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PARK Science

Integrating Research and Resource Management in the National Parks

National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science
Office of Education and Outreach

ESTIMATING CARBON STOCKS AND THE EFFECTS OF MANAGEMENT IN THE WORLD'S TALLEST FOREST

ALSO IN THIS ISSUE

- Planning for sea-level rise
- Animal inventories at El Malpais and Pipe Spring
- Visitor preferences for tamarisk control at Canyonlands
- Potential effects of climate warming on park visitation in Alaska
- Vegetation mapping analysis at Great Basin

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Park Science is a research and resource management journal of the U.S. National Park Service. It reports the implications of recent and ongoing natural and social science and related cultural research for park planning, management, and policy. Seasonal issues are published usually in spring and fall, with a thematic issue that explores a topic in depth published annually in summer or winter. The publication serves a broad audience of national park and protected area managers and scientists and provides public outreach. It is funded by the Associate Director for Natural Resource Stewardship and Science.

Articles are field-oriented accounts of applied research and resource management topics that are presented in nontechnical language. They translate scientific findings into usable knowledge for park planning and the development of sound management practices for natural resources and visitor enjoyment. The editor and board or subject-matter experts review content for clarity, completeness, usefulness, scientific and technical soundness, and relevance to NPS policy.

From the Editor

Another kind of sequestration

To be among forests as grand as those at Redwood National and State Parks is a rare treat for most of us. I've had the pleasure of this experience a couple of times and I enjoy being reminded of it by our cover photo and the related article. Naturally I marvel at the size of these behemoths, but the architecture of the species also strikes me as peculiar and wondrous. What structure supports such tremendous weight, resists breaking, survives most fires, and facilitates growth commonly to heights of 200–300 feet? How do the internal hydraulics overcome what must be nearly nine atmospheres of pressure to transport water and nutrients from the forest floor to the uppermost branches? What ecological niches are made possible by leaves and limbs lofted so high aboveground? Science, of course, has answers for these questions and for many more that not only provide basic information about our natural world but also link that information to resource management actions and affect conservation policy.

One question I never thought to ask is the subject of our cover article: What is the ability of the coast forest to sequester carbon, and how do we go about estimating it? This riddle is both practical and symbolic as resource managers, scientists, and policymakers look for a silver lining on the cloud of climate change. As you know, atmospheric carbon dioxide concentrations and temperatures are rising. One way to slow this process is to prevent carbon from entering the atmosphere by withdrawing it and storing it. Considering the vast amounts of vegetation, organic soils, wetlands, and other carbon-containing resources in parks and protected areas, these places are profoundly involved in carbon sequestration and can be managed to help preserve the carbon stocks. This ecological service is particularly acute at Redwood National and State Parks where the forest ecosystem stores carbon as densely as almost any place on Earth and has the potential to do more as second-growth forests continue to undergo restoration there. This is yet another awe-inspiring aspect of a very unusual forest, and the story of estimating this capability is equally fascinating.

—Jeff Selleck, Editor

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Contents

NPS/REBECCA BEAVERS

NPS PHOTO

NPS PHOTO

DEPARTMENTS

ON THE COVER

Redwood National and State Parks are home to the world's tallest trees, the coast redwood (*Sequoia sempervirens*). Beginning on page 20, researchers describe their efforts to measure and monitor carbon in these complex forests, showing how forest restoration may allow for greater ecosystem carbon storage over the long term.

USGS/PHILIP VAN MANTGEM

FEATURES

CASE STUDIES

RONDA STRAUCH

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UPCOMING ISSUES

Fall 2013
Seasonal issue. December release.
Contributor's deadline: 15 October

Winter 2013–2014
Theme issue: Biodiversity science in the national parks. February 2014 release.
Contributor's deadline: 15 November

Spring 2014
Seasonal issue. May release.
Contributor's deadline: 15 February

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RESEARCH REPORTS

In This Issue

NATIONAL PARKS AND OTHER PROTECTED AREAS DISCUSSED IN THIS ISSUE

Abbreviations

NHP	National Historical Park
NM	National Monument
NP	National Park
N Pres	National Preserve
NRA	National Recreation Area
NS	National Seashore
Pres	Preserve

Index

Features

Planning for the impact of sea-level rise on U.S. national parks

By Maria Caffrey and Rebecca Beavers

Abstract
Rising sea levels present a challenge for National Park System managers over the next century as they incorporate the latest sea-level rise information, including regional parameters when available, into individual park management plans. Rates of sea-level change vary throughout the National Park System, so the National Park Service (NPS) cannot define a single rate applicable to all parks. This complicates park planning and requires interpretation of research and modeling results. In this article we discuss many of the latest developments in sea-level rise research, including the drivers of sea-level change, global sea-level projections for this century, and what these mean for park managers. We also explain why tide gauge data in some regions have recorded decreasing mean sea levels and why potential storm surge should be included in planning.

Key words
eustasy, glacial melting, isostasy, storm surge, thermal expansion

GLOBAL MEAN SEA LEVELS have been rising since the last ice age approximately 20,000 years ago (Archer and Rahmstorf 2010; IPCC 2007). Relative to the past two to three thousand years, the rate of rise has increased significantly and is projected to increase at an accelerating pace throughout the 21st century because of climate change (IPCC 2007). In 2007 the Intergovernmental Panel on Climate Change's (IPCC) fourth assessment report projected that global mean sea levels will rise 18–59 cm (7–23 in) by 2100; however, these projections have been criticized as being conservative, lacking data, and failing to take into account dynamic changes in large, land-based ice sheets (Rahmstorf et al. 2007; Horton et al. 2008; Overpeck and Weiss 2009; Rahmstorf 2010). The aim of this article is to introduce three major sources of sea-level change, describe related complexities and uncertainties in projecting sea-level rise, and discuss how the National Park Service can best manage for climate change in the coastal zone.

Sources of sea-level rise

Changes in sea level can occur as a result of numerous drivers. Steric sea-level change is driven by a change in water density, thermosteric changes are the result of changes in temperature, and halosteric change is caused by changes in salinity. The term "eustasy" is commonly used in the literature to describe sea-level change that is the result of a change in water volume (Sverdrup et al. 2003; Milne et al. 2009). Global mean sea level responds to a number of environmental sources that result in a change in eustasy that the IPCC (2007) has broadly categorized as continental ice cap and sheet melt, ocean thermal expansion, and shifts in terrestrial storage capacity (table 1).

Increasing temperatures (driven by increased atmospheric loading of carbon dioxide [CO_2]) are melting glaciers and ice sheets (IPCC 2007). Melting of the Greenland ice sheet alone raised the global mean sea level by an average 0.21 ± 0.07 mm/yr (0.01 ± 0.002 in/yr) from 1993 to 2003 (IPCC 2007). Archer and Rahmstorf (2010) calculated that if both the entire Greenland and Antarctic ice sheets were to melt, global sea levels would rise by around 65 m (210 ft), although global climate models suggest that the complete loss of continental ice sheets is extremely unlikely. Overpeck et al. (2006) predicted that a rise in relative sea level is more likely to be on the order of several meters. The disparity in these projections highlights the difficulty of modeling the contributions of melting ice to sea-level rise (SLR). In addition to the melting occurring near the poles, melting ice caps and glaciers in other regions such as the Himalayas and the Andes are contributing to increasing sea levels. Additionally the IPCC (2007) expects that the eastern section of the Antarctic ice sheet could increase in size over this century because of projected changes in precipitation and runoff, although it is highly unlikely that this will result in lower global rates of sea-level rise.

Ocean thermal expansion is an increase in volume (and decrease in density) of ocean waters (Wigley and Raper 1987). While the rate of thermal expansion is expected to vary with temperature, there is a degree of thermal inertia, or lag time, between the time when CO_2-driven warming is observed and subsequent thermal expansion occurs. Domingues et al. (2008) found that ocean thermal expansion in the upper 700 m (2,297 ft) of the water column has overtaken the melting of the Antarctic and Greenland ice sheets as the second largest contributor (ahead of the melting of glaciers and ice caps) to rising sea levels over the past 10 years. Rates of thermal expansion are the subject of active research with many questions remaining about rates of thermal inertia. Vermeer

Table 1. Rates of sea-level rise by source

Source of Sea-Level Rise	Rate of Rise in mm (in) per Year 1961–2003[1]	 1993–2003
Thermal expansion	0.42 ± 0.12 (0.02 ± 0.004)	1.6 ± 0.5 (0.06 ± 0.02)
Glaciers and ice caps	0.50 ± 0.18 (0.02 ± 0.01)	0.77 ± 0.22 (0.03 ± 0.01)
Greenland ice sheet	0.05 ± 0.12 (0.002 ± 0.004)	0.21 ± 0.07 (0.01 ± 0.03)
Antarctic ice sheet	0.14 ± 0.41 (0.006 ± 0.02)	0.21 ± 0.35 (0.01 ± 0.01)
Sum of individual climate contributions listed above	1.1 ± 0.5 (0.04 ± 0.02)	2.8 ± 0.7 (0.11 ± 0.03)
Observed total SLR	1.8 ± 0.5 (0.07 ± 0.02)	3.1 ± 0.7 (0.12 ± 0.03)
Difference (observed minus sum of estimated climate contributions)[2]	0.7 ± 0.7 (0.03 ± 0.03)	0.3 ± 1.0 (0.01 ± 0.04)

Note: Sea-level rise data are from the fourth IPCC climate assessment report (IPCC 2007).

[1]Numbers prior to 1993 are from tide gauges; those after 1993 are from satellite altimetry.

[2]Differences between observed and estimated climate contributions represent other contributors to sea-level rise, such as increased runoff from land (discussed above). Differences can also occur because of sampling error.

and Rahmstorf (2009) calculate that thermal expansion will contribute 55–70% of eustatic rise by 2100. Rates of ocean thermal expansion have already caused an increase in sea-level rise from 0.42 ± 0.12 mm/yr (0.02 ± 0.005 in/yr) over the last 40 years (1961–2003), compared with 1.6 ± 0.05 mm/yr (0.06 ± 0.002) at the end of the 20th century (1993–2003; IPCC 2007).

Changes in precipitation are predicted to alter the balance between water storage on land and that in the oceans (Church et al. 2008; Llovel et al. 2011). In addition to greater precipitation over some oceans, some terrestrial regions can expect greater precipitation, resulting in increased runoff. In other regions, drought conditions will lead to a greater human reliance on freshwater aquifers, creating an opportunity for greater saltwater intrusion and local reductions in sea levels, as we explain below. In comparison with the ice sheets, changes in the amount of precipitation over the oceans have not been studied intensively (Koster et al. 2000). However, overall change in terrestrial water storage is not expected to generate anywhere near the same level of eustatic rise in sea levels as that created by the melting of continental ice caps and sheets and thermal expansion (Milly et al. 2003).

A number of scientists have modeled how sea levels could rise in the future (table 2, next page). The fourth IPCC climate assessment report projected sea levels based on climate scenarios ranging up to a maximum warming of 5.2°C (9.4°F) by 2100 (IPCC 2007). This estimate *does not* consider the full range of IPCC temperature scenarios that predict a maximum 6.4°C (11.5°F) temperature increase under their most fossil-fuel-intensive A1FI scenario (Archer and Rahmstorf 2010). Updated SLR predictions in the fifth IPCC assessment report, expected in late 2013, will use the new Representative Concentration Pathways (RCPs; van Vuuren et al. 2011) of four greenhouse gas concentration trajectories. These are also based on radiative forcing values and will eventually replace the Special Report on Emissions Scenarios (SRES) trajectories that were used in the third and fourth IPCC reports (IPCC 2000, 2001, 2007). For example, Jevrejeva et al. (2012) predicted rates of sea-level rise using RCPs that estimated that these rates will be almost double those predicted by the IPCC in their fourth report.

More recently Parris et al. (2012) released their sea-level rise scenarios as part of the U.S. National Climate Assessment. Their projections are global sea-level scenarios that provide greater detail regarding the state of the scientific literature along with scales of confidence. While research like Parris et al. (2012) is very useful in planning, it usually provides specific numbers only for end-of-the-century sea-level rise. The full data set has not been published, making it difficult to graph how sea level could change throughout the century. Park managers often need to cite specific numbers in their planning documents that are based on a 5-year, 20-year, or somewhat longer time horizon. Furthermore, these values do not include projected storm surge values on top of SLR data, which could further complicate the management of coastal lands (UKCP09; Burkett 2012). The potential impacts of increased storm surges on coastal parks like Cape Hatteras and Cape Lookout National Seashores could potentially engulf the entire park units (fig. 1, page 10; Sallenger et al. 2012).

Historical sea-level change data

Predicting how sea level will affect coastal park units is complex. In addition to considering eustatic sea-level rise, researchers must take into account changes in *isostasy* (the raising or lowering of land levels when a mass, such as a glacier or ice sheet, is lost or added). In Alaska where land-based ice is melting rapidly, tide gauge data suggest that mean relative sea levels in six southeastern coastal parks (Sitka and Klondike Gold Rush National Historical Parks; Glacier Bay, Lake Clark, and Katmai National Parks and Preserves; and Kenai Fjords National Park) have decreased. In

Table 2. Projected magnitudes of global mean sea-level rise by 2100

Published Source	Range m	Range ft	Methods
Intergovernmental Panel on Climate Change (IPCC 2007)	0.18–0.59	0.59–1.94	Data published as part of the fourth IPCC climate assessment report "Climate change 2007: The physical science basis." The IPCC is the most commonly cited source of SLR projections, which are based on six scenario families defined in IPCC 2000. Projections are modeled based on a maximum global mean temperature increase of 5.2°C (9.4°F) by 2100.
Rahmstorf (2007)	0.50–1.40	1.64–4.59	One of the first sources to apply semiempirical modeling to project future sea-level rise. This model connects rates of warming to sea-level rise in which rates of rise are expected to be proportional to global mean surface temperature.
Horton et al. (2008)	0.54–0.89	1.77–2.92	Calculated using a semiempirical global model based using IPCC scenarios (scenarios A1B, A2, and B1).
Pfeffer et al. (2008)	0.80–2.00	2.62–6.56	Projections based on kinematic scenarios of increased ice dynamics (based on projected differences in the breakup and melt rate of ice).
Grinsted et al. (2010)	0.90–1.30	2.95–4.27	Analyzed the last 2,000 years of global temperatures and sea levels in order to model future sea levels based on IPCC scenario A1B.
Vermeer and Rahmstorf (2009)	0.75–1.90	2.46–6.23	Based on observed data from 1880 to 2000. Modeled global sea-level rise using all IPCC scenarios.
United Kingdom Climate Projections 2009 (UKCP09)	0.23–1.90	0.75–6.23	United Kingdom report using calculations based on IPCC scenarios A1B, A1FI, and B1. Also includes their own H++ scenario modeled after a 2.4 m (7.9 ft) rate of sea-level rise per century during the last interglacial period. The H++ is considered unlikely in most regions. Model projections predict rates of sea-level rise around the UK only. Model outcomes are not weighted based on observations.
Jevrejeva et al. (2012)	0.57–1.10	1.87–3.61	Modeled global sea-level rise using four new representative concentration pathways of radiative forcing scenarios. *Note:* Range values reported here represent median confidence limits. Upper and lower confidence limits project a maximum rise of 1.65 m (5.41 ft) and lowest rise of 0.36 m (1.18 ft) based on 95% and 5% confidence limits, respectively.
Meehl et al. (2012)	0.25–1.45	0.81–4.76	Numbers were calculated based on RCP2.6, RCP4.5, and RCP8.5 (van Vuuren et al. 2011). Lower-range numbers are based on RCP2.6 SLR anomalies (taking into account contributions from continental ice coupled with thermal expansion). The higher-range number was calculated using a semiempirical method for RCP8.5.

Notes: The summary is of projections published since the IPCC fourth climate assessment report (IPCC 2007).

The range of sea-level rise estimates shown here is calculated relative to the mean tide levels for the period 1990–2000.

Sitka, tide gauge data reveal that sea level has decreased by 2.11 ± 0.29 mm/yr (0.08 ± 0.01 in/yr) from 1924 to 2011 (Zervas 2009) because rates of eustatic rise did not exceed the rate of isostatic rebound of the land. This is a sharp contrast to parks in the continental United States, such as those in southern Louisiana, where compaction of Mississippi River delta sediments leads to a high rate of relative sea-level rise, for example at Jean Lafitte National Historical Park and Preserve. The tide gauge nearest the park indicates sea level has risen by 9.07 ± 0.49 mm/yr (0.36 ± 0.02 in/yr) from 1947 to 2012 (Zervas 2009).

Local tide gauge trend data can be used in conjunction with sea-level rise models to determine how sea level has changed in the past, although rates of rise over the last century have varied spatially and temporally. The U.S. Army Corps of Engineers has released a "Sea Level Change Calculator" (USACE 2013) that uses tide gauge data as part of their sea-level change calculations, which predict the amount of sea-level change going forward. Unfortunately, many national parks do not contain a tide gauge, which can be a hindrance in using the USACE calculations. Including Alaska where sea-level change data are limited as well as complicated by long-term uplift, 92% of coastal U.S. national parks have experienced an increase in sea level over the past century based on National Oceanic and Atmospheric Administration (NOAA) tide gauge data recorded either in or near coastal national parks. However, mean water levels are not rising uniformly; for example, variations in tide gauge distribution, water temperatures, salinity, and ice masses have been discussed as potential drivers of recently identified increases in sea level over a 60-year period at an area known as the "Northeast hotspot" (along the Atlantic coast from North Carolina to Massachusetts). Sallenger et al. (2012) found that rates of sea-level rise in this area are three to four times greater than global SLR rates from 1950 to 2009. However, others have hypothesized that this apparent difference could be an artifact associated with a lack of tide gauges both spatially and temporally (Chambers et al. 2012). The Northeast hotspot is a good example of how dynamic the science concerning sea-level rise can be.

Choosing among these sources of data can be a dilemma for park managers as they plan for climate change. Tide gauges are needed to determine historical rates of

Including Alaska . . . 92% of coastal U.S. national parks have experienced an increase in sea level over the past century based on NOAA tide gauge data recorded either in or near coastal national parks.

sea-level rise, but most are located outside the parks and may not contain a very long record (a minimum 30-year record is necessary to determine historical relative sea level). The NOAA National Water Level Observation Network (NWLON) manages a network of 175 long-term gauges. The data from these stations are used to analyze rates of relative sea-level rise. However, extensive spatial "gaps" in the tide gauge network make it difficult to calculate regional SLR trends for all coastal parks. Historical data (either paleoenvironmental proxy data or tide gauge data exclusively) should not be used exclusively to predict local to regional changes in sea level; however, the lack of these data sets makes it difficult to create and test any models that are necessary for predicting future sea-level changes. Filling in these gaps in the tide gauge network is the first step toward a more comprehensive monitoring network that could be essential to identifying whether regions such as the proposed Northeast hotspot are experiencing greater-than-average rates of sea-level rise (Chambers et al. 2012).

The NPS Oceans and Coastal Resources Branch and the NPS Climate Change Response Program are working with NOAA to help close these information gaps. An optimal solution would be to expand the network by installing permanent tide stations in all of the gap areas, but this is currently cost-prohibitive. As an alternative, a pilot project was implemented at Assateague Island National Seashore (Maryland) where a temporary tide station was installed in a gap area. This station will collect water-level data for one year from which a local tidal datum will be established. The equipment can then be removed and installed in another park with a gap. The newly established local tidal datum will be correlated with the closest NWLON stations, enabling the park to take advantage of NWLON's long-term water level and trend data.

In most cases we do not recommend that park managers use tide gauges outside national parks to extrapolate potential rates of sea-level rise. In addition to not taking into account future changes in the rate of sea-level rise, the accuracy of these results will vary depending on how close the gauges are to the park, basin shape and size, and length of the record. We need to add more tide gauges in coastal national parks to measure these trends and help protect our coastal natural and cultural resources over the long term.

Storm surge

In addition to evaluating various drivers of relative sea-level change, park planners and managers need to consider projected storm surge values added to sea-level rise magnitudes, which could further complicate the management of coastal lands (UKCP09). Storm surges occurring at coastal parks like Cape Hatteras and Cape Lookout National Seashores (North Carolina) will continue to change the land- and seascapes of these areas, with the potential to completely submerge them (fig. 1, next page). The likelihood of increased storm intensity added to increasing rates of sea-level rise makes predicting the reach of future storm surges especially difficult.

More than 100 national parks are vulnerable to the combined effects of sea-level rise and storm surge, and Strauss et al. (2012) calculate that in the United States approximately 3.7 million people live within the zone of a projected 1 m (3.3 ft) sea-level rise. However, based on the projected amounts of sea-level rise by 2100 (table 2), this may be a conservative estimate. Such estimates do not take into account how storm surge on top of increased relative sea levels will spread into areas previously untouched by storms. Tebaldi et al. (2012) estimate that by 2050, some locations in the United States will experience century-scale storm surges annually. In many locations today we have accepted that century-scale stormwater levels can now be expected decadally (Tebaldi et al. 2012).

Thus, the state of the science for storm surge prediction is even more uncertain than it is for sea-level rise. Accordingly, in summer 2013 we began a new study that aims to provide sea-level rise and storm surge data for 105 coastal parks. We are using local tide gauge data in conjunction with NOAA sea, lake, and overland surge from hurricanes (SLOSH) data to predict how the parks could be affected by climate change–related factors over this century. The results of this research will be discussed in context with other project data (e.g., the coastal vulnerability index described in Pendleton et al. 2010) to inform NPS planning in foundation documents and general management plans. The results of this research are expected to be published in the academic literature along with a full park-by-park report by 2016.

Disaster response and adaptation planning

In 2012, Hurricane Sandy catapulted the subject of sea-level rise and storm surge into national debate when it struck the Atlantic coast in October. The National Park Service expected flooding at about 40 coastal parks. Sandy was at hurricane strength when it made landfall near Atlantic City, New Jersey, south of Fire Island National Seashore (New York) and Gateway National Recreation Area (New York and New Jersey). Flooding was greatest in the New York City area. Docks at Statue of Liberty National Monument were destroyed and historical structures suffered severe water damage from the powerful storm surge (fig. 2).

In response, the National Park Service established a rapid review team to identify sustainability and natural and cultural resource priorities for recovery and reconstruction projects. This team is still active and meets regularly to ensure that the proposed projects are carried out effectively. For example, work at the Sandy Hook Unit of Gateway National Recreation Area in New Jersey and at the Statue of Liberty will elevate critical building systems such as boilers and electrical panels above expected future stormwater levels. Other infrastructure will be designed to be submersible or to withstand storm surges. Related work is under way to record the

Figure 1. The map depicts areas potentially inundated by sea-level rise and increases in storm surge severity in the vicinity of Cape Hatteras and Cape Lookout National Seashores, North Carolina. The dark brown areas will be submerged by a 1 m (3 ft) rise in sea level. Red-to-yellow areas would be affected by additional flooding caused by storm surges from 1 to 4 m (3 to 13 ft).

SOURCES: PROJECTIONS ARE BASED ON WEISS ET AL. (2011); BASE MAPS PROVIDED BY ESRI

Figure 2. Damaged by Hurricane Sandy in November 2012, this dock at Liberty Island in New York Harbor is a stark reminder of the combined effects of sea-level rise and storm surge. As sea levels increase, severity of potential storm surge also increases. Storm surge is often overlooked by managers planning for the impacts of sea-level rise.

NPS/REBECCA BEAVERS

elevation of all assets affected by Sandy. Federal regulations require that post-storm reconstruction meet a 1–2 ft (0.3–0.6 m) safety factor above projected sea-level rise and storm surge levels (Hurricane Sandy Rebuilding Task Force 2013). The elevation inventory is necessary to ensure proper planning and compliance. This experience is helping the National Park Service to develop adaptation strategies and decision frameworks that will assist other coastal national parks that must respond to and recover from a future storm.

Caffrey and Beavers (2008) described the coastal adaptation strategies of *retreat* (e.g., Cape Hatteras lighthouse, Cape Hatteras National Seashore) and *fortify in place* (e.g., Fort Massachusetts, Gulf Islands National Seashore, Mississippi) for major coastal historical infrastructure such as lighthouses and forts. One challenge related to climate change that has yet to be fully articulated and addressed is the imminent loss of some of our cultural heritage to sea-level rise or storm surge and the resulting coastal erosion. For example, vulnerable archeological sites on Jamestown Island in Colonial National Historical Park (Virginia) are at risk from a rising water table because of sea-level rise. Sea-level rise causes the water table to rise when overlying freshwater is forced upward by more dense salt water that intrudes into coastal aquifers. Once these sites are saturated, traditional archeological excavation and documentation techniques cannot be used for a variety of reasons (e.g., excavation pits become flooded and artifacts can become damaged or destroyed by the water). Likewise, archeological sites along the Chukchi Sea in northwestern Alaska at Cape Krusenstern National Monument and Bering Land

Bridge National Preserve are experiencing sea-level rise and increased coastal erosion because of diminished sea ice that once protected these sites during coastal storms (Manley and Lestak 2012). Park managers are prioritizing archeological studies at the most highly vulnerable locations so they can maximize documentation of these sites before they are claimed by the sea.

The National Park Service plays a leading role in developing innovative strategies for coastal parks to adapt to sea-level rise and storm surge, and coastal storms are opportunities to apply highly focused responses. Major storms, floods, tsunamis, and even fires in the coastal zone are often followed by special recovery funding to refurbish infrastructure and mitigate future vulnerabilities. Yet the National Park Service needs to ensure that future recovery and rehabilitation projects also address needs to protect habitat (Stabeneau et al. 2011; Nielsen and Dudley 2013) and cultural resource sites. For example, we need to plan for opportunities to simultaneously relocate structures away from eroding shorelines, facilitating the natural development of future habitat. We should act as soon as possible, because even with this level and scope of adaptation planning we may have only a limited amount of time to protect, move, or adapt vulnerable infrastructure and document irreplaceable cultural resources.

Scenario planning

The rates of sea-level change are dramatically different across the diverse geography of the National Park System. A single rate of sea-level rise cannot be defined for all parks. Local to regional information on sea-level change, in addition to global estimates, is needed in order to develop sea-level rise projections that are relevant for coastal planning and management. To manage parks despite these uncertainties, the National Park Service is using scenario planning to develop and test adaptive strategies under a variety of plausible climate futures (Weeks et al. 2011). Scenario planning is a "living process." Information such as site-specific storm surges and SLR vulnerability assessments are needed and must be updated, for example after major coastal storms, to feed into the process in order for coastal park planning to be effective (Pendleton et al. 2010).

The National Park Service will continue to support use of the best available science for coastal management decisions. We are closely monitoring states' recommendations. In places such as California, Florida, and New York, we are incorporating the local or regional, geographically specific findings into NPS scenario and other planning. By collaborating with various partners, we are optimistic that the National Park Service will be able to adapt to and mitigate many of the impacts of sea-level rise and storm surge over the 21st century.

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Bat research and interpretive programming

Increasing public interest in Pipe Spring National Monument

By John R. Taylor, Andrea Bornemeier, Amber Van Alfen, and Cameron Jack

Key words
acoustical survey, bats, interpretive programs, mist-netting, Sonobat

WATER PLAYS THE STARRING role in the history of Pipe Spring National Monument in northern Arizona. The natural springs that emerge here are one of the few stable water sources in an arid strip of desert sandwiched between Grand Canyon and Zion National Parks. Wildlife, prehistoric people, Paiute Indians, Mormon pioneers, and national park visitors have all used this oasis as a life-sustaining rest area. Here the Sevier fault routes groundwater from an adjacent aquifer to the surface, where three springs emerge from the sandstone.

Mormon pioneers developed the springs around 1880, catching the water in basins or ponds and diverting it for irrigation and for cattle and sheep. They also constructed a fortress-like structure directly over the main spring. Known as Winsor Castle (fig. 1), this historical building is symbolic of the struggle over water rights that ensued and is a central feature in the story of Pipe Spring National Monument.

The ponds continue to provide a constant supply of water for livestock and irrigation for the gardens and fruit trees that reflect the park's rich history. These open water sources also benefit local wildlife. At least 21 species of squirrels, rats, shrews, and mice are present in the area, all of which are food sources for coyotes, bobcats, badgers, and foxes (Bogan and Haymond 2001). Red-tailed hawks and great horned owls also spend time in the trees surrounding the ponds in hopes of gaining an easy meal. Additionally, bats rely on the ponds as a place to hunt insects.

Repairs

Over time the stone masonry of Winsor Castle and the nearby ponds has deteriorated and is in need of repair. Cracks in the mortar and leaks in the clay-bottom pond basins have led to water loss in the surrounding soil, muddied the area, and left less water for garden and orchard maintenance. The ponds need to be drained in order to fix these problems. While the repairs are important for the park, draining the ponds in summer when water is especially critical to wildlife could be devastating. Unfortunately summer is also the best time of year to carry out the rehabilitation work. This conflict prompted park staff to review options for timing of the construction to minimize the associated impacts it would have on wildlife.

Of all the wildlife that depend on the ponds, bats are the only ones that need an open water source with a calm surface. While a small squirrel or fox may be able to obtain water from a small puddle or stream, bats require a water source that will allow them to drink on the wing. Furthermore, the bat diversity at Pipe Spring National Monument is represented by some of the largest and smallest bats in the states of Utah and Arizona. Much like airplanes, big bats require a larger flyway when drinking, while the smaller, more agile bats can often drink from small cattle troughs. Park

Figure 1. Historical Winsor Castle and one of two ponds at Pipe Spring National Monument where the bat surveys and interpretive activities took place.

NPS PHOTO

officials decided that if the ponds required draining for maintenance, these repairs should be made when bats' use of the ponds is at its lowest level for the year.

Need for bat research

Bat surveys at the monument have been going on for more than 30 years, but nearly all of this work has taken place from June through August, the peak time for bat activity (Kim and Johnson 2004; Johnson 2005; and Tyburec 2011). To identify optimal timing for pond maintenance, surveys needed to begin in early spring, when migratory bat species arrive at the monument, and continue until late November, when their activity sharply declines. Our intention was to determine whether some bat species use the ponds year-round or they only rely on the ponds in the hot summer months. We also wanted to know how much seasonal variation in use by different bat species exists.

