recovered" frog populations have a better chance of persisting in the long term than those facing the additional stressor of introduced fish.

Options to help ameliorate the impacts of *Bd* include developing protected populations and prophylactic or remedial disease treatment (Woodhams et al. 2011). Because sustainable conservation in the wild is dependent on long-term population persistence, successful disease mitigation would include managing already infected populations by decreasing pathogenicity and host susceptibility so that co-evolution with those potentially lethal pathogens can occur (Woodhams et al. 2011). Currently, researchers are working to identify mechanisms of disease suppression and develop adaptive management strategies to be tested in field trials with natural populations. Antimicrobial skin peptides and microbes that inhibit infection by *Bd* have been identified from the skin of mountain yellow-legged frogs and may be useful tools for increasing the frog's resistance to *Bd* (Harris et al. 2009, Woodhams et al. 2007).

## Pollution

Exposure to pesticides transported to the Sierra Nevada by prevailing winds from California's Central Valley has been hypothesized as a cause of population declines of native amphibians, primarily along the west slope of the Sierra Nevada (Davidson 2004, Fellers et al. 2004). Pesticide residues from Central Valley agricultural areas have been found in samples of air, snow, surface water, lake sediments, amphibians, and fish across the Sierra Nevada (Cory et. al 1970, Fellers et al. 2004, Hageman et al. 2006, McConnell et al. 1998), and windborne contaminants have been linked to patterns of decline of mountain yellow-legged frogs (Davidson and Knapp 2007). However, a recent study that compared concentrations of historically and currently used pesticides with the population status of mountain yellow-legged frogs in the southern Sierra Nevada found no association between frog population status and measured pesticide concentrations (Bradford et al. 2011). In addition,

in both the Sierra Nevada and southern Cascade Range, pesticide concentrations in water and amphibian tissue were consistently below concentrations found to be toxic to amphibians (Bradford et al. 2011, Davidson et al. 2012). Low concentrations of pesticides could, however, interact with other stressors and contribute to adverse effects on frogs. For example, the pesticide carbaryl was found to reduce production of amphibian skin peptides that inhibit the growth of *Bd*, the fungus that causes chytridiomycosis (Davidson et al. 2007). For practical reasons, most field studies continue to focus on single stressors; however, the few studies assessing interactive effects of low-level contaminants with other stressors highlight the need for additional multifactor studies.

## Terrestrial Influences on Lakes

Mountain lakes receive organic material from two different sources: primary production that occurs within the system's boundaries (autochthonous sources) and primary production imported from the terrestrial watershed (allochthonous sources). It is important to understand the relative significance of those sources for particular lakes and how those relationships may change over time given disturbances and stressors. Determining the source and magnitude of inputs will help to predict the trajectories of variables important to management objectives (McIntire et al. 2007). New research using multiple techniques including stable isotopes has shown that terrestrial organic material provides major support for both benthic and pelagic organisms in small lakes (Bade et al. 2007, Cole et al. 2011). General disturbance ecology suggests that periodic human-induced watershed and riparian disturbances that emulate natural disturbance regimes may initiate renewal processes that are required for long-term sustainability of aquatic ecosystems (Kreutzweiser et al. 2012). Consequently, disturbance-based forest management approaches such as those described in PSW-GTR-220 (North et al. 2009) may result in long-term positive feedbacks in forested lake systems in the Sierra Nevada.

However, Spencer et al. (2003) cautioned that expanded fire activity and other treatments associated with fuel reductions could deliver nutrients to lakes and other surface waters that may already be threatened by eutrophication. Their study focused on the northern Rocky Mountains, where the reference fire regime ranged from decades to several hundred years. They suggested that there was potential to shift regimes from infrequent pulses of nutrients to an annual loading from the combination of wildfires, prescribed burns, pile burning, and agricultural burning. Another stressor to consider is the nutrient loading through long-distance aerial transport (see chapter 8.1, "Air Quality." They noted the need to evaluate impacts of large wildfires on aquatic ecosystems. Accordingly, an important avenue for scientific research is

to better understand linkages between terrestrial and aquatic ecosystems, including lakes, especially in the context of large wildfires and various stressors (see chapter 1.5, "Research Gaps: Adaptive Management to Cross-Cutting Issues").

