

CALIFORNIA FISH AND GAME

“Conservation of Wild Life Through Education”

Volume 100

Winter 2014

Number 1

Special Native Plant Issue



Gilia tricolor, CDFW Photo by Jeb Bjerke

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The United States Department of Agriculture says:

“The free marketing of wild game leads swiftly to extermination.”

—Yearbook, 1910, page 254

Published Quarterly by the California Department of Fish and Wildlife.

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SAN FRANCISCO, OCTOBER, 1914

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California Fish and Game

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Notes from the Editor

One hundred years is a long time. It is a very long time in the context of professional journals. As the Editor-in-Chief of *California Fish and Game*, I am very pleased that this journal is celebrating with a series of special issues that will comprise Volume 100. I am also proud to be responsible for the production of Volume 100, but I am especially proud of the work that is being accomplished by those that have taken on the responsibilities of recruiting authors, seeking reviews, corresponding with the authors, passing judgment on the acceptability of contributions for publication in our centennial year, and for assistance with reading proofs. Those individuals truly served as Corresponding Editors, a task usually handled by the Editor-in-Chief. As a result, they deserve a world of thanks for the effort put forth in producing volume 100, and each Corresponding Editor will be identified appropriately as each special issue is published this coming year.

This issue includes a special contribution from Governor Jerry Brown recognizing the centennial year of publication of *California Fish and Game*. As noted by Governor Brown, much has changed in the last century, and challenges to conservation seem to increase in unending ways. Also included is an introduction written by Dan Gluesenkamp of the California Native Plant Society and Chuck Bonham, Director of the California Department of Fish and Wildlife (CDFW), in which they recognize the relevance of an entire issue of the journal dedicated to the biology and conservation of plants.

Responsibility for the successful production of volume 100(1), our first issue this year, lies largely with Ms. Cherilyn Burton, a Senior Environmental Scientist in the Native Plant Program at CDFW. Ms. Burton earned her B.S. in Biological Sciences with a concentration in conservation from California State University Sacramento, and her M.S. in Ecology from the University of California, Davis. Cherilyn has 13 years of experience as a biologist in the environmental field, having worked previously with the California Department of Transportation and a private environmental consulting firm, where she acquired a wide diversity of experience assessing habitats and populations of many sensitive species, with a primary focus on plants. In her current position, Ms. Burton works on a variety of tasks related primarily to native plant conservation and recovery, with an emphasis on state-listed plant taxa. She also represents CDFW on a multi-agency team responsible for implementing the U.S. Fish and Wildlife Service Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon. Thank you, Cherilyn, for the effort and dedication expended while producing the first-ever issue of *California Fish and Game* dedicated solely to botanical subjects.

Following the recent retirement of Associate Editor Liam Davis and the resignation of Associate Editor Paul Hofmann, I am pleased to announce that three incoming Associate Editors will join the editorial staff of *California Fish and Game*. Ms. Laura Patterson works in the Wildlife Division of CDFW, where she has assumed the position of statewide coordinator for the conservation of amphibians and reptiles; she will handle manuscripts addressing the biology of those vertebrate classes. Laura earned a B.S. in Wildlife and Fisheries Biology from the University of California, Davis and a M.S. in Biological Conservation from California State University, Sacramento, where her research focused on red-eared sliders and western pond turtles at a Central Valley site. She has worked for the State of California since 1996, where she conducted surveys for a broad range of wildlife species and specialized in environmental compliance for much of that time.

Mr. Levi Souza also has agreed to serve as an Associate Editor. Levi is an Environmental Scientist in the Upland Game Program of the Wildlife Branch, with duties tending toward migratory upland game bird species, and is the state-wide coordinator for band-tailed pigeons. Levi earned B.S. and M.S. degrees from the University of California, Davis, and has broad experience in vertebrate ecology, including investigations of habitat suitability, ecotoxins, population dynamics, and behavioral responses of wildlife to disturbance. He is interested in building harvest-independent wildlife monitoring programs that bridge the gap between game and non-game species and are explicitly tied to regulation changes. Mr. Souza will handle manuscripts dealing primarily with the ecology, conservation, and management of upland game and avian species.

Finally, I am pleased to announce that Ms. Cherilyn Burton also will assume the role of Associate Editor. Her background was described earlier in this column, and Cherilyn will be responsible for manuscripts addressing the ecology, conservation, and management of plant taxa and vegetation communities. Welcome Laura, Levi, and Cherilyn, and thank you for volunteering for these newfound opportunities. Additional thanks are extended to Paul Hofmann and Liam Davis, who collectively served as Associate Editors for >25 years. And, I extend a special thanks to current Associate Editors Jeff Villepique, Steve Parmenter, Scott Osborn, Dave Lentz, Kevin Shaffer, Pete Kalvass, Nina Kogut, Jim Harrington, and Joel Trumbo for their many years of volunteer service. Collectively, the Associate Editors are the reason that *California Fish and Game* continues to improve with each issue.

Vernon C. Bleich
Editor-in-Chief
California Fish and Game

Congratulatory remarks regarding *California Fish and Game* volume 100

EDMUND G. BROWN JR., GOVERNOR

Volume 1, Number 1 of *California Fish and Game* was published in October 1914. This volume is the Centennial Anniversary of the journal.

In 1914, Germany had a Kaiser, Russia had a Czar, and California's population was approximately 2.7 million people. Climate change was neither known nor discussed. In 2014, California has more than 38 million people and climate change is one of the most pressing issues of our time.

In his 1914 introduction, former Governor George Pardee emphasized that California's wildlife ought to be conserved for the public benefit, and used "at such times and in such quantities and under such conditions as the public needs require, but without unnecessary waste or destruction." He wrote that the conservation of wildlife was dependent upon the continued existence of suitable habitat, including forests and waters, and noted that as a natural resource, water is the most necessary of them all. Since Pardee penned those words, little has changed with respect to the importance of water.

Today, all of California is in some form of drought, ranging from moderate to severe. In fact, our hydrologic cycles and natural conditions are changing. The northern Sierra snowpack is decreasing, reducing natural water storage, and altering runoff patterns. The temperatures of the ocean and rivers are rising.

Something too is occurring in the state's fish and wildlife populations. The spring and fall arrivals of some migratory birds are changing. Small mammals in Yosemite National Park and in the Sierra are found at different elevational ranges compared to one hundred years ago. Butterflies in the Central Valley are arriving earlier in the spring. Data show that of the 358 at risk bird species in California, 128 (38 %) are vulnerable to climate change, particularly those dependent on wetlands. Native fish are also threatened, particularly species such as salmon and steelhead that are found at the southernmost part of their historic range. Our state's bighorn sheep are facing drier weather and less available food and water.

Drought, climate change, and an ever-growing number of people will severely impact our fish and wildlife resources. That is why the conservation measures that we undertake now must produce real and lasting results. And, our commitment to enacting such conservation measures must be resolute. Stewardship — the idea of holding our fish, wildlife and natural resources in trust and passing them along in a better state to future generations — must be our fundamental ethic. The lush, natural bounty of California depends on our success.

California's roots run deep in science, wildlife management, and conservation. The 100 years of *California Fish and Game* are a treasure chest of recorded leadership in these fields, and will remain a guide for the future.

My deepest appreciation to all those who have made this journal possible.

Introduction to the special native plant issue

This very special issue of *California Fish and Game* is the first to be dedicated exclusively to California's special plants and spectacular flora. It is an historic volume, suited to celebrate the 100th anniversary of a journal that, like the Fish and Game Commission and the California Department of Fish and Wildlife, have long embraced a commitment to understanding and conserving California's plants and plant communities. Indeed, the very first issue of this journal featured an essay by former Governor George C. Pardee clearly explaining that:

“Everybody therefore, who believes — as almost everybody does — in the conservation of fish and game must, if he takes a second thought, believe just as thoroughly in the conservation ...of our forests and our waters. The fish and game conservationist therefore must be a forest and water and waste places conservationist, whether they be publicly or privately owned. For it is just as much an injury to the public, including the hunter, if our forests be destroyed no matter whether they be publicly or privately owned.”

A century ago, when this journal published its first issue, California was a very different place. In 1914, automobiles were still something of a novelty and California was still largely wild. Though American and European botanists had already learned much about California, in 1914 much of the state's botany remained a mystery. In many ways it was a key period in California biology and botany: Jepson was working on his *Flora*, legendary botanists were describing new species by the score, and *California Fish and Game* was initiated as a scientific journal to capture and share the exciting discoveries that were being made every day.

Today we live in a very different California. Our land today contains about 35 million more humans than it did century ago, including thousands of dedicated scientists and conservation professionals. In the intervening years we have learned much about our flora, and today we are particularly privileged to live during a renewed renaissance of discovery.