The surveys began in September 2011 and ended in November 2012. Over this time we made acoustic recordings of bat activity at the ponds for one night every two weeks. This gave us the data we would need to determine species and numbers of bats. This technique involves coupling a full-spectrum ultrasonic bat detector (Petterson D240X, Pettersson Elektronick) to an H2 (Zoom) digital voice recorder. We placed the detector 3 m (9 ft) aboveground by strapping it to a large cottonwood tree and facing it toward the ponds. It was protected from the elements by a PVC housing constructed from a large electrical junction box. The audio signal was routed through a 4-meter-long (12 ft) audio cable to the recorder, which was housed in a weather-resistant toolbox at the base of the tree. This setup allowed both the recorder and the detector to run off of a stable power source and facilitated data downloading and periodic changing of digital storage media (8 GB flash memory) used by the recorder.

The bat detection system was turned on at sunset and run continuously through the night to provide approximately 12 hours of monitoring. Analysis of the data was achieved using Sonobat 3.03 software (USWest 2010). We note that acoustic identification such as this is probabilistic, and not as reliable as identifying bats through morphological or genetic methods. For this reason we conducted mist-netting events each month, as close to the new moon phase as possible. A typical survey consisted of deploying three nets, 6–9 m (19–30 ft) in length, in the following array: one net on the sidewalk passing between the two ponds and two nets around the pond perimeter (fig. 2). We opened the nets at sundown and closed them three hours later. During this time handlers carefully removed bats caught in the nets, took measurements, weighed and identified them to species, examined them for parasites, and released them. Bats were also inspected for signs of white-nose syndrome, a fungal infection that is sweeping across the nation, reducing bat populations. The bat handlers took precautions to guard against spreading disease between bats by using disposable gloves for each capture. Furthermore, all research crew members were vaccinated for rabies and were trained in bat handling.

A focal point for interpretive programs

Netting generated a fair amount of excitement among park staff and visitors, and it quickly became apparent these periodic events could serve as a foundation for new interpretive programs to increase public awareness of and appreciation for bats. Thus the park interpretive team quickly went about creating posters and sending out e-mails advertising these public events.

In the survey's first year a group of college students who were visiting as part of the Partners in Parks program attended one of the mist-netting events. A number of Boy Scouts participated in the evening activities to fulfill requirements for the mammal study merit badge. We also began to bring in droves of introductory biology students from Southern Utah University (SUU) who, despite varying career interests, might benefit from engaging in the scientific research. Others served as interns under the SUU–National Park Service Intergovernmental Internship Cooperative. We soon expanded these public netting events to include other topics, such as "Bats and the Night Sky," which paired an evening of astronomy with bat natural history. "Bats and Bugs" soon followed, allowing the public to view insects caught at the same time as bats. Finally, "Bats and Salamanders" allowed participants to net salamanders from the ponds and learn about their life histories.

In all, approximately 600 participants enjoyed an evening under the stars learning about bats and a variety of other topics. We drew participants from nearly 100 miles (161 km) away, not to mention travelers who just happened to be at the monument and decided to wait for the evening programs. These engaging experiences help visitors connect to the monument in a very personal way (fig. 3).

Results of bat research

One of the most astounding features of Pipe Spring National Monument is its bat species diversity (table 1, page 18; fig. 4, facing page; and fig. 5, page 19). In summer we often captured more than 20 bats per night, representing eight to nine species. Though we caught fewer bats in spring and fall, we often documented different species with each capture. For example, one evening we netted only four individual bats, yet remarkably all four were different species. As expected, summer months were the most species-rich; diversity plummeted by mid-November and remained low until the following May (table 1, fig. 5).

Netting events also resulted in capturing one of the largest and the smallest bat species known in the area. The smallest bat species in the United States, the western pipistrelle, has a light yellow or grayish coat with dark black wings, ears, and rostrum (tail area) (Whitaker 1998). Forearm measurements typically range from 27 to 33 millimeters (1.1–1.3 in); by comparison, forearm measurements of the much larger pallid bat are typically twice (50–60 mm [2.0–2.4 in]) that length. This type of observation created an outstanding backdrop for the interpretive programs and facilitated discussions about how each species interacts differently with its environment. Accordingly, program attendees experienced the great variety of bats firsthand and seemed to develop an appreciation for the area's biodiversity.

Though 18 of the 28 bat species known in Arizona were identified at Pipe Spring, some were more prevalent than others. Our acoustic and netting data suggest that far more western pipistrelles and Mexican free-tailed bats use the ponds than do any other species (acoustically 22% and 24%, respectively; see fig. 4). Pallid bats, fringe-tailed myotises, California myotises, and big brown bats are the next most plentiful species. The remainder are in relatively small abundance.

We also learned that Pipe Spring National Monument may serve as a migratory stopover for spotted and western mastiff bats. These species were captured or detected only in late spring; they then disappeared and reappeared in August or September. This pattern also fit for Allen's big-eared bats; however, earlier mark-and-capture research of this species associated it with a nearby day roost that could also possibly have served as a maternity roost. Acoustic and mist-netting data also suggest that a number of species were transitory in their

Figure 2. Researchers erected a mist net adjacent to one of the ponds for the evening bat surveys. The need for pond repair stimulated the surveys, with the goal of determining the best time of year for facilities management work.

JOHN R. TAYLOR

Figure 3. Park visitors inspect a bat netted during one of the popular interpretive programs that coincided with the bat surveys at Pipe Spring.

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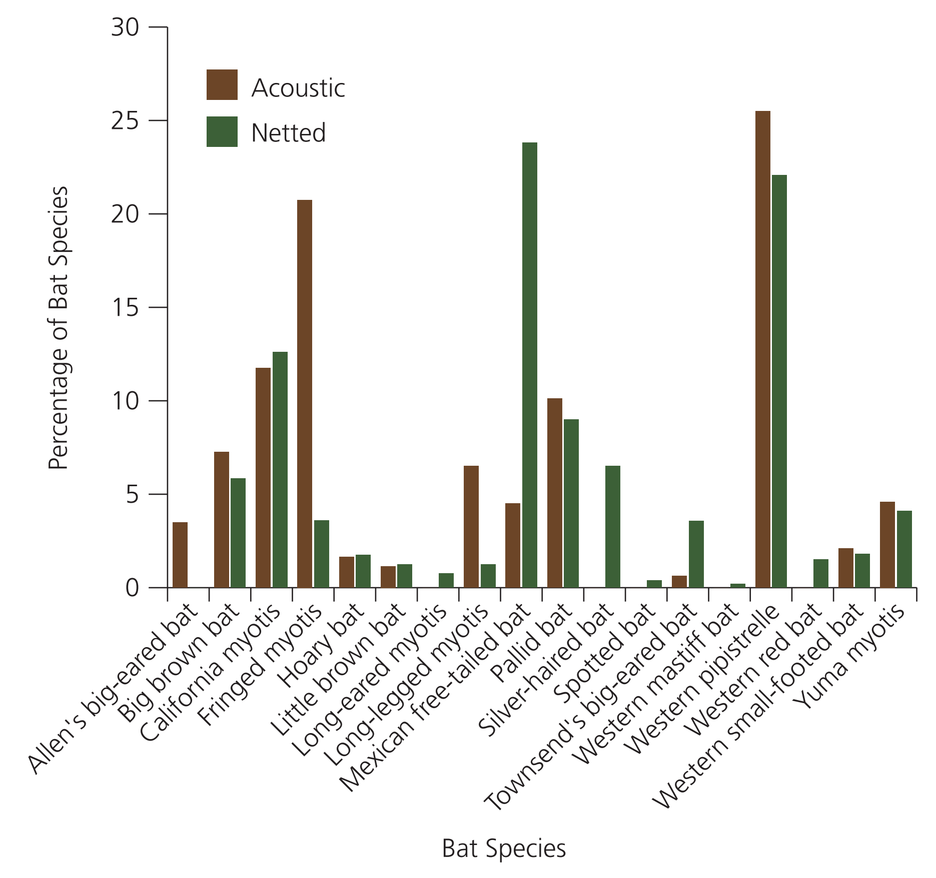

Figure 4. Percentage of bat species netted and detected acoustically during 2011–2012 surveys at Pipe Spring.

use of the ponds, as they were present one month, gone the next, and then reappeared the following month (see table 1 for long-eared myotis and Townsend's big-eared bat). This come-and-go pattern may suggest that Pipe Spring is one of several areas used by these bat species in their overall foraging habitat on the Arizona strip. In contrast, pallid bats, California myotises, fringe-tailed myotises, Yuma

Table 1. Bat species detected acoustically (✓) and number of individuals net-captured by month, 2011–2012, Pipe Spring National Monument, Arizona

Species	Mar		Apr		May		Jun		Jul		Aug		Sep		Oct		Nov		Total Netted
Allen's big-eared bat (*Idionycteris phyllotis*)		0		0	✓	1	✓	1	✓	4	✓	1		0		0		0	7
Big brown bat (*Eptesicus fuscus*)		0	✓	0	✓	5	✓	7	✓	0	✓	1	✓	0	✓	1		0	14
California myotis (*Myotis californicus*)	✓	0	✓	0	✓	6	✓	4	✓	3	✓	3	✓	3	✓	7		0	26
Fringed myotis (*Myotis thysanodes*)		0	✓	1	✓	13	✓	8	✓	4	✓	5	✓	7	✓	3		0	41
Hoary bat (*Lasiurus cinereus*)		0		0	✓	1	✓	1		1	✓	0	✓	0	✓	0	✓	0	3
Little brown bat (*Myotis lucifugus*)		0		0	✓	0	✓	0	✓	0	✓	1	✓	1	✓	0		0	2
Long-eared myotis (*Myotis evotis*)	✓	0		0	✓	0	✓	0	✓	0		0	✓	0		0		0	0
Long-legged myotis (*Myotis volans*)		0		0	✓	7	✓	1	✓	2	✓	1	✓	0	✓	2		0	13
Mexican free-tailed bat (*Tadarida brasiliensis*)		3	✓	1	✓	0	✓	1	✓	0	✓	0	✓	2	✓	3		0	10
Pallid bat (*Antrozous pallidus*)		0	✓	1	✓	4	✓	6	✓	6	✓	2	✓	0	✓	1		0	20
Silver-haired bat (*Lasionycteris noctivagans*)		0		0	✓	0	✓	0	✓	0	✓	0	✓	0	✓	0		0	0
Spotted bat (*Euderma maculatum*)		0		0	✓	0		0		0		0	✓	0	✓	0		0	0
Townsend's big-eared bat (*Corynorhinus townsendii*)		0		0	✓	0	✓	1		0	✓	0	✓	0	✓	0		0	1
Western mastiff bat (*Eumops perotis*)		0		0	✓	0		0		0	✓	0	✓	0	✓	0		0	0
Western pipistrelle (*Pipistrellus hesperus*)	✓	1	✓	1	✓	8	✓	9	✓	19	✓	9	✓	3	✓	1	✓	0	51
Western red bat (*Lasiurus blossevillii*)		0		0	✓	0	✓	0	✓	0	✓	0	✓	0	✓	0		0	0
Western small-footed bat (*Myotis ciliolabrum*)		0		0	✓	0	✓	3	✓	1	✓	0	✓	0	✓	1		0	5
Yuma myotis (*Myotis yumanensis*)	✓	0	✓	0	✓	2	✓	0	✓	1	✓	3	✓	1	✓	1	✓	1	9
Total bats net-captured	4		4		47		42		41		26		17		20		1		202
Total bat species captured	2		4		9		11		9		9		6		9		1		18
Percentage diversity[1]	11		22		50		61		50		50		33		50		6		100

[1]Figured as bat species captured divided by 18 acoustically detected species.

myotises, and western pipistrelles appear to stay in the area year-round.

One particularly frustrating facet of this study was the acoustic detection of species we were never able to capture (see fig. 5). Acoustic records indicate that spotted bats, western mastiff bats, silver-haired bats, long-eared myotises, and western red bats are all present in the area. Some of these bats share commonalities in that they are high-altitude, fast-flying species that typically forage well aboveground.

However, both silver-haired bats and long-eared myotises do not fit this description, and we had high expectations of capturing these species at this location. Western red bats, on the other hand, have never been captured in this or the surrounding area. The inability of researchers to net this species has led us to think that acoustic reference files labeled in the Sonobat software as belonging to western red bat could actually be other species, such as western pipistrelle, with similar acoustic attributes.

Outcomes

What began as a straightforward research project blossomed into the synergistic development of engaging interpretive programs. The research provided the National Park Service with the timing information it needed to maintain the ponds and nearby facilities that bats and other wildlife are known to use. The inventory details which bat species use the monument and how this use changes throughout the year. Finally, the National Park Service plans to continue this

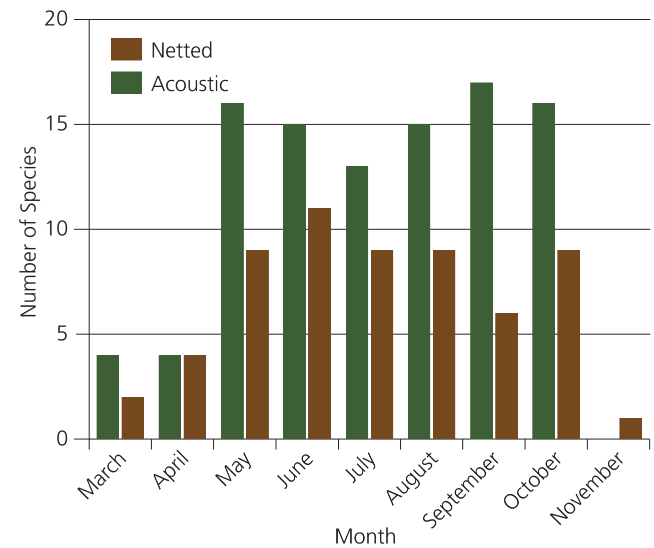

Figure 5. Number of bat species netted and detected acoustically during 2011–2012 surveys at Pipe Spring.

monitoring program, along with the especially meaningful educational opportunities not typically experienced by park visitors.

One college student who attended the bats and salamanders program wrote a paper explaining the "profound effect" the experience had had on him. It helped him decide that a career in science may be exactly what he has been searching for. Another student wrote to say she had never seen a bat before and loved learning about them and listening to their calls. By the end of the night she had found herself referring to them as "cute little guys" and realized that, as she said, "this is exactly why I've come to college . . . to explore new things." The possibilities of expanding interpretive programs in conjunction with park research are being realized in Pipe Spring National Monument and are a bright and refreshing way to engage the childlike curiosity in all park visitors.

So what became of the ponds and their maintenance? We have found that bats' use of the ponds is steady, beginning in May and lasting through October. Bat activity and diversity drop off sharply by mid-November and stay low until spring. Pipe Spring National Monument appears to be an important water and food source for bats and should be treated with care. Maintenance that requires pond drainage should occur in November, when daytime temperatures average 13°C (56°F) and nighttime lows are around freezing, and should be avoided from May through late October, especially during birthing periods for particular wildlife species. In the summer of 2012 the rock walls surrounding the ponds were excavated, reinforced, and reconstructed. However, this work did not require the ponds to be drained. Repair of the pond basins is scheduled for the next couple of years should funding be available.

The excitement generated by this program has spread to other nearby parks, which also see this format as offering great possibilities. This field season we are doing bat inventory work and similar interpretive programs at nearby Zion and Bryce Canyon National Parks and Cedar Breaks National Monument. Unlike Pipe Spring, these parks experience much higher visitation, which necessitates limiting the number of participants. Nevertheless, the goal is the same: to help visitors connect physically and emotionally with the public lands they love. We hope these connections will last a lifetime, influencing participants to continue to protect and care for their national parks.

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PS

Case Studies

Estimating ecosystem carbon stocks at Redwood National and State Parks

By Phillip van Mantgem, Mary Ann Madej, Joseph Seney, and Janelle Deshais

Abstract

Accounting for ecosystem carbon is increasingly important for park managers. In this case study we present our efforts to estimate carbon stocks and the effects of management on carbon stocks for Redwood National and State Parks in northern California. Using currently available information, we estimate that on average these parks' soils contain approximately 89 tons of carbon per acre (200 Mg C per ha), while vegetation contains about 130 tons C per acre (300 Mg C per ha). Restoration activities at the parks (logging-road removal, second-growth forest management) were shown to initially reduce ecosystem carbon, but may provide for enhanced ecosystem carbon storage over the long term. We highlight currently available tools that could be used to estimate ecosystem carbon at other units of the National Park System.

Key words

carbon accounting, climate change, management, mitigation

RAPID CLIMATE CHANGE IS forcing fundamental changes in the stewardship of protected areas. Emissions of greenhouse gases (primarily carbon dioxide, CO_2) into the atmosphere have led to increases in global temperatures of 1.1°F (0.6°C) over the past 50 years (IPCC 2007). Warming trends are expected to exacerbate the effects of other ecosystem stressors, such as air pollution, exotic species (including introduced diseases), and disruptions of historical disturbance regimes. Much greater impacts from climate change are almost certain in coming decades, although predicting the exact conditions for a particular location is beyond our ability.

How should we manage natural areas in the face of these threats? It may be possible to encourage landscapes that can adapt to change (e.g., by altering fire management practices; Nydick and Sydoriak 2011) or are better able to withstand changing conditions (for examples see Millar et al. 2007). At the same time it is becoming increasingly important to prevent natural areas from contributing to greenhouse gas emissions. This represents an aspect of mitigation that may be new to National Park Service (NPS) managers, and one that could fit into the NPS Climate Change Response Strategy (NPS 2010).

Figure 1 (left). Visitors on a guided hike contemplate tree height and forest structure at Redwood National and State Parks.

NPS PHOTO

Terrestrial ecosystems store vast amounts of carbon, on the order of 2,200 to 2,800 billion tons C (2,000 to 2,500 billion Mg; 1 Mg = 1 megagram = 10^6 g = 1 metric ton) (Houghton 2007). By comparison, the atmosphere is estimated to contain approximately 880 billion tons C (800 billion Mg C). Much of the terrestrial carbon is found in soil and is relatively insensitive to most, but not all, land management practices occurring in national parks (see "Road removal" below). But in some terrestrial ecosystems, particularly forests, a large proportion of ecosystem carbon is stored in vegetation (Bonan 2008). The carbon pool (or "stock") held in live vegetation is vulnerable to sudden release following major disturbances such as drought, insect outbreaks, and fire (Kurz et al. 2008). As live vegetation dies and decomposes, the carbon held in once-living biomass is eventually released back into the environment and contributes to further climatic changes. Protecting forested landscapes in national parks is especially important, as some of these sites may hold extremely large amounts of carbon (e.g., old-growth forests).

Reducing greenhouse gas emissions in national park operations is already a priority (e.g., NPS Climate Friendly Parks Program), but managing ecosystem carbon stocks is a relatively new consideration. In some cases maintaining carbon stocks will be in direct conflict with other management goals, for example removing invasive species such as tamarisk and Russian olive trees, which may contain substantial carbon. Often, the connection between management actions and their ultimate effects on carbon stocks is less clear. For example, prescribed fire may directly release large amounts of CO_2 via combustion and tree mortality. But burning may result in a landscape more resistant to future wildfire, which could otherwise release large amounts of carbon (this carbon accounting may not apply over large scales; see Campbell et al. 2011). Considering management outcomes for carbon stocks is likely to become more common in the future, but tools necessary to do so are still under development.

A first step in understanding management effects on ecosystem carbon stocks is to inventory and monitor these stocks, although they are notoriously difficult to measure. Given limited budgets and staff, how can park managers assess ecosystem carbon stocks and their changes over time? In this case study we present our estimates of ecosystem carbon stocks in soils and vegetation at Redwood National and State Parks ("the parks"), California. We also consider changes to these stocks directly linked to park management (and some of the uncertainties associated with our estimates). We describe the methods we used with the intention that our work might be useful to managers interested in similar assessments.

Redwood National and State Parks: A brief history

Redwood National and State Parks share a joint mission to protect coastal redwood ecosystems along California's northern coast. The parks contain the largest area of unlogged redwood forest, home to the world's tallest tree, the coast redwood (*Sequoia sempervirens*, fig. 1, page 20, and cover). This tree species can reach massive sizes and its wood decomposes very slowly, so old forests containing these trees can contain very high levels of vegetative biomass (and therefore carbon). For example, the highest-ever biomass (per unit area) in any forest was recorded in an old-growth redwood stand approximately 100 miles (177 km) south of the parks (Busing and Fujimori 2005). However, approximately half of the land base of the parks (roughly 79,000 acres or 32,000 ha) is composed of second-growth forests that were heavily logged prior to park ownership. The second-growth areas pose management problems, as tractor logging and associated road building (standard forest practice at the time of harvest) severely damaged watersheds and resulted in forests that are only slowly regenerating.

Although Redwood National and State Parks are relatively small, estimating carbon stocks and changes to these stocks poses three challenges. First, the parks span several geologic, climatic, and soil conditions, with habitats ranging from estuaries, freshwater rivers, and coastal dunes to grasslands, open oak woodlands, and coniferous forests. Second, old-growth redwood forests contain some of the highest concentrations of biological carbon of any terrestrial system, making accurate carbon assessments difficult to obtain. Third, landscape-scale restoration treatments that have been in operation since 1978 to restore damaged watersheds and forests have had major effects on carbon stocks (see "Management Effects," page 24). With this diversity of history and legacy issues and physical and biological systems, Redwood National and State Parks can be thought of as a microcosm for many other areas in the western United States, making the approaches we present here potentially useful to other parks.

Taking stock of carbon stocks (where's the carbon?)

Soils

First-order estimates of soil carbon at most parks can be derived from data in existing soil surveys or ecological inventories (as a starting point see http://nature.nps.gov/geology/soils/SRI.cfm). We used data from the recently completed Soil Survey of Redwood National and State Parks to estimate soil carbon in these parks (USDA-NRCS 2008). Eighty-seven soil map units and 442 soil components are mapped in the parks. To measure soil carbon, scientists need to know organic and inorganic carbon contents, bulk density, percentage of rock fragments, and thickness of horizons for each soil component to a depth of about 5 to 6.5 feet (1.5 to 2 m). All these soil properties are available for each soil component in contemporary soil survey reports.

Average carbon content varied among soil map units and soil components, ranging from 5 tons per acre (11 Mg per ha) in floodplain soils with little vegetation cover to 209 tons per acre (468 Mg per ha) in moist redwood forests with a thick herbaceous understory. Overall, approximately 13 million tons (12 million Mg) of carbon is stored in soils of Redwood National and State Parks, or an average of 95 tons per acre (213 Mg per ha) (fig. 2). A comparison of the soil organic carbon stock values of different vegetation types in the parks shows that soil carbon stocks generally decrease with increasing landscape instability and distance from the ocean (which relates to plant productivity).

Vegetation

We combined cover data from the parks' vegetation map with estimates of carbon content for vegetation types from publicly available online tools. Specifically, we used the U.S. Forest Service (USFS) Carbon On-Line Estimator (COLE) (NCASI 2011), NASA's carbon modeling tool, and estimates for live forest carbon provided by the NASA–Carnegie Ames Stanford Approach (CASA) (Potter et al. 2008). COLE uses USFS Forest Inventory and Analysis (FIA) data and standard allometric equations (describing the relationships between tree size and shape) to estimate carbon by forest type (i.e., species composition and age class). Users can define the scope of the FIA data from local to national, although sample sizes (number of FIA plots used) may be very small for specific locales. We used data at the county level, representing a trade-off between locally derived data and sample size (we used a sample size of approximately 20 FIA plots per major forest type). The CASA model uses remotely sensed vegetation cover data with FIA-derived estimates of carbon content per unit area of vegetation type throughout the continental United States.

Based on COLE, carbon held in vegetation at Redwood National and State Parks was estimated to be 19 million tons C (17 million Mg) (average = 133 tons per acre or 299 Mg C per ha), of which 13 million tons (12 million Mg) was standing wood (live and dead) (12 million tons C [11 million Mg] was live C only) (fig. 3). The per area estimates of forest carbon are somewhat lower than has been reported elsewhere for coastal redwood forests (Gonzalez et al. 2010), likely because of the high representation of relatively young recovering forests at the parks. The CASA model gave a much higher estimate of live forest car-

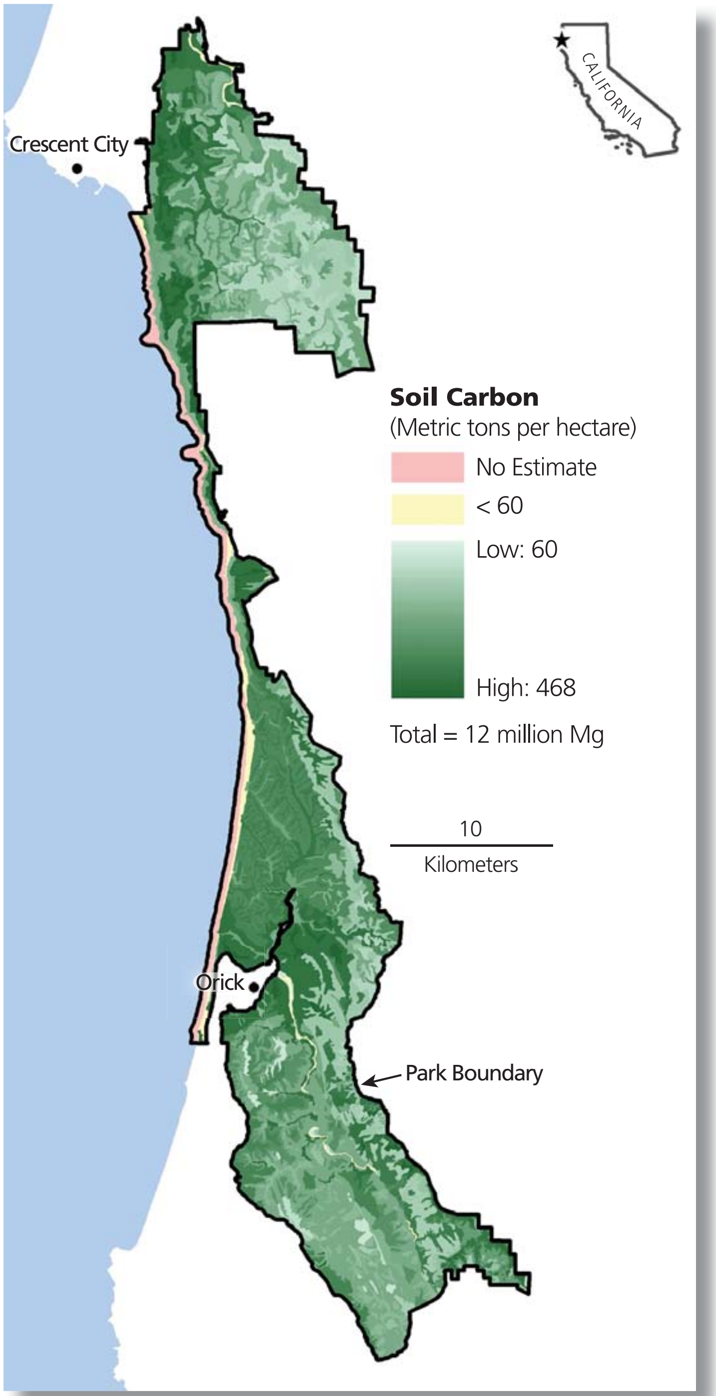

Figure 2. Map of soil carbon stocks at Redwood National and State Parks.

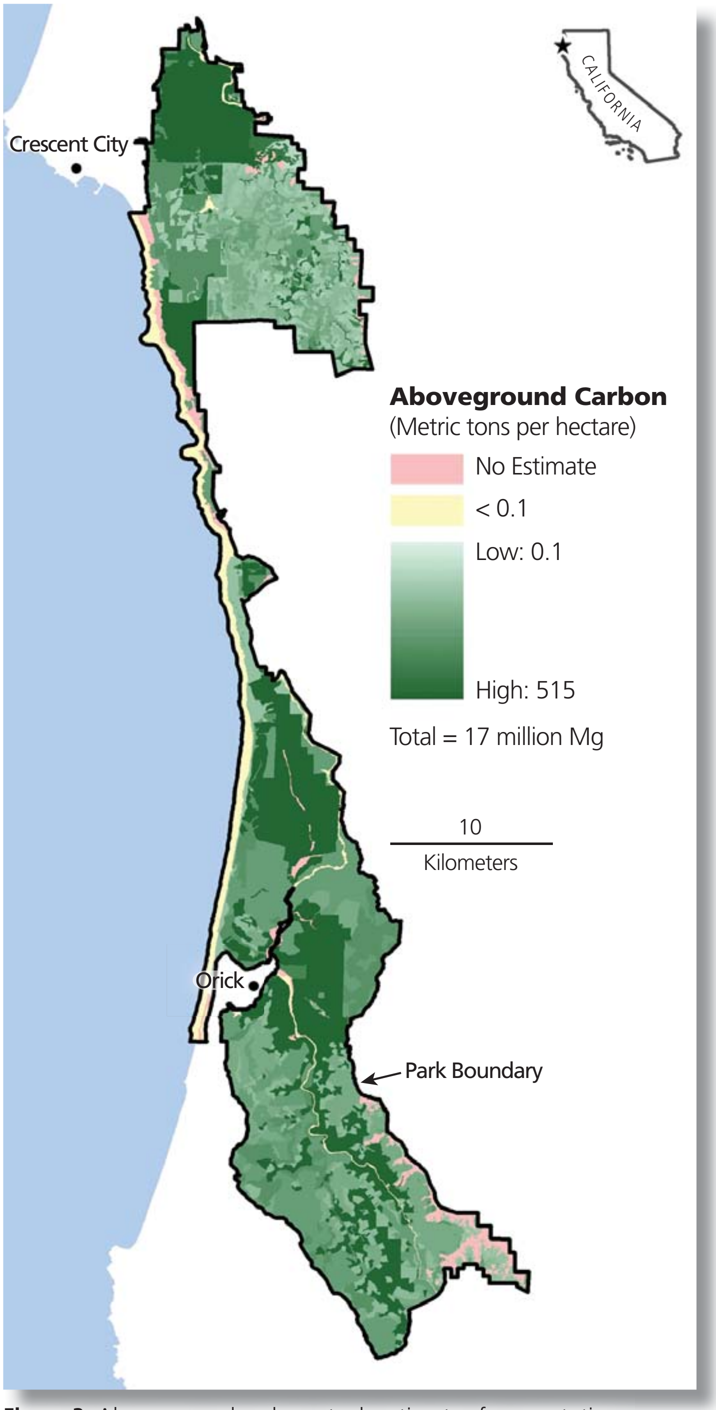

Figure 3. Aboveground carbon stock estimates for vegetation at Redwood National and State Parks. We derived estimates from lookup tables generated in COLE (see text) and applied these estimates to the parks' best available vegetation data.

bon, 65 million tons C (59 million Mg C), over five times the amount of the COLE estimate. The largest discrepancy between these models was for old-growth redwood forests (e.g., maximum live C, COLE = 185 tons per acre [415 Mg C per ha], CASA = 1,229 tons per acre [2,756 Mg per ha]). We suspect that CASA may overestimate carbon in old-growth redwood forests, as it was tuned to a forest stand at Humboldt Redwoods State Park that contains the highest carbon density ever measured (National Park Service, Patrick Gonzalez, climate change scientist, personal communication, 12 May 2013).