## Metrics for Lake Assessments

With multiple serious threats facing lake ecosystems in the Sierra Nevada (fig. 2), it is valuable to understand the degree to which these systems are altered. A metric using the ratio of the number of taxa observed (O) at a site to that expected (E) to occur at the site in the absence of anthropogenic impacts (abbreviated as O/E) has been applied to lakes with fish in the Sierra Nevada (Knapp et al. 2005). The authors found that amphibians, reptiles, benthic invertebrates, and zooplankton have relatively low resistance to fish introductions, but all taxa recover when lakes revert to a fishless condition. This metric proved effective at assessing resistance and resilience of expected native taxa in Sierra Nevada lakes to one threat (introduced fish), and it may be effective at assessing other threats. Since then, Van Sickle (2008) has suggested an alternative statistical approach that could be more sensitive

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Figure 2—Dusy Basin, Kings Canyon National Park.

to anthropogenic stressors; this approach uses BC, an adaptation of Bray-Curtis distance, instead of O/E to assess compositional dissimilarity between an observed and expected assemblage.

Although comparisons to reference systems are important, assessment of long-term trends in lake dynamics is another important component for understanding the patterns of change associated with both natural and human-caused stressors (e.g., Kowalewski et al. 2013, McIntire et al. 2007). Use of both paleolimnological evidence and long-term monitoring provide insight into mechanisms for observed changes and provide more accurate baselines for potential restoration efforts (Kowelewski et al. 2013). For example, a 16-year study at Crater Lake, Oregon, revealed that changes in lake productivity (and thus water clarity) were likely driven by long-term climatic changes that regulate the supply of allochthonous nutrients into the lake (McIntire et al. 2007).

**Recent research has shown that successful restorations will likely depend on the elimination of introduced fish, regardless of whether *Bd* is present and how virulent it is if it is present, and habitat conditions that help frogs to withstand the effects of climate change and disease.**

Although metrics based upon species composition may be useful in assessing the overall condition of the aquatic biota, priority will likely be given to restoration of lakes in the Sierra Nevada and southern Cascade Range to support populations of amphibian species, such as the Sierra Nevada and southern mountain yellow-legged frogs and the Cascades frog. Recent research has shown that successful restorations will likely depend on the elimination of introduced fish, regardless of whether *Bd* is present and how virulent it is if it is present, and habitat conditions that help frogs to withstand the effects of climate change and disease.

**Box 6.4-1**

## Recent and Pending Studies on Frogs and Chytrid Disease

Pending research may suggest additional considerations to help frogs withstand the novel threat posed by the fungal disease chytridiomycosis, which is caused by *Batrachochytrium dendrobatidis* (*Bd*). For example, a recent lab study comparing the effects of *Bd* isolated from two localities in northern California found dramatic differences in the virulence of the isolates on Cascades frogs (*Rana cascadae*), with one isolate causing nearly complete mortality in test animals within 6 weeks and the other causing a mortality rate only slightly higher than that of unexposed animals.[3] This result suggests that in addition to environmental conditions and local host population size, the local strain of *Bd* may also play an important role in determining host population dynamics.

[3] Piovia-Scott et al., 2013. Unpublished data. On file with: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Redwood Sciences Lab, 1700 Bayview Drive, Arcata, CA 95521.

*(continued on next page)*

Furthermore, preliminary field testing of supplementing microbes on the skin of mountain yellow-legged frogs have shown promising results in boosting survival with the disease.[4]

A recent study by the Pacific Southwest Research Station and University of California–Davis collaborators found that all the remnant populations of Cascades frogs in the southern Cascades survive with *Bd* and occur in habitats where adult frogs commonly move among different water bodies for breeding, summer feeding, and overwintering. When the frogs are in the breeding sites, they tend to have high prevalence and loads of *Bd*, but when they move to streams and channels, loads of *Bd* are dramatically reduced or eliminated.[5] One hypothesis is that the movement away from breeding sites where the disease may thrive allows the frogs to behaviorally eliminate the disease, and thus, the population is able to persist. If this is true, restoration of lake basins with additional stream and meadow habitats nearby may be most effective where *Bd* is a concern.

[4] Vredenburg, V.T.; Briggs, C.J.; Harris, R. 2011. Host-pathogen dynamics of amphibian chytridiomycosis: the role of the skin microbe in health and disease. In: Olsen, L., ed. Fungal diseases: an emerging threat to human, animal and plant health: workshop summary. Washington, DC: National Academies Press: 342–354.

[5] Pope et al. 2013. Unpublished data in final report to the Lassen National Forest (ISA #12-05-06-02) on the status of the Cascades frog (*Rana cascadae*) in the southern Cascade Mountains of California. On file with: U.S. Department of Agriculture, Forest Service, Lassen National Forest, 2550 Riverside Drive, Susanville, CA 96130.