In recent decades, we have developed powerful tools for deciphering and analyzing the genetic composition of California plants, providing an unparalleled understanding of evolutionary relationships. As a result we are discovering hundreds of new species; in some cases we are finding that plants formerly considered to be a single species are not even close relatives! Simultaneously, we are experiencing a computer-driven revolution in data sharing and visualization; we are able to generate and share powerful maps that combine information unimagined in 1914, and apply powerful software to assessing conservation needs and prioritizing solutions. As these inspiring discoveries teach us more about the remarkable state we call home, they also further increase our commitment to conserving this legacy for future generations. As we learn more about why California is special, not just for us but for the whole planet, we strengthen our pledge to dedicate our minds, our hands and our hearts to conserving it.

How will California change in the next 100 years? We expect that it will be a lot warmer, and sea levels are likely to be higher. In all likelihood we will have discovered hundreds of new plant species. Though, because our state already has both the highest amount of biodiversity and also ranks highest in loss of biodiversity, we will very likely have lost many species, too. We can expect to see the beginning of population stabilization, though total human population will likely reach 50 million before it shows signs of declining. It is also very likely that our science and our profound commitment to saving this place will have given us powerful new tools for conservation. What is not certain is whether these tools will be sufficient to make a big enough difference, soon enough. Will California still have the spring wild flower displays that feed specialized pollinators and attract photographers from throughout the world? Will we succeed in saving Joshua trees, the wild desert, the delicate and sensitive alpine flower fields, and all of the other very special and very fragile parts that make our home so special? We cannot predict with certainty what the answers will be, but the good news is that we have a proven track record of conservation, we have incredibly powerful tools to employ, and thousands of acres of protected areas, all to help us conserve our state's unparalleled biodiversity.

When viewed as a whole, the papers that have been included in this special issue tell the story of a special community of people who strive to understand, celebrate, and conserve California's plants. We hope you will enjoy their good work, appreciate their dedication, and join their effort to preserve a unique and special land.

Charlton H. Bonham
Director, California Department of Fish and Wildlife

Dan Gluesenkamp
Executive Director, California Native Plant Society

Past, present, and future of native plant conservation within the California Department of Fish and Wildlife

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Native plant conservation is an essential part of the California Department of Fish and Wildlife's (CDFW) mission to manage California's diverse natural resources. The awareness of the need to conserve native plants gained a strong foothold in the 1960s and 1970s due to activities of the California Native Plant Society as well as enactment of conservation-minded legislation, such as the Native Plant Protection Act and the California Environmental Quality Act. CDFW responded to this surge in attention to native plant conservation by creating staff positions and programs to help address the need for botanical expertise throughout California. CDFW botanical staff and programs have been, and will continue to be, an essential component to maintaining the biodiversity of California's native flora.

Key words: California Department of Fish and Wildlife, legislation, native plant conservation, plant programs

The California Department of Fish and Wildlife's (CDFW) mission to manage California's diverse natural resources for their ecological values and public benefits can be daunting. Among the 6,500 native plant taxa in California (Baldwin et al. 2012), approximately 2,300 are considered to be of conservation concern (CNDDB 2014); this is a staggering number of plants for CDFW to monitor and manage. In this paper, we explore the journey native plant conservation has taken within California, from the initial awareness of native plant conservation brought forward by the California Native Plant Society, to enactment of conservation-minded legislation, and the subsequent role that CDFW botany programs and staff have played, and continue to play, in the regulation of sensitive plant taxa and promotion of native plant conservation.

NATIVE PLANT CONSERVATION AWARENESS (1960s AND 1970s)

Much of the initial awareness of the need to protect native plants and vegetation communities in California grew from the work of the California Native Plant Society (CNPS) in the late 1960s and 1970s. In 1968, G. Ledyard Stebbins, CNPS president at the time, began keeping a card file of information on native plants that appeared to have a limited distribution in California based primarily on information from Philip Munz's *A California Flora* (Powell 1975, York et al. 1982). CNPS used this card file as a starting point for soliciting the knowledge of local botanists on plant rarity, and subsequently compiled several informal rare plant lists. Culmination of a major CNPS effort to update and expand rare plant information occurred in 1974 when G. Ledyard Stebbins, along with the Chairman of the newly formed CNPS Rare Plant Committee, Roman Gankin, and Director of the CNPS Rare Plant Project, W. Robert Powell, organized a meeting with amateur and professional botanists to evaluate and refine the list of plant taxa of conservation concern and manually map localities for each plant record (Powell 1975, Smith 1986, York et al. 1982). As a result, the first *Inventory of Rare and Endangered Vascular Plants of California* (CNPS *Inventory*) was published in 1974, with a total of 1,393 plant taxa (Smith 1986). The CNPS *Inventory* was the first of its kind in California, and possibly the nation, and helped raise awareness of the need for native plant conservation.

LEGISLATION

This awareness of the need for native plant conservation was instrumental in the passage of the Native Plant Protection Act (NPPA) in 1977, which was the first California legislation designed to protect native plants (Cochrane 1988). The NPPA was the first California law, and one of the first in the country, to formally designate plant taxa as rare or endangered. In 1984, plant protection was expanded beyond the NPPA when plants were included for protection under the revised California Endangered Species Act (CESA) (Cochrane 1988). CESA did not replace the NPPA, but added the designations of "threatened" and "candidate" species to the existing rare and endangered designations from the NPPA (Cochrane 1988). CESA also provided a more formalized method to petition the Fish and Game Commission to add, delete, or change the status of a taxon, and provided a consultation process for projects subject to the California Environmental Quality Act (CEQA), a key piece of legislation that was passed in 1970 (Cochrane 1986, 1988).

While the NPPA and CESA are important laws, they apply only to the 218 plant taxa that are officially designated as rare, threatened, endangered, or as a candidate for such listing by the state of California (CNDDDB 2014). There are >2,500 additional plant taxa that are in need of conservation and management in California, which explains in part why CEQA is one of the most important laws for plant protection. CEQA provides for the protection of taxa and natural communities that may not have been formally listed under state law or the federal Endangered Species Act, but that can be shown to meet the definition of rare, threatened, or endangered (CEQA 2005, Wagner 2006). CEQA requires local and state agencies to examine the environmental impacts of proposed projects during the planning process, publicly disclose those impacts, identify project alternatives, and implement feasible mitigation measures (Cochrane 1986, Wagner 2006). The CEQA process is often the only means of protecting plant taxa that are of conservation concern in California, but not officially designated as rare, threatened, endangered, or as a candidate for such listing under California law.

Another noteworthy piece of California legislation for native plant conservation is the Natural Community Conservation Planning Act (NCCPA), which was enacted in 1991 (NCCPA 1991, Pollak 2001). The intent of the NCCPA is to promote long-term protection of plants, animals, and natural communities through landscape-level planning while still allowing compatible land use and economic activity (NCCPA 1991, Morey and Ikeda 2001, Pollak 2001). The NCCPA is designed to provide for conservation of ecosystems, including all of their components whether they are rare or common, rather than a strategy of addressing impacts to rare taxa on a piecemeal, individual development project basis (Pollak 2001). By preserving larger areas of important habitat, it is hoped that rare taxa will survive long-term and common taxa will not become rare in the future.

CDFW BOTANY PROGRAMS AND STAFF

Due to the size of California, CDFW has staff in centralized headquarters in Sacramento, and in regional offices. Currently, there are three main programs at CDFW headquarters that actively work to support native plant conservation: the Native Plant Program, the California Natural Diversity Database (CNDDB), and the Vegetation Classification and Mapping Program (VegCAMP). A unifying purpose of these three programs is to provide regional CDFW staff with the knowledge, data, and tools that they need to support “on the ground” native plant conservation.

Native Plant Program.—CDFW responded to the passage of the NPPA by hiring its first botanist at headquarters in 1978, and subsequently formed the Endangered Plant Project (EPP) (S. Rae, Musci Natural Resource Assessment, personal communication). The EPP, which later became the Endangered Plant Program, implemented the NPPA by coordinating and carrying out listing and protection activities for plants (Cochrane 1986). Among the first tasks undertaken by the EPP were to update maps that depicted rare and endangered plant distributions, develop and expand rare plant status reports, develop a standardized field survey form, help fund conservation projects, and help with the establishment of reserves for rare plant preservation such as the Pine Hill Ecological Reserve (S. Rae, Musci Natural Resource Assessment, personal communication). However, with limited staff to address the multitude of rare plant issues that needed attention, a large focus of the EPP was to support regional CDFW biologists with projects or problems concerning listed plant taxa and to increase awareness of the need for plant conservation throughout California (Cochrane 1988).