Management effects on carbon stocks at Redwood National and State Parks

Restoring degraded landscapes is a primary mission of Redwood National and State Parks. Precisely because these management activities are designed to influence the parks' ecosystems at large scales, they also have the potential to meaningfully influence ecosystem carbon stocks. Important programs in this context at national parks are fire management and mechanical fuel treatments; at Redwood National and State Parks two other programs have larger influences on carbon storage, road removal and forest thinning. While the immediate effect of these activities is the release of carbon from the removal of vegetation, we were interested in the long-term effects of these programs.

Road removal

Since 1978, Redwood National Park has been decommissioning or removing legacy logging roads, which contribute high sediment loads to salmon-bearing rivers. Such work commonly results in ecological benefits, but it also produces CO_2 through the use of heavy equipment and vegetation removal. We examined 135 park project reports and contracts covering the period 1979 to 2009 to determine volumes of road fill excavated from stream channels, volumes of material reshaped and transported on road prisms, and hours of heavy equipment work (Madej et al. 2013).

We contacted heavy equipment vendors (for bulldozers, dump trucks, etc.) to estimate fuel consumption rates. We used park reports to calculate work hours. Forests cut along the road corridor contributed to carbon emissions through decomposition. Timber harvest records and historical aerial photographs provided the ages of second-growth forests adjacent to the decommissioned road reaches. We estimated the carbon content of various stand ages for these second-growth redwood forests using COLE, based on county-level FIA records. Carbon savings from reforestation (carbon content of vegetation regrowth) were based on COLE estimates for California red alder forests, a typical early successional forest type in the parks.

Using this method, we estimated a total carbon cost for treating 264 miles (425 km) of road to be 25,000 tons C (23,000 Mg C), with increasing emissions from vegetation removal in later years as forests matured (fig. 4). Total savings as of 2009 were 75,000 tons C (68,000 Mg C). Savings ultimately may be greater; we currently cannot account for potential soil carbon savings from landslide risk reduction. The ratio of cost to savings will vary by ecosystem type and road-removal methodology, but the carbon-budget methodology outlined here should be transferable to other systems.

Second-growth forest thinning

The typical vegetation in second-growth forests at Redwood National and State Parks is dense, even-aged Douglas fir (*Pseudotsuga menziesii*) stands with simple canopy structure and little understory development. The parks' vegetation management staff is applying thinning treatments to accelerate the development of these forests to mature, old-growth conditions (where forests contain trees from a range of sizes and ages, dominated by coast sequoia). While thinning will likely help achieve second-growth restoration goals in terms of forest structure (size, arrangement, and tree species composition), the consequences for forest carbon are not clear. Forest thinning, by definition, will remove carbon from the system. However, the enhanced growth of remaining trees may offset these losses. Additional carbon offsets are possible because the small trees that are removed are typically used as biofuels, replacing fossil fuels for electricity generation. Long-term storage of larger harvested wood is possible with some durable forest products (e.g., building materials, furniture). Is carbon sequestration compatible with these management actions?

It is becoming increasingly important to prevent natural areas from contributing to greenhouse gas emissions.

We used forest inventory data from the parks and a standard forest development model (FVS, http://www.fs.fed.us/fmsc/fvs/) to project the outcome of current thinning prescriptions: no action, low-intensity thinning (25% basal area removal), and moderate-intensity thinning (40% basal area removal). In all prescriptions coast redwood is not removed. These projections suggest that over the long term, increased tree growth in treated stands may allow thinned and unthinned stands eventually to contain similar forest carbon

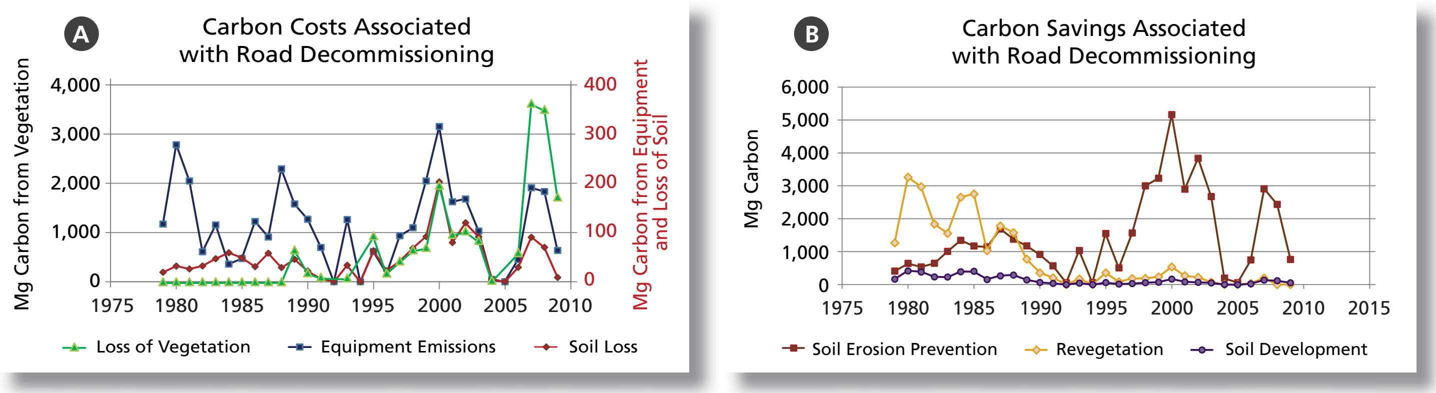

Figure 4. Carbon costs (A) and savings (B) associated with road decommissioning in Redwood National and State Parks as of 2009 (from Madej et al. 2013).

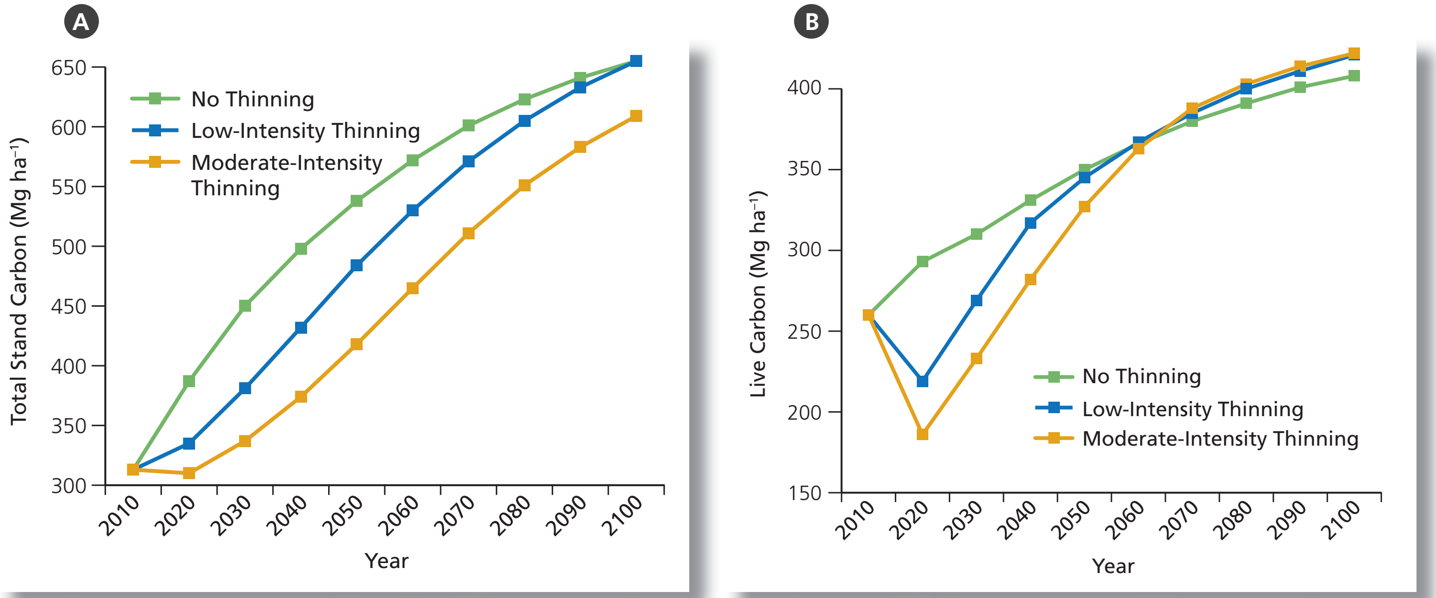

Figure 5. Average projected total (A) and live (B) forest carbon for baseline (no thinning), 25% (low intensity) and 40% (moderate thinning) basal area removal treatments in Redwood National and State Parks second-growth forests.

stocks (fig. 5). Unusual for the National Park System, contractors cover the cost of the project by selling the harvested materials as biofuels or as durable wood products.

Conclusions

The consideration of ecosystem carbon stocks is important as national parks seek to reduce greenhouse gas emissions (e.g., Climate Friendly Parks Program). Protecting current ecosystem carbon stocks may be the primary consideration for managers in this context. However, management actions may have substantial intended and unintended effects on carbon stocks.

Accounting for carbon emissions from park operations is relatively simple compared with measuring ecosystem carbon stocks and management effects on these stocks, and obtaining precise estimates requires increasingly substantial amounts of effort and expense. Our first-order estimates required roughly 100 hours of staff time, after the data were assembled and quality checked. However, once these data are in place, multiple tools are available for managers who wish to evaluate parkwide biological carbon stocks.

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An exploration of the human dimensions of riparian tamarisk control in Canyonlands National Park, Utah

By Robyn L. Ceurvorst and E. Clay Allred

NPS PHOTO

Figure 1. Invasive tamarisk vegetation dominates much of the shore of the Green and Colorado Rivers in Canyonlands National Park. Human dimensions research sought to understand visitors' knowledge of tamarisk, support for its removal, norms and preferences for control methods, and the need for more interpretation of invasive species control and ecological restoration.

Abstract

We examined human dimension aspects such as visitor knowledge, acceptability, and social implications of invasive alien species management in Canyonlands National Park river corridors. Tamarisk control methods applied in riparian park visitation areas support restoration of natural resource landscapes and high-quality visitor experiences. River users (n = 330) were questioned about their knowledge of tamarisk and preferences for tamarisk management on the Green and Colorado rivers within the park. We examined overall self-assessed knowledge of tamarisk, norms for different control method application options (e.g., cut-stump, tamarisk beetle, prescribed fire, mechanical), soundscape implications, and desire for increased interpretation regarding tamarisk and related management. Findings revealed (1) a lack of overall knowledge of tamarisk; (2) weak acceptability and agreement among park visitors for removal by cutting, biological defoliation, and burning; (3) variation of acceptability of and agreement with the location of a proposed application method; (4) sensitivity among respondents related to soundscape impacts on wilderness settings; (5) and a strong desire for more interpretation of tamarisk management. Many respondents stated they supported tamarisk removal for reasons that align with ecological health. A discussion of social, management, and future research implications concludes the article.

Key words

interpretation, invasive species, land management practices, restoration, riparian recreation

DURING A PARK EXPERIENCE, what do visitors think about ecological resource management practices used to control invasive species? This is a question we sought to answer related to tamarisk control along river corridors in Canyonlands National Park in Utah.

Tamarisk is a prevalent invasive alien plant genus found commonly on the waterways of the Colorado Plateau in the western United States. To survive dry desert climates, tamarisk grows close to water sources and forms thick groves along riverine corridors such as the Colorado and Green rivers (fig. 1). Some public land management agencies, such as the National Park Service (NPS) and the Bureau of Land Management (BLM), have employed numerous efforts and resources to control invasive plant species and restore areas to a more natural state. Executive Order 13112 mandates federal agencies, where practicable and permitted by law, to take actions including preventing the introduction of invasive species, detecting and responding rapidly to and controlling populations of such species in a cost-effective and environmentally sound manner, and providing for restoration of native species and habitat conditions in ecosystems that have been invaded (Williams 2005). Methods used to control tamarisk have included manual removal (pulling trees and cutting stumps), mechanical (mulching trees), chemical control (foliar herbicide application), biological control (the release of the tamarisk leaf beetle, *Diorhabda elongate*), and prescribed fire (Belote et al. 2010; Harms and Hiebert 2006).

While diverse methods are used to control tamarisk, public natural resource management decisions may need to consider policy and social factors tied to visitor experiences. The National Park Service (NPS) mission, for example, strives to preserve park resources and values for visitor enjoyment (USDOI 2006). Studies have acknowledged that invasive species' presence along river corridors could alter opportunities for shade, shore access, safety elements, access to cultural sites, scenic viewing, and opportunities for viewing wildlife during river-based recreation experiences (Belote et al. 2010). Few studies have addressed the human dimensions of managing invasive species, such as stakeholder knowledge of ecological aspects of public lands, support for or opposition to invasive species control methods, and need for interpretation regarding these areas of public land management (Hultine et al. 2010). More research is also needed regarding human dimensions of invasive species management along river corridors closely tied to communities dependent on recreation and tourism uses of the river resource. This article examines river users' knowledge of tamarisk, desire and reasons for removal, acceptability of control methods, potential for disagreement about acceptable control methods, implications for visitor experience setting and soundscape, and preferences for additional tamarisk management interpretation and education along the Green and Colorado river corridors.

Invasive alien species management

The introduction and spread of invasive alien species is one of the major threats to environments worldwide. These species can alter habitat structure and reduce native species diversity (Belote et al. 2010; Daab and Flint 2010). Riparian ecosystems are vulnerable because they provide many opportunities for new species to become established through natural and human disturbances (Brown and Peet 2003; Tabacchi et al. 2005). Anthropogenic impacts on river ecosystems can include altered flow regimes, historical land use, and introduction of invasive species for purposes such as erosion control. These impacts can alter ecosystems' competitive hierarchies and favor species with different life-history traits (Tickner et al. 2001).

A plant genus on the Colorado Plateau that may have benefited from the alteration of riverine environments is tamarisk, or salt cedar (*Tamarix* spp.). Tamarisk was first introduced in the United States as an ornamental plant in the 1800s. Shortly thereafter, it was introduced on western rivers to provide ecosystem services such as erosion control (Stromberg et al. 2009). Today tamarisk is one of the most successful invasive alien plant species at outcompeting natives in riparian areas. It is the third most prevalent woody riparian plant in the western United States (Friedman et al. 2005); with life-history traits that allow it to endure higher soil salinity, heat, and excessive drought, tamarisk has the ability to outcompete native cottonwoods and willows (Di Tomaso 1998).

Research on invasive alien plant species includes impacts on native ecosystems and efficacy of potential for control methods on public lands and river corridors (see D'Antonio and Meyerson 2002, for example). Studies have identified the need for more research regarding the social implications of invasive alien species management on and around public lands and waterways (Friedman et al. 2005), such as the impacts of invasive alien plant management on public outdoor recreation areas and the visitor experience (D'Antonio and Meyerson 2002). For example, public land management agencies like the National Park Service seek an understanding of recreation-based stakeholders' input coupled with natural resource–based research to inform planning and decision making. Research regarding topics such as park visitors' preference for invasive alien plant management in riparian recreation corridors could provide managers with more information on the level of agreement among visitors about managing prevalent invasive species.

We were unable to make an exhaustive comparison of all possible tamarisk control methods (e.g., chemical) at Canyonlands National Park because of the limited selection of methods available to the National Park Service in this management setting. Therefore, we focused on an assessment and comparison of norms for the mechanical, cut-stump, burning, and tamarisk leaf beetle methods. Although tamarisk beetle release is not permitted by the National Park Service, the beetle control option is presented in this research site.

Conceptual background

Normative research

Human dimensions of natural resource management research includes the study of social norms, which provide descriptive and evaluative information necessary for managers to identify goals and set standards (Vaske and Whittaker 2004). Past recreation research has defined norms as standards that individuals use for evaluating actions, or conditions caused by actions, as good or bad, better or worse (Shelby et al. 1996). Norms are held by individuals personally, and the aggregate of personal norms can be considered social norms. Managers have used normative data reported by various research studies to understand and describe acceptable conditions, standards, or actions for management of public land- or water-based recreation areas (Vaske and Donnelly 2002).

One application of normative research is to compare norms in different settings. This application has been used to compare indicators, such as visitor encounters on high-use and backcountry trails, ecological conditions at wilderness campsites, biophysical conditions vis-à-vis river flows, and boat encounters on whitewater river trips (Whittaker and Shelby 2002). Various studies have helped managers determine standards for indicators like social carrying capacity. The river user norms addressed in this research may be of most importance to the National Park Service because of policy requiring that it protect visitor experiences within its jurisdictional areas. The study area for this research, along river corridors flowing mostly through Canyonlands National Park, included stretches of remote backcountry and proposed wilderness. In these areas, management decisions cannot be based solely on control methods that are most effective for ecological health. Even though the rivers themselves are not proposed wilderness, areas adjacent to them are. Thus, special consideration must be given to protecting social values for recreation experiences, such as tranquillity, solitude, and natural condition, that river users may desire when visiting these areas.

Managing parks and similar protected areas with the objective of preserving natural soundscapes is becoming an important aspect of public land and waterway management (Ambrose and Burson 2004). With various human-caused noises from aircraft, vehicles on roads, infrastructure maintenance, and park visitors, natural soundscapes are increasingly scarce resources (Park et al. 2009). Visitors in places like national parks want to experience natural quiet and not human-caused noise. Past research has revealed that 91% of visitors are drawn to national parks to enjoy natural soundscapes (Ambrose and Burson 2004; Marin et al. 2011). In general, visitors increasingly exposed to unnatural noise may find it an imposition on a naturally quiet, nature-based experience. This study also addresses river user acceptability (e.g., norms) and agreement (e.g., potential for conflict) regarding soundscapes and invasive alien species control in river recreation areas on the Green and Colorado rivers through Canyonlands National Park. Normative research may help managers set standards that are used in management-by-objective or indicator-based planning and management frameworks.

Management frameworks

Management frameworks are stepwise, sometimes iterative planning processes used to solve complex problems on public lands. Within these planning processes, managers can incorporate descriptive and evaluative aspects into management actions that include ecological, social, economic, cultural, and managerial dimensions. Public input from stakeholders about thresholds or standards for various indicator dimensions is frequently a part of the decision-making process. Management frameworks commonly implemented for this purpose have included, for example, Limits of Acceptable Change (LAC) (Stankey 1988), Visitor Impact Management (VIM) (Kuss et al. 1990), Visitor Experience and Resource Protection (VERP) (Manning 2001), and the Recreation Opportunity Spectrum (ROS) (Manning 2011). Little research exists regarding norms for tamarisk management that may be used in indicator-based planning and management frameworks and subsequent management strategy decision making or action. Our research addressed the norms and preferences of river users to aid in public land planning and management. Visitors may value aspects of the park experience such as scenic quality, access to campsites, shade, and soundscape rather than being knowledgeable of ecological phenomena such as invasive species presence and management. The potential for stakeholder disagreement should be considered when implementing management actions that could impact visitor experiences. This study considers the level of agreement among visitors regarding the acceptability of tamarisk control methods in park river corridors.

Research questions and hypotheses

This research addressed river users' knowledge, preferences, and norms regarding tamarisk management and interpretation along the Green and Colorado river corridors in Canyonlands National Park, Utah. We report on the following research questions in this article: (1) What level of overall, self-assessed knowledge about tamarisk removal exists among visitors? (2) Do visitors desire tamarisk removal? (3) How does acceptability vary among tamarisk control methods? (4) Does the acceptability of particular control methods differ depending on the location of application (within campsites or not)? (5) How much do visitors agree with one another regarding these evaluations? (6) How do visitors evaluate soundscape implications of tamarisk removal? (7) Is more interpretation warranted? The research project followed a line of research questions rather than hypotheses, as there are few studies on which to base a comparison. However, exploratory hypotheses (H) could have included the following:

H1: Visitors (e.g., river users) will have different mean acceptability ratings (e.g., norms) for tamarisk control methods.

H2: Visitors will be less accepting of control methods that could cause more impacts or impositions on their park experience.

H3: Acceptability evaluations (e.g., norms) will differ depending on the location of control method application.

H4: Visitors will have less agreement for rating the acceptability of control methods that could cause more impacts or impositions on their recreation experience.

H5: Visitors will be less accepting of the use of chainsaws for invasive species control in wilderness-type settings versus settings closer to a town or higher-use settings.

H6: Visitors will have more agreement regarding the use of chainsaws in proposed wilderness areas along the Green River corridor.

H7: More tamarisk management interpretation will be desired by visitors.

Methods

Study site and data collection
This research focused on river recreation areas along both the Green and Colorado rivers. Both river sections flow through areas of Canyonlands National Park, Bureau of Land Management (BLM) lands, and some private lands. River users participated in private flat-water (i.e., no rapids) boating trips (e.g., nonguided canoeing) starting at Mineral Bottom on the Green River and arriving at Spanish Bottom approximately 84 kilometers (52 mi) downstream. The flat-water river float section generally concludes at Spanish Bottom located shortly after the Green River flows into the Colorado River, immediately before the first white-water rapid of class IV Cataract Canyon. Researchers approached respondents to answer questionnaires after the completion of their river recreation experience while returning by jet boat shuttle approximately 81 kilometers (50 mi) upstream on the Colorado River to the Potash Boat Ramp take-out in Moab, Utah.

The area of study along the Green and Colorado rivers experiences approximately 2,000 annual river users. Lands adjacent to the river corridor in Canyonlands National Park are proposed wilderness areas. This section of the Green River generally receives nonmotorized, private boater use, whereas this section of the Colorado River experiences more frequent commercial jet-boat tours. Researchers gathered 330 completed questionnaires from river recreationists during the river recreation season, which spanned the months of April to October 2011. An unusually high-water year was experienced by researchers during data collection, which made access to the area unavailable for several weeks in May when the National Park Service and other commercial companies strongly recommended that recreationists not float the river for safety reasons. Data collection followed a systematic random sampling scheme, accounting for factors such as different days and times of use (e.g., low and high use) and varying influxes of river-based activities (e.g., commercial rafting season, private canoeing season) throughout the 2011 river season.

Researchers randomly varied the times of day and day of the week for administering surveys to every fifth person on the shuttle to control for selection bias and allow for generalization to the corridors' population of river users with 95% confidence that data were not found by chance (Salant and Dillman 1994). Visitors were not offered an incentive and were asked to complete the survey in confidential circumstances while riding the return jet-boat shuttle immediately after the trip. The on-site survey approach helped to control for memory loss and allowed for continued viewing of areas (matched with provided photographs and locations confirmed by researchers when questioned) where tamarisk control methods had been implemented along the river corridors. Refusal rates were less than 10% (90% participation rate) and a lack of nonresponse bias existed for this study.

Variables
The self-assessment of river users' knowledge of tamarisk used a single item of measurement on a four-point scale of 0, "no knowledge"; 1, "some knowledge"; 2, "advanced knowledge"; and 3, "expert knowledge." To serve as a baseline of information, four photos were placed on the questionnaire depicting each tamarisk control method. Photos included mechanical control methods: (1) large earthmoving machinery digging up woody debris, (2) smoke and flames rising from a burning tamarisk stand with burnt plants within view, (3) cut-stump treatment with laborers present using handsaws and chainsaws, and (4) browned tamarisk defoliated by tamarisk leaf beetle colonization.

Normative research has previously used image-capture technology, such as photographs, which may allow respondents to comprehend conditions more comprehensively than providing solely a written description of the indicator to be evaluated (Ceurvorst and Needham 2012; Manning and Freimund 2004; Manning et al. 1996; Moyle and Croy 2007; Shelby and Harris 1985; Shelby et al. 2003). User norms for the acceptability of tamarisk control methods in campsites and between campsites, for soundscapes, and in different river corridor settings were found through aggregated evaluative responses. Questions regarding the acceptability of control methods were evaluated on a scale of acceptability ranging from −2, "very unacceptable," to +2, "very acceptable," with 0, "neither," as a neutral point.

Respondents answered a closed-ended question concerning whether or not they wanted tamarisk to be removed from the river recreation area. Respondents were prompted with an open-ended question to elaborate on the main reason they did or did not want tamarisk to be removed. Finally, the questionnaire assessed whether or not more interpretation and education about tamarisk were needed by asking respondents for their preferences.

Results

Knowledge of tamarisk, support for its removal, and desire for more interpretation
Table 1 summarizes visitor level of tamarisk knowledge, preferences for its removal, desire for more interpretation of tamarisk, and saw type preference. Few respondents indicated having "advanced knowledge" (17%)

or “expert knowledge” (3%). Overall, most river users (80%) assessed their knowledge at low levels (e.g., some or no knowledge).

Most river users (88%) would like tamarisk to be removed from the river corridors. Many respondents (62%) stated they supported tamarisk removal for biocentric-based reasons (e.g., those that align with ecological health or the benefit of nature). For example, written comments from respondents expressed support for tamarisk removal because the plant is invasive or is not supportive of a healthy, native riverine ecosystem. Some respondents (9%) reasoned in favor of tamarisk removal for recreation-specific reasons (e.g., access to shore for recreation or safety). The remainder of respondents (29%) in favor of tamarisk removal did not articulate reasons for supporting its removal from the corridor. Few respondents (6%) provided reasons for opposing tamarisk removal. When asked to provide reasons why they did not want tamarisk to be removed, respondents provided open-ended sentiments, such as wanting to “leave nature alone,” thinking the tamarisk removal “task was too large,” and believing that “tamarisk was not a problem.”

Since some interpretation and education materials on tamarisk exist at river access ramps, park visitor centers, and local community businesses, river users were asked about their desire for additional education and interpretation regarding tamarisk and tamarisk management in the questionnaire. Most respondents (84%) reported that they would prefer more educational or interpretive information regarding tamarisk (table 1). This finding offers public land managers a nonintrusive and effective way to inform the public about management actions. Offering additional education could assist public land managers in influencing awareness and social acceptability of tamarisk management.

Table 1. Visitor tamarisk knowledge, removal preference, and desire for more interpretation

Knowledge level	None: 23%	Some: 57%	Advanced: 17%	Expert: 3%
Removal preference	Remove: 88%		Do not remove: 12%	
More interpretation	Desired: 84%		Not desired: 16%	
Saw type preference	Chain saw: 62%		Handsaws: 38%	

Table 2. Comparison of visitor acceptability and agreement levels for saw use

	Green River	Colorado River
Acceptability (norms) for chain saw noise[1]	0.33	0.49
Normative agreement for chain saw noise[2]	0.25	0.31

[1]The mean is the sum of the individual values for each respondent divided by the number of cases. Evaluation is on a scale ranging from −2, “very unacceptable,” to +2, “very acceptable,” with 0, “neither,” as a neutral point.

[2]The potential for conflict (PCI_2) is measured on a scale ranging from 0, “minimum potential conflict,” to 1, “maximum potential conflict.”

Table 3. Visitor norms for tamarisk control methods and application location

Tamarisk Control Method and Location	Mean[1]	PCI_2[2]	Standard Deviation	*p*-value[3]
Burn between camps	0.62	0.40	1.21	0.001
Burn in camps	0.41	0.45	1.26	0.001
Cut-stump between camps	0.97	0.23	1.00	0.072
Cut-stump in camps	0.93	0.25	1.05	0.072
Beetle between camps	0.95	0.33	1.19	0.001
Beetle in camps	0.86	0.36	1.24	0.001
Mechanical between camps	0.02	0.49	1.36	0.116
Mechanical in camps	0.05	0.48	1.34	0.116

[1]The mean is the sum of the individual values for each respondent divided by the number of cases. Evaluation is on a scale ranging from −2, “very unacceptable,” to +2, “very acceptable,” with 0, “neither,” as a neutral point.

[2]The potential for conflict (PCI_2) is measured on a scale ranging from 0, “minimum potential conflict,” to 1, “maximum potential conflict.”

[3]Between camp and in and adjacent to camp values are paired-samples *t*-test analyses.

Preferences for saw use and norms for chainsaw noise

Sixty-two percent of respondents indicated that they would prefer the use of chainsaws over handsaws for tamarisk removal (table 1). While the use of a chainsaw would alter the soundscape and potentially infringe upon visitor experience, river users in this sample evaluated the use of chainsaws as acceptable on both the Green and Colorado rivers. On the scale of acceptability for chainsaw noise from −2, “very unacceptable,” to +2, “very acceptable,” the average evaluation of acceptability (e.g., norm) on the Colorado River was 0.49. Chainsaw noise on the Green River was found to be slightly less acceptable, with a norm of 0.33 (table 1).

Norms for noise on the Green and Colorado rivers

Chainsaw noise produced while removing tamarisk on the Colorado River was found to be more acceptable than hand-sawing, and had a score of 0.25 in agreement among users, whereas the Green River had less respondent agreement, with a PCI_2 of 0.31 (table 2). Visitors rated chainsaw noise as less acceptable in the more wilderness setting of the Green River than in the higher-use areas on the Colorado River; however, chainsaw noise for the removal of tamarisk was found acceptable to river users regardless of location applied.