## Management Implications

- Removal of introduced fish is an important strategy to help priority amphibians species withstand combined stressors including climate change and disease.
- Chytridiomycosis continues to be a major conservation concern for frogs in mountain lake ecosystems. Researchers are working to develop treatment options to help infected populations.

## Literature Cited

**Armstrong, T.W.; Knapp, R.A. 2004.** Response by trout populations in alpine lakes to an experimental halt to stocking. Canadian Journal of Fisheries and Aquatic Sciences. 61: 2025–2037.

**Bade, D.L.; Carpenter, S.R.; Cole, J.J.; Pace, M.L.; Kritzberg, E.; Van de Bogert, M.C.; Cory, R.M.; McKnight, D.M. 2007.** Sources and fates of dissolved organic carbon in lakes as determined by whole-lake carbon isotope additions. Biogeochemistry. 84(2): 115–129.

**Bosch, J.; Martinez-Solano, I.; Garcia-Paris, M. 2001.** Evidence of a chytrid fungus infection involved in the decline of the common midwife toad (*Alytes obstetricans*) in protected areas of central Spain. Biological Conservation. 97: 331–337.

**Bradford, D.F. 1989.** Allotopic distribution of native frogs and introduced fishes in high Sierra Nevada lakes of California: implication of the negative effect of fish introductions. Copeia. 1989(3): 775–778.

**Bradford, D.F.; Knapp, R.A.; Sparling, D.W.; Nash, M.S.; Stanley, K.A.; Tallent-Halsell, N.G.; McConnell, L.L.; Simonich, S.M. 2011.** Pesticide distributions and population declines of California, USA, alpine frogs, *Rana muscosa* and *Rana sierrae*. Environmental Toxicology and Chemistry. 30(3): 682–691.

**Briggs, C.J.; Knapp, R.A.; Vredenburg, V.T. 2010.** Enzootic and epizootic dynamics of the chytrid fungal pathogen of amphibians. Proceedings of the National Academy of Sciences of the United States of America. 107: 9695–9700.

**Caires, A.M.; Chandra, S.; Hayford, B.L.; Wittmann, M.E. 2013.** Four decades of change: dramatic loss of zoobenthos in an oligotrophic lake exhibiting gradual eutrophication. Freshwater Science. 32(3): 692–705.

**California Department of Fish and Game [CDFG]. 2011.** A status review of the mountain yellow-legged frog (*Rana muscosa* and *Rana sierrae*). A report to the California Fish and Game Commission. 52 p. http://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=40357. (17 December 2013).

**Cayan, D.R.; Maurer, E.P.; Dettinger, M.D.; Tyree, M.; Hayhoe, K. 2008.** Climate change scenarios for the California region. Climatic Change. 87: S21–S42.

**Cole, J.J.; Carpenter, S.R.; Kitchell, J.; Pace, M.L.; Solomon, C.T.; Weidel, B. 2011.** Strong evidence for terrestrial support of zooplankton in small lakes based on stable isotopes of carbon, nitrogen, and hydrogen. Proceedings of the National Academy of Sciences of the United States of America. 108(5): 1975–1980.

**Cory, L.; Fjeld, P.; Serat, W. 1970.** Distributional patterns of DDT residues in the Sierra Nevada mountains. Pesticides Monitoring Journal. 3: 204–211.

**Cucherouset, J.; Olden, J.D. 2011.** A conspectus of the ecological impacts of invasive freshwater fishes. Fisheries. 36: 215–230.

**Davidson, C. 2004.** Declining downwind: amphibian population declines in California and historic pesticide use. Ecological Applications. 14: 1892–1902.

**Davidson, C.; Benard, M.F.; Shaffer, H.B.; Parker, J.M.; O'Leary, C.; Conlon, J.M.; Rollins-Smith, L.A. 2007.** Effects of chytrid and carbaryl exposure on survival, growth and skin peptide defenses in foothill yellow-legged frogs. Environmental Science & Technology. 41(5): 1771–1776.

**Davidson, C.; Knapp, R.A. 2007.** Multiple stressors and amphibian declines: dual impacts of pesticides and fish on yellow-legged frogs. Ecological Applications. 17: 587–597.

**Davidson, C.; Stanley, K.; Simonich, S.M. 2012.** Contaminant residues and declines of the Cascades frog (*Rana cascadae*) in the California Cascades, USA. Environmental Toxicology and Chemistry. 31: 1895–1902.