The Endangered Plant Program was renamed several times in the 1990s and 2000s, becoming the Plant Conservation Program, the Species Conservation and Recovery Program, and the Rare Plant Program. In 2010, the program was again renamed, and became the Native Plant Program, highlighting the need to conserve and manage all of California’s native plant species. The Native Plant Program is now part of CDFW’s Habitat Conservation Planning Branch, and coordinates statewide plant conservation efforts. The Native Plant Program still provides support to regional CDFW biologists, but the program also issues permits to take state-listed plants for scientific, educational, or management purposes; assists with management of grants for plant conservation and research; evaluates petitions to list new plant species under CESA; and assists CDFW and the public with various other plant-related issues.

California Natural Diversity Database (CNDDB).—While native plant conservation was gaining a foothold in California with the passage of the NPPA, The Nature Conservancy was establishing Natural Heritage Programs throughout the nation; there are

now >80 such programs in the western hemisphere. The purpose of these programs is to collect and manage data on the status and distribution of species and ecosystems that are of conservation concern. In 1979, the California Natural Diversity Database (CNDDDB) program was established as California's Natural Heritage Program, and in 1981 CDFW gained responsibility for CNDDDB as part of the legislated California Significant Natural Areas Program (Significant Natural Areas Program 1981, Bittman 2001).

The CNDDDB program inventories the status and locations of plants, animals, and natural communities of conservation concern within California. Within its first year, the CNDDDB botany program began collaborating with the CNPS Rare Plant Program. The CNDDDB botany program used CNPS rare plant data as the starting point for CNDDDB's rare plant dataset. Such data sharing is an excellent example of CDFW successfully collaborating with other organizations for the benefit of native taxa, and was the beginning of a lasting and mutually beneficial relationship between CDFW and CNPS. Over the years, the relationship between the CNDDDB botany program and the CNPS rare plant program became more streamlined, with less duplication of effort. The CNDDDB botany program now takes the lead on mapping and managing rare plant data and the CNPS rare plant program takes the lead on evaluating and tracking the rarity status of California plant taxa.

The CNDDDB program is part of CDFW's Biogeographic Data Branch, and helps to conserve California's biotic diversity by providing tools, and information on special-status taxa, to decision-makers involved with land-use and resource management activities. In the first years of the CNDDDB program, location data for rare taxa were hand drawn onto topographic maps that were then digitally processed into a CAD/CAM (computer aided design/computer aided manufacturing) system. In 1990, the CNDDDB program became the first Natural Heritage Program in the nation to enter its rare taxa location data into a Geographic Information System (GIS) (Bittman 2001, 2014). The CNDDDB is now composed of a GIS linked to a database that contains information on occurrences of rare taxa. The CNDDDB program distributes these data to its subscribers via an internet-based application called RareFind, which allows users to query the CNDDDB and generate reports. RareFind is linked to a map viewer called BIOS, which allows users to view and analyze spatial data.

The CNDDDB program also distributes GIS data for use in ArcGIS and ArcView programs. When the CNDDDB began utilizing GIS to map plant occurrences in 1990, it consisted of fewer than 18,000 occurrence records for rare plants, animals, and natural communities (Bittman and York 1988, Bittman 2014). The CNDDDB has now grown to >73,000 occurrence records for rare plants and animals, making California's Natural Heritage Program one of the largest and most complex programs in the Natural Heritage Program network. The CNDDDB is used by CDFW personnel, the Natural Heritage Program network, and over 500 subscribers including federal and state agencies, private consulting firms, researchers, and project planners.

Vegetation Classification and Mapping Program (VegCAMP).—The natural communities portion of CNDDDB worked from 1979 to 1995 to maintain an inventory of rare California natural communities. However, following the publication of the *Manual of California Vegetation* (Sawyer and Keeler-Wolf 1995), CDFW realized the value of a detailed statewide vegetation classification and map to identify all natural communities of conservation significance (T. Keeler-Wolf, CDFW, personal communication). VegCAMP was created in 2005 to meet this need, and is staffed with personnel from the natural communities portion of CNDDDB and the former Significant Natural Areas Program.

VegCAMP maintains a standardized vegetation classification and mapping system by developing vegetation mapping rules for California in accordance with federal and international standards. VegCAMP and the CNPS Vegetation Program produce field sampling protocols, hold workshops to teach field sampling and mapping, and update *A Manual of California Vegetation* (Sawyer et al. 2009), California's authoritative guide to plant communities. VegCAMP also maps smaller areas of vegetation such as on CDFW lands, and contracts for field data collection and mapping of larger areas. As with many CDFW programs, VegCAMP also provides support to regional CDFW staff by helping to assess the definitions of, and impacts to, rare natural communities during environmental review of proposed projects (D. Hickson, CDFW, personal communication).

Regional CDFW staff.—In many ways, it is the regional CDFW staff that have the greatest impact on native plant conservation. CDFW has six terrestrial regions, and each covers a separate geographic area of California. Prior to the early 1990s, wildlife biologists dealt with all plant-related issues in the regions in addition to their day-to-day wildlife-related duties. It was not until 1991 that the first regional plant ecologists were hired, with each CDFW region having at least one plant ecologist (J. Horenstein, CDFW, personal communication). Hiring of these plant ecologists was an important step for CDFW in recognizing and fulfilling the need for botanical expertise at the regional level.

While job classifications within CDFW have been modified since the first regional plant ecologists were hired, there are still CDFW staff in each region with botanical expertise. Those individuals perform a wide variety of tasks and deal with issues such as environmental review, land management, population monitoring, enforcement, research, and conservation planning. Among the most important tasks of CDFW regional staff is the cultivation of relationships with local governments, agencies, and private landowners to help ensure the protection of rare plants and natural communities. CDFW regional staff also comment on CEQA documents with regard to impacts that projects have on native plants and natural communities, and recommend actions that will avoid, minimize, or mitigate impacts to California's sensitive plant taxa.

THE FUTURE OF BOTANY WITHIN CDFW

It is the combination of strong CDFW plant programs at headquarters and knowledgeable CDFW regional staff that continues to distinguish CDFW's contributions to native plant conservation in California. As conservation concerns change within California, however, so must the response of CDFW and its botanists. The decades-long evolution of CDFW responsibilities was formally recognized in 2013 with the change in name from the California Department of Fish and Game to the California Department of Fish and Wildlife. This important symbolic change served to emphasize that CDFW is responsible not just for the management of fish and game species, but for the management and conservation of all wildlife, including native plants.

Many California plant taxa have been severely impacted by a history of human-induced change, and many of these taxa continue to be threatened by changes in land use, invasive species, hydrological changes, altered fire regimes, pathogens, genetic factors, pollution, or other degradations of natural habitat. Some California plant taxa are now extinct, and others may be doomed to extinction from impacts that have already taken place. As new threats to native plant species emerge, CDFW programs and staff continue to shift their focus to try to address these threats. As outdoor marijuana cultivation increased in northern

California, staff in the Northern Region of CDFW that previously were associated with timber harvest review shifted their focus to the new concerns and rising threats associated with marijuana cultivation. As renewable energy development is fast-tracked within California, the CDFW Climate Science and Renewable Energy Program moved to evaluate and address these threats to desert ecosystems, and a Desert Renewable Energy Conservation Plan is currently in development.

In the future, CDFW will confront what may be the biggest threat to the botanical resources of the state yet: a changing climate. CDFW demonstrated its commitment to reducing the impacts of climate change on California's resources, including plants, by forming the Climate Science Program in 2008. Nevertheless, climate change continues to pose a looming threat that will certainly be difficult, and perhaps impossible, to fully mitigate. The Biogeographic Data Branch, in conjunction with the California Landscape Conservation Cooperative, completed a climate change vulnerability assessment in 2012 for 156 rare plant taxa in California, and that assesses the potential impacts of climate change on those taxa (Anacker et al. 2013). The results of such vulnerability assessments can help guide future monitoring, management, and conservation plans for rare plants.

Managing and responding to the multitude of threats to California's biological diversity will require a continued commitment to California's botanical programs, and a renewed commitment to maintain botanical expertise among CDFW offices throughout California. From monitoring and managing plant populations on CDFW lands, to conducting research on emerging threats, and coordinating conservation activities with landowners and local agencies, there is still much that needs to be done.

Many people and organizations have contributed to the awareness and progression of native plant conservation within California over the past 45 years. A combination of enthusiastic, dedicated botanists in CNPS and the enactment of conservation legislation helped pave the way for the establishment of botanical programs within CDFW. CDFW botany programs and staff are essential aspects of California's commitment to conserve and manage natural resources and the habitats upon which they depend, for their ecological values and for their use and enjoyment by the public, now and into the future.