Norms and potential for conflict

Table 3 compares statistically significant differences in norms for application of the four different tamarisk control methods

Table 4. Differences in the potential for conflict over tamarisk control methods

Areas Respective of Camps	Burn Between	Burn In	Cut stump Between	Cut stump In	Beetle Between	Beetle In	Mechanical Between	Mechanical In
	Tamarisk Control Method and Location of Application							
Burn between camps	—	1.04	3.81	3.36	1.45	0.83	1.99	1.79
Burn in camps	1.04	—	5.19	4.73	2.47	1.81	1.06	0.86
Cut-stump between camps	3.81	5.19	—	0.50	2.00	2.60	6.12	5.80
Cut-stump in camps	3.36	4.73	0.50	—	1.57	2.18	5.68	5.36
Beetle between camps	1.45	2.47	2.00	1.57	—	0.56	3.31	3.10
Beetle in camps	0.83	1.81	2.60	2.18	0.56	—	2.65	2.46
Mechanical between camps	1.99	1.06	6.12	5.68	3.31	2.65	—	0.13
Mechanical in camps	1.79	0.86	5.80	5.36	3.10	2.46	0.13	—

Note: Values >1.96 represent the difference between the methods' potential for conflict (PCI_2) values (Vaske et al. 2010). Values are Bonferroni corrected.

applied either between campsites or within or adjacent to campsites accessible from the river study area. Respondents reported that burning, cut-stump, and beetle tamarisk control methods were acceptable in areas both between river-accessible camps and within or adjacent to campsites along the Green and Colorado river corridors. Although a positive mean acceptability level of 0.02 for between camps and 0.05 for within or adjacent to camps was reported for the mechanical removal method, no statistical significance was found.

The potential for conflict (PCI_2, a measure of agreement with a particular control method) over tamarisk control methods implemented within or adjacent to river campsites resulted in PCI_2 values of 0.48 for mechanical removal, 0.45 for burning, 0.36 for salt cedar beetle, and 0.25 for the cut-stump method (table 3). Results indicate that the cut-stump and salt cedar beetle removal methods have the least potential for conflict when implemented within or adjacent to river-based campsites. The potential for disagreement over tamarisk control methods implemented between campsites resulted in PCI_2 values of 0.49 for mechanical removal, 0.40 for burning, 0.33 for salt cedar beetle, and 0.23 for the cut-stump method. These results indicate the cut-stump and salt cedar beetle removal methods have the least potential for causing conflicting social acceptability among visitors when implemented between camps.

Table 3 shows visitors' difference in acceptability ratings (norms) for tamarisk control methods in different settings. Differences in acceptability for burning and tamarisk leaf beetle methods depending on the location of application (e.g., between camps versus in or adjacent to campsites) had a statistically significant result. Both the cut-stump and mechanical removal methods did not result in statistically significant values for differences in application within or adjacent to campsites versus between campsites. In other words, for the mechanical and burning methods, river users did not draw much of a distinction between methods in and between camps (discussed in the previous paragraph); rather they focused on whether the method was acceptable regardless of where it was applied.

For the burning, cut-stump, and beetle control methods, respondents held more agreement for burning, cut-stump, and beetle methods when applied between camps rather than within camps. Respondents had less agreement regarding the acceptability of the mechanical method. In other words, the mechanical method had the largest standard deviation (1.36 for between camps and 1.34 for within or adjacent to camps) out of all the control method options (table 3). The most agreement or smallest standard deviation among respondents was found for the cut-stump and salt cedar beetle removal methods when implemented regardless of the location (e.g., between or within river-based campsites). We observed a general pattern that as the acceptability (e.g., higher mean value) of the control method increased, the potential for disagreement decreased among respondents regardless of whether the method was applied within or between camps.

The differences between potential for conflict for tamarisk control methods between river campsites versus within or adjacent to campsites were found using the PCI_2 difference (d) equation (table 4). In other words this equation compares the PCI_2 values of variables to determine if there is a statistically significant difference between the chosen variables. If the result of this equation is $d > 1.96$, the difference between the compared values is statistically significant at $a = 0.05$. The d values comparing the differences between control methods and location of application are shown in table 3.

Table 4 provides an exploratory approach into a comparison of multiple variables regarding tamarisk management methods and application locations and caution should be exercised regarding the use of this information. For example, the greatest distance in potential for conflict values was found between the cut-stump and the mechanical methods. The opposite is true for potential for conflict distance values between the burning and beetle methods. As a general pattern, more distance existed when the method was applied in campsites versus between campsites. Although this could mean river users may be more sensitive to management disturbances directly affecting their river recreation experiences, a confident conclusion cannot be made based on this study for several reasons. For example, the general pattern of river users being more sensitive or having more varied norms regarding tamarisk control methods implemented within river campsites differs depending on the nature of the control method implemented. Burning tamarisk, for instance, may cause more smoke and pose a safety threat to recreationists using the site. Mechanical removal may cause excessive amounts of noise impeding on the natural soundscape, and large machinery may result in an intolerable imposition on the viewscape of freshly cut stumps.

Discussion and recommendations

These findings have implications for management consideration and further examination. First, visitors who lack knowledge of tamarisk desire more information. River users' interest in receiving additional education should be addressed by public land managers, as outlined in EO 13112 (Williams 2005). In addition to mandating the control of invasive alien species, EO 13112 requires federal land management agencies to educate the public where possible and practical. Examples of this education include interpretive talks by rangers, increased or improved signage, engagement of interested volunteer groups in providing education opportunities, and informative multimedia approaches (e.g., Web site, video, brochures, and river permit packet information) for visitors and other stakeholders. Additional study is warranted as to specific reasoning for and the relationship between level of knowledge and desire for more interpretation. For example, are visitors mostly concerned about enjoying themselves in the outdoors and are they not aware of encroaching invasive species phenomena? Are river users more concerned about loss of beach space for tents, kitchens, or sports; loss of access to riverbanks, eddies, or trails; or loss of larger trees that provide better shade and boat anchoring than ecological decline from a monoculture invasive species? Social desirability could bias respondent concern toward ecological issues, rather than honest and practical reasons for tamarisk removal.

Although visitors had a low level of knowledge, a majority wanted tamarisk removed and many knew it compromised ecological health. Researchers and the survey, however, did not provide respondents with preamble material suggesting tamarisk was an exotic and spreads quickly in riparian areas. An assessment of whether river users knew about the specifics of tamarisk invasion, removal, and site restoration should be conducted. More depth in understanding stakeholder knowledge of tamarisk could be gleaned from a series of questions about knowledge, providing a baseline of information about tamarisk and examining how value orientations relate to knowledge and support for tamarisk removal. Because of the limited and exploratory nature of this study, these research improvements were not addressed. Future research could address these variables, analyze their influences and the potential for disagreement with different control methods, and broaden the scope to a more regional or landscape scale (e.g., areas where tamarisk is prevalent, an entire river corridor, or the entire Colorado Plateau). Combining ecological data with social data could be beneficial for planning and management in these areas.

Second, the vast majority of respondents found burning, use of the tamarisk leaf beetle, and the cut-stump method acceptable; however, acceptability ratings for the mechanical method were not statistically significant. The cut-stump method and use of the tamarisk leaf beetle had the highest acceptability and most agreement among users. Respondents agreed the least in their acceptability ratings for the burning and mechanical methods. Similar trends have been found in potential-for-conflict research where, for example, as degree of acceptability of a proposed action decreases, agreement in ratings also decreases.

Previous findings have also shown less agreement among acceptability levels for more heated issues or in situations where it may be difficult to express a norm or rating of how people feel conditions should be or which management actions should be taken. Managers should consider the implications of visitor confusion about unknown tamarisk management methods or resistance from stakeholder groups when implementing actions evaluated with lower mean acceptability and less agreement or a higher level for potential disagreement among visitors.

Third, results additionally revealed different responses to the location of tamarisk management within the proposed wilderness area and for soundscape considerations. As a general pattern, river users were more sensitive or had more varied norms regarding their acceptability ratings for tamarisk control methods implemented within river campsites. Visitor acceptability differed depending on the nature of the control method implemented—in other words, the more impact the control method imposed on the visitor experience, the less acceptable the method was rated or the less it was agreed upon. For example, visitors held the least agreement and acceptability for the burning and mechanical methods within campsites perhaps because of the costs, access, air quality, scenic, and soundscape impacts a large piece of machinery could impose on or around campsites. Burning tamarisk, for instance, may cause more smoke and pose a safety threat to recreationists using the site. Mechanical tamarisk removal, for instance, may cause excessive amounts of noise impinging on the natural soundscape, and large machinery may result in an intolerable imposition on scenery because of freshly cut stumps. More in-depth inquiries could be made regarding the reason responses are given. Managers should exercise caution if using burning and mechanical removal. Respondents indicated less support, less acceptability, and more disagreement about norms for these removal methods. Respondents additionally held more disagreement regarding the acceptability of using these methods within, as opposed to between, campsites. Future research could further assess reasons for differences in stakeholder responses and differences in situational variables, such as location of implementation or other site attributes, and compare them in other locations that experience various levels and types of use.

River users expressed a preference for use of chainsaws over handsaws to remove tamarisk. As in previous studies (see Manning et al. 2006, for example), however, chainsaw noise was less acceptable to most respondents along the wilderness setting of the Green River than in the areas not managed as wilderness on the Colorado River. Contrary to previous soundscape studies in some national parks, respondents in this study found chainsaw noise acceptable for tamarisk removal regardless of location applied, for example close to or within visitation areas (Manning et al. 2006). As in other more in-depth soundscape research, this study neither offered an audio example of chainsaws nor asked about preferences for decibel levels in these settings—topics for further study.

Alteration of scenic views may be important in considering tamarisk control because of the dominant role tamarisk plays in riparian ecosystems. Removing the prevalent tamarisk invasive species from riparian areas could significantly alter the riverbank scenery. Future studies could further focus on visitor opinions regarding scenic quality related to removal of invasive species. For example, an assessment could include displaying before-and-after images of a restoration management site to gain respondents' scenery preferences or asking for input on unique attributes that comprise high-quality scenic viewing opportunities. In this study photographs of the different control method applications were shown to respondents above the line of questioning about rating the acceptability of each tamarisk control method.

This study addressed each aspect of tamarisk management as stand-alone variables and did not address the relationship or influences of knowledge, preferences, and norms. Further research regarding invasive species management might include statistical approaches (e.g., path analysis, cluster analysis) to analyze differences in or influences among variables such as user demographics, activity groups, stakeholder segments, recreation sites, or other social physiological variables relevant to managing recreation resource areas. Likewise, one could argue that the acceptability ratings are merely a social convention of an emerging norm rather than an established norm with the management of this particular invasive species. This is because respondent acceptability levels for any control method was not particularly high and more knowledge regarding tamarisk was desired.

Fourth, our findings may help managers understand norms for river recreationists but do not address any other stakeholders. Public land managers may want to address other stakeholders, such as different recreation-based user groups, commercial outfitters who use river corridors for economic gain, grazing permit holders, river managers, private landowners in or dependent on river corridors, and adjacent communities dependent on rivers with invasive species. In addition tamarisk is the only plant genus addressed in this study. These findings do not consider norms or acceptability for the other diverse gamut of invasive alien species (e.g., Russian olive or thistle) and related control methods available for implementation in the various national park ecosystems and settings. A more comprehensive examination of these topical areas could broaden managers' understanding of how the public responds to invasive alien species management to reduce the potential for conflict situations such as polarization among the public, creating costly measures in decision-making processes. An assessment of respondents' value orientations, or where respondents' values are on a range or continuum from anthropocentric (e.g., managing river corridors to benefit human use) to biocentric (e.g., managing for the benefit of ecosystems and nature), or using other scales based on stakeholders' basic beliefs, may increase understanding of responses to invasive species management and restoration. Overall, more information could be gathered for broader generalization as well as for reasons why visitors rated each method at varying levels of acceptability, which could help managers prioritize areas targeted for tamarisk removal.

Finally, researchers could extend more attention to issues that complement tamarisk management in river corridors. After managers implement the control or removal of prevalent invasive species such as tamarisk, other invasive species may immediately succeed, outcompete, and invade the area because of optimal growing conditions in the ecosystem (e.g., more sunlight and availability of nutrients in the soil). Future studies should address the effectiveness of follow-up restoration techniques that could increase success of native plant succession and support a natural ecosystem state as dictated by public land management policy. A focus of these future studies could be on other alien species associated with populations of tamarisk, such as Russian knapweed (*Rhaponticum repens*, previously called *Centaurea repens*). Finally, future research should further examine the multitude of social implications and human dimensions tied to invasive species control and restoration, thus broadening the scope to other recreation-based areas and beyond.

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Research Reports

Potential effects of warming climate on visitor use in three Alaskan national parks

By Christine M. Albano, Courtney L. Angelo, Ronda L. Strauch, and Lindsey L. Thurman

Known for its vast system of glaciers, the Alaska Range is home to Mount McKinley—a key attraction for visitors to Denali National Park. Warming climate may affect the timing and duration of the visitor season at national parks in Alaska and also the natural wonders visitors come to see.

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Abstract

Alaska's national parks draw millions of people annually to enjoy wildlife, breathtaking scenery, and recreational adventure. Visitor use is highly seasonal and occurs primarily during the summer months when temperatures are warm and daylight is long. Climate is an important consideration when planning a trip to Alaska's national parks because of the great distances and associated costs of travel for many visitors. As a result of projected climate warming, peak visitor season of use in Alaska's national parks may expand. To examine the potential effects of warming climate on park visitor season of use, we used regression analyses to quantify the relationship between historical (1980–2009) visitor use and monthly temperatures for three Alaskan national parks and identified the monthly mean temperatures at which the peak visitor season of use occurred in each park. We compared these contemporary temperatures with projected future average monthly mean temperatures for 2040–2049 and 2090–2099 to provide context for how visitation might be affected by warming climate. Based on historical relationships among temperature, visitor use, and increased temperatures associated with climate change, our analysis suggests that peak season of visitor use could expand into May and September depending on the park, the climate scenario, and the time period. As a consequence of a warming climate, planning by the National Park Service and other stakeholders may need to consider this transition in temperatures and the potential for an extended peak season of visitor use, along with other climate-related changes (e.g., extreme weather), climate-induced environmental changes, and shifts in recreational opportunities that will likely accompany climate change.

Key words

Alaska, climate change, national park, temperature, visitor use

ALASKA'S NATIONAL PARKS DRAW PEOPLE FROM ALL over the world for wildlife viewing, breathtaking scenery, and recreational opportunities, including hiking, backpacking, mountain climbing, boating, hunting, and fishing. In 2012 these parks received more than 2.4 million visitors (NPS 2013*a*) and in 2011 they generated $237 million in state economic benefit, a conservative estimate because of challenges in capturing the full spending attributed to visiting national parks in Alaska (Cui et al. 2013). National parks provide a large portion of nature-based tourism. Regional climate directly affects this tourism by influencing the activities of visitors and contributing to the quality of the visitor experience (Amelung et al. 2007). The climatic influence on visitation is most evident in the northern national parks found in Alaska, wherein the majority of visits occur during the warmer months of summer when weather and daylight are conducive to recreational activities. Shifts in the length and quality of the warm season caused by climate change will likely alter visitation to national parks in Alaska (Suffling and Scott 2002) and provide a key consideration for planning recreation and tourism activities and related services (Scott and Lemieux 2010).

Relatively rapid climate change in Alaska poses a significant challenge to ecological conservation and management and to land use planning (NPS 2012*a*). Alaska's climate has warmed over the last 50 years at an average rate of more than twice that of the rest of the United States (USGCRP 2009). During this time, annual mean air temperatures (hereafter referred to as "temperature") throughout the state increased by 3.4°F (1.9°C) (USGCRP 2009). The greatest increases in Alaska were seen in the winter, with temperatures rising by 6.3°F (3.5°C) (USGCRP 2009). Total precipitation also increased in all seasons except summer at the end of the 20th century throughout the state outside of the Arctic region (Stafford et al. 2000). By the middle of the 21st century, annual precipitation is expected to increase and annual mean temperatures are expected to be 3.6° to 7°F (1.9°–3.9°C) higher than at present with a longer summer growing season (USGCRP 2009). Thus, climate change will continue to affect ecological, hydrological, and human systems in a profound way throughout Alaska (USGCRP 2009). Impacts on glacial and permafrost extent, storm severity, sea-level rise, subsistence living, severity and extent of forest fires, insect outbreaks, and general disruption to ecosystem processes and functions will continue to challenge scientists and planners (USGCRP 2009). All of these factors play a role in the safety, frequency of visits, and enjoyment of Alaska's national parks.

The commitments associated with cost of travel, perceived isolation, and distance from the rest of the United States likely compel potential visitors to plan their vacations for times that maximize their chance of predictably good weather, which has been seen in other mountainous regions (Parks Canada 2004; Scott et al. 2007). National Park Service recreational visitor statistics for many U.S. national parks show that visitor use is related more often to regionally pleasant weather patterns than to institutional seasonality associated with school- and work-related vacation periods. For example, visitor use is highest during the spring and fall months in some parks located in the southwestern warm desert, where temperatures can be extreme in summer and winter. Alternatively, visitor use can be fairly consistent year-round in the Hawaiian Islands, where temperatures are generally pleasant throughout the year. Visitor use in the Rocky Mountains is often constrained to the summer months when temperatures typically exceed the likelihood of freezing conditions (NPS 2012*b*; figs. 1A, 1B, and 1C).

Scientists have established links between climate change and shifts in the timing of visitor use, with some parks already experiencing more visitor use earlier in the season than has been

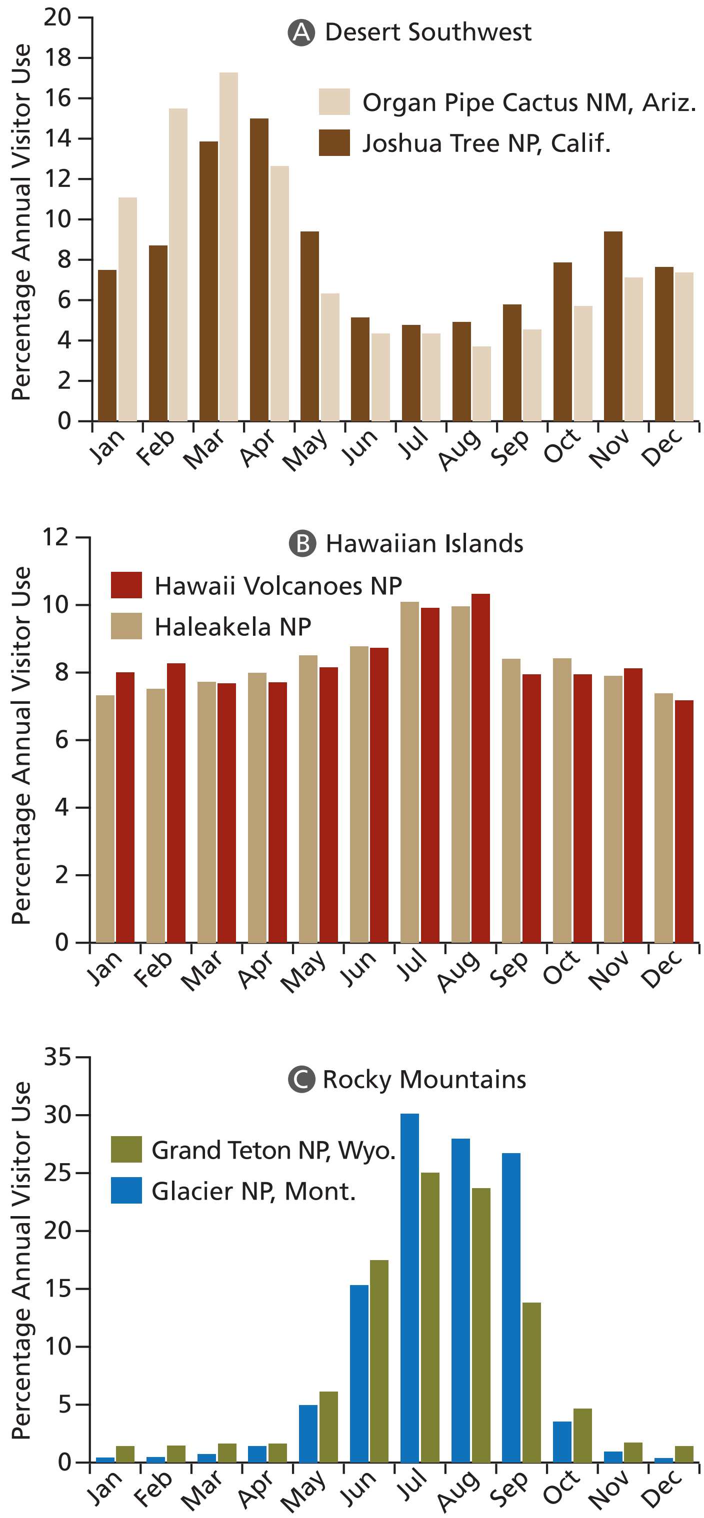

Figure 1. Seasonality of recreational visitor use in U.S. national parks (NP) and national monuments (NM) in regionally distinct areas: (A) southwestern deserts, (B) Hawaiian Islands, and (C) Rocky Mountains.

Figure 2. The study involved three Alaskan national parks—Gates of the Arctic (red), Denali (green), and Katmai (blue)—distributed over markedly different latitudes to gauge the influence of projected climate changes on visitor season of use.

observed historically (Buckley and Foushee 2012). Climate change is expected to expand periods of climate conditions conducive to visitor use at higher latitudes (Scott et al. 2004; Amelung et al. 2007), which may lead to more visitor use at times that are currently considered shoulder seasons (Scott et al. 2007). While many climate-related factors may directly or indirectly influence visitor use, temperature has been shown to be a stronger predictor of national park visits than other climate variables, such as precipitation (Richardson and Loomis 2004; Scott et al. 2007). Indeed, simple temperature-based models of snow accumulation and melt can simulate observations as well as, if not better than, more complex models that include other climate variables (Franz et al. 2008). Because of the strong seasonality of visitor use in Alaska's national parks and the expected changes in climate, we conducted a study to identify how historical visitor use relates to temperature to provide context for how future visitor season of use may change at each of three Alaskan national parks under three different climate change scenarios.

Methods

We selected three Alaskan national parks for study: Gates of the Arctic, Denali, and Katmai (fig. 2). We chose these parks because they are distributed across a latitudinal gradient within the state and may experience different magnitudes of climate change effects. To analyze the potential impacts of climate change on visitor season of use, we first examined the relationship between temperature and recreational visitor use (hereafter referred to as "visitor use," defined by the National Park Service as entries of persons onto lands or waters it administers; NPS 2013*b*). We used historical (1980–2009) monthly temperature data (decadal averages of monthly mean temperatures, averaged across the three historical decades) downscaled by SNAP (Scenarios Network for Alaska and Arctic Planning) (based on Climate Research Unit of the University of East Anglia time-series data, version 3.1; SNAP 2012) to a 2 km (1.2 mi) resolution. These data were then averaged across the entire park. To characterize historical visitor use, we used monthly recreational visitor use data from each of the parks (NPS 2012*b*) for the 1980–2009 period. Because average monthly park visitor use tended to increase over the study period, we calculated the percentage of annual visitor use that occurred in each month of a given year to standardize the monthly frequency of use across time.

Examination of plots of temperature and visitor use indicated a nonlinear relationship between these two variables. Following Scott et al. (2007), we used regression analysis to fit the data using a third-order polynomial equation to quantify the relationship between temperature and visitor use at each park. For our analysis we defined peak season as those months when >10% of annual visitor use occurred, as this was a natural break in the data for all three parks that appeared to distinguish peak season from shoulder seasons. We used the fitted regression equation to estimate the average monthly mean temperature at which 10% of annual visitor use occurred for each park and used this temperature as a point of reference to provide context for how the visitor peak season of use may change in the future given projected average monthly mean temperatures for 2040–2049 and 2090–2099, representing mid- and end-of-century conditions.

Future temperatures were derived from an average of five top-ranked global circulation models that perform best across Alaska and the Arctic (Walsh et al. 2008) under three emission scenarios adopted by the International Panel on Climate Change (IPCC; Nakićenović et al. 2000). The A2 scenario assumes a world with high population growth and slow technological and socioeconomic change, resulting in an increased rate of carbon dioxide emissions relative to today. The A1B scenario assumes rapid economic growth, new and efficient technologies, and finding a balance between fossil fuels and alternative sources of energy, resulting in a trajectory in carbon dioxide emissions similar to that of today. The B1 scenario represents the most optimistic case, in which carbon dioxide emissions level off at mid-century when population growth begins to decline, and governments emphasize global environmental sustainability through changes in economic and social structures (Nakićenović et al. 2000). As with the historical average monthly mean temperature data, we averaged

Table 1. Regression analyses between average percentage of annual visitor use and average monthly mean temperature for the 1980–2009 period in three Alaskan national parks

National Park	Equation	r^2	Temperature (°C) at 10% Annual Visitor Use
Gates of the Arctic	$Y = 0.002x^3 + 0.092x^2 + 1.2058x + 4.9768$	0.89	3.28 (37.9°F)
Denali	$Y = 0.0022x^3 + 0.084x^2 + 0.9714x + 3.2085$	0.96	4.77 (40.6°F)
Katmai	$Y = 0.0118x^3 + 0.0524x^2 - 0.0783x + 1.7909$	0.96	7.39 (45.3°F)

downscaled (based on Coupled Model Intercomparison Project model outputs for IPCC's Fourth Assessment Report; SNAP 2012) 2 km (1.2 mi) resolution data across each of the parks.

Results

From 2000 to 2009, Denali had the highest average number of annual visitors (386,805), Katmai had an intermediate number (56,237), and Gates of the Arctic had the lowest number (8,954). There is remarkable visitor seasonality in these national parks (fig. 3). Cool-season (October–April) visits represent a small percentage of annual visits: 1% for Gates of the Arctic, 6% for Denali, and 13% for Katmai. Visitor use at all three parks corresponded closely to high monthly mean temperatures (fig. 4). Peak season occurs in the warm summer months of June, July, and August when average monthly mean temperatures are typically greater than 50°F (10°C), while few people visit from October to May. September is a month of moderate visitation at Katmai. Regression analyses indicated strong relationships between temperature and visitor use at all three parks (table 1, fig. 4). Based on these regressions, the 1980–2009 average monthly mean temperatures associated with peak season were 38°F (3.3°C), 41°F (4.8°C), and 45°F (7.4°C) for Gates of the Arctic, Denali, and Katmai, respectively (table 1, fig. 4). These temperature values provided a point of reference for temperatures at which most visitation occurs in these parks and function as a baseline for comparing future projections of temperature change and its effects on peak visitor season of use.

Over the coming century, average monthly mean temperatures are projected to rise substantially in each of these parks, especially during the shoulder months in spring and fall, as well as in winter, regardless of emission scenario. By the 2040s, the projected average monthly mean temperature at Gates of the Arctic is expected to be similar to the historical average (1980–2009) in June and July, but temperatures will increase in August for each climate scenario (fig. 5A). Average monthly mean temperatures in May and September are projected to approach the 38°F (3.3°C) point of reference for Gates of the Arctic by the 2080s in the A1B and A2 scenarios. Fall and winter average monthly mean temperatures are also substantially warmer by the 2090s in all scenarios.

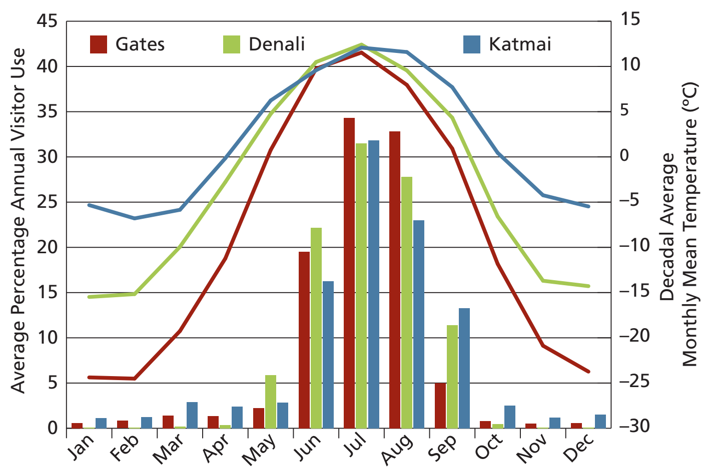

Figure 3. Average percent annual visitor use (bars) and average monthly mean temperatures (°C) (lines) by month at Gates of the Arctic (red), Denali (green), and Katmai (blue) National Parks. Values represent averages for 1980 to 2009.

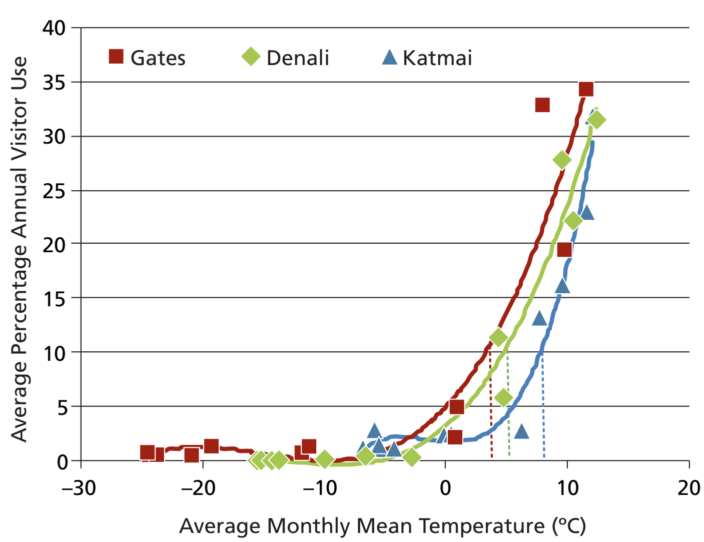

Figure 4. Relation of average percentage annual visitor use and average monthly mean temperatures (°C) (1980–2009) at Gates of the Arctic, Denali, and Katmai National Parks. The solid lines show the fitted regression line. The vertical dotted lines indicate points of reference for peak visitor season of use, defined as >10% annual visitor use. Points of reference correspond to 3.28°, 4.77°, and 7.39°C for Gates of the Arctic, Denali, and Katmai, respectively.