**Ding, J.Q.; Mack, R.N.; Lu, P.; Ren, M.; Huang, H. 2008.** China's booming economy is sparking and accelerating biological invasions. Bioscience. 58: 317–324.

**Epanchin, P.N.; Knapp, R.A.; Lawler, S.P. 2010.** Non-native trout impact an alpine-nesting bird by altering aquatic-insect subsidies. Ecology. 91: 2406–2415.

**Fellers, G.M.; McConnell, L.L.; Pratt, D.; Datta, S. 2004.** Pesticides in mountain yellow-legged frogs (*Rana muscosa*) from the Sierra Nevada mountains of California, USA. Environmental Toxicology and Chemistry. 23: 2170–2177.

**Finlay, J.C.; Vredenburg, V.T. 2007.** Introduced trout sever trophic connections in watersheds: consequences for a declining amphibian. Ecology. 88: 2187–2198.

**Greig, H.S.; Kratina, P.; Thompson, P.L.; Palen, W.J.; Richardson, J.S.; Shurin, J.B. 2012.** Warming, eutrophication, and predator loss amplify subsidies between aquatic and terrestrial ecosystems. Global Change Biology. 18(2): 504–514.

**Grinnell, J.; Storer T.I. 1924.** Animal life in the Yosemite. Berkeley, CA: University of California Press. 752 p.

**Hageman, K.J.; Simonich, S.L.; Campbell, D.H.; Wilson, G.R.; Landers, D.H. 2006.** Atmospheric deposition of current-use and historic-use pesticides in snow at national parks in the Western United States. Environmental Science & Technology. 40(10): 3174–3180.

**Harris, R.N.; Brucker, R.M.; Walke, J.B.; Becker, M.H.; Schwantes, C.R.; Flaherty, D.C.; Lam, B.A.; Woodhams, D.C.; Briggs, C.J.; Vredenburg, V.T.; Minbiole, K.P.C. 2009.** Skin microbes on frogs prevent morbidity and mortality caused by a lethal skin fungus. Isme Journal. 3(7): 818–824.

**Harsch, M.A.; Hulme, P.E.; Mcglone, M.S.; Duncan, R.P. 2009.** Are treelines advancing? A global meta-analysis oftreeline response to climate warming. Ecology Letters. 12: 1040–1049.

**Johnson, L.E.; Ricciardi A.; Carlton J.T. 2001.** Overland dispersal of aquatic invasive species: a risk assessment of transient recreational boating. Ecological Applications. 11: 1789–1799.

**Johnson, P.T.J.; Olden, J.D.; Vander Zanden, M.J. 2008.** Dam invaders: impoundments facilitate biological invasions into freshwaters. Frontiers in Ecology and the Environment. 6: 357–363.

**Karatayev, A.Y.; Claudi, R.; Lucy, F.E. 2012.** History of Dreissena research and the ICAIS gateway to aquatic invasions science. Aquatic Invasions 7: 1–5.

**Knapp, R.A. 1996.** Non-native trout in natural lakes of the Sierra Nevada: an analysis of their distribution and impacts on native aquatic biota. In: Sierra Nevada ecosystem project: final report to Congress. Vol. III: Assessments, commissioned reports, and background information. Report No. 38. Davis, CA: Centers for Water and Wildland Resources, University of California–Davis: 363–407.

**Knapp, R.A. 2005.** Effects of nonnative fish and habitat characteristics on lentic herpetofauna in Yosemite National Park, USA. Biological Conservation. 121: 265–279.

**Knapp, R.A.; Boiano, D.M.; Vredenburg, V.T. 2007.** Removal of nonnative fish results in population expansion of a declining amphibian (mountain yellow-legged frog, *Rana muscosa*). Biological Conservation. 135: 11–20.

**Knapp, R.A.; Briggs, C.J.; Smith, T.C.; Maurer, J.R. 2011.** Nowhere to hide: impact of a temperature-sensitive amphibian pathogen along an elevation gradient in the temperate zone. Ecosphere. 2: 1–26.

**Knapp, R.A.; Hawkins, C.P.; Ladau, J.; McClory, J.G. 2005.** Fauna of Yosemite National Park lakes has low resistance but high resilience to fish introductions. Ecological Applications. 15: 835–847.

**Knapp, R.A.; Matthews, K.R. 2000.** Non-native fish introductions and the decline of the mountain yellow-legged frog from within protected areas. Conservation Biology. 14: 428–438.

**Knapp, R.A.; Matthews, K.R.; Sarnelle, O. 2001.** Resistance and resilience of alpine lake fauna to fish introductions. Ecological Monographs. 71: 401–421.