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Characterizing habitat suitability for disturbance-dependent rare plants of gabbro soils

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Assessing habitat quality to identify the highest priority sites for conservation planning is a complex task, and requires an understanding of which habitat attributes are most important for species population success. I explored the relative importance of biotic, abiotic, spatial, or disturbance-related habitat attributes to the population abundance of four edaphic-endemic, disturbance-dependent rare plant species. Variable selection provided a way to evaluate the relative importance of ecologically relevant groups of habitat attributes. Overall, biotic and disturbance history variables were the best predictors of population abundance for all four gabbro rare plant species, while spatial and abiotic variables were not found to be strong drivers of population abundance. Habitat quality for the four rare plants evaluated here may be best characterized by the associated species in the vegetation community, and an appropriate disturbance regime is a key component to maintain populations over time.

Key words: rare plants, disturbance, edaphic endemic, AIC, model selection, gabbro

Understanding the habitat attributes that shape habitat suitability for species has been a continuing theme in ecological research for the past century (Grinnell 1914, 1917; Kruckeberg 1954; Hutchinson 1959; Hirzel and Lelay 2008), and is an essential step in identifying high priority sites for conservation planning (Prendergast et al. 1999). Assessing habitat quality is a complex task that requires an analysis of the relationship between species distributions, population viability, and habitat conditions (Johnson 2007). New statistical methods allow for analyses of large datasets with many variables, which can for the first time begin to untangle the complex relationships between suites of habitat attributes and habitat quality (e.g., Schlesinger et al. 2008).

Determining the components of habitat quality for disturbance-dependent plants, such as species adapted to fire-driven systems, poses a special challenge because these species rely on transient habitat types (e.g., early-successional) and habitat quality varies during succession. Sites important for long-term conservation may fail to be identified because they appear unoccupied even though the species is actually present in the seedbank. The conservation of currently occupied sites may not guarantee successful conservation of the species over long time periods if the site loses required habitat qualities (Drechsler et al. 2009). A matrix of suitable habitats at different successional stages may be necessary for conservation; however, identifying suitable but unoccupied habitat is challenging (Quintana-Ascencio 1998). The distributions of early-successional species appear to be driven largely by random disturbance events such as fire (Vuilleumier et al. 2007); however, abiotic, biotic, and spatial factors also play a role in shaping the distributions of disturbance-dependent rare plants (Maliakal-Witt et al. 2005, Moretti et al. 2008).

Habitat quality has been measured for plants in a number of ways including (1) biotic community composition (e.g., Munzbergova 2004); (2) site history (e.g., Ross et al. 2002, Evans et al. 2008); (3) physical environmental gradients (e.g., Whittaker 1960, Woodward and Williams 1987); and (4) habitat spatial configuration (e.g., Wolf and Harrison 2001, Boyle et al. 2002). Four main ecological and conservation paradigms provide frameworks for assessing habitat quality based on vegetation classification, site history and succession, ecological niches, and patch dynamics. These frameworks are not mutually exclusive, and more than one may have explanatory power for predicting the distribution and abundance of species in a system.

Vegetation classification categorizes habitats based on dominant or indicator plant species (Sawyer et al. 2009). The use of vegetation classification in conservation emphasizes the importance of biotic habitat attributes, including community composition and associated species, as measures of habitat suitability. Vegetation type is commonly used to predict species presence in wildlife-habitat relationships programs (Morrison et al. 1998), and has recently been applied in large conservation planning efforts in which vegetation type is used as a surrogate to capture plant and animal biodiversity across the landscape (e.g., Reyers et al. 2007). Biotic community attributes such as associated species (Munzbergova 2004, Elmendorf and Moore 2008) have been shown to predict plant species presence, abundance, and reproductive output.

Site history represents a process-based conservation paradigm that emphasizes the roles of disturbance, restoration, and the maintenance of successional processes for habitat suitability (Pickett et al. 2009). Site history and disturbance have been identified as key components of species occupancy patterns in fragmented landscapes (Bastin and Thomas 1999, Ross et al. 2002). Furthermore, local-scale disturbance, such as edge effects, can explain species distribution and abundance better than habitat or community models in some systems (Harrison 1997, Benitez-Malvido 1998). Evans et al. (2008) showed that time since last fire can have a direct effect on the demography of disturbance-dependent plants; therefore, management to ensure proper disturbance regimes may be an essential component of a conservation strategy for such species.

The niche modeling paradigm predicts that physical environmental gradients define the habitat areas in the landscape where a species has the potential to occur (potential niche), and biotic interactions further limit the habitat in which the species actually occurs (realized niche: Grinnell 1914, 1917; Hutchinson 1959). Numerous studies have found that physical gradients, including climate, topography, and soils, are strong predictors of plant distributions

(Whittaker 1960, Kruckeberg 1984, Woodward and Williams 1987, Wolf and Harrison 2001). A large body of recent work has focused on the use of ecological niche modeling to understand species distributions and inform conservation (Elith and Leathwick 2009).

The patch dynamic paradigm, shown in metapopulation (Levins 1969, Moilanen and Hanski 1998) and mainland-island (MacArthur and Wilson 1967) models, predicts population presence and persistence based on the spatial configuration of habitat. Conservation actions based on this framework emphasize the role of large, connected areas to maintain habitat quality and allow population persistence over time. However, studies have shown that small, isolated preserves may be valuable for conserving species with narrow distributions, particularly if there is little remaining habitat (Lesica and Allendorf 1992, Shafer 1995), and small, isolated patches of rare plants may persist well over time in small preserves (Lawson et al. 2008). Despite that, reproductive output in small or fragmented plant populations may be reduced, even in species that naturally occur in small, patchy populations (Groom 2001, Wolf and Harrison 2001). Furthermore, small reserves may suffer from area-related changes in habitat quality (Hokit and Branch 2003). Early-successional species that rely on fire to maintain their populations may be particularly sensitive to habitat fragmentation if it results in changes to the fire return interval (Leach and Givnish 1996).

This study was conducted in an area of gabbroic soils surrounding Pine Hill in El Dorado County, California. The Pine Hill Preserve system was established in 2001 to conserve rare-plant habitat in this region, and there is currently an effort underway to identify habitat for addition to the preserve system (USFWS 2002). Land prices in this area can vary substantially depending on location, compounding the need for a tool to assess relative conservation value (i.e., habitat quality) of different sites to set conservation priorities for the best allocation of limited conservation funds. Because the focus of conservation actions often differs depending on which paradigm lens (vegetation classification, site history and succession, ecological niches, or patch dynamics) is used to view the system, exploring the relative contribution of each framework to explain observed species distributions may help identify the best conservation and management approach for a species.

METHODS

Study system.—This study was conducted within a 104-km² area of mafic, gabbroic soils surrounding Pine Hill in western El Dorado County, California (hereinafter referred to as the PHC), which is bisected on the southern end by US Hwy 50, and is fragmented by residential development. Gabbroic soils are considered intermediate to serpentine and metamorphic rock, and sometimes support unique plant communities (Alexander 1991). The PHC has a high botanical diversity, including eight rare plants, four of which are gabbro endemics. This study focused on four of the PHC rare plant species that occur primarily in chaparral openings and rely on fire or other disturbances to maintain populations over time: two gabbro endemics, *Wyethia reticulata* Greene (Asteraceae) and *Ceanothus roderickii* W. Knight (Rhamnaceae), and two species endemic to gabbro and serpentine substrates in the Sierra Nevada foothills, *Calystegia stebbinsii* Brummit (Convolvulaceae) and *Packera layneae* (Greene) W.A. Weber & A. Love (Asteraceae). Although they all co-occur within the same habitats, these four species have slightly different distributions, life history characters, and mechanisms for persisting through successional cycles (Table 1).

TABLE 1.—Life-history characters of the four gabbro rare plants at Pine Hill, California. Rarity status is abbreviated as follows: FE=federally listed endangered; FT=federally listed threatened; SE=state-listed endangered; SR=state-listed rare; BLM-S=Bureau of Land Management sensitive.