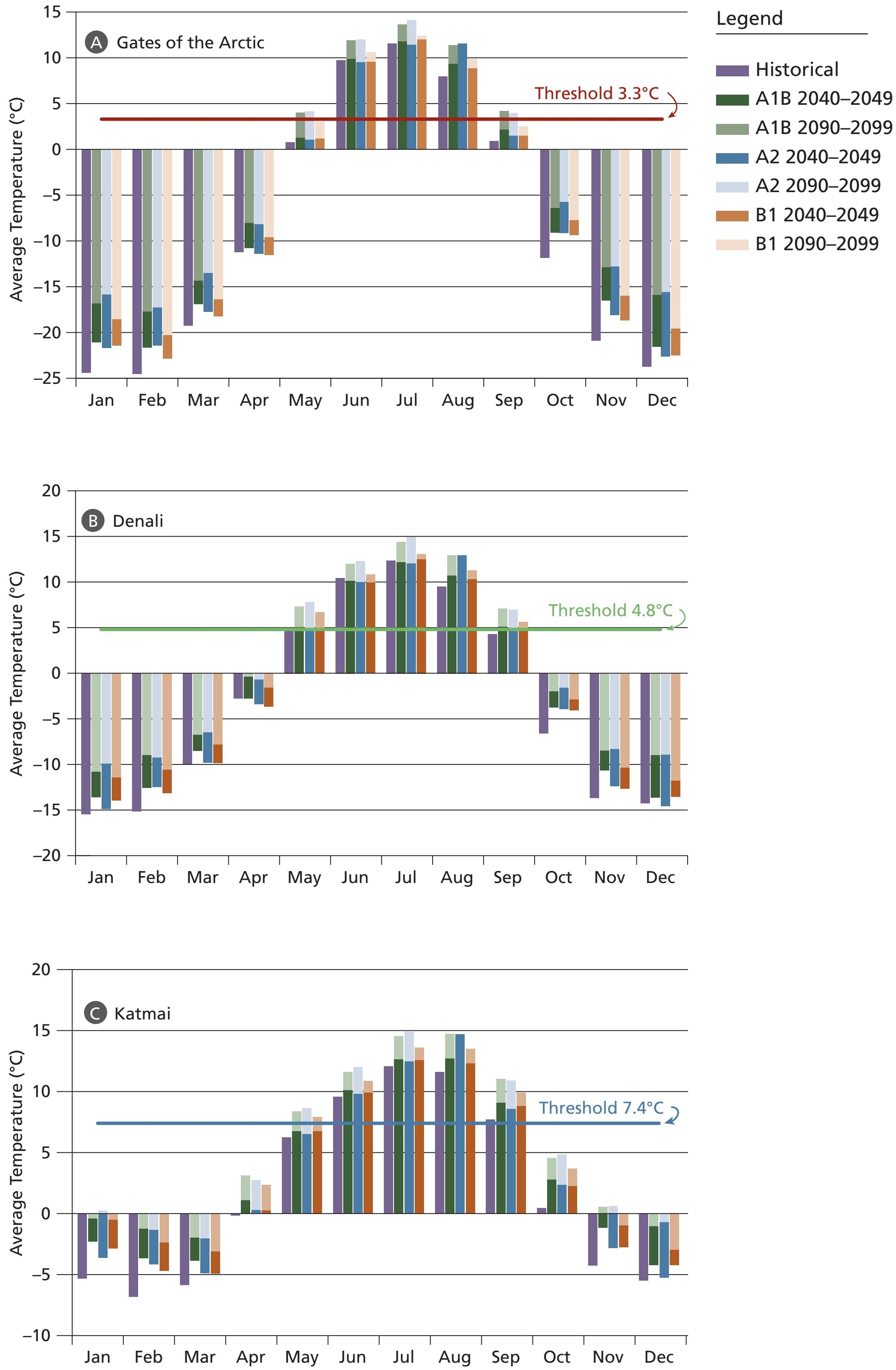

Figure 5. Historical (1980–2009) in purple and projected average monthly mean temperatures for the 2040s (darker color) and 2090s (lighter color) with three greenhouse gas emission scenarios (A1B, A2, and B1) for (A) Gates of the Arctic, (B) Denali, and (C) Katmai National Parks. Points of reference for peak visitor season of use (threshold) are shown as horizontal lines in each graph.

Similarly, by the 2040s Denali is projected to experience temperatures near the park's temperature point of reference of 41°F (4.8°C) under all emission scenarios from May through September and will be well above the point of reference by the 2090s (fig. 5B, previous page). During April and October historically, temperatures have been well below freezing, but are projected to be closer to 32°F (0°C) at Denali, and winters will also be substantially warmer by the 2090s, particularly under the A2 and A1B emission scenarios. Changes are less pronounced under the B1 emission scenario, which represents a leveling off of human-source emissions by mid-century.

By the 2040s, average monthly mean temperatures at Katmai are projected to be above the 45°F (7.4°C) point of reference for visitor use from June through September (fig. 5C, previous page). By the 2090s, May temperatures also rise above the Katmai point of reference, and even April and October averages are well above freezing temperatures, which has not been the case historically. Furthermore, average monthly mean temperatures during the rest of the year (November through March) are projected to be near or just below freezing by the 2090s, which represents substantial warming compared with historical conditions.

Discussion

Our analysis may help the parks to anticipate management needs under future climate change by providing context for understanding how temperature may relate to future visitor use. While short-term changes in climate have relevance for contemporary tourism planning, long-term climate change projections provide strategic relevance to park managers and the tourism industry (Scott et al. 2007). Based on historical relationships between temperature and visitor use and projected changes in temperature over the coming century, our research suggests that peak season of visitor use could expand by up to two months depending on the park, the climate scenario, and the time period analyzed. As temperatures in months currently considered shoulder seasons (e.g., May and September) become more similar to temperatures during the current peak season, we expect an increase in the percentage of annual visitation during these months, provided that other climatic, ecological, and social factors are conducive to this increase.

While climate is strongly linked to visitor use in Alaska's national parks, it acts in combination with other factors to determine seasonality and amount of visitor use and annual visitation trends. For example, human population growth and socioeconomic conditions may influence overall visitor use regardless of the environmental and climate conditions of these parks. Moreover, while some studies indicate that temperature more strongly affects visitor use than precipitation in mountainous regions (e.g., Richardson and Loomis 2004; Scott et al. 2007), changes in precipitation patterns associated with climate change may still affect conditions such that they are more uncertain or unpleasant during different times of year than has been the case with historical precipitation patterns. Twenty-first century changes in visitation may also be influenced by perceived diminishment of natural wonders (e.g., glaciers) as the climate warms; thus visitation may increase in the near future as people desire to see sights or events before they decline or disappear in the latter part of the century because of climate change. Although park managers may expect an increase in visitation over the next 20–30 years, visitors may view new conditions as having less value, which could result in a negative impact on visitation by the end of the century (Scott et al. 2007).

Our analysis may help the parks to anticipate management needs under future climate change by providing context for understanding how temperature may relate to future visitor use.

Numerous trade-offs are associated with the potential for increasing visitor use in Alaska's national parks. As a consequence of climate warming, planning by the National Park Service, park concessioners, and neighboring communities may need to consider potential changes in the timing, duration, and amount of visitor use. For example, park facilities (e.g., trails, lodging, roads, waste management, and water systems) may require additional maintenance and more frequent upgrades than in the past. Visitors may also expect park facilities to be operational earlier and later than historically prescribed, necessitating the hiring of seasonal staff earlier and for a longer duration. This will result in increased costs for operations and staffing that may be offset by increased economic benefits from user fees authorized under the Federal Lands Recreation Enhancement Act of 2004. Local businesses, particularly those in adjacent communities, could see increased revenue from park-related activities. For example, access to Katmai and Gates of the Arctic National Parks is largely restricted to airplane transport provided by local communities. An increase in visitor use in the shoulder seasons could generate more revenue for these local businesses, provided that they have the capacity and desire to expand these services.

Visitors hike among aspen trees on Taiga Trail in Denali National Park. Warming climate may extend the "shoulder" seasons for park visitation at Denali and other Alaskan national parks.

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Ecological changes and shifts in recreational opportunities that will likely accompany climate change may alter the seasonality in which peak recreation and wildlife viewing are possible. Phenological shifts associated with the timing of migratory or foraging patterns in fish, birds, and wildlife may disrupt the predictability of viewing opportunities (Taylor 2007; Post and Forchhammer 2008). For example, in Katmai, timing and locations of visitor use are closely tied to opportunities for viewing bears. Timing and locations of bears are related to summer salmon runs. It is unclear how climate will influence these complex ecological patterns and processes that will ultimately lead to shifts in visitor use trends. Thus, more research is needed to determine visitor responses to climate-induced ecological changes in Alaska's national parks.

Expected changes in the frequency, timing, and severity of extreme weather events causing fires, flooding, landslides, and avalanches may pose new hazards to visitor and staff safety, requiring heightened awareness of those working and playing in the national parks (Suffling and Scott 2002). Infrastructure damage caused by these events or from other climate-related changes, such as thawing permafrost, can result in increased maintenance and repair expenses. In addition, extreme events may dramatically alter the composition and structure of park ecosystems, such as vegetation in debris flows and inundated floodplains, with potentially long-lasting effects.

Conclusions

As indicated in this and other studies (see the introductory paragraphs to this article), climate change has the potential to alter visitor use patterns as well as the scenic, recreational, cultural, and ecological values for which the parks were designated. In this context, our research suggests that park managers may experience new challenges in balancing visitor support and conservation of natural and cultural resources in a warming world. Our temperature-based assessment provides a first approximation of potential changes in visitor use, but does not account for other factors that could influence visitation in the future, such as transportation costs, enhanced park facilities, indirect effects of climate change on park resources, and many other climate and non-climate factors.

The uncertainty associated with future visitor use patterns leads to several considerations for managers and the tourism industry. Should parks invest in building new infrastructure for access to wildlife and other viewing opportunities to accommodate ecological shifts associated with climate change? As thawing permafrost causes damage to structures of cultural significance or other types of infrastructure, should parks invest in stabilizing or moving these features? If climate change results in more seemingly pleasant conditions that attract a greater number of visitors or shifts in seasonal use patterns, should parks accommodate these visitors in places that typically experience low or short-duration human traffic? Future research aimed at planning for climate change impacts on the National Park System should incorporate climatic, environmental, and social data for a holistic evaluation of projected change and adaptive capacity. Conserving these areas will be a challenge for all stakeholders, but provides an opportunity to engage the public in understanding changing climate and the continued management and protection of our valuable national parks.

Acknowledgments

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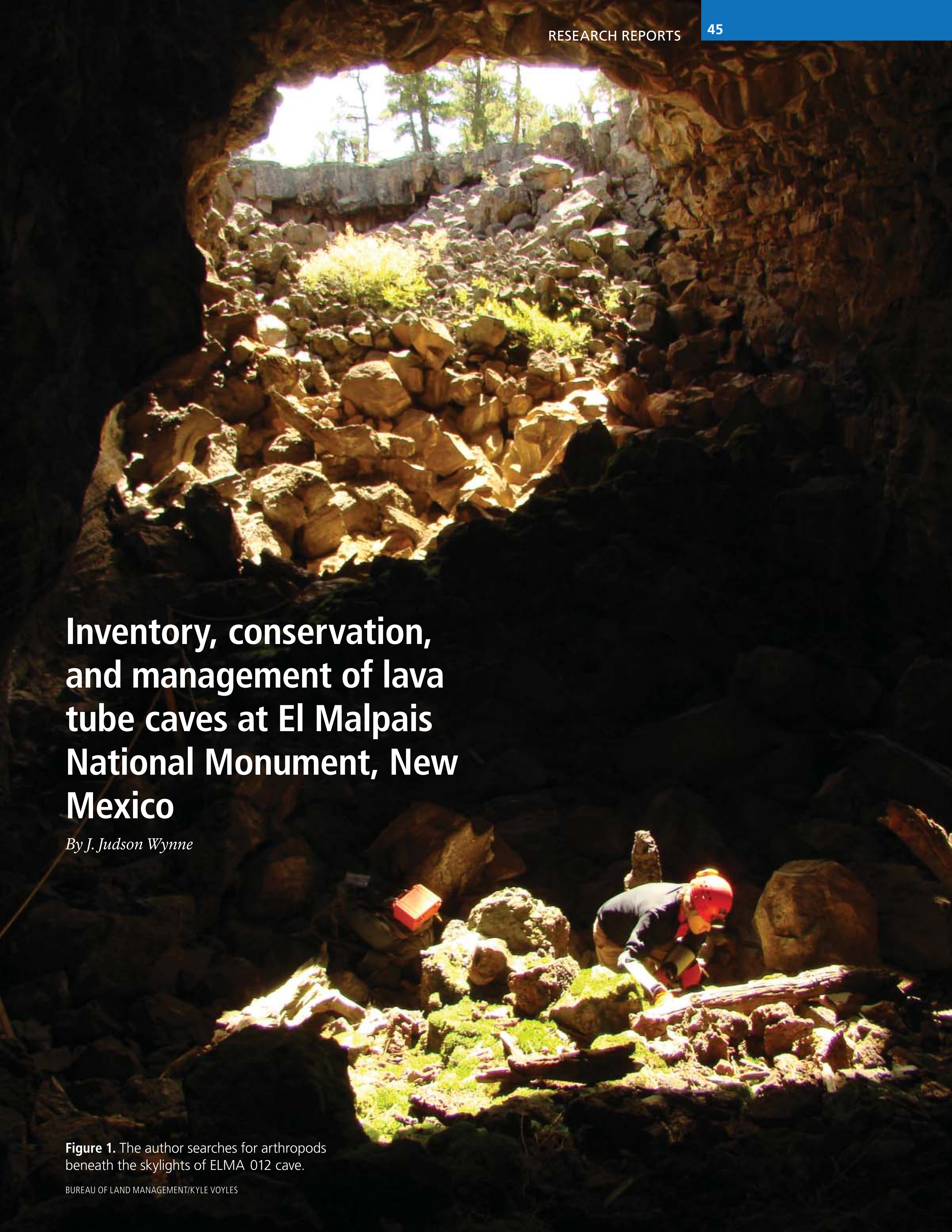

Inventory, conservation, and management of lava tube caves at El Malpais National Monument, New Mexico

By J. Judson Wynne

Figure 1. The author searches for arthropods beneath the skylights of ELMA 012 cave.

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Abstract

Lava tube caves at El Malpais National Monument have received little scientific attention with regard to their bat and arthropod populations. From an all taxa biological inventory of 11 caves, I identified seven new species of cave-dwelling arthropods (including two potential troglobites) and range expansions of two parasitoidal wasps. The presence of unique microhabitats, tree root "curtains" hanging from the ceilings, and moss gardens in cave entrances resulted in higher species richness of arthropods at four caves. For bats, I confirmed continued use of one large bat hibernaculum cave and one significant bat maternity roost. While several recommendations have been made to better conserve and manage sensitive cave resources, additional research and monitoring will be required for the long-term management and protection of several caves. Finally, I introduce three new terms to cave biology: two for entrance-dwelling animals (eisodophiles and eisodoxenes) and one for animals that hunt deep within or near the entrances of caves (xenosylles).

Key words

cave, cave biology, cave-dwelling arthropods, cave-roosting bats, conservation, eisodophile, eisodoxene, Monument, land management, lava tubes, xenosylle

LOCATED IN WESTERN NEW MEXICO, EL MALPAIS National Monument encompasses approximately 1,522 km^2 [~588 mi^2]. Featuring at least eight major volcanic eruptions ranging in age from 100,000 to 3,000 years old (Cascadden et al. 1997), the national monument is a dramatic landscape comprising vast expanses of pahoehoe and ʻaʻā lava flows, cinder cones, ice caves, and at least 290 lava tube caves (fig. 1, previous page). Despite the large number of lava tube caves, this region has received little scientific attention with regard to bat and arthropod populations that occur within these features.

Bats are often considered keystone species of cave ecosystems. When bats populate caves in large numbers, they transport a significant amount of organic material (as guano) from the surface into the cave. Although bats have been studied throughout most of the western United States, how these animals use caves remains underresearched. Bat maternity roosts (sites where female bats rear their pups) and hibernacula (winter hibernation sites) are highly sensitive to human disturbance (Brown et al. 1993; McCracken 1989; Elliott 2000; Hamilton-Smith and Eberhard 2000). With the westward advance of white-nose syndrome, a disease responsible for the mortality of more than five million bats in eastern North America (USFWS 2012), inventory and monitoring of all roost sites will be critically important to the long-term management of bats at El Malpais National Monument.

Other animals of high conservation and management value are arthropods that occur exclusively in caves. Prior to this study, at least five cave-adapted arthropods (presumed sensitive species) were known from six lava tube caves at El Malpais (Northup and Welbourn 1997). Many troglomorphic (cave-adapted) animals are endemic to a single cave or region (Reddell 1994; Culver et al. 2000; Christman et al. 2005) and are generally characterized by low population numbers (Mitchell 1970). Additionally, numerous human-induced impacts threaten subterranean ecosystem health and the very persistence of cave-obligate species. Many cave-obligate species are therefore considered imperiled. Nonnative species introductions (Elliott 1992; Reeves 1999; Taylor et al. 2003; Howarth et al. 2007), global climate change (Chevaldonné and Lejeune 2003; Badino 2004), and recreational use (Culver 1986; Howarth and Stone 1993; Pulido-Bosch et al. 1997) are among the impacts that present challenges for the long-term management of cave-obligate arthropod populations at El Malpais.

An all taxa biological inventory focusing on bats, cave-dwelling arthropods, and other vertebrates was not only important to characterizing the fauna that use El Malpais lava tubes, but also was required to provide resource managers with the information necessary to best conserve and manage these sensitive resources. My objectives for this study were to (1) catalog all taxa using caves, including the identification of endemic and sensitive cave-adapted invertebrates, (2) apply and examine a systematic sampling protocol for inventorying arthropods, (3) draw comparisons across the national monument to gain inference into patterns of invertebrate species distributions, biodiversity, biogeography, and endemism, and (4) provide recommendations to enhance management of El Malpais lava tube caves. I addressed objectives 1 and 4 in this article and will address objectives 2 and 3 in subsequent publications.

Methodology

During 7–15 October 2007 and 8–15 October 2008, research teams and I conducted two site visits per cave at 10 caves at El Malpais National Monument and 1 cave on adjacent Bureau of Land Management lands. We scheduled site visits around deployment and collection of baited pitfall traps for sampling cave-dwelling arthropods. At the monument's request, I used cave codes rather than actual cave names for all caves on National Park Service lands. A copy of this report, which includes a table of cave names with associated cave codes, is on file with monument headquarters in Grants, New Mexico, and the National Cave and Karst Research Institute, Carlsbad, New Mexico.

Arthropod sampling

I used both opportunistic and systematic sampling to search for arthropods. During each cave visit, a team of three researchers uniformly applied three techniques: opportunistic collecting, baited pitfall trapping, and timed searches. For opportunistic collecting, the team collected invertebrates encountered as they walked between sampling stations while deploying and removing pitfall traps and conducting timed searches.

Because I wanted to maximize the number of invertebrate species detected, we sampled each cave from its entrance (i.e., drip line) to the back of the cave. Using available cartographic cave maps, teams applied an interval sampling approach whereby 10% of each cave's length was used as the sampling interval (e.g., for a 100 m–long cave [328 ft], sampling interval was every 10 m [33 ft]). All sampling stations were plotted on each cave map. Three sampling stations (one at either wall and one at the cave centerline) were established at each sampling interval. Fewer than three sampling stations per sampling interval were established in only two cases: (1) when the cave passageway width was ≤5 m (16 ft), one station was designated in the best available location and (2) when exposed lava floors were encountered and no materials were available to aid in countersinking the trap, the sampling station was skipped.

At each sampling station we deployed one pitfall trap and conducted two timed searches. Traps consisted of two 907 g (32 oz) plastic containers (13.5 cm high, 10.8 cm diameter rim, and 8.9 cm base [5.3 in high, 4.3 in diameter, 3.5 in base]) placed inside one another, with bait (a teaspoon of peanut butter) placed in the outer container and holes punched in the base of the inner container. This design attracts arthropods and keeps most animals separated from the bait. With the assistance of field technicians, I buried containers to the rim where possible, built rock ramps to the trap rim in other cases, and covered all traps with a caprock. The team conducted timed searches within a 1 m (3.3 ft) radius of each pitfall trap station for a period of 1 to 3 minutes before traps were deployed and prior to checking traps (modified from Peck 1976). Each search was concluded after 1 minute if no arthropods were detected and continued for a total of 3 minutes when arthropods were observed.

Because some caves contain unique microhabitats that support distinct arthropod communities and endemic populations, I augmented this sampling protocol by conducting direct intuitive searches in those areas. Unique microhabitats at El Malpais include moss gardens (refer to Northup and Welbourn 1997) at cave

Figure 2. Both moss gardens (top row) and root curtains (bottom row) are important microhabitats and support new and unique arthropod species. At least three lava tube caves contain moss gardens within cave entrances and beneath skylights, and two caves contain plant root curtains within cave deep zones at El Malpais National Monument.

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entrances and beneath skylights (i.e., holes in the ground formed by the partial collapse of the cave roof), and tree root "curtains" hanging from the ceilings in cave deep zones (fig. 2). Within each unique microhabitat we spent one hour (3 observers × ~20 minutes) searching for arthropods. Specifically, we searched tree root curtains hanging from the ceilings in two caves (one hour per cave) and moss gardens in two caves (one hour per cave).

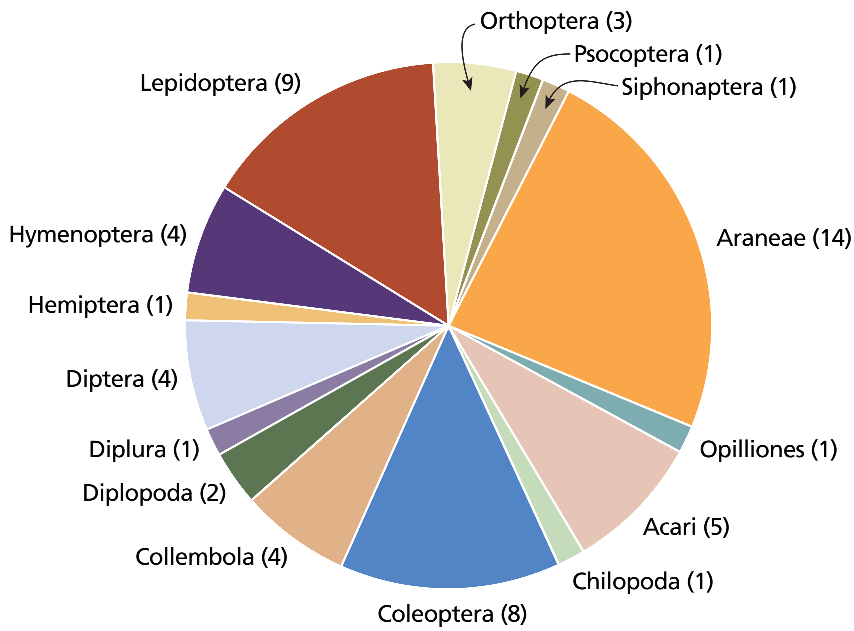
Arthropod identification

For arthropod groups actively being studied by taxonomic specialists, I sent either specimens or high-resolution images of specimens to various taxonomic experts for identification. Otherwise, we used existing keys to identify specimens to the lowest taxonomic level possible.

Bat sampling

I visited and inventoried three known bat roosts: a Mexican free-tailed bat (*Tadarida brasiliensis*) maternity colony, a Townsend's big-eared bat (*Corynorhinus townsendii*) maternity roost, and a Townsend's big-eared bat hibernaculum. Additionally, for sites where bat use was unknown, these caves were surveyed for midfall use by bats. Research teams scanned ceilings and walls throughout the length of each cave, and searched for any evi-

Figure 3. Number of arthropod morphospecies detected by order (including classes Chilopoda and Diplopoda) at El Malpais National Monument in 2007 and 2008. The surveys were conducted at 11 caves: ELMA-062, ELMA-008, ELMA-110, ELMA-262, ELMA-012, ELMA-054, ELMA-029, ELMA-303, ELMA-315, ELMA-061, and Hummingbird.



dence of bats (e.g., guano). When bats were encountered, I attempted to identify them to species visually. No bats were handled during this study.

Documenting other vertebrates
Within each cave, I searched for and recorded the presence of all other vertebrate species. Sign of other vertebrates included direct observation, scat, feathers, and skeletal remains.

Cave specificity functional groups
I divided El Malpais cave-dwelling taxa into nine cave specificity functional groups. The following functional group terminology was taken from Barr (1968) and Howarth (1983): (1) *troglobites*, obligate cave dwellers who can only complete their life cycle within the cave environment; (2) *troglophiles*, species that occur facultatively within caves and complete their life cycles there, but also exist in similar surface microhabitats; (3) *trogloxenes*, taxa that frequently use caves for shelter but forage on the surface; and (4) *accidentals*, morphospecies occurring within caves, but which cannot survive within the cave environment. Additionally, because this project sampled cave entrances for arthropods and documented other vertebrate (i.e., non-bat taxa) use of caves, I propose three additional groups for categorizing cave-dwellers: (5) *eisodophiles*, species facultatively using cave entrances and twilight zones (areas where light faintly penetrates into the cave, but is sufficient for humans to see) that may complete their life cycles there, but also occur in similar partially sheltered surface environments; (6) *eisodoxenes*, animals that frequently use cave entrances and twilight zones for shelter but return to the surface to forage; and (7) *xenosylles*, surface-dwelling animals that hunt deep within caves or in the cave entrances. For eisodophiles and eisodoxenes, the etymology of the first half of the terms, *eísodo*, is from the Greek word *eísodos*, "entrance," while the second halves were derived from the same naming convention used for functional groups 2 and 3: *philos*, Greek for "love" or "desire," and *xenos*, Greek for "guest." *Xenosylle* is a combination of *xenos* and *sylléktis*, Greek for "collector." Finally, (8) *parasites*, the special-case group, describes parasitic arthropods detected in caves due to the presence of their host (e.g., bats or other vertebrates); and (9) *unknown* is used for animals for which information is lacking to reasonably place them in one of the eight other groups.

Results

My work resulted in the identification of at least 66 morphospecies (groups distinguished from others based upon morphological characteristics), including 59 arthropods (representing at least 13 orders and two classes; fig. 3), three bats, and four other vertebrates. Appendix A, available online at http://www.nature.nps.gov/ParkScience/archive/PDF/Article_PDFs/ParkScience30(1)Summer2013_A1–A12_Wynne_3653.pdf, shares the entire list of inventoried species and provides explanations for cave specificity functional group designations.

Arthropods
Cave specificity functional groups for arthropods consisted of one troglobite, two questionable troglobites, five troglophiles, 18

questionable troglophiles, seven trogloxenes, one questionable trogloxene, one accidental, 21 eisodophiles, one parasite, and two unknowns (fig. 4). At least seven new species were discovered and two range expansions were documented. New species discoveries include one potentially cave-adapted spider (family Theridiidae, *Theridion* n.sp.?); a mite (family Histiostomatidae, *Histiostoma* n.sp.); two springtails (order Collembola, *Drepanura* n.sp. and *Pogonognathellus* n.sp.); one new cricket species (*Ceuthophilus* cf *apache* n.sp.); one beetle (family Carabidae, *Rhadine* n.sp., *perlevis* species-group); and a new species of potentially cave-adapted planthopper (order Hemiptera: superfamily Fulgoroidea; Fulgoroidea n.sp.?; refer to fig. 5 for images of select new species). Additionally, I confirmed the persistence of the troglomorphic bristletail (order Diplura: family Campodeidae; Campodeidae n.sp.) within the deep zone of ELMA-054. Northup and Welbourn (1997) identified this as both a troglobite and an "undetermined species." Working with dipluran taxonomist Dr. R. Thomas Allen (Academy of Natural Sciences, Drexel University, Philadelphia, Pennsylvania), we confirmed this animal as a new species in 2013. The likely new species of planthopper was detected within the cave deep zones on roots protruding from the ceiling of ELMA-315 and ELMA-303; this animal has reduced eyes in its nymphal stage and may be troglomorphic. Also, this work resulted in the range expansions of two species of parasitoidal wasps (fig. 6, next page) (family Tiphiidae, *Tiphia andersoni* and *T. nona*; Allen 1971). Both tiphiids were in a torpor and collected from beneath rocks within the moss gardens of ELMA-008 and ELMA-012. Given the season (midfall) and their behavior, I suggest these wasps may have been preparing to hibernate within the moss gardens.