**Knight, T.M.; McCoy, M.W.; Chase, J.M.; McCoy, K.A.; Holt, R.D. 2005.** Trophic cascades across ecosystems. Nature. 437: 880–883.

**Kostow, K. 2009.** Factors that contribute to the ecological risks of salmon and steelhead hatchery programs and some mitigating strategies. Reviews in Fish Biology and Fisheries. 19(1): 9–31.

**Kowalewski, G.A.; Kornijów, R.; McGowan, S.; Woszczyk, M.; Suchora, M.; Bałaga, K.; Kaczorowska, A.; Gąsiorowski, M.; Szeroczyńska, K.; Wasiłowska, A. 2013.** Persistence of protected, vulnerable macrophyte species in a small, shallow eutrophic lake (eastern Poland) over the past two centuries: implications for lake management and conservation. Aquatic Botany. 106(0): 1–13.

**Kreutzweiser, D.P.; Sibley, P.K.; Richardson, J.S.; Gordon, A.M. 2012.** Introduction and a theoretical basis for using disturbance by forest management activities to sustain aquatic ecosystems. Freshwater Science. 31: 224–231.

**Lacan, I.; Matthews, K.; Feldman, K. 2008.** Interaction of an introduced predator with future effects of climate change in the recruitment dynamics of the imperiled Sierra Nevada yellow-legged frog (*Rana sierrae*). Herpetological Conservation and Biology. 3: 211–223.

**Lips, K.; Mendelson, J.I.; Munoz-Alonso, A.; Canseco-Márquez, L.; Mulcahy, Daniel G. 2004.** Amphibian population declines in montane southern Mexico: resurveys of historical localities. Biological Conservation. 119: 555–564.

**Luckman, B.; Kavanagh, T. 2000.** Impact of climate fluctuations on mountain environments in the Canadian Rockies. Ambio. 29: 371–380.

**Matthews, K.R.; Knapp, R.A.; Pope, K.L. 2002.** Garter snake distributions in high-elevation aquatic ecosystems: is there a link with declining amphibian populations and nonnative trout introductions? Journal of Herpetology. 36: 16–22.

**McCarthy, J.M., Hein, C. L.; Olden, J.D.; Vander Zanden, M.J. 2006.** Coupling long-term studies with meta-analysis to investigate impacts of nonnative crayfish on zoobenthic communities. Freshwater Biology. 51: 224–235

**McConnell, L.L.; LeNoir, J.S.; Datta, S.; Seiber, J.N. 1998.** Wet deposition of current-use pesticides in the Sierra Nevada mountain range, California, USA. Environmental Toxicology and Chemistry. 10: 1908–1916.

**McIntire, C.D.; Larson, G.L.; Truitt, R.E. 2007.** Seasonal and interannual variability in the taxonomic composition and production dynamics of phytoplankton assemblages in Crater Lake, Oregon. Hydrobiologia. 574: 179–204.

**Muths, E.; Corn, P.S.; Pessier, A.P.; Green, D.E. 2003.** Evidence for disease-related amphibian decline in Colorado. Biological Conservation. 110: 357–365.

**Nakano S.; Murakami M. 2001.** Reciprocal subsidies: dynamic interdependence between terrestrial and aquatic food webs. Proceedings of the National Academy of Sciences of the United States of America. 98: 166–170.

**Nilsson, E.; Solomon, C.T.; Wilson, K.A.; Willis, T.V.; Larget, B.; Vander Zanden, M.J. 2012.** Effects of an invasive crayfish on trophic relationships in north-temperate lake food webs. Freshwater Biology. 57(1): 10–23.

**Pope, K.L. 2008.** Assessing changes in amphibian population dynamics following experimental manipulations of introduced fish. Conservation Biology. 22: 1572–1581.

**Pope, K.L.; Garwood, J.M.; Welsh, H.H.; Lawler S.P. 2008.** Evidence of indirect impacts of introduced trout on native amphibians via facilitation of a shared predator. Biological Conservation. 141: 1321–1331.

**Pope, K.L.; Piovia-Scott, J.; Lawler, S.P. 2009.** Changes in aquatic insect emergence in response to whole-lake experimental manipulations of introduced trout. Freshwater Biology. 54: 982–993.

**Rachowicz, L.J.; Knapp, R.A.; Morgan, J.A.T.; Stice, M.J.; Vredenburg, V.T.; Parker, J.M.; Briggs, C.J. 2006.** Emerging infectious disease as a proximate cause of amphibian mass mortality. Ecology. 87(7): 1671–1683.