Species	Common name	Family	Rarity status	Known substrates ^a	Counties of occurrence ^a	Dispersal mechanism	Life-form	Response to fire	Vegetative persistence during inter-fire period	Seed-bank
<i>Calystegia stebbinsii</i>	Stebbins' morning-glory	Convolvulaceae	FE, SE	gabbro, serpentine	El Dorado, Nevada	gravity ^b	perennial herb	increased germination ^{c,f}	no	yes ^{e,t}
<i>Ceanothus roderickii</i>	Pine Hill ceanothus	Rhamnaceae	FE, SR	gabbro	El Dorado	gravity	shrub	increased germination ^{f,g}	limited	yes ^{f,g}
<i>Packera layneae</i>	Layne's ragwort	Asteraceae	FT, SR	gabbro, serpentine, metamorphic	El Dorado, Tuolumne, Yuba, Butte, Placer	wind	perennial herb, clonal ^c	resprouts ^h	yes	no ^h
<i>Wyethia reticulata</i>	El Dorado County mule ears	Asteraceae	BLM-S	gabbro	El Dorado	gravity	perennial herb, clonal ^d	resprouts, increased flowering ^{d,g}	yes	no ^f

^aCalifornia Natural Diversity Database (2014)

^bAnts have also been observed carrying the seed (M. Gogol-Prokurat, personal observation), although their role as a dispersal agent is unknown

^cD. Ayres, UC Davis, and L. Fety, Bureau of Land Management, unpublished data

^dAyres and Ryan (1997)

^eNosal (1997)

^fAyres (2011)

^gBoyd (1987)

^hD. Ayres and L. Fety, personal observation

Sampling strategy.—Plot locations were randomly stratified across chaparral and oak woodland vegetation types on public lands using ArcGIS and aerial photographs. Because it was not possible to achieve 30 presence plots for each of the rare plants using random sampling, additional plots were haphazardly stratified across rare plant patches observed in the field. All rare plant plots were separated by >100 m. Relevé data on environmental conditions representing biotic community composition, abiotic attributes, spatial configuration, and disturbance history (Table 2), as well as rare plant presence and population size were collected at each location in nested 10 m² and 400 m² plots during 2005, 2006, and 2007.

Abiotic variables included topographic position, soil color and texture, and coarse fragments (rock and litter layer). Topographic position was measured using a standard compass. Although mineral content has been shown to influence rare plant presence and abundance on serpentine soils (Kruckeberg 1954, Wolf and Harrison 2001), a recent study of the mineral content of gabbro soils found no significant differences in chemical composition between sites that were occupied and unoccupied by rare plants in the PHC (Alexander 2011). I chose simple measures of soil properties that are easy to implement in the field, including color, texture, and rockiness. Soil color (i.e., redness) represents properties such as Fe content (Schwertman 1993), and was measured using Munsell's (2000) color chart. Soil texture was measured using a simplified key to soil types adapted from Brewer and McCann (1982), which was converted to an ordinal scale representing clay content. The percent cover of coarse fragments of various types (litter, boulders and bedrock, stones, and gravel) was estimated in each plot.

I collected a full species list of plants present in the 10 m² plot to capture microhabitat conditions directly surrounding the rare plant occurrences, and a list of the dominant species within each layer (tree, shrub, herb) in the 400 m² plot to characterize the larger vegetation community in which the rare plants occurred. Each plot was visited at least once early in the season (March–May) and once late-season (June–August) to ensure a full species list.

TABLE 2.—Explanatory variables used in multiple linear regression models for *Calystegia stebbinsii* (CAST), *Ceanothus roderickii* (CERO), *Packera layneae* (PALA), and *Wyethia reticulata* (WYRE), Pine Hill, California.

	Variable	Description	
Spatial Factors	Connectivity metric	CAST and PALA connectivity = (Σ area/distance of all patches of plants within 300 m of a plot) CERO and WYRE connectivity = (1/distance to nearest patch)	
	Location	UTMN UTM Northing, NAD83 UTME UTM Easting, NAD83	
	Habitat area	Vegetation patch area Alliance-level vegetation patch, minimum mapping unit 0.4-0.8 ha Local patch area vegetation patch, minimum mapping unit <0.4-0.8 ha, patches separated by anthropogenic linear features such as roads	
Biotic Factors	Vegetation community composition	PCA1 PCA2 PCA3 PCA axes of arcsine-square root transformed cover values of the dominant species present in the 400-m ² plots. Chaparral shrubland was associated with PCA1 in a positive direction, while sites characterized by species indicating recent fire were associated with PCA1 in a negative direction. Xeric chaparral and grassland habitats were associated with PCA2 in a positive direction, while mesic woodland was associated with PCA2 in a negative direction. Chaparral shrubland was associated with PCA3 in a positive direction, and mesic chaparral and woodland species were associated with PCA3 in a negative direction.	
		Nat. Rich. Native richness = # native species in 10-m ² plot	
		Non-native cover Percent non-native cover = total non-native cover / total vegetation cover	
		Epiph. lichen Presence of epiphytic lichen	
	Vegetation structure	Tall Cover % tall cover (>4 m) in 10-m ² plot Medium cover % medium cover (0.5-4 m) in 10-m ² plot Low cover % low cover (<0.5 m) in 10-m ² plot	
	Abiotic Factors	Topography	Elevation Elevation (feet) Slope Slope (degrees) Aspect Northness = cosine (aspect in radians)
Soil properties		Soil Redness measured using Munsell's color chart Clay content Ranking of clay content based on simplified key to soil types adapted from Brewer and McCann (1982)	
Coarse fragments		Litter % litter cover in 10-m ² plot Boulder and Bedrock % boulder (>60 cm diameter) and bedrock cover in 10-m ² plot Stoniness % cover stones (7.5-60 cm) in 10-m ² plot Gravel % cover gravel (2 mm-7.5 cm) in 10-m ² plot	
Disturbance		Edge effects	Distance to edge distance between the plot and the nearest paved road or developed area >0.4 ha
		Site history	Clearing presence of grading or clearing noted in plot Fire presence of recent fire noted in plot
		Successional status	regeneration % of total shrubs regenerating (<3 years old) senescence % of total shrubs present with >25% dead branches

The abundance (number of individuals within the 400 m² plot) of each rare plant was counted in each plot. If there were >1000 individuals present, abundance was estimated by counting the number of individuals in a subplot and multiplying by occupied area. Plot locations were recorded and the perimeter of the rare plant patches sampled were field mapped using a Garmin 12XL GPS unit (accurate to <4 m). Additional rare plant patches throughout the PHC were field mapped for the development of connectivity metrics. For data analysis, abundance within 400 m² was chosen as the primary measure of population size because it represents an intermediate between total number of individuals per population, density of individuals (plants/m²), and total patch area (m²) that is standardized and easily repeatable.

Data analysis.—I used an Akaike's Information Criteria (AIC) model selection approach to determine which spatial, biotic, abiotic, and disturbance history variables best predicted population abundance of each of the four rare plant species using multiple linear regression. AIC allows for selection of the most parsimonious models based on goodness-of-fit (r^2) by including a penalty for model complexity (number of variables) to discourage model overfitting. I first tested for correlations among explanatory variables to ensure that none were highly collinear (Pearson's $r > 0.7$). Model selection was a two-step process: (1) identifying core variables with the greatest explanatory power by assessing models with all possible combinations of variables within each core group, and (2) building final models based on the selected core variables. For each species, the best final models with $\Delta AIC \leq 2$ were selected. In all cases, the AICc, which includes a correction for small sample sizes was used (Burnham and Anderson 2002). All analyses were run using *Statistica* 6 (StatSoft, Inc. 2003).

Plots were classified post-sampling according to the vegetation types identified by Wilson et al. (2009), which described four main vegetation types within the PHC: grassland, xeric chaparral, mesic chaparral, and oak woodland. Woodland plots were dominated by oaks (primarily *Quercus wislizeni* or *Q. kelloggii*) or by foothill pine (*Pinus sabiniana*). The two chaparral types were generally dominated by white-leaf manzanita (*Arctostaphylos viscida*) or chamise (*Adenostoma fasciculatum*), and were differentiated by the prevalence of four mesic indicator species: Brainerd's sedge (*Carex brainerdii*), redbud (*Cercis occidentalis*), toyon (*Heteromeles arbutifolia*), and hoary coffeeberry (*Frangula californica* ssp. *tomentella*). Vegetation stands including those species with cumulative cover values of >3% were classified as mesic chaparral, while stands in which those species were absent, or present with a cumulative cover value of <3%, were classified as xeric chaparral. Grassland plots were those with <10% cover trees or shrubs.

RESULTS

Overall, biotic and disturbance history variables had the strongest explanatory power for predicting population abundance for all species (Table 3). Biotic variables, including vegetation community composition and vegetation structure, were the strongest predictors of abundance for all species except *P. layneae*, for which disturbance history was the best predictor. Percent cover of gravel was the only abiotic variable that was a predictor of abundance for three of the four species. Spatial and abiotic variables identified as core variables during the first model selection step generally added little to model explanatory power based on AIC model selection, and were sometimes excluded from the final models (Table 3). Variables excluded from all models using AIC model selection included connectivity, local patch area, native richness, elevation, slope, aspect, percent cover litter, boulder and bedrock, and percent shrub regeneration.