A, B, AND D: NORTHERN ARIZONA UNIVERSITY/JUT WYNNE; C: BUREAU OF LAND MANAGEMENT/KYLE VOYLES

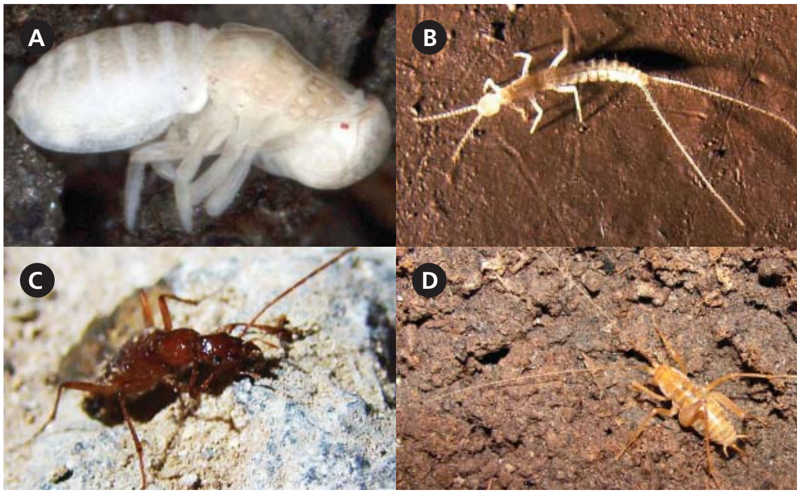
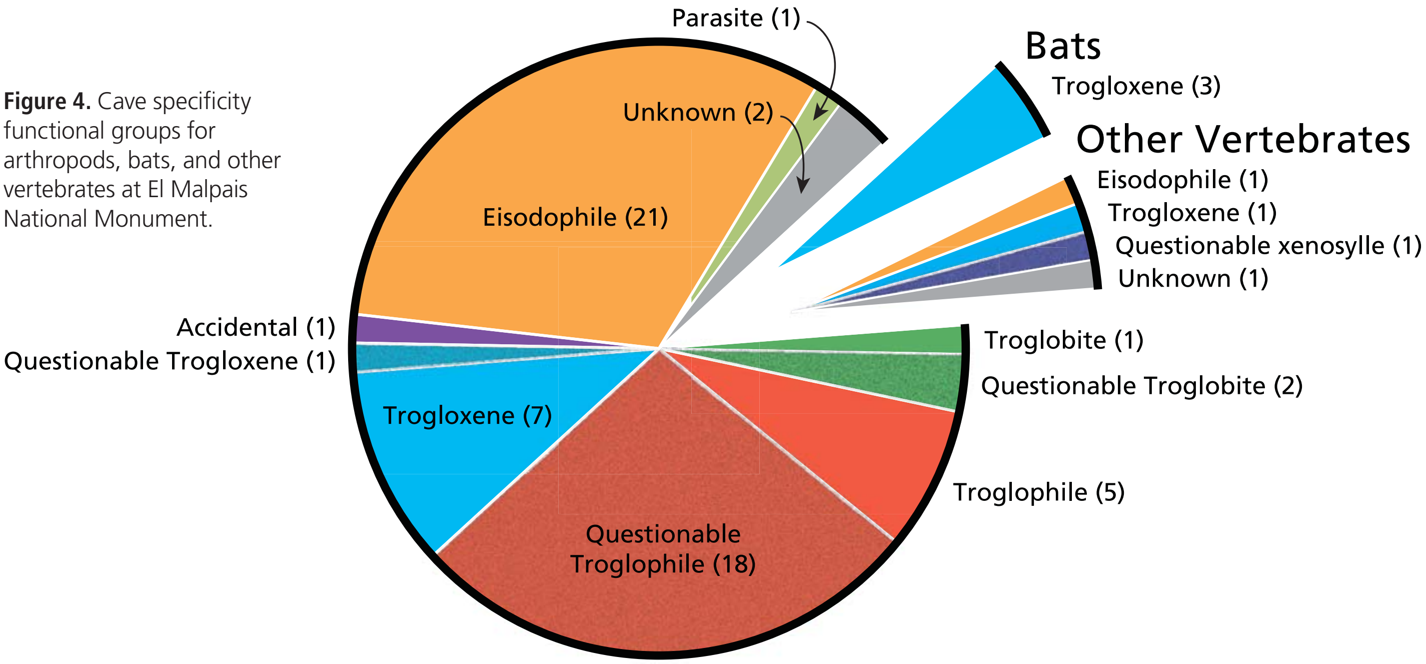


Figure 5. (A) New species of a potentially troglobitic planthopper (order Hemiptera: superfamily Fulgoroidea: Fulgoroidea n.sp.?; body length ~1.5 mm); (B) new species of troglobitic bristletail (order Diplura: family Campodeidae; from Northup and Welbourn 1997; length 2.5 mm); (C) new species of trogloxenic Carabid beetle (*Rhadine* n.sp., *perlevis* species-group; length 15 mm); and (D) new species of trogloxenic cave cricket (*Ceuthophilus* cf *apache* n.sp.; length 25 mm).

Figure 4. Cave specificity functional groups for arthropods, bats, and other vertebrates at El Malpais National Monument.



Caves with the highest arthropod species richness, in rank order, were ELMA-315 (n = 22), ELMA-012 (n = 16), ELMA-303 (n = 15), ELMA-008 (n = 15), ELMA-062 (n = 13), and ELMA-262 (n = 11) (table 1). ELMA-315 and ELMA-303, which had the highest species richness, contain extensive root curtains protruding through ceiling fissures within the cave deep zones. For ELMA-012 and ELMA-008, richness is driven by the large number of species detected within moss gardens at cave entrances and beneath skylights. ELMA-062 supports a large Mexican free-tailed bat maternity roost; because significant nutrients via guano have been transported into this cave, this likely contributed to the high number of morphospecies. In 2007, logistical constraints prevented my team from sampling the moss gardens within the entrance of ELMA-029 and from further sampling ELMA-110 (which supports a bat maternity roost). Thus, I suggest both of these caves likely support more arthropods (in terms of richness and abundance) than are included in this report.

All new species reported here were identified as "new" by taxonomic specialists. Several of these new species ultimately will be formally described and the results published in scientific journals.

Figure 6. *Tiphia andersoni* Allen, 1971. This specimen was collected via a direct intuitive search of moss gardens (beneath the large skylights) of ELMA-012. Prior to this work, this parasitoidal wasp was known to occur in central Mexico, north into southeastern and north-central Arizona (Allen 1971). Detection of this animal in New Mexico represents an expansion of its known range.

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Bats

During the 2007 surveys, I observed five hibernating Townsend's big-eared bats in the deep zone of ELMA-054 and one torpid big brown bat (*Eptesicus fuscus*) roosting near the entrance (figs. 7A and 7B, respectively). Additionally, ELMA-062 continues to support a maternity colony of Mexican free-tailed bats. On 10 October 2007, I observed thousands of individuals of this species roosting approximately 75 m (246 ft) within the cave (fig. 7C); however, when I returned four days later to remove arthropod traps, that number dropped to less than 100. Also in October 2007, I did not observe any Townsend's big-eared bats in residence at ELMA-110; once I arrived, this roost was already abandoned for the year. Relatively fresh guano in the main section of the cave (beginning at the northeasternmost skylight, extending to the northeastern ward) suggests they were using this area before they relocated to their winter roosts. During an unrelated study in 2005 and 2006, I observed a maternity roost of this species in both the main cave and tunnel segments directly southwest of the main section of this cave. It seems the colony uses several areas in the tunnel segments and within the main cave passageway during the breeding season. Given the sampling period in early October (after breeding), I was unable to ascertain whether or not additional summer roosts exist on the national monument. However, aside from the two known maternity roosts, I did not observe any significant deposits of fresh guano (suggestive of a large summer roost) in any of the other caves. Thus, I have no evidence to suggest additional large summer roosts occur in the caves sampled. All bats were considered trogloxenes.

Other vertebrates

I documented small-carnivore (questionable xenosylle) scat, likely ringtail (*Bassariscus astutus*), skunk (*Conepatus* sp.), or raccoon (*Procyon lotor*), in ELMA-054 and ELMA-110. Skunks and raccoons often prey upon infirm bats or bat pups that have fallen from the ceiling (Winkler and Adams 1972), and ringtails are commonly known to hunt bats roosting on cave walls. I found a fully articulated ringtail skeleton near the terminus of the northern extent of ELMA-303. I sent photographs of the skeleton to Eastern Tennessee State University paleontologist Dr. Jim Mead. In an e-mail exchange with him on 4 March 2013, he suggested the remains were between 1,000 and 10,000 years old. This animal may have entered the cave to hunt bats, became disoriented, and, unable to find its way back to the entrance, died in the cave. Given its age, I did not consider this animal's remains as part of this inventory. Recent packrat (*Neotoma* sp.; *N. mexicana* and/or *N. albigula*; refer to Bogan et al. 2007) activity was evident in both ELMA-062 and ELMA-061; packrats are considered trogloxenes. Also, I found the carcass of a gopher snake (*Pituophis catenifer*) in the twilight zone of ELMA-061. The snake was wrapped around

Table 1. Observed morphospecies richness for arthropods, bats, and other vertebrates at El Malpais National Monument caves

Cave	Arthropods	Bats	Other Vertebrates
[μ]ELMA-008	15	—	—
[μ]ELMA-012	16	—	—
ELMA-029	—	—	—
[β]ELMA-054	2	2	1
ELMA-061	1	—	2
[β]ELMA-062	13	1	1
[β]ELMA-110	4	1	—
[μ]ELMA-262	11	—	1
[ρ]ELMA-303	15	—	—
[ρ]ELMA-315	22	—	—
Hummingbird	3	—	—

Notes: Some species were detected in two or more caves.
[μ]Moss gardens occurred beneath skylights and within entrances of these caves.
[β]Confirmed bat roosts.
[ρ]Caves with extensive root curtains protruding through the ceiling within the deep zone.

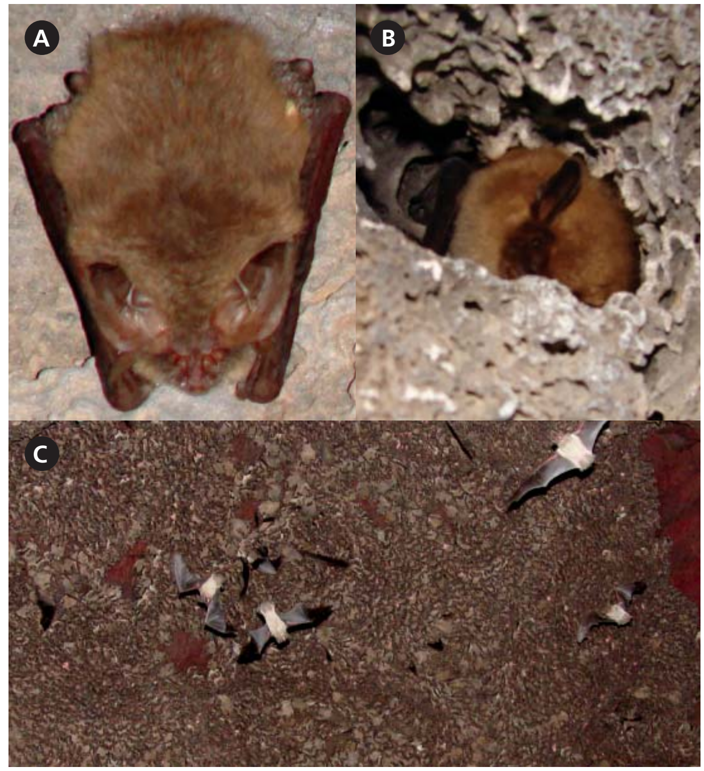

Figure 7. (A) Hibernating Townsend's big-eared bat and (B) big brown bat aroused from torpor at ELMA-054. (C) Late-season maternity colony of Mexican free-tailed bats within ELMA-062. Note that the "rough" surface in this panel is actually tightly clustered roosting bats. For scale, the average wingspan of Mexican free-tailed bats is 301 mm (12 in) (refer to Wilkins 1989).

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a stick and had several lacerations along the length of its body; I suggest a park visitor probably killed this animal. Because I do not know whether the snake was killed in the cave or it was brought into the cave postmortem, its use of the cave is "unknown." Finally, a barn owl (*Tyto alba*; eisodophile) was spooked as my team entered ELMA-262. This owl was roosting near the entrance and then flew to a skylight where it exited the cave.

Conservation and management

This work resulted in the identification of seven new species of cave-dwelling arthropods (including two potential troglobites), range expansions of two parasitoidal wasps, and two caves containing significant root curtains hanging from the ceiling. The presence of root curtains and moss gardens has been shown to be an important driver of high arthropod richness at El Malpais lava tube caves. For bats, I confirmed continued use of one hibernaculum cave and two significant bat maternity roosts. Although all of these caves will require further study, these findings have been useful in highlighting future management directions and research needs.

Arthropods

Four of the new species reported here are dependent on caves for most or all of their life cycle. The potentially new troglomorphic spider (*Theridion* n.sp.?) and the planthopper (Fulgoroidea n.sp.?) are likely to be restricted to the cave environment, while the cricket (*Ceuthophilus* cf *apache* n.sp.) and beetle (*Rhadine* n.sp., *perlevis* species-group) are trogloxenes. Unfortunately, only one specimen of *Theridion* n.sp.? was detected and collected; additional specimens will be required to describe this animal and determine whether it is indeed troglomorphic. In the two caves with root curtains, I identified at least one potential troglobite, a planthopper (Fulgoroidea n.sp.?). Unfortunately, all specimens collected were nymphs, and adults are required to confirm both cave adaptation and whether or not it is a new species. The remaining three newly discovered species likely occur in surface habitats as well as caves. The two new springtail species (*Drepanura* n.sp. and *Pogonognathellus* n.sp.) are edaphic (soil-dwelling) organisms. *Histiostoma* n.sp. is a very small mite (600–900 µm [0.6–0.9 mm] in length) and is found in association with other insects. Because the deutonymph (early life stage of mites) hitchhikes on larger-bodied insects for dispersal between habitats, this mite may have been transported into the cave by another animal. Both springtails and mites will require further study to determine their affinities for caves and the ecological roles they play in the cave environment.

To address questions concerning population dynamics and distribution patterns of these new arthropod species, additional surveys at caves known or likely to support these animals will be required. This information will be necessary to develop resource management plans to best protect these species and their habitats. All of these new species should be considered important finds because they expand our knowledge of the natural history of El Malpais National Monument and, by extension, the state of New Mexico.

Bats

ELMA-054 supports the largest known hibernaculum of Townsend's big-eared bats on the monument, while ELMA-110 supports the largest known maternity roost of this species. Wynne (2006) counted 100 Townsend's big-eared bats hibernating in the deepest section of ELMA-054. ELMA-110 supports a maternity roost of Townsend's big-eared bats, estimated at 50 individuals in 2006 (Wynne, unpublished data). ELMA-110 has been closed to park visitors for several years while bats are in residence. As a result of this study and the 2006 site visit, ELMA-054 is now closed during the hibernation period (October through mid-April).

Based on our knowledge of Townsend's big-eared bats in other areas, I suggest the same population uses both roosts. In Oklahoma, movements of these bats between maternity roosts and hibernacula averaged 11.6 km (7.2 mi) (range 3.1 to 39.7 km [1.9–24.7 mi], n = 3 individuals; Humphrey and Kunz 1976). Dobkin et al. (1995) documented Townsend's big-eared bats traveling distances ranging from 5 to 24 km (3–15 mi) from summer roost to foraging sites in Oregon. Additionally, Pierson et al. (1999) suggested that this species was in decline throughout its range. The straight-line distance from ELMA-110 to ELMA-054 is 10.5 km (6.5 mi).

Given that this species is likely to be the most affected by white-nose syndrome on the monument, knowledge of this bat's habits, movements, and roost locations will enhance its management and protection. I recommend conducting a radio tagging and telemetry study of Townsend's big-eared bats and their use of these two roosts. For such a project, radio tagging of bats should occur late in the maternity season (late August to early September). This project would enable monument personnel to (1) establish baseline estimates of population size and structure to begin monitoring this species and its two known roosts, (2) determine if the same population is using both ELMA-110 and ELMA-054, (3) potentially identify additional Townsend's big-eared bat roosts by tracking bat movements with telemetry, and (4) make informed decisions regarding potential cave closures and protection of this species.

Scientists and managers know little about the winter habitat requirements of year-round bat residents at El Malpais. Thus, more surveys are needed, particularly winter bat inventories, to identify additional hibernacula. I recommend annual to biennial monitoring of ELMA-054, as well as newly identified hibernacula and long-term microclimatic monitoring in caves supporting hibernating bats. In light of the westward advance of white-nose syndrome and global climate change, this information may be useful in guiding management decisions to protect bat populations in the future. Additionally, information gathered by such an endeavor may be informative for developing similar monitoring strategies for other management units of the National Park System in the southwestern United States.

Deep zones and unique habitats

All deep zone environments that support or have the potential to support cave-adapted animals should be considered high-priority sites for conservation and management. Deep zones are characterized where environmental conditions (e.g., complete darkness, temperature, relative humidity, moisture, airflow) remain relatively stable over time (refer to Howarth 1980 and 1982). When nutrients are added to this equation (via root curtains protruding from the ceiling, bat guano, or dissolved organic material percolating through rock), these areas should be intensively sampled for troglomorphic arthropods. For example, Howarth et al. (2007) stressed the importance of roots in caves for conserving troglomorphic arthropods in Hawaiian lava tubes.

Three caves on the monument meet these criteria. ELMA-315 and ELMA-303 contain deep zones with extensive root curtains hanging from the ceiling. During the arthropod sampling period, these caves were among the warmest on the monument (ELMA-315: mean temperature 12.4°C [54.3°F], standard deviation 0.5°C [0.9°F]; ELMA-303: mean temperature 11.9°C [53.4°F], standard deviation 0.7°C [1.3°F]). I know of no other caves in the region that support this microhabitat type. Additionally, ELMA-110 has the most extensive deep zone microhabitat known on the monument. At the terminus of this cave, water percolates through fissures into the cave chamber. I recommend conducting additional surveys in all of these caves using a bait sampling and direct intuitive search sample design (*sensu* Wynne 2010 and Wynne et al. 2012). These inventories, conducted during the most produc-

This work resulted in the identification of seven new species of cave-dwelling arthropods (including two potential troglobites), range expansions of two parasitoidal wasps, and two caves containing significant root curtains hanging from the ceiling.

tive times of year (i.e., spring and summer), would likely result in the detection of additional troglomorphic arthropods.

Because cave-adapted animals are cryptic, they are often difficult to detect and researchers must conduct numerous site visits to obtain even a baseline knowledge of community composition. For example, Krejca and Weckerly (2007) reported that 10 to 22 site visits were required to detect three endangered arthropods known to occur in Texas caves. Although it is not directly applicable to terrestrial cave-dwelling invertebrates, Culver et al. (2004) reported that Sket (1981) discovered a new stygobite (an aquatic cave-adapted animal belonging to a new genus) after more than 100 collecting trips to a cave in Slovenia. During this study, I identified two potential troglobites and detected only one of five troglobites originally identified by Northup and Welbourn (1997). This not only underscores the inefficiency in our abilities to effectively detect cave-adapted animals but also emphasizes the need for additional inventory work in deep zone microhabitats.

ELMA-054 is home to a troglomorphic bristletail (order Diplura: family Campodeidae). It has been detected on the mud floors of a small chamber at the terminus of this cave. This animal may prove to be a narrow-range endemic (occurring in this cave and nowhere else on the planet). To best protect this animal and its habitat, in 2013 monument personnel permanently closed the deepest section of ELMA-054 to all recreational use.

Another important and highly sensitive microhabitat is moss gardens. These areas have been identified as relict habitats of the last glacial maximum (approximately 20,000 years ago) and support species now restricted to this environment at both El Malpais (Northup and Welbourn 1997) and in Oregon (Benedict 1979). Species richness for both ELMA-012 and ELMA-008 was driven by the large number of species detected within moss gardens at cave entrances and beneath skylights. Roughly 25% of the arthropods detected during the Northup and Welbourn (1997) study were found within moss gardens.

Because moss gardens are considered relict habitats and have been shown to support large numbers of species, this microhabitat should be afforded the highest level of protection. In 2013 ELMA-012 was closed to recreational use. Moss gardens within ELMA-008 have been roped off and signage has been posted indicating the fragility of these habitats. Based on my observations of both caves since 2005, this approach seems to be deterring foot traffic in these areas. However, some of the posts supporting the rope have fallen. More frequent maintenance of the posts and ropes, and adding more signage in ELMA-008, are relatively inexpensive measures that may serve to better protect these important microhabitats. Should ELMA-012 reopen in the future, I recommend using the same management and maintenance approach described for ELMA-008.

I did not detect any arthropods in ELMA-029 because I did not have an opportunity to sample the moss gardens in the cave entrance (as Northup and Welbourn [1997] did during their work). I observed no signs of recent human use or visitation when I was there in 2007. Given its remote location (approximately 1.6 km [1 mi] from an unmaintained dirt road), this cave and its moss gardens are likely well protected.

ELMA-029 also contains the most significant ice deposit on the monument, with a meters-thick ice sheet extending from near the entrance to the back of the cave. Cave interior and deep zone temperatures fluctuated from near to below freezing (mean temperature = 0.141°C [32.25 °F], standard deviation = 1.21°C [2.18°F]) during the arthropod sampling period. Although this cave is not suitable habitat for most arthropod species, it is possible that ice crawlers (order Notoptera: family Grylloblattidae) occur there and in other ice caves on private lands adjacent to the monument. These animals are known to occur in caves at both Oregon Caves and Lava Beds National Monuments (Jarvis and Whiting 2006) and would be a significant discovery at El Malpais. If ice crawlers exist within this cave, these animals would likely be relict species of the last glacial maximum. I suggest surveys for ice crawlers be conducted at ELMA-029, as well as at other ice caves in the El Malpais region.

Future directions

The information presented here provides a solid foundation on which to continue building knowledge of cave natural resources on the national monument, and has already proven useful in managing these resources. Additional studies targeting the use of lava tubes by cave-roosting bats, the distributional extent of known troglomorphic arthropods in caves or groups of caves, additional sampling for several of the new species discussed here, and further study of cave deep zones, root curtains, moss gardens, and cave ice sheets will be required to obtain the data necessary for optimal conservation and management of lava tube cave biological resources at El Malpais National Monument. My hope is that some of the protocols presented here and the recommendations made will be useful in the development and implementation of a monitoring framework that may be used to gauge the response of sensitive cave-dwelling animals to recreational use, invasive species, global climate change, and white-nose syndrome.

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About the author

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APPENDIX A IS AVAILABLE ONLINE AT HTTP://WWW.NATURE.NPS.GOV/PARKSCIENCE/ARCHIVE/PDF/ARTICLEPDFS/PARKSCIENCE30(1)SUMMER2013_A1–A12_WYNNE_3653.PDF

PS

Research Report

APPENDIX A

Annotated list of cave-dwelling taxa

By J. Judson Wynne

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Editor's note: The following is an online-only supplement to the research report "Inventory, conservation, and management of lava tube caves at El Malpais National Monument, New Mexico," by J. Judson Wynne. It can be cited as Wynne, J. J. 2013. Appendix A: Annotated list of cave-dwelling taxa. [Online supplement.] Park Science 30(1)Appendix A:1–12. Available online at http://www.nature.nps.gov/ParkScience/archive/PDF/Article_PDFs/ParkScience30(1)Summer2013_A1-A12_Wynne_3653.pdf.

Author's notes: In cases where members of a given morphospecies were detected only in entrances and twilight zones, I erred cautiously and referred to them as "eisodophiles." In cases where both the location of the detection and known information concerning the morphospecies supported the likelihood of an animal being "troglophillic," but I was still uncertain, I categorized the animal as a "questionable troglophile." Additionally, when a morphospecies was found only in the deep zone of a cave (or several individuals of a morphospecies occurred only within the deep zone) but troglomorphic characters were lacking, I also referred to it as "questionable troglophile."

THERE WERE SEVERAL CASES WHERE INDIVIDUALS EVADED CAPTURE BUT WERE BELIEVED TO represent a distinct arthropod morphospecies for a given cave. Because this information is of limited value in this article, arthropod morphospecies groups for which specimens are lacking were not included. However, this information has been integrated into a larger El Malpais morphospecies database and will be analyzed and the results reported in additional publications.

For arthropod groups actively being studied, I either sent specimens or high-resolution images of specimens to taxonomic specialists for identification or verification of my identifications. These experts include Rolf Aalbu, Department of Entomology, California Academy of Sciences, San Francisco, California (Coleoptera: Tenebrionidae); R. Thomas Allen, The Academy of Natural Sciences of Drexel University, Philadelphia, Pennsylvania (Diplura); Max Barclay, Natural History Museum, London (Coleoptera), and Thomas Barr (deceased), formerly with Department of Biology, University of Kentucky, Lexington, Kentucky (Coleoptera: Carabidae); Ernest Bernard, Department of Entomology, The University of Tennessee, Knoxville (Collembola); Jostein Kjaerandsen, Museum of Zoology, Lund University, Lund, Sweden (Diptera: Mycetophilidae); Sarah Oliveira, Department of Biology, University of São Paulo, Brazil (Diptera: Mycetophilidae); Theodore Cohn (deceased), formerly with Department of Zoology, San Diego State University, California (Orthoptera: Rhamphidophoridae); Lynn Kimsey, Department of Entomology, University of California, Davis (Hymenoptera: Tiphiinae); Robert Johnson, School of Life Sciences, Arizona State University, Tempe (Formicidae); Edward Mockford, Department of Biology, University of Illinois, Normal (Psocoptera); Glené Mynhardt, Department of Evolution, Ecology, and Organismal Biology, The Ohio State University, Columbus (Coleoptera: Ptinidae); Barry O'Connor, Department of Ecology and Evolutionary Biology, University of Michigan, Ann Arbor (Acari); Stewart Peck, Department of Biology, Carleton University, Ottawa, Ontario, Canada (Coleoptera: Leiodidae); Pierre Paquin, Cave and Endangered Invertebrate Research, SWCA Environmental Consultants, Austin, Texas (Araneae); William Shear, Department of Biology, Hampden-Sydney College, Hampden Sydney, Virginia (Myriapods and Opiliones); and Harald Schillhammer, Department of Entomology, Naturhistorische Museum, Vienna, Austria (Coleoptera: Staphylinidae). For all other specimens, Colorado Plateau Museum of Arthropod Biodiversity staff and I identified the specimens to the lowest taxonomic level possible using available taxonomic keys.

"Det." following each species or morphospecies designation is the abbreviation for the Latin *dēterminēvit* or "determined by."

Phylum Arthropoda

Class Arachnida

Order Araneae

Family Araneidae

***Metellina mimetoides* Chamberlin & Ivie, 1941**. Det. P. Paquin. Eisodophile.

One adult female was collected via timed search in the twilight zone of ELMA-262. Additionally, one juvenile specimen that may represent this species was collected via timed search at the entrance of ELMA-262.

Family Linyphiidae

Note: Numerous troglobitic and troglophillic forms of this family are known globally (e.g., Ruzicka 1998; Deltshev and Curcic 2002; Miller 2005).

Linyphiidae sp. Det. P. Paquin. Eisodophile.

One juvenile specimen was collected opportunistically near the entrance of ELMA-262. Another juvenile was collected via timed search in the twilight zone of ELMA-303.

***Lepthyphantes* sp**. Det. P. Paquin. Eisodophile.

Two female specimens were collected by timed searches at the entrance of ELMA-012; one female specimen was collected using direct intuitive searches in the moss gardens of ELMA-008.

***Porrhomma* sp. 1**. Det. P. Paquin. Troglophile?

One female specimen was collected using direct intuitive searches within root curtains in the deep zone of ELMA-315.

***Porrhomma* sp. 2**. Det. P. Paquin. Troglophile?

One female specimen was collected using direct intuitive searches within root curtains in the deep zone of ELMA-303. P. Paquin (personal communication, e-mail, 23 March 2007) suggests it differs from *Porrhomma* sp. 1.

Family Liocranidae

Liocranidae sp. Det. P. Paquin. Troglophile?

One juvenile specimen was collected via timed search in the deep zone of ELMA-012.

Family Nesticidae

Note: Nesticidae has an impressive cave fauna globally (Hedin 1997; Cokendolpher and Reddell 2001; Snowman et al. 2010).

Nesticidae sp. Det. P. Paquin. Eisodophile.

One juvenile specimen was collected opportunistically from the twilight zone of ELMA-012.

***Eidmanella pallida* (Emerton, 1875)**. Det. P. Paquin. Troglophile.

Three females were collected using direct intuitive searches within root curtains in the deep zone of ELMA-315. Two juvenile specimens (identified as Nesticidae sp.) were collected using direct intuitive searches within root curtains in the deep zone of this cave. While unconfirmed, these juveniles may also be *Eidmanella pallida*.

Note: Reddell and Cokendolpher (2004) consider this species a troglophile in Texas caves.

Family Pholcidae

Note: Both troglophillic and troglobitic forms of this family are known globally (e.g., Gertsch and Peck 1992; Deeleman-Reinhold 1993; Chen et al. 2011; Ferreira et al. 2011).

***Psilochorus* sp. 1**. Det. P. Paquin. Troglophile?

One male and two females were collected using direct intuitive searches within root curtains in the deep zone of ELMA-303. Three males were collected via timed searches (n = 2) and pitfall trapping (n = 1) at the entrance and beneath the skylights of ELMA-008. One female specimen was identified via timed search in the twilight zone, directly below the entrance of ELMA-315. Two males were collected by timed searches in both the entrance and deep zone of ELMA-012, and two individuals (1 male, 1 female) were collected via timed search from the entrance in Hummingbird Cave.

***Psilochorus* sp. 2**. Det. P. Paquin. Troglophile?

Two individuals were collected opportunistically in the deep zone near the bat maternity roost in ELMA-062. One adult female specimen was designated as a different species from *Psilochorus* sp. 1 (P. Paquin, personal communication, e-mail, 4 December 2009). Additionally, one juvenile specimen identified as *Psilochorus* was collected from the same cave. I suggest it is probably the same morphospecies because an adult female was identified in the same area as the juvenile specimen.

Family Theridiidae

***Achaearanea porteri* (Banks, 1896)**. Det. P. Paquin. Troglophile.

Two females were collected using timed searches, one near the entrance and the other in the twilight zone of ELMA-303. Three females were collected via direct intuitive searching (n = 2) in root curtains and with timed searches (n = 1) in the deep zone of ELMA-315.

Note: Cokendolpher and Reddell (2001) consider this species a troglophile in Texas caves.

***Nesticodes rufipes* (Lucas, 1846)**. Det. P. Paquin. Troglophile.

Three adult females were collected using direct intuitive searches in root curtains from the deep zone of ELMA-315.

Note: Because all were located within the same location and none had characters suggestive of troglomorphism, I consider this spider a troglophile. Additionally, theridiid spiders have been widely documented globally as being both troglophiles and troglobites (e.g., Ferreira and Martins 1998; Ruzicka 1998; Dippenaar-Schoeman and Myburgh 2009).

***Steatoda* sp**. Det. P. Paquin. Unknown.

One juvenile specimen was collected from a pitfall trap within the twilight zone of ELMA-062.

***Theridion* n.sp**.? Det. P. Paquin. Troglobite?

One adult female was collected via timed search in the twilight zone of ELMA-262. P. Paquin (personal communication, e-mail, 23 March 2007) suggests this may be a new species, and potentially has cave-adapted characteristics.

Order Opiliones

Family Sclerosomatidae

***Leiobunum townsendii* Weed, 1893**. Det. W. Shear. Trogloxene.

This harvestman (n = 13) was identified from ELMA-012, ELMA-062, ELMA-008, ELMA-262, ELMA-303, ELMA-315, and Hummingbird Cave. It was collected via direct intuitive search in the moss gardens of ELMA-008 and ELMA-012 and in root curtains in the deep zone of both ELMA-303 and ELMA-315. It was collected both opportunistically and via timed search in the entrances and twilight zones of ELMA-062 and Hummingbird Cave. W. Shear (personal communication, e-mail, 12 April 2009) suggests this group in western North America requires major revision. It is possible multiple species exist across the southwestern United States, or greater North America. However, until it is revised, the accepted name provided here will be used.