**Richards, D.C. 2002.** The New Zealand mudsnail invades the Western United States. Aquatic Nuisance Species. 4: 42–44.

**Sahoo, G.B.; Schladow, S.G.; Reuter, J.E.; Coats, R.; Dettinger, M.; Riverson, J.; Wolfe, B.; Costa-Cabral, M. 2012.** The response of Lake Tahoe to climate change. Climatic Change: 1–25.

**Sanzone, D.M.; Meyer, J.L.; Marti, E.; Gardiner, E.P.; Tank, J.L.; Grimm, N.B. 2003.** Carbon and nitrogen transfer from a desert stream to riparian predators. Oecologia. 134: 238–250.

**Schindler, D.E.; Knapp, R.A.; Leavitt, P.R. 2001.** Alteration of nutrient cycles and algal production resulting from fish introductions in mountain lakes. Ecosystems. 4: 308–321.

**Schneider, P.; Hook, S.J.; Radocinski, R.G.; Corlett, G.K.; Hulley, G.C.; Schladow, S.G.; Steissberg, T.E. 2009.** Satellite observations indicate rapid warming trend for lakes in California and Nevada. Geophysical Research Letters. 36(22): L22402.

**Simberloff, D.I.; Parker, M.; Windle, P.N. 2005.** Introduced species: policy, management, and future research needs. Frontiers in Ecology and the Environment. 3: 12–20.

**Simon, K.S.; Townsend, C.R. 2003.** Impacts of freshwater invaders at different levels of ecological organisation, with emphasis on salmonids and ecosystem consequences. Freshwater Biology. 48: 982–994.

**Spencer, C.N.; Gabel, K.O.; Hauer, F.R. 2003.** Wildfire effects on stream food webs and nutrient dynamics in Glacier National Park, USA. Forest Ecology and Management. 178: 141–153.

**Vadeboncoeur, Y.; Jeppesen, E.; Vander Zanden, M.J.; Schierup, H.H.; Christoffersen, K.; Lodge D.M. 2003.** From Greenland to green lakes: cultural eutrophication and the loss of benthic pathways in lakes. Limnology and Oceanography. 48: 1408–1418.

**Vander Zanden, M.J.; Olden, J.D. 2008.** A management framework for preventing the secondary spread of aquatic invasive species. Canadian Journal of Fisheries and Aquatic Sciences. 65: 1512–1522.

**Van Sickle, J. 2008.** An index of compositional dissimilarity between observed and expected assemblages. Journal of the North American Benthological Society 27: 227–235.

**Vredenburg, V.T. 2004.** Reversing introduced species effects: experimental removal of introduced fish leads to rapid recovery of a declining frog. Proceedings of the National Academy of Sciences of the United States of America. 101: 7646–7650.

**Vredenburg, V.T.; Knapp, R.A.; Tunstall, T.S.; Briggs, C.J. 2010.** Dynamics of an emerging disease drive large-scale amphibian population extinctions. Proceedings of the National Academy of Sciences of the United States of America. 107: 9689–9694.

**Williamson, C.E.; Saros, J.E.; Schindler, D.W. 2009.** Sentinels of change. Science. 323: 887–888.

**Woodhams, D.C.; Bosch, J.; Briggs, C.J.; Cashins, S.; Davis, L.R.; Lauer, A.; Muths, E.; Puschendorf, R.; Schmidt, B.R.; Sheafor, B.; Voyles, J. 2011.** Mitigating amphibian disease: strategies to maintain wild populations and control chytridiomycosis. Frontiers in Zoology. 8(1): 8.

**Woodhams, D.C.; Vredenburg, V.T.; Simon, M.A.; Billheimer, D.; Shakhtour, B.; Shyr, Y.; Briggs, C.J.; Rollins-Smith, L.A.; Harris, R.N. 2007.** Symbiotic bacteria contribute to innate immune defenses of the threatened mountain yellow-legged frog, *Rana muscosa*. Biological Conservation. 138(3–4): 390–398.

**Young, C.A.; Escobar-Arias, M.I.; Fernandes, M.; Joyce, B.; Kiparsky, M.; Mount, J.F.; Mehta, V.K.; Purkey, D.; Viers, J.H.; Yates, D. 2009.** Modeling the hydrology of climate change in California's Sierra Nevada for subwatershed scale adaptation. Journal of the American Water Resources Association. 45(6): 1409–1423.