TABLE 3.—Univariate contribution of core variables to predict population abundance in linear regression models for *Calystegia stebbinsii* (CAST), *Ceanothus roderickii* (CERO), *Packera layneae* (PALA), and *Wyethia reticulata* (WYRE), Pine Hill, California. Values for variables that were not identified as core variables are blank. Core variables included in all final models determined by AIC model selection are shown in bold. Core variables excluded from final models when all variables were considered are italicized.

			Proportion of variance explained (r^2)				
			CAST	CERO	PALA	WYRE	
Spatial Factors	Connectivity metric	<i>CAST and PALA</i> <i>CERO and WYRE</i>					
	Location	UTMN		0.28	0.04		
	Habitat area	UTME Vegetation patch area <i>Local patch area</i>	<0.01		0.01 0.15	0.12	
Biotic Factors	Vegetation community composition	PCA1 PCA2 PCA3	0.16	0.3	0.04 0.13		
		<i>Nat. Rich.</i> Non-native cover		0.01			
		Epiph. lichen	0.18				
	Vegetation structure	Tall Cover Medium cover	0.03			0.19	
		Low cover				0.28	
Abiotic Factors	Topography	<i>Elevation</i> <i>Slope</i> Aspect		0.13	0.05		
	Soil properties	Soil Redness Clay content			<0.01 0.02	0.11 0.14	
	Coarse fragments	<i>Litter</i> <i>Boulder and Bedrock</i> Stoniness				0.01	
		Gravel	0.15		0.1	0.1	
	Disturbance	Edge effects	Distance to edge		0.19	0.39	
		Site history	Clearing Fire	0.15			0.29
Successional status		<i>regeneration</i> <i>senescence</i>		0.23			

C. stebbinsii.—Models including vegetation community (PCA1), tall cover, epiphytic lichen presence, and recent fire best predicted *C. stebbinsii* population abundance. Abiotic and spatial core variables were excluded from all final models using AIC.

C. roderickii.—Models including vegetation community (PCA1), percent non-native cover, and shrub decadence best predicted *C. roderickii* abundance. Distance to edge and location (UTMN) were also included in some final models, but did not significantly improve model fit ($\Delta r^2 < 0.02$).

P. layneae.—Distance to edge was the dominant predictor of *P. layneae* abundance ($r^2 = 0.39$), and gravel cover and vegetation community (PCA1 and PCA2) further improved model fit ($0.05 \leq \Delta r^2 \leq 0.14$). Clay content, location (UTMN), and aspect were also included in some final models but did not significantly improve model fit ($\Delta r^2 \leq 0.02$).

W. reticulata.—Low vegetation cover, medium vegetation cover, soil redness, and stoniness were included in all final models. The inclusion of gravel cover, vegetation patch area, and clearing improved model fit somewhat ($\Delta r^2 \leq 0.05$).

Grouping plots into the Wilson et al. (2009) vegetation types based on indicator species further showed that vegetation community was a predictor of species presence for all species (Figure 1). The majority of *C. stebbinsii* and *C. roderickii* plots were found in xeric chaparral. Other species closely associated with xeric chaparral included *Eriodictyon californicum*, *Eriophyllum lanatum*, *Helianthemum* sp., and *Allium* sp. (Table 4). *P. layneae* and *W. reticulata* were more closely associated with mesic chaparral, which included associated species *Cercis occidentalis*, *Rhamnus ilicifolia*, *Frangula californica* ssp. *tomentella*, *Polygala cornuta*, and *Galium porrigens* (Table 4). *W. reticulata* was also associated with woodland, and was the only of the rare species closely associated with *Calochortus albus*, *Quercus wislizeni*, *Heteromeles arbutifolia*, *Toxicodendron diversilobum*, and *Lepechinia calycina* (Table 4).

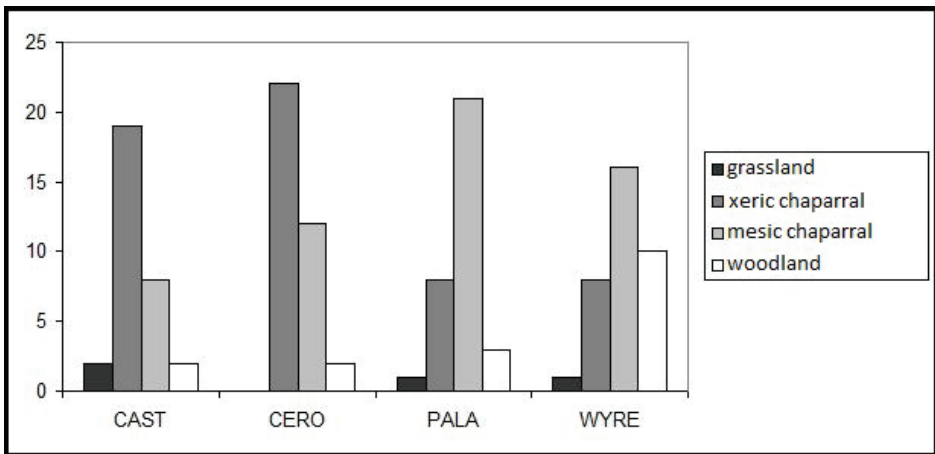


FIGURE 1.—Number of rare plant plots for *Calystegia stebbinsii* (CAST), *Ceanothus roderickii* (CERO), *Packera layneae* (PALA), and *Wyethia reticulata* (WYRE) assigned to the Wilson et al. (2009) Pine Hill, California gabbro vegetation types: grassland, xeric chaparral, mesic chaparral, and woodland.

TABLE 4.—List of species most frequently observed co-occurring with *Calystegia stebbinsii* (CAST), *Ceanothus roderickii* (CERO), *Packera layneae* (PALA), or *Wyethia reticulata* (WYRE) within 10-m² plots, Pine Hill, California. The 30 species most frequently associated with each rare species are in bold font. Incidence (Inc) is the percentage of plots occupied by each rare plant in which the associated species also was present. Cover (Cov) is the average percent cover of the associated species within those plots.

Scientific Name	CAST		CERO		PALA		WYRE	
	Inc	Cov	Inc	Cov	Inc	Cov	Inc	Cov
<i>Acmispon americanus</i>	29%	<1	42%	<1	24%	<1	23%	<1
<i>Adenostoma fasciculatum</i>	61%	2	67%	3	55%	6	83%	8
<i>Aira caryophylla</i>	77%	<1	88%	<1	83%	<1	87%	<1
<i>Allium</i> sp. ^a	39%	<1	39%	<1	17%	<1	10%	<1
<i>Anagallis arvensis</i>	29%	<1	42%	<1	41%	<1	33%	<1
<i>Arctostaphylos viscida</i>	61%	3	67%	4	83%	6	83%	6
<i>Brachypodium distachyon</i>	87%	2	94%	2	76%	1	57%	1
<i>Bromus madritensis</i>	29%	<1	42%	<1	76%	<1	70%	<1
<i>Calochortus albus</i>	13%	<1	6%	<1	10%	<1	40%	<1
<i>Calystegia stebbinsii</i>	100%	1	61%	2	28%	1	17%	1
<i>Carex brainerdii</i>	32%	2	42%	2	41%	3	30%	1
<i>Ceanothus lemmonii</i>	29%	1	33%	2	45%	3	53%	4
<i>Ceanothus roderickii</i>	71%	5	100%	6	52%	5	30%	3
<i>Centaurium tenuiflorum</i>	23%	<1	21%	<1	17%	<1	3%	<1
<i>Cercis occidentalis</i>	19%	1	33%	2	59%	1	33%	2
<i>Chlorogalum grandiflorum</i>	68%	1	85%	1	72%	1	30%	1
<i>Elymus multisetus</i>	19%	1	30%	<1	41%	<1	10%	<1
<i>Eriodictyon californicum</i>	26%	1	21%	<1	28%	1	7%	<1
<i>Eriophyllum lanatum</i>	23%	<1	24%	<1	14%	<1	20%	<1
<i>Festuca myuros</i>	68%	1	76%	1	76%	2	63%	1
<i>Frangula californica</i> var. <i>tomentella</i>	10%	2	18%	1	41%	1	43%	2
<i>Galium porrigens</i>	35%	<1	33%	<1	83%	<1	97%	<1
<i>Galium</i> sp. ^b	58%	<1	58%	<1	69%	<1	60%	<1
<i>Gastroidium phleoides</i>	94%	<1	100%	<1	86%	<1	80%	<1
<i>Helianthemum</i> sp. ^c	45%	1	30%	<1	24%	<1	17%	<1
<i>Hemizonella minima</i>	19%	<1	30%	<1	31%	<1	43%	<1
<i>Hesperolinon micranthum</i>	68%	<1	76%	<1	69%	<1	53%	<1
<i>Heteromeles arbutifolia</i>	0%	0	12%	1	21%	1	50%	3
<i>Hypericum concinnum</i>	29%	<1	24%	<1	34%	<1	33%	<1
<i>Hypochoeris glabra</i>	39%	<1	42%	<1	52%	<1	27%	<1
<i>Leontodon saxitilis</i>	87%	<1	94%	1	86%	1	47%	1
<i>Lepechinia calycina</i>	6%	<1	24%	1	21%	1	37%	3
<i>Logfia gallica</i>	94%	<1	91%	<1	72%	<1	53%	<1
<i>Navarretia filicaulis</i>	16%	<1	27%	<1	31%	<1	30%	<1
<i>Pinus sabiniana</i>	23%	2	18%	2	34%	1	37%	6
<i>Polygala cornuta</i>	10%	1	21%	<1	38%	1	57%	1
<i>Quercus wislizeni</i>	13%	<1	15%	<1	28%	1	53%	7
<i>Rhamnus ilicifolia</i>	23%	1	36%	1	55%	1	80%	2
<i>Salvia sonomensis</i>	90%	3	100%	3	100%	6	83%	5
<i>Sanicula bipinnatifida</i>	74%	1	85%	<1	76%	<1	60%	<1
<i>Packera layneae</i>	26%	1	39%	1	100%	1	30%	1
<i>Toxicodendron</i> <i>diversilobum</i>	3%	<1	3%	<1	10%	2	50%	3
<i>Wyethia reticulata</i>	13%	6	24%	5	34%	2	100%	9