Subclass Acari

Order Sarcoptiformes

Family Histiostomatidae

***Histiostoma* n.sp**. Det. B. O'Connor. Troglophile?

Two deutonymphs were collected during timed searches in the deep zone of ELMA-315. B. O'Connor indicates this is an undescribed species. This animal is similar to *H. pierrestrinati* described from Carlsbad Cavern (B. O'Connor, personal communication, e-mail, 3 August 2012).

Order Trombidiformes

Family Bdellidae

Bdellidae sp. Det. B. O'Connor. Troglophile?

One specimen was collected by direct intuitive searches in the deep zone of ELMA-303. The palpi were damaged during collection, so lower-level taxonomic identification was not possible. B. O'Connor (personal communication, e-mail, 3 August 2012) indicates this family contains predators of soil, leaf litter, and littoral zones.

Family Erythraeidae

***Erythraeus* sp**.? Det. B. O'Connor. Eisodophile.

Five specimens were captured via pitfall trapping from the twilight zone of ELMA-008. B. O'Connor (personal communication, e-mail, 3 August 2012) indicates this genus is known from the Southwest, but no species are described. Additional analysis will be required to identify these specimens to a lower taxonomic level.

Family Rhagiididae

Rhagiididae sp. Det. B. O'Connor. Troglophile?

One specimen was collected by direct intuitive searches in the dark zone of ELMA-012. The specimen was damaged and could not be identified beyond family level.

Family Smarididae

***Phanolophus* sp**. Det. B. O'Connor. Unknown.

One specimen was collected via pitfall trapping at the entrance of ELMA-012. This family of predatory mites has not been studied in North America (B. O'Connor, personal communication, e-mail, 3 August 2012).

Subphylum Myriapoda

Class Chilopoda

Order Lithobiomorpha

Family Gosibiidae

Gosibiidae sp. Det. B. Shear. Troglophile?

One specimen was collected using direct intuitive searches from root curtains in the deep zone of ELMA-303. Additional specimens will be required to identify this centipede beyond the family level (W. Shear, personal communication, e-mail, 9 October 2009).

Class Diplopoda

Order Chordeumatida

Family Contylidae

***Austrotyla* sp**.? Det. W. Shear. Eisodophile.

This specimen (n = 1), identified to genus level by W. Shear, was collected via direct intuitive search from the moss gardens of ELMA-008. Additional specimens will be required to identify this animal to a lower taxonomic level.

***Austrotyla* cf *coloradensis* (Chamberlin, 1910)**. Det. W. Shear. Troglophile?

One specimen was collected using direct intuitive searches from root curtains in the deep zone of ELMA-315. This is a tentative species designation because of a lack of material. Additional specimens will be required to confirm this species designation.

Class Entognatha

Order Collembola

Note: The two new collembolan species will be included in a paper describing several new cave-dwelling Collembola species from the southwestern United States.

Family Entomobryidae

***Drepanura* n.sp**. Det. E. Bernard. Troglophile?

One specimen was collecting using pitfall trapping near the entrance of ELMA-008. E. Bernard (personal communication, e-mail, 15 July 2010) indicates this specimen represents a new species.

***Entomobrya guthriei* Mills, 1931**. Det. E. Bernard. Troglophile?

Five specimens were collected via pitfall trapping from the twilight zone to the deep cave zone of ELMA-110.

Entomobrya zona? Christiansen & Bellinger, 1980. Det. E. Bernard. Troglophile?

All specimens were collected in the entrances and twilight zones of ELMA-012 (n = 28) and ELMA-008 (n = 4). Seven specimens were collected using direct intuitive searches from moss gardens beneath the skylights of ELMA-012. All of the remaining specimens were captured using pitfall trapping. They likely represent *E. zona*. E. Bernard (personal communication, e-mail, 15 July 2010) made this tentative species designation, but indicated the specimens are not a "sure fit" for this species.

Family Tomoceridae

***Pogonognathellus* n.sp**. Det. E. Bernard. Eisodophile.

All specimens were collected via direct intuitive searches from the moss gardens of ELMA-008 (n = 10) and opportunistic collecting of ELMA-012 (n = 2). E. Bernard (personal communication, e-mail, 15 July 2010) suggests these specimens represent a new species.

Order Diplura

Family Campodeidae

Campodeidae n.sp. Det. J. Wynne and T. Allen. Troglobite.

This animal was first reported by Northup and Welbourn (1997). Five specimens were collected using direct intuitive searches from the "mud room" at the terminus of ELMA-054. Dipluran taxonomist Dr. Thomas Allen has these specimens and has confirmed this as a new species (personal communication, e-mail, 5 May 2013). I will be working with him to describe this new species.

Class Insecta

Order Coleoptera

Family Carabidae

***Rhadine* n.sp. *perlevis* species-group**. Det. T. Barr. Trogloxene.

These carabid beetles (n = 25) were identified primarily by pitfall trapping (but also with opportunistic collecting and timed searches) from ELMA-062, ELMA-110, ELMA-262, ELMA-303, and ELMA-315. This animal was observed from the twilight to deep zones of most caves. These specimens were initially sent to Dr. Thomas Barr for identification. T. Barr (personal communication, e-mail, 12 June 2009) suggested the specimens represent a new species and they belong to the *perlevis* species-group of *Rhadine*. Dr. Barr passed away in April 2011. The specimens are now at the Carnegie Museum of Natural History in Pittsburgh, Pennsylvania, and are awaiting formal description. Dr. Kipling Will, Essig Museum of Entomology, University of California, Berkeley, is coordinating this effort.

Family Cryptophagidae

Cryptophagidae sp. Det. M. Barclay. Eisodophile.

One specimen was collected via pitfall trapping from the entrance of ELMA-062. Additional work will be required to identify this specimen to a lower taxonomic level.

Family Leiodidae

***Dissochaetus arizonensis* Hatch, 1933.** Det. S. Peck. Accidental.

This leiodid beetle was collected from cave entrances of ELMA-012 (n = 1) and ELMA-062 (n = 1), while specimens from ELMA-315 (n = 2) were detected in the cave deep zone; all were captured using baited pitfall traps.

Note: S. Peck (personal communication, e-mail, 28 February 2013) suggests this species is an accidental because there are no data to suggest it is a regular cave dweller or that it reproduces in caves.

Family Melyridae

***Listrus* sp**. Det. M. Barclay. Eisodophile.

This coleopteran was captured via pitfall trapping (n = 1) in the twilight zone of ELMA-262. This specimen will require further study.

Family Ptinidae

***Niptus ventriculus* LeConte, 1859.** Det. G. Mynhardt. Troglophile.

Five spider beetle specimens were collected via pitfall trapping in ELMA-008 (n = 1), ELMA-012 (n = 2), and ELMA-262 (n = 1) and by opportunistic collecting in ELMA-062 (n = 1). Four specimens were collected in the cave entrances, while one specimen was collected in the twilight zone.

Note: Spilman (1968) documented this species in packrat middens, while Aalbu (2005) indicated larvae and potentially adults feed on the scat of packrats. Given that habitat exists for these spider beetles and that they complete a portion of their life cycle underground, I consider this animal a troglophile.

Family Staphylinidae

Staphylinidae sp. Det. J. Wynne. Eisodophile.

One individual was collected from the entrance of ELMA-012 during time searches. Additional work will be required to identify this specimen to a lower taxonomic level.

Subfamily Tachyporinae

***Sepedophilus* sp**. Det. H. Schillhammer. Eisodophile.

Three individuals were collected from the twilight zone of ELMA-062 (n = 2) and entrance of ELMA-315 (n = 1). Each specimen was detected using a different technique from the others: opportunistic collecting, timed searches, and pitfall traps. H. Schillhammer (personal communication, e-mail, 19 April 2013) suggests this genus is generally not associated with caves.

Family Tenebrionidae

***Neobaphion planipennis* (LeConte, 1866).** Det. R. Aalbu. Trogloxene.

Four individuals were collected opportunistically and via timed search from ELMA-062 (n = 3) and using direct intuitive searches in ELMA-303 (n = 1). In ELMA-062 this species was observed in the dark zone and beneath a skylight entrance; the individual in ELMA-303 was collected from the deep zone.

Note: Aalbu et al. (2012) consider this species an occasional trogloxene in ELMA-062.

Order Diptera

Family Culicidae

Culicidae sp. Det. J. Wynne. Trogloxene.

One culicid fly was collected opportunistically from the entrance of ELMA-012 and one via timed search in the deep zone of ELMA-315. Additional work will be required to identify this specimen to a lower taxonomic level.

Note: Reeves et al. (2000) and Makiya and Taguchi (1982) identified mosquitoes as trogloxenes.

Family Mycetophilidae

***Mycetophila* sp**. Det. J. Kjaerandsen and S. Oliveira. Trogloxene?

One specimen was collected using direct intuitive searches from the root curtains in the deep zone of ELMA-303. Additional work will be required to identify this specimen to a lower taxonomic level.

Note: Peck (1981) considered a morphospecies of this genus and five morphospecies of this family to be trogloxenes from two caves (>2,134 m [7,000 ft] elevation) in the Uinta Mountains, Utah. Additionally, from caves in Grand Canyon National Park, Peck (1980) considered a morphospecies of this genus to be a trogloxene.

Family Phoridae

Phoridae sp. Det. J. Wynne. Eisodophile.

Eight specimens were collected from pitfall traps at the entrance of ELMA-062 (n = 7) and in the twilight zone of ELMA-008 (n = 1). One individual was collected using direct intuitive searches in the moss gardens beneath skylights of ELMA-012. Additional work will be required to identify these specimens to a lower taxonomic level.

Family Sciaridae

Sciaridae sp. Det. J. Wynne. Eisodophile.

Twenty-one specimens were collected via opportunistic collecting, pitfall trapping, and timed searches from the entrance to the middle of ELMA-062; one specimen was collected using direct intuitive searches from the moss gardens beneath a skylight of ELMA-008; and three specimens were collected opportunistically from the entrance of ELMA-061. Additional work will be required to identify these specimens to a lower taxonomic level.

Order Hemiptera

Infraorder Fulgoromorpha

Superfamily Fulgoroidea

Fulgoroidea n.sp.? Det. J. Wynne. Troglobite?

Nymphal-stage planthoppers were collected using direct intuitive searches in root curtains from the deep zones of ELMA-303 and ELMA-315. Adults will be required to confirm troglomorphism, identify to a lower taxonomic level, and determine new species status.

Order Hymenoptera

Family Formicidae

***Liometopum* sp**. Det. R. Johnson. Eisodophile.

One undetermined *Liometopum* specimen was collected using direct intuitive searches in the moss gardens of ELMA-008.

***Pheidole* sp**. Det. R. Johnson. Eisodophile.

Two minor workers (R. Johnson, personal communication, e-mail, 10 December 2010) were collected via pitfall trapping near the entrance and at close proximity to the moss gardens of ELMA-008.

Family Tiphiidae

Note: All specimens of both tiphiid wasp species were found in a torpor beneath rocks; given the time of season, I suggest these individuals were in the early stages of hibernation and were likely using moss gardens as winter habitat.

***Tiphia andersoni* Allen, 1971**. Det. L. Kimsey. Eisodophile.

One female specimen was collected using direct intuitive searches in moss gardens (beneath large skylights) of both ELMA-012 and ELMA-008. Historically, this wasp is known to occur in central Mexico as well as southeastern and north-central Arizona (Allen 1971). This animal was not known to occur in New Mexico and thus represents a range expansion.

***Tiphia nona* Allen, 1965.** Det. L. Kimsey. Eisodophile.

One female specimen was collected using direct intuitive searches in the moss gardens of ELMA-008. Previously it was known from central Mexico, southeastern Arizona to the southern extent of the Mogollon Rim, and one locality in southwestern Kansas (Allen 1971). This animal was not known to occur in New Mexico and thus represents a range expansion.

Order Lepidoptera

Note: None of the larval specimens were reared in the lab and I was unable to locate a key for Lepidoptera larvae. Thus, all lepidopteran specimens have been sorted into operational taxonomic units, and further identifications were not possible before this article was published. This level of identification is acceptable for community-level as well as other analyses, which will be the subject of additional scientific publications.

Lepidoptera sp. 1. Det. J. Wynne. Troglophile?
Three larval specimens were collected with pitfall traps (n = 2) and via direct intuitive searches (n = 1) from the root curtains within the deep zone of ELMA-315.

Lepidoptera sp. 2. Det. J. Wynne. Troglophile?
Four larval specimens were collected using direct intuitive searches of the root curtains within the deep zone of ELMA-315 (n = 3) and ELMA-303 (n = 1).

Lepidoptera sp. 3. Det. J. Wynne. Troglophile?
One larval specimen was collected using direct intuitive searches of the root curtains within the deep zone of ELMA-315.

Lepidoptera sp. 4. Det. J. Wynne. Troglophile?
One larval specimen was collected using direct intuitive searches of the root curtains within the deep zone of ELMA-315.

Lepidoptera sp. 5. Det. J. Wynne. Troglophile?
One larval specimen was collected using direct intuitive searches of the root curtains within the deep zone of ELMA-303.

Lepidoptera sp. 6. Det. J. Wynne. Eisodophile.
One adult moth was collected during a timed search in the entrance of ELMA-262.

Lepidoptera sp. 7. Det. J. Wynne. Eisodophile.
One adult moth (different from Lepidoptera sp. 6) was collected during a timed search in the entrance of ELMA-012.

Family Tenididae

Tenididae sp. 1. Det. J. Wynne. Eisodophile.
One micro-lepidopteran was collected opportunistically in ELMA-262.

Tenididae sp. 2. Det. J. Wynne. Eisodophile.
One micro-lepidopteran (different from *Tenididae* sp. 1) was found in a pitfall trap in the twilight zone of ELMA-008.

Order Orthoptera

Family Rhaphidophoridae

***Ceuthophilus* sp**. Det. T. Cohn. Trogloxene.

One juvenile male was captured via pitfall trapping from the entrance of ELMA-010. Given this animal's immature state, it was not possible to identify it to a lower taxonomic level.

***Ceuthophilus* cf *apache* n.sp**. Det. T. Cohn. Trogloxene.

T. Cohn (personal communication, e-mail, 21 March 2011) indicated this was a new *Ceuthophilus* species, which is similar to *Ceuthophilus* cf *apache*. We collected one adult male and one adult female from ELMA-062, two adult males from ELMA-303, and one adult male from ELMA-315. This morphospecies was detected using opportunistic collecting, pitfall trapping and timed searches, and occurred from the entrances to each cave's dark/deep zone.

***Ceuthophilus* (*Geotettix*) *polingi* Hubbell, 1936**. Det. T. Cohn. Trogloxene.

T. Cohn and A. Swanson identified all specimens in this group. We collected two adult females and four adult males from ELMA-262, one adult male from Hummingbird Cave, one adult male from ELMA-012, one adult male from ELMA-054, one adult female and two adult males from ELMA-303, and two adult females from ELMA-315. This species was detected using opportunistic collecting, pitfall trapping, and timed searching, and occurred from the entrances to each cave's dark/deep zone. T. Cohn (personal communication, e-mail, 21 March 2011) suggested this animal was considered rare until recently; we now know it is widespread in its range, but probably restricted to caves and animal burrows.

Order Psocoptera

Family Psyllipsocidae

***Psyllipsocus ramburii* Selys Longchamps, 1872**. Det. E. Mockford. Troglophile.

This species was identified from ELMA-062 (n = 2), ELMA-262 (n = 1), and ELMA-315 (n = 6). With the exception of one individual collected opportunistically, all were detected in pitfall traps and from cave entrances to the dark/deep zones.

Note: This species is known to occur in caves globally (E. Mockford, e-mail, 1 February 2013). E. Mockford and I (unpublished data) recently confirmed this species on Easter Island, South Pacific Ocean, as well as from a cave on Grand Canyon–Parashant National Monument, Arizona.

Order Siphonaptera

Family Pulicidae

Pulicidae sp. Det. J. Wynne. Parasite.

Nine specimens were collected from ELMA-315. I found no evidence of recent rodent activity within either cave. However, the presence of fleas suggests recent vertebrate use. Additional work will be required to identify these specimens to a lower taxonomic level.

Phylum Chordata

Subphylum Vertebrata

Class Reptilia

Family Colubridae

***Pituophis catenifer* (Blainville, 1835)**. Det. J. Wynne. Unknown.

A gopher snake carcass was found in the twilight zone of ELMA-061. This individual had numerous lacerations along the length of its body. A park visitor probably killed the snake. Because I am uncertain whether the snake was killed in the cave or brought into the cave postmortem, its functional group status is "unknown."

Class Mammalia

Order Chiroptera

Family Vespertilionidae

***Corynorhinus townsendii* Cooper, 1837**. Det. J. Wynne. Trogloxene.

This bat has been documented hibernating in ELMA-054 since 2005 (Wynne 2006). A maternity roost exists at ELMA-110. This maternity roost has been documented both in the tunnel section prior to the main section of the cave and in the twilight zone of the cave's main section.

***Eptesicus fuscus* (Palisot de Beauvois, 1796)**. Det. J. Wynne. Trogloxene.

One torpid big brown bat was observed near the entrance of ELMA-054.

Family Molossidae

***Tadarida brasiliensis* (I. Geoffroy, 1824)**. Det. J. Wynne. Trogloxene.

A long-established maternity roost of Mexican free-tailed bats exists in ELMA-062. We observed bats in residence during the October 2007 work.

Order Rodentia

Family Muridae

***Neotoma* sp**. Det. J. Wynne. Trogloxene.

Evidence of *Neotoma* sp. was documented at both ELMA-062 and ELMA-061. Both *N. mexicana* and *N. albigula* have been confirmed on the monument (Bogan et al. 2007). Either or both of these species likely use these caves.

Order Carnivora

Unknown family, genus, and species. Xenosylle?

Small carnivore scat was observed at the entrance of ELMA-054 and in the twilight zone of ELMA-110. Because we neither observed small carnivores nor saw them hunting bats within either cave, "questionable xenosylle" is most appropriate.

Class Aves

Order Tytonidae

***Tyto alba* (Scopoli, 1769)**. Det. J. Wynne. Eisodophile.

A barn owl was spooked as the team entered ELMA-262. The animal was observed within the main entrance and flew deeper into the cave toward the next collapse pit entrance, where it exited the cave.

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PS

Landscape conservation forecasting™ for Great Basin National Park

By Louis Provencher, Tanya Anderson, Greg Low, Bryan Hamilton, Tod Williams, and Ben Roberts

Figure 1. This view of the North Fork Bigwash Abyss comprises the Limber Bristlecone Pine and Aspen Mixed Conifer biophysical settings in the mid to late succession stage. These vegetation classes are common in the southern half of the park. Both settings represent moderate departures from ecological reference conditions. Our analyses indicate the Aspen Mixed Conifer vegetation class would benefit from treatments to thin the conifer component.

Abstract

Great Basin National Park and The Nature Conservancy collaboratively mapped the park's biophysical settings (potential natural communities), calculated ecological departure from reference conditions (pre-European settlement), and simulated management actions to restore and prevent future degradation of natural communities. Among 21 mapped biophysical settings, 9 were slightly departed from reference conditions, 10 were moderately departed, and only 2 smaller biophysical settings were highly departed. The primary causes of ecological departure were lack of the earliest succession classes, overrepresentation by late succession classes, conifer encroachment in shrublands, and invasion of nonnative cheatgrass. Fifty-year simulations with no active management revealed that 10 of 22 biophysical settings required active management. Two active management scenarios were simulated for 50 years: maximum management causing reduction of ecological departure without fiscal budget constraints and preferred management causing reduction of ecological departure within the constraints of a realistic budget. Simulations of cost-effective management actions achieved lower ecological departure for all 10 focal biophysical settings at a total cost of approximately $3.6 million over 50 years. Many actions were implemented fully in the first years of simulation. The aspen–subalpine conifer and limber-bristlecone pine–mesic biophysical settings achieved the greatest improvement relative to dollars invested.

Key words

biophysical setting, desired future condition, ecological departure, Fire Regime Condition Class (FRCC), management scenarios, remote sensing, return on investment, state-and-transition modeling, treatment areas

THE FUNDAMENTAL PURPOSE OF THE NATIONAL Park Service is to maintain, conserve, and restore park resources and the processes that sustain them, to the greatest extent practicable, to a condition minimally influenced by human actions, and to provide for visitor enjoyment of the same. Great Basin National Park was established to preserve unimpaired a representative segment of the Great Basin (described below). The park is to be managed so as to maintain the greatest degree of biological diversity and ecosystem integrity.

A century of fire exclusion combined with livestock grazing, nonnative plant species invasion, and stream diversions has resulted in large-scale conversion of many native vegetative ecosystems across the Great Basin and in the national park (Blackburn and Tueller 1970; Pyne 2004). Although livestock grazing permits were progressively retired and most water diversions were eliminated and restored since the park's creation in 1986, nonnative annual grass invasion persists at lower elevations, whereas fire-sensitive conifers dominate in sagebrush shrublands and aspen forests as a legacy of fire exclusion. Occurrence of the majority of wildlife classified as species of management concern relates to habitat loss and degradation by conifer encroachment historically caused by fire exclusion, altered flows from former water diversions, and livestock grazing. Maintenance, protection, and restoration of sagebrush, aspen, riparian, and ponderosa pine vegetation community types and their dependent wildlife populations are of high priority for park management (fig. 1, previous pages, and fig. 2, right).

Identification of desired future conditions, which can be difficult for parks to define, and scenario modeling to achieve them are important steps in implementing restoration actions. Desired future conditions may be based on enabling legislation but more often are derived from some conceptualization of "pre-Columbian" condition. This conceptualization is often biased and difficult to quantify at a landscape scale.

In 2005 natural resources staff mapped the Fire Regime Condition Class (FRCC) of all the park's major vegetation types using existing data, soil surveys, and GIS (geographic information system) software. Fire Regime Condition Class is a categorical measure of ecological departure from the reference condition (Hann and Bunnell 2001). Our initial work quantified desired future conditions in conjunction with fuels planning, complied with federal fire management guidelines, and produced a science-based fire management plan.

At this time, The Nature Conservancy (TNC) approached the park to collaboratively achieve the following goals: (1) map more accurately the park's vegetation based on field-interpreted high-resolution satellite imagery; (2) develop computer models of all major potential natural communities that would allow characterization of their reference conditions (i.e., the pre-European settlement condition of the landscape); (3) evaluate current and projected future departure from reference conditions, including FRCC; and (4) simulate cost-effective management scenarios that would reduce future departure from reference conditions. We highlight mapping and management results from this collaborative effort.

Figure 2. This aspen riparian vegetation represents the Montane Riparian biophysical setting and occurs in the park's drainages with perennial streams. Typical dominant woody species are aspen, cottonwood, willows, white fir, and wood rose. Overall this system exhibits low ecological departure, but the analysis found numerous areas in the park with higher ecological departure values because of conifer encroachment.

Study methods

The national park is located within the central Great Basin physiographic region of alternating north-south–trending mountains and valleys on the southern Snake Range in eastern Nevada close to the Utah border (Wheeler Peak's 3,982 m [13,063 ft] summit: 38° 59′ 9″ N; 114° 18′ 50″ W). The park is about 31,161 ha (77,000 ac) in size, most of which is above the 30 cm (12 in) precipitation zone, with large expanses above 3,048 m (10,000 ft) in elevation. It encompasses a wide diversity of Great Basin ecological systems, ranging from desert upland shrublands to subalpine bristlecone pines to alpine terrain. Because the park is relatively far from the rain shadow of the Sierra Nevada and relatively close to humid atmospheric circulation from the Gulf of California, precipitation patterns combine with topographic relief and metamorphic and carbonate geology to produce an abundance of plant communities, wildlife, and aquatic habitat types from both the eastern and western Great Basin.

We implemented TNC's Landscape Conservation Forecasting™ methodology (fig. 3; Low et al. 2010), which combines remote sensing of potential natural communities (hereafter, biophysical setting), calculation of ecological departure (also known as Fire Regime Condition), computer simulations to forecast future condition of biophysical settings under minimum alternative management scenarios, and calculation of return-on-investment analysis to assess which strategies would most efficiently return park

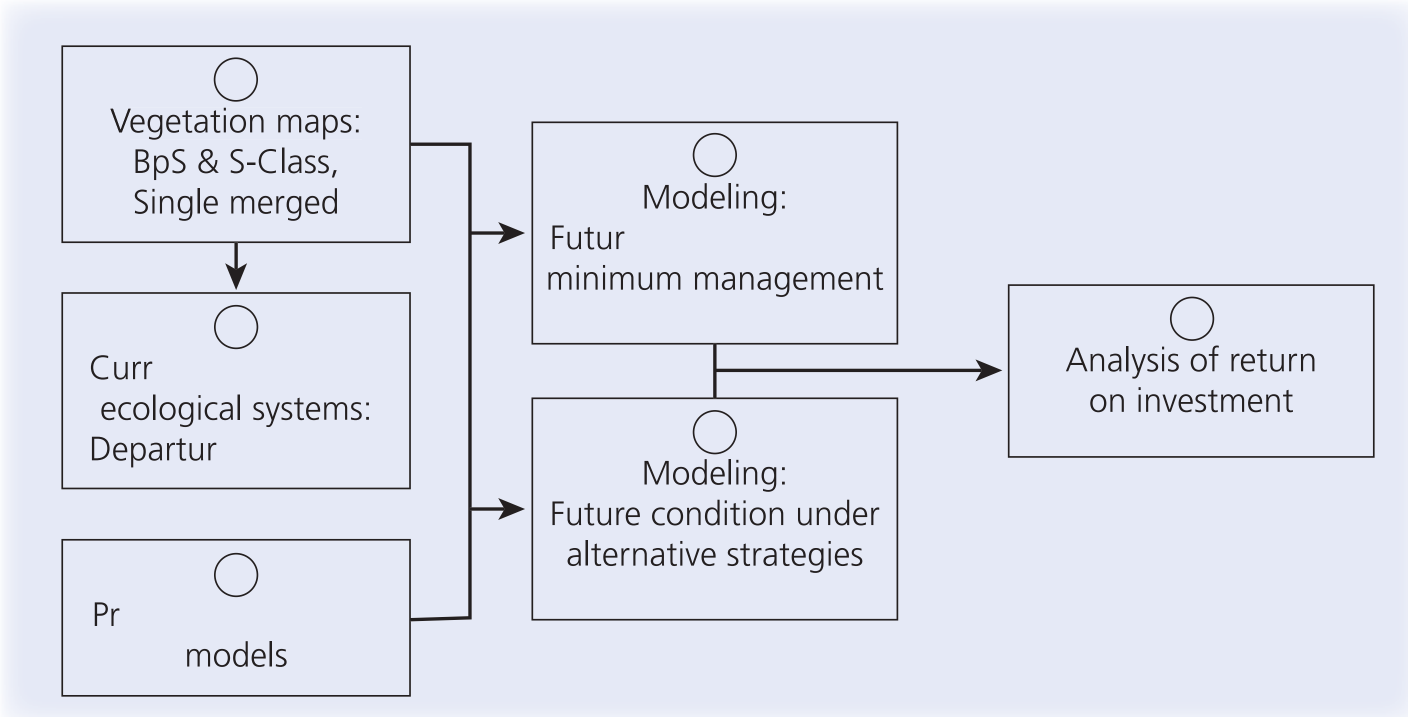

Figure 3. Diagram of the Landscape Conservation Forecasting™ method. This method was formerly called Enhanced-Conservation Action Planning in Low et al. (2010). BpS = biophysical setting, S-Class = succession class, NRV = natural range of variability.

ecosystems to reference conditions. Return-on-investment analysis was designed only to inform, but not finalize, decision making because managers might choose different priorities dictated by regulatory factors, public perception, and funding opportunities.

Remote sensing was completed using the software Imagine® from Leica Geosystems to conduct an unsupervised classification of QuickBird imagery (pixels are 2.4 m [7.9 ft] multispectral imagery) captured in 2007. The unsupervised classification of satellite imagery is described in Provencher et al. (2008) and Low et al. (2010) and is summarized here. A first unsupervised classification of the imagery was conducted to identify unique spectral classes prior to field surveys. To support interpretation of spectral classes (Lillesand and Kiefer 2000), we conducted an initial field survey to establish training plots (59) and to record rapid driving and hiking observations (around 2,000, including two photographs per observation) in July 2009. At a minimum, only the identity of the biophysical setting and vegetation class was needed at a location, although simple ground-cover and landform observations were also recorded. After the first field survey, iterative unsupervised classifications with manual editing were used to create draft maps of biophysical settings and vegetation classes, which were verified and improved during a second field trip in October 2009. Nearly every sector of the project area was visited by truck or on foot to interpret unique spectral classes. A final map of current vegetation classes was used to calculate ecological departure scores.

One state-and-transition computer model was developed for each biophysical setting using the Vegetation Dynamics Development Tool (VDDT, by ESSA Technologies, Ltd.; Beukema et al. 2003) to estimate the reference condition for ecological departure calculation and to simulate the effect of proposed management actions. A state-and-transition model is a discrete, box-and-arrow representation of the continuous variation in vegetation composition and structure of an ecological system (Rumpff et al. 2010). Different boxes in the model belong either to different states or to different phases within a state. States are formally defined in rangeland literature (Rumpff et al. 2010) as persistent vegetation and soils per potential ecological sites that can be represented in a diagram with two or more boxes (phases of the same state). Different states are separated by "thresholds." A threshold implies that substantial management action would be required to restore ecosystem structure and function. Relatively reversible changes (e.g., fire, flooding, drought, insect outbreaks, and others), unlike thresholds, operate between phases within a state. Models are typically represented by succession classes, ranging from herbaceous vegetation to increasing woody species dominance where the dominant woody vegetation might be shrubs or trees. We included uncharacteristic vegetation classes (classes outside the reference condition) and simulated future conditions under alternative management strategies and scenarios (Low et al. 2010). State-and-transition models included each vegetation class's responses to natural and uncharacteristic disturbances, and multiple pathways and success or failure rates associated with various management actions.

Using local and regional data, temporal variability in fire activity, climate, insect and disease outbreaks, nonnative plant and native tree species invasions, and stream discharge were imported in simulations as annual time series that modified (suppressed or increased) the base parameters in each biophysical setting's model (Low et al. 2010; Provencher et al. 2010). Five replicates of each time series were imported into the VDDT database. For fire, each replicate was created by resampling 75 yearly values from a standardized time series of area burned (i.e., the annual value divided by the temporal average) obtained from 1980 to 2008 for the park and from four adjacent mountain landscapes of the same size as the park. The historical time series of the Palmer Drought Severity Index was similarly resampled five times to generate five replicates representing the variability for mortality caused by drought and insect outbreaks and for nonnative annual grass and native tree invasion. Historical discharge data from Lamoille Canyon in the Ruby Mountains were used to generate temporal variability for 7-, 20-, and 100-year flood events. Often, the introduced variability was high, especially for fire activity.

Two landscape-scale metrics were used to summarize the ecological condition of each biophysical setting: ecological departure and high-risk vegetation classes. Ecological departure incorporates species composition, vegetation structure, and disturbance regimes to estimate a biophysical setting's departure from its natural range of variability (also known as reference condition and Historic Range of Variation) over the whole landscape. The natural range of variability is the relative amount (%) of each vegetation class in a landscape expected to occur in a biophysical setting under natural disturbance regimes and actual climate. The natural range of variability is determined by simulating a natural disturbance regime for each biophysical setting until the proportion of each reference vegetation class reaches equilibrium, or 1,000 years if equilibrium is not reached (Hann and Bunnell 2001; Provencher et al. 2008). In this project, ecological departure was based only on vegetation responses to reference and current disturbance regimes and not on departure from historical fire regimes. The park also has a paucity of data on historical fire regimes.[1] Ecological departure is scored on a scale of 0–100%, where 0% represents the natural range of variability and 100% represents total departure. Further, Fire Regime Condition Class is used by federal agencies to group ecological departure scores into three classes: FRCC 1 represents biophysical settings with low (<34%) departure; FRCC 2 indicates moderate (34–66%) departure; and FRCC 3 indicates high (>66%) departure (Hann and Bunnell 2001).

[1]Technically, ecological departure is a measure of dissimilarity between the natural range of variability and the current vegetation class distribution obtained from remote sensing (Provencher et al. 2008).

Table 1. Descriptions of management scenarios for Great Basin National Park

Management Scenarios
Minimum Management
A control scenario that included only natural disturbances, unmanaged nonnative species invasion, and fire suppression management. Fire suppression by agencies was simulated by reducing natural, reference fire return intervals using time series that reflected current fire events from the immediate and nearby areas. Fire event data were obtained from the Federal Fire Occurrence Web site. In essence, this scenario can be considered a no-treatment control, but does not represent current management.
Maximum Management
This scenario allocated unlimited management funds with the goal of reducing ecological departure and high-risk vegetation classes to the greatest extent possible. Management strategies were applied in an attempt to reduce ecological departure significantly or to maintain high-risk vegetation classes below 10% of the area of the biophysical setting. This scenario assumed no financial or other resource constraints on strategy implementation (i.e., annual agency budgets were typically exceeded).
Preferred Management
The preferred management scenario was the result of management strategies identified by stakeholders. It was usually effective at reducing ecological departure and high-risk vegetation classes while recognizing anticipated agency budgets, management funding availability, and regulatory constraints. Strategies that produced the highest return on investment were sought.

Ecological departure assumes that all uncharacteristic classes are equal to managers. We developed a separate designation and calculation of high-risk vegetation classes because it is possible to reduce ecological departure while increasing the percentage of undesirable classes (for example, nonnative annual grassland). A high-risk class was defined as an uncharacteristic vegetation class that met at least two of the following three criteria: (1) ≥5% cover of invasive nonnative species, (2) expensive to restore, and (3) a direct pathway to one of these classes (invaded or very expensive to restore). Park staff modified the definition of high-risk class to include the loss of aspen clones to other biophysical settings, such as mixed conifer and montane sagebrush steppe, which is vegetation conversion (Debyle et al. 1987; Mueggler 1988; Kay 1997, 2001; Bartos and Campbell 1998).

Using computer-based models, the likely future condition of the focal biophysical setting was assessed after 50 years under three primary scenarios to achieve these objectives: Minimum Management, Maximum Management, and Preferred Management (table 1). We used an ecological return-on-investment metric to determine which of the scenarios (Maximum or Preferred) produced the greatest ecological benefits per dollar invested across

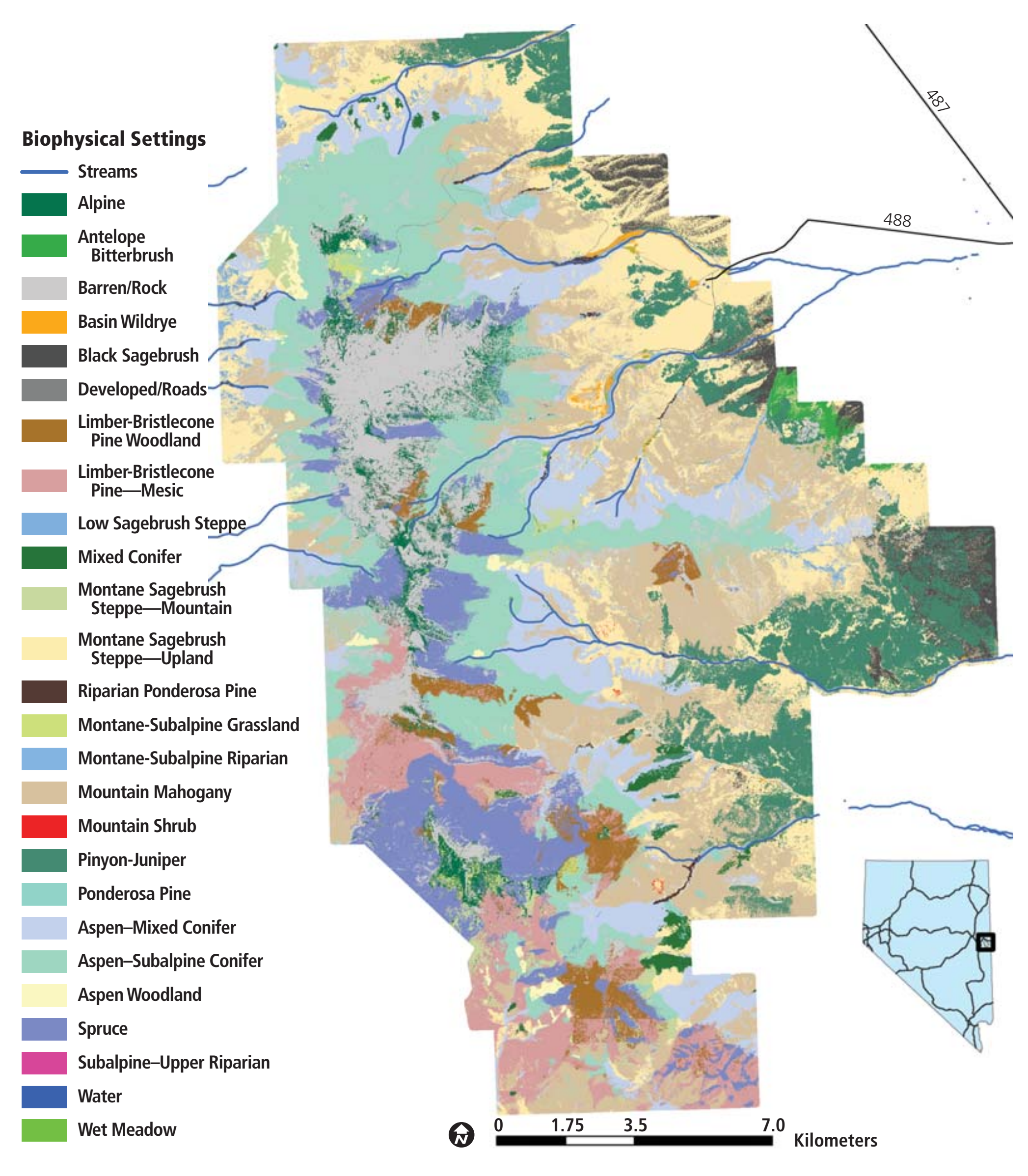

Figure 4. Map of Great Basin National Park's potential natural communities, also called biophysical settings.

SOURCE: THE NATURE CONSERVANCY

or Preferred Management scenario in the last year of the simulation, multiplied by total area of the biophysical setting, divided respectively by total cost of each scenario over the duration of the simulation (here 50 years; Low et al. 2010). Inflation was not included in the calculation of cost; however, this would probably not have affected the relative comparison of return-on-investment values among scenarios, as inflation would have applied equally to all management activities.

Findings

We mapped 21 biophysical settings (fig. 4). The natural range of variability of each biophysical setting is presented in table 2 (next page) based on simulations of presettlement conditions. The number of reference vegetation classes varies with the complexity of the biophysical setting.

Nine biophysical settings were slightly departed from the natural range of variability, 10 were moderately departed, and only 2 smaller biophysical settings were highly departed (table 3, next page). The primary cause of ecological departure was sagebrush biophysical settings that lacked the earliest succession classes and aspen-conifer biophysical settings that were overrepresented by late succession classes. Two small biophysical settings (antelope bitterbrush and basin wildrye) were highly departed primarily because of the presence of conifer encroachment and nonnative cheatgrass.

Twelve biophysical settings were not targeted for active management in the park because they were projected to benefit from periodic wildfires embedded in the computer simulations

multiple scenarios within each biophysical setting, and across targeted biophysical settings in relation to Minimum Management. We first created this metric for a western Nevada project with the Bureau of Land Management (Low et al. 2010) and have since used it in about 10 landscapes in the western United States and once for eastern Tennessee's Cherokee National Forest. The ecological systemwide return-on-investment metric is the sum of change in ecological departure, high-risk vegetation classes (if applicable), and vegetation conversion classes (if applicable) between the Minimum Management scenario and the Maximum

Table 2. Natural range of variability (NRV) for biophysical settings of Great Basin National Park

Biophysical Setting	NRV (%)				
	A[1]	B	C	D	E
Alpine	1	99			
Antelope Bitterbrush	21	44	21	7	7
Aspen Woodland	16	41	33	10	
Aspen–Mixed Conifer	19	43	24	9	5
Aspen–Subalpine Conifer	12	33	47	8	
Basin Wildrye	18	63	19		
Black Sagebrush	17	47	24	10	2
Limber-Bristlecone Pine	9	12	78		
Limber-Bristlecone Pine—moist	17	47	36		
Low Sagebrush Steppe	25	56	19		
Mixed Conifer	11	19	24	23	23
Montane Riparian	21	36	43		
Montane Sagebrush Steppe—mountain	21	44	22	10	3
Montane Sagebrush Steppe—upland	21	44	22	10	3
Montane-Subalpine Grassland	4	30	66		
Mountain Mahogany	8	13	15	23	41
Mountain Shrub	7	23	41	29	
Pinyon-Juniper Woodland	2	6	26	65	
Ponderosa Pine	11	2	29	57	1
Riparian Ponderosa Pine	26	9	47	17	1
Spruce	18	36	2	43	
Subalpine Riparian	13	58	29		
Wet Meadow	5	38	58		

Note: By definition, the "Uncharacteristic" classes are equal to zero (not shown).

[1]Standard LANDFIRE coding for the five-box vegetation model: A = early-development; B = mid-development, closed; C = mid-development, open; D = late-development, open; and E = late-development, closed. This terminology was often modified.

Table 3. Ecological departure and percentage of high-risk classes of Great Basin National Park's biophysical settings

Biophysical Setting	Area		Ecological Departure (%)	High-Risk Classes (%)
	ac	ha		
Alpine-Subalpine				
Alpine	1,690	684	0.1	0
Aspen–Subalpine Conifer	11,320	4,581	60	7
Limber-Bristlecone Pine Woodland	1,991	482	16	0
Limber-Bristlecone Pine Woodland—mesic	4,500	1,821	48	0
Montane-Subalpine Grassland	269	109	16	0
Spruce	5,770	2,335	36	0
Mid-Elevation Forests				
Aspen Woodland	571	231	27	16
Aspen–Mixed Conifer	8,110	3,282	66	6
Mixed Conifer	591	239	32	0
Ponderosa Pine	250	101	54	0
Shrublands				
Antelope Bitterbrush	341	138	74	28
Basin Wildrye	269	109	68	43
Black Sagebrush	1,880	761	60	39
Low Sagebrush Steppe	420	170	61	0
Montane Sagebrush Steppe—mountain	939	380	30	2
Montane Sagebrush Steppe—upland	12,711	5,144	56	21
Mountain Mahogany	14,050	5,686	23	0
Pinyon-Juniper	6,951	2,813	11	10
Riparian and Wet Meadows				
Montane Riparian	450	182	26	3
Riparian Ponderosa Pine	171	69	34	0
Wet Meadow	89	36	49	0

Note: Ecological departure scores were classified as good (0–33%, Class 1, green), fair (34–66%, Class 2, orange), and poor (>66%, Class 3, red). Levels of high-risk classes by ecological systems were ranked as low (0%, dark green), medium (1–10%, light green), high (11–30%, orange), and very high (>30%, red).

(table 4, page 63). These natural communities included curl-leaf mountain mahogany, pinyon-juniper woodland, spruce, limber-bristlecone pine, montane sagebrush steppe–subalpine sites, mixed conifer, aspen woodland, montane-subalpine grassland, ponderosa pine, riparian ponderosa pine, mountain shrub, and subalpine riparian. The subalpine riparian biophysical setting was very small and further subsumed within the montane riparian biophysical setting.

Ten biophysical settings with high departure from reference conditions were chosen for 50-year simulations of specific management actions within budgetary constraints (table 4). Key ecological issues included the following:

- Sagebrush biophysical settings (montane sagebrush–upland sites, black sagebrush, low sagebrush steppe, antelope bitterbrush, basin wildrye): lack of early succession classes, pinyon-juniper encroachment, and prediction of increased cheatgrass cover.
- Aspen-conifer biophysical settings (aspen–subalpine conifer and aspen–mixed conifer): high percentage of conversion to conifers and permanent loss of aspen clones.
- Riparian, wet meadow, and basin wildrye systems: invasion by exotic forbs.

Table 4. Current and predicted future (under minimum management) ecological departure and high-risk vegetation classes of ecological systems of Great Basin National Park

	Ecological Departure		High-Risk Classes	
Biophysical Setting	**Current Condition**	**Minimum Management (50 years)**	**Current Condition**	**Minimum Management (50 years)**
Alpine-Subalpine				
Alpine	0	1	0	0
Aspen-Subalpine Conifer	60	27	7	20
Limber-Bristlecone Pine Woodland	16	17	0	0
Limber-Bristlecone Pine—mesic	48	42	0	0
Montane-Subalpine Grassland	16	4	0	0
Spruce	36	23	0	0
Mid-Elevation Forests				
Aspen Woodland	27	10	16	11
Aspen–Mixed Conifer	66	33	6	12
Mixed Conifer	32	10	0	0
Ponderosa Pine	54	25	0	5
Shrublands				
Antelope Bitterbrush	74	62	28	44
Basin Wildrye	68	70	43	64
Black Sagebrush	60	55	39	40
Low Sagebrush Steppe	61	27	0	1
Montane Sagebrush Steppe—mountain	30	8	2	2
Montane Sagebrush Steppe—upland	56	41	21	30
Mountain Mahogany	23	19	0	4
Pinyon-Juniper	11	16	10	14
Riparian and Wet Meadows				
Montane Riparian	26	40	3	36
Riparian Ponderosa Pine	34	31	0	0
Wet Meadow	49	40	0	36

Note: Systems in boldface type were selected for active management analyses. Ecological departure scores were classified as good (0–33%, Class 1, green), fair (34–66%, Class 2, yellow), and poor (>66%, Class 3, red). Levels of high-risk classes to ecological systems were ranked as low (0%, dark green), medium (1–10%, light green), high (11–30%, yellow), and very high (>30%, red).

- Mesic[2] limber-bristlecone pine: high percentage of late succession classes at the expense of mostly mid-succession forests.

A variety of strategies were modeled. Multiple strategies were required for most biophysical settings, except aspen systems, in the Preferred Management scenario (table 5, page 64):

- Sagebrush management strategies included prescribed fire to restore early succession classes; chainsaw lopping of encroached conifer trees; chainsaw thinning of late succession classes of tree-encroached sagebrush, variously combined with chipping, mastication, pile burning, herbicide, and seeding of native species; and varied applications of herbicide and native seeding to uncharacteristic vegetation classes.
- Aspen-conifer management strategies included prescribed fire to prevent transition to conifers and loss of aspen clones (for example, fig. 5, page 66).
- The mesic limber-bristlecone pine forest management strategy included prescribed fire to reduce the area of late succession classes and increase that of early and mid-succession classes (fig. 5).

[2]Unlike ancient (>2,000 years old) limber and bristlecone pines growing on bedrock and skeletal soils, mesic forests show soil depth, soil moisture, and closed canopies, and generally do not exceed 1,000 years of age mainly because of death caused by heart rot.

Table 5. Treatment details (area and duration) for the preferred management scenario strategy for managed biophysical settings in Great Basin National Park

	Biophysical Setting*									
Treatment	**AB**	**AMC**	**ASC**	**BS**	**BW**	**LBm**	**LSS**	**MR**	**MSu**	**WM**
Chainsaw lopping	3.2 ha (7.9 ac) 5 yr			4.0 ha (9.9 ac) 3 yr					20.2 ha (49.9 ac) 50 yr	
Chainsaw thinning	1.6 ha (4 ac) 5 yr									
Exotic control					1.2 ha (3.0 ac) 50 yr			2.4 ha (5.9 ac) 50 yr		0.4 ha (1.0 ac) 50 yr
Herbicide	4.9 ha (12.1 ac) 5 yr								20.2 ha (49.9 ac) 50 yr	
Masticate + herbicide + seed				40.5 ha (100.1 ac) 3 yr	3.2 ha (7.9 ac) 3 yr					
Prescribed fire	1.6 ha (4 ac) 5 yr	131.5 ha (324.9 ac) 10 yr	1,194 ha (2,950 ac) 3 yr	70.8 ha (175.0 ac) 3 yr	1.2 ha (3.0 ac) 3 yr	224.6 ha (555.0 ac) 3 yr	16.2 ha (40.0 ac) 3 yr			
"Free" prescribed fire[1]			368.3 ha (910.1 ac) 3 yr			78.9 ha (195.0 ac) 3 yr				
Seed				4.0 ha (9.9 ac) 3 yr						
Spot herbicide + seed for SA/DP/AG	2.0 ha (4.9 ac) 5 yr									
Thin + seed					4.5 ha (11.1 ac) 3 yr					
Thin + spot herbicide + seed	3.2 ha (7.9 ac) 5 yr			40.5 ha (100.1 ac) 3 yr	0.8 ha (2.0 ac) 3 yr				20.2 ha (49.9 ac) 50 yr	
Weed Inventory					4.0 ha (10 ac) 50 yr			3.2 ha (8 ac) 50yr		1.62 ha (4 ac) 50 yr

**Biophysical settings:* AB = Antelope Bitterbrush, AMC = Aspen–Mixed Conifer, ASC = Aspen–Subalpine Conifer, BS = Black Sagebrush, BW = Basin Wildrye, LBm = Limber-Bristlecone Pine—mesic, LSS = Low Sagebrush Steppe, MR = Montane Riparian, MSu = Montane Sagebrush Steppe—upland, WM = Wet Meadow.

[1]"Free" prescribed fire is prescribed fire that is allowed to spread at little cost to biophysical settings situated below alpine vegetation and rock, but above aspen–mixed conifer vegetation where ignitions are conducted.

- Riparian and wet meadow management strategies included cyclic weed inventory and spot application of herbicides.

Computer simulations of cost-effective management actions achieved lower ecological departure for all 10 focal natural communities. The total cost of implementation across all communities was around $3.6 million over 50 years (table 6, page 65). Many actions, however, were implemented fully in the first years of simulation (table 5).

The return on investment for aspen–subalpine conifer and mesic limber-bristlecone pine biophysical settings was significantly higher than that of other biophysical settings because the 95% confidence intervals between the two groups of biophysical settings did not overlap (table 6). Both biophysical settings rapidly responded to burning and recruitment into underrepresented early succession classes at very low costs that were uniquely caused by letting fire applied at full cost to lower-elevation aspen–mixed conifer to climb uphill until reaching barren cover and alpine terrain at nearly no extra cost to the park. Other biophysical settings did not benefit from such low implementation costs and the security of natural fire breaks.

Discussion

Landscape Conservation Forecasting™ is a useful science-based process for determining desired future conditions of plant com-

Table 6. Summary of ecological forecasts for management scenarios in 10 biophysical settings of Great Basin National Park

Biophysical setting	Ecological Departure (%): Current Condition	Ecological Departure (%): Minimum Management (50 years)	Ecological Departure (%): Preferred Management (50 years)	High-Risk Classes: Current Condition	High-Risk Classes: Minimum Management (50 years)	High-Risk Classes: Preferred Management (50 years)	Area: ac	Area: ha	Preferred Management average annual cost of implementation	Return on Investment (± 95% confidence interval)	Preferred Management average total cost (50 years)
Alpine-Subalpine											
Aspen–Subalpine Conifer	60	27	11	7	20	10	11,320	4,581	$108,918	92 ± 10	$326,755
Limber-Bristlecone Pine—mesic	48	42	24	—	—	—	4,500	1,821	$29,284	89 ± 11	$87,854
Mid-Elevation Forests											
Aspen–Mixed Conifer	66	33	22	6	12	8	8,110	3,282	$82,834	15 ± 9	$828,340
Shrublands											
Antelope Bitterbrush	74	62	35	28	44	10	341	138	$20,866	20 ± 6	$104,330
Basin Wildrye	68	70	35	43	64	24	269	109	$12,371 first 3 yr $1,192 last 47 yr	22 ± 3	$93,159
Black Sagebrush	60	55	34	39	40	19	1,880	761	$113,877	23 ± 3	$341,633
Low Sagebrush Steppe	61	27	16	0	1	1	420	170	$10,616	14 ± 2	$31,850
Montane Sagebrush Steppe—upland	56	41	29	21	30	11	12,711	5,144	$33,422	24 ± 3	$1,671,115
Riparian and Wet Meadows											
Montane Riparian	26	40	20	3	36	5	450	182	$1,806	25 ± 3	$90,301
Wet Meadow	49	40	28	0	36	9	89	36	$399	18 ± 4	$19,930
Total											**$3,595,268**

Note: Ecological departure scores were classified as good (0–33%, Class 1, green), fair (34–66%, Class 2, yellow), and poor (>66%, Class 3, red). Levels of high-risk classes to ecological systems were ranked as low (0%, dark green), medium (1–10%, light green), high (11–30%, yellow), and very high (>30%, red).

munities and their current ecological departure. The method's scenario simulations quantify the effects of current and future planning decisions into the future at landscape scale, assess the environmental costs and benefits of proposed resource restoration projects, and calculate the probability of success prior to initiation of any action. The inclusion of numerous management tools in the models allows assessment of many options and timelines for restoring plant communities to their desired future conditions. The results of this Landscape Conservation Forecasting™ effort will form the basis for the park's landscape-scale vegetation management plan and future funding proposals.

We generated detailed management prescriptions (table 5) and recommended treatment area maps for each focal biophysical setting of the park (fig. 5). Recommendations are ambitious and require more than $3 million to achieve predicted results over a 50-year period (table 6). It was determined that the park should focus first on prescribed burning of the aspen–subalpine conifer and limber-bristlecone pine mesic biophysical settings based on return-on-investment analysis alone (table 6; fig. 5). However, this represents a total cost of about $400,000 and requires time for planning. Immediate need for action to prevent loss of critical wildlife habitat in antelope bitterbrush and basin wildrye areas will allow park staff to make progress concomitantly in those biophysical settings while securing funding for aggressive restoration of the park's high elevations.

The park's fire management plan had always allowed the full range of management responses, including wildland fire use, prior to this project. Unfortunately, many areas had previously been considered FRCC 3, which required suppression. Similarly, the park had undertaken previous restoration projects based on professional knowledge on a case-by-case basis. This project provided valuable science-based and ground-verified data that enabled the park to update the FRCC map, allowing greater areas of wildland fire use as well as a better understanding of the consequences of and responses to wildland fire. Within months of updating the FRCC map, a naturally ignited fire was allowed to burn on the summit of Granite Peak until weather conditions put it out. This project also allows the park to develop a comprehensive vegetation management plan that identifies and prioritizes key management areas.

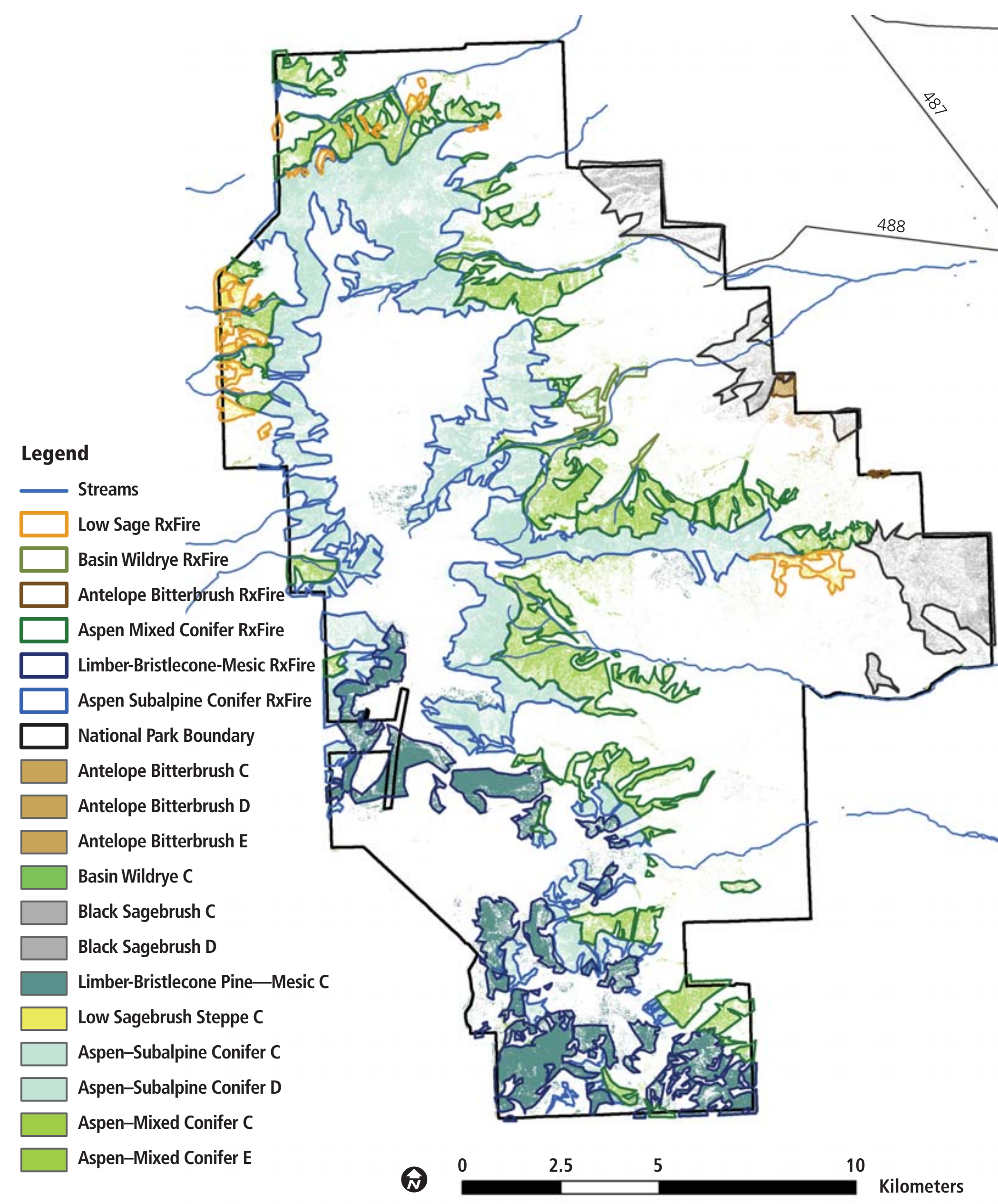

Figure 5. Map of recommended prescribed fire areas for different ecological systems. Not all areas need to be burned to reach low ecological departure from the natural range of variability.

Management implications

Application of Landscape Conservation Forecasting™ to assist natural resources management in other national parks is straightforward, although not easy, as it requires natural resource experts and contractors who can (1) conduct comprehensive high-resolution (≤5 m [16 ft] recommended) remote sensing of biophysical settings and vegetation classes (reference and all uncharacteristic classes), (2) build and run local state-and-transition computer models using the latest state-and-transition software platform, (3) collaboratively develop and run management scenarios with local stakeholders, and (4) produce and statistically analyze ecological departure and return-on-investment results that are understandable to managers (e.g., Provencher et al. 2010). Based on experience in the dry intermountain West, areas greater than about 20,234 ha (50,000 ac) are recommended for application of the method to ensure that ecological disturbances can create the different reference vegetation classes used to estimate ecological departure—as a rule of thumb, longer fire return intervals require larger project areas. The Nature Conservancy collaboratively implemented this method in 15 other landscapes with the Bureau of Land Management, U.S. Forest Service, and private industry.

Landscape Conservation Forecasting™ has also been implemented with climate change effects on model disturbance rates with the Bureau of Land Management in western Nevada, for the revision of Nevada's State Wildlife Action Plan, and currently as a supplemental project with Great Basin National Park. Climate change effects as predicted by global circulation models are incorporated through temporal time series acting as forcing factors; however, we are innovating in this field of state-and-transition modeling because (1) the climate change literature on ecological system range shifts for the intermountain West is very young and one field study does not support the predicted speed of modeled range shifts (Kelly and Goulden 2008; Rehfeldt et al. 2006); (2) field studies are mostly lacking; and (3) the literature offers no quantitative guidance on how predicted temperature, precipitation, and greenhouse gas concentration changes will affect local ecological processes. Therefore, updated state-and-transition models incorporating local climate change effects require formulating hypotheses that local land managers accept as highly probable but not definitive.

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