^a*Allium peninsulare* or *A. sanbornii*

^b*Galium murale*, *G. parisiense*, or *G. divaricatum*

^c*Helianthemum suffrutescens* or *H. scoparium*

DISCUSSION

Overall, biotic community composition and disturbance history were the strongest predictors of species population abundance for the four gabbro rare plant species, while spatial habitat configuration and abiotic environmental variables were less important. The method of variable selection used provided a way to evaluate the relative importance of ecologically relevant groups of habitat attributes in predicting population abundance, which may be useful in ranking habitat quality (Johnson 2007), as well as identifying possible management actions that could improve habitat quality.

Biotic community composition was a strong predictor of population presence or abundance for all four rare species, supporting the findings of Munzbergova (2004) and Elmendorf and Moore (2008), that species composition in the vegetation community can be used to predict habitat suitability for plant species. Classifying sites by vegetation community type using indicator species was also informative for predicting habitat suitability. The dominant species most closely associated with all four rare plants include *Adenostoma fasciculatum*, *Arctostaphylos viscida*, and *Salvia sonomensis*, defined as the *Arctostaphylos viscida* Alliance or the *Arctostaphylos viscida* - *Adenostoma fasciculatum* / *Salvia sonomensis* Association in the Manual of California Vegetation (Sawyer 2009). Wilson et al. (2009) further separated the PHC vegetation into four vegetation types: xeric chaparral, mesic chaparral, woodland, and grassland.

Results presented herein showed strong associations between the rare plant populations and specific vegetation types and indicator species (Figure 1, Table 4), suggesting that using vegetation community data may be a promising method to determine habitat suitability in the field, particularly to identify suitable but unoccupied sites. Further research is needed to determine whether long-lived associated species could be used as indicator species to identify sites in later successional stages where a seedbank may be present. The distribution of these rare plant populations across the different vegetation types identified in the PHC shows that although the species are often found together in the landscape and treated together as a group for conservation planning, they require different microhabitats for successful conservation and may require a matrix of conservation sites at different successional stages.

Disturbance history was a strong predictor of population abundance at occupied sites, confirming the need for active monitoring and management of appropriate disturbance regimes to sustain viable populations of these species. The response of disturbance-dependent species to different types of disturbance may vary based on individual species traits (Menges 2007), such as reproductive strategy (Franklin et al. 2004, Clarke and Dorji 2008) or competitive ability (Moretti et al. 2008). Surprisingly, fire was a strong predictor of population abundance for only one species, *C. stebbinsii*, although all four species are fire-adapted and were expected to be associated with recently burned areas. This may be due to sampling error caused by the current management practice of fire suppression; there are many more cleared areas than burned areas available for sampling within the PHC. Recently burned habitats were limited to a few small accidental ignition sites and several larger controlled burns that were located in areas with little to no surrounding development, whereas clearings for fire breaks were present throughout the entire region. Ayres (2011) reported that *C. stebbinsii* is relatively short-lived and its populations begin to decline before

being shaded out during succession, which may explain its absence on older firebreaks. However, the importance of fire for *C. stebbinsii* despite the greater availability of cleared habitats indicates that mechanical clearing may not be an adequate replacement for fire to sustain populations of this species. The importance of clearing, rather than fire, as a predictor for the other three species should be viewed with caution, and further research into the relative effects of fire and grading as management tools to provide the disturbance required for these species is needed.

Abiotic habitat attributes including topographic position and soil properties were generally poor predictors of species population abundance within the PHC. This was somewhat unexpected because physical gradients are generally considered strong drivers of plant distributions, and studies have shown that the distributions of edaphic-endemic species are influenced by substrate properties (Kruckeberg 1954, Wolf and Harrison 2001). However, biotic community attributes may provide a better representation of the niche than physical gradients alone because they provide an indirect measure of key abiotic attributes (Elmendorf and Moore 2008) while incorporating species interactions that are not captured using abiotic variables alone (Kruckeberg 1954, Veblen and Young 2009). Here, abiotic variables did show a stronger contribution to population abundance models when considered alone, but had reduced explanatory power or were excluded based on AIC model selection when biotic variables were also considered. Furthermore, all surveys in this study were done within the limited geographic area of the PHC. Abiotic variables may better predict the distributions at broader spatial scales, while biotic variables better differentiate microhabitat differences between sites within the PHC.

Spatial population models have been used to understand habitat occupancy patterns of plants in some systems (Boyle et al. 2002), and habitat configuration may be an important predictor of population size and reproductive output of rare plants (Wolf and Harrison 2001). In this study, spatial configuration had only very weak effects in the population abundance models. Habitat area had little relative importance in predicting species abundance; however, I did find significant correlations between habitat area and population patch area for three of the four species, indicating that habitat area played some role in shaping overall population size. The importance of spatial configuration can be masked by site history and successional dynamics in disturbance-driven systems as shown by Hodgson et al. (2009), and spatial configuration may be less important than local habitat attributes in determining local population abundance (Adriaens et al. 2009). However, spatial configuration may still play an important role in long-term species occupancy patterns, even when habitat quality is taken into account (McVinish and Pollet 2013).

Structure, composition and associated species in the vegetation community were the best indicators of habitat quality for the four disturbance-dependent gabbro rare plants evaluated here, indicating that the vegetation classification paradigm may provide the most information about habitat suitability in this system. Vegetation attributes can be relatively easily collected in the field and should be used in habitat assessments when prioritizing sites for conservation. Disturbance history was also an important component of habitat quality at occupied sites; therefore, management of disturbance and successional stage at suitable sites will be an important component of the conservation strategy. Future research should investigate the use of indicator species in the vegetation community as predictors of suitable but unoccupied habitat for these disturbance-dependent rare plants.

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Status survey for endangered Bakersfield cactus

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Bakersfield cactus (*Opuntia basilaris* var. *treleasei*) is endemic to the southeastern corner of the San Joaquin Valley, California. Populations of Bakersfield cactus continue to be lost due to conversion of habitat to agricultural and urban uses. We conducted a status survey of sites with Bakersfield cactus based on occurrence records from the California Natural Diversity Database (CNDDDB). Of the 39 sites in the CNDDDB (30 presumed extant and 9 presumed extirpated), we visited 33 and examined aerial photography or conducted aerial surveys of the remaining sites. Based on our surveys, 25 populations were confirmed extant, 11 are believed to be extirpated, and the status of 3 could not be determined. Furthermore, two previously unreported populations were documented, and six undocumented translocated populations were identified. Of 33 sites with Bakersfield cactus, 27 occur entirely or partly on private lands. For the 27 naturally occurring extant populations, estimated size ranges from 2 to over 11,000 plants, but 16 (59%) populations have 100 plants or fewer. Habitat conditions within extant populations ranged from relatively undisturbed to highly disturbed, and remaining populations are fragmented and generally occur on small parcels. Only four entire populations and parts of eight others are permanently conserved. Based on the reduced number of extant populations and the reduced number of plants within many populations, Bakersfield cactus appears to be declining on multiple landscape scales. Conservation needs include the permanent conservation of additional populations, increased protections from impacts, vegetation management within populations, increasing the number of populations and the number of plants within populations, a population viability analysis, surveys for additional populations, and outreach programs.

Key words: Bakersfield cactus, endangered species, habitat loss, *Opuntia basilaris* var. *treleasei*, San Joaquin Valley, status survey

Bakersfield cactus (*Opuntia basilaris* var. *treleasei*) is endemic to the southeastern corner of the San Joaquin Valley of central California. This taxon historically occurred from just north of Bakersfield south to the Wheeler Ridge area at the southern end of the valley; cactus populations may have been more or less continuous within this area. Many sites with Bakersfield cactus have been converted to agricultural or urban uses (U.S. Fish and Wildlife Service 1998, 2011). Extant Bakersfield cactus populations are fragmented and generally occur on small parcels. Although some of these parcels are protected lands (e.g., California Department of Fish and Wildlife [CDFW], Tejon Ranch Conservancy), an increasing number are surrounded by incompatible land uses (e.g., urban development) and are subject to frequent disturbance from destructive trespass activities (e.g., off-highway vehicle use, dumping, and fires). Also, some of the extant cactus populations are on private lands where developments are planned for the future. Thus, populations of Bakersfield cactus continue to be lost, and habitat conditions are being degraded for some remaining populations.

The status of most of the remaining Bakersfield cactus populations has not been assessed for over two decades. According to the California Natural Diversity Database (CNDDDB; CNDDDB 2010), the most recent large-scale status survey was conducted by Moe (1989), and updates were provided on a few specific populations between 1989 and 2002 (CNDDDB 2010). Periodic status surveys for listed species are necessary to determine whether populations are still extant and to assess current site conditions throughout the range. This information is crucial for devising appropriate conservation and management actions to prevent extinction or facilitate recovery of such species.

Our goal was to conduct surveys to determine the current status of Bakersfield cactus throughout its range. Specific objectives were to (1) document the presence or absence of Bakersfield cactus at locations listed in occurrence records in the CNDDDB; (2) estimate the number of individuals present in extant populations; (3) assess current habitat conditions at each site to determine whether habitat improvement measures might be necessary to enhance the potential viability of each population; (4) identify actual and potential threats to the populations at each site; and (5) develop recommendations for the conservation and recovery of Bakersfield cactus.

MATERIALS AND METHODS

Study area and occurrence locations.—All known locations for Bakersfield cactus occur within Kern County, California (Figure 1). It occurs primarily in chenopod scrub and grassland habitats in the southeastern portion of the San Joaquin Valley (U.S. Fish and Wildlife Service 1998). Bakersfield cactus also has been found in some areas with blue oak (*Quercus douglasii*) woodlands.

Prior to conducting field work, we compiled a list of sites where Bakersfield cactus was known to occur, based on historic or recent observations. The initial list was generated based on Element Occurrences (EO) in the CNDDDB (CNDDDB 2010), which listed 45 Bakersfield cactus occurrences. However, six of these were on the east side of the Tehachapi Mountains and preliminary genetic analyses indicated that these may not be *Opuntia basilaris* var. *treleasei* (P. Smith, California State University-Bakersfield, personal communication). Of the remaining 39 occurrences, 29 were presumed extant, and nine were known to be, or thought to have been, extirpated (Figure 1). Additional locations were added to the list based on our personal knowledge as well as information from individuals (e.g., local biologists and land owners) who knew of cactus occurrences not yet reported to the CNDDDB.

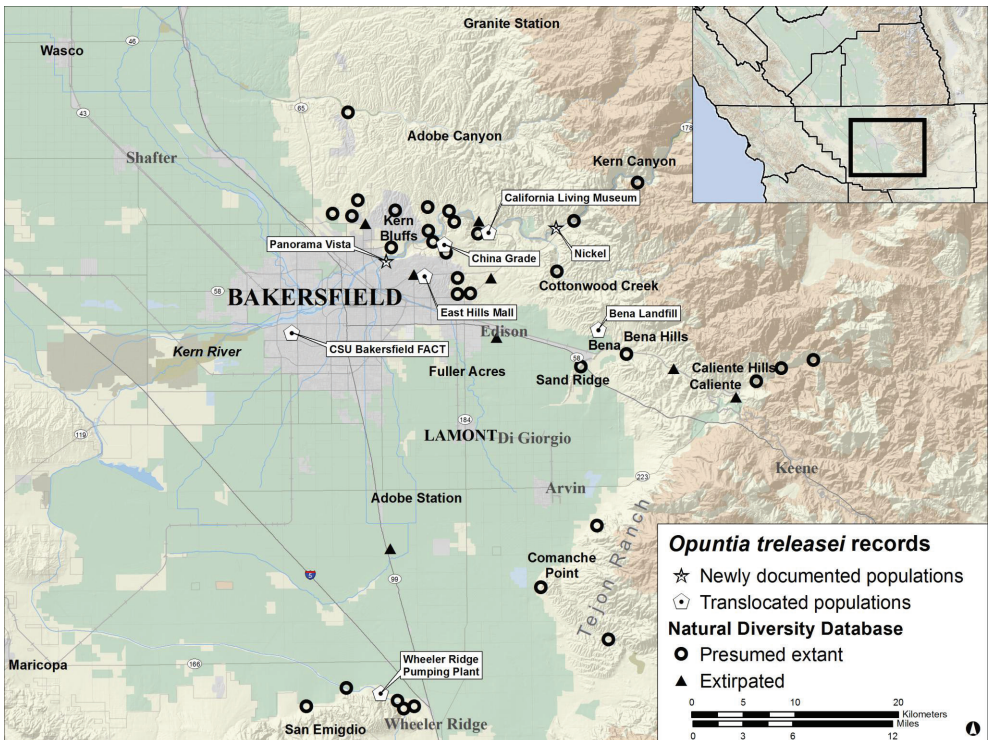


FIGURE 1.—California Natural Diversity Data Base occurrence records for Bakersfield cactus and locations of new and translocated populations in the San Joaquin Valley, California.

Next, we attempted to identify landowners for each of the locations. This was achieved by superimposing a GIS layer of Kern County parcels on a GIS layer of cactus occurrence records and aerial photographs to determine whether habitat was still present on parcels with known occurrence records. For parcels with habitat, we used the Assessor Parcel Number (APN) and the program Parcel Quest (<http://www.parcelquest.com>) to obtain landowner information from county property records. For many of the locations, the cactus population occurred on multiple parcels, and we attempted to contact all landowners to secure permission to access their property to survey for cactus.

Survey protocol.—During site visits, we attempted to survey as much of each EO or new site as possible. In particular, we attempted to visit each parcel with a different owner. Parcels under different ownership frequently were subject to different land uses or management that produced differential effects on cactus presence and condition. At least two field biologists conducted each survey. Surveys were conducted on foot and information relevant to each population was recorded (Appendix I).

Some sites could not be accessed, primarily because they were on private lands and attempts to secure permission from landowners to visit those sites were not successful. To the extent possible, those sites were examined from public roads, adjacent public lands, or adjacent private lands for which access had been granted. When these approaches were not possible or were insufficient for thoroughly inspecting a site, we surveyed from the air to determine whether suitable habitat for Bakersfield cactus was present and, if so, to see

if we could observe any cactus on the sites. We flew over these sites in a chartered, fixed-wing, single-engine aircraft. Two biologists and a pilot conducted surveys from an altitude of approximately 300–600 m. Each site was circled for whatever time was necessary to conduct a thorough inspection.

RESULTS

Site visits and population status.—Sites with Bakersfield cactus were visited during March 2010–May 2011. Detailed information on each site can be found in Cypher et al. (2011b:Appendix B). Time spent at each site varied from about one to six hours; some of the larger populations were visited multiple times in order to achieve more complete survey coverage.

Of the 39 EOs listed in the CNDDDB, we visited all, or portions, of 33, and confirmed that Bakersfield cactus was present at 25 (Table 1). We did not find cactus at eight sites. Although the putative determination is that these eight populations are extirpated, habitat was still present at these sites and additional visits are warranted to confirm our findings. We were unable to access six EOs that were on private lands. Based on an examination of aerial imagery and also on aerial surveys conducted on 18 May 2010 and 4 May 2011, no habitat remained at three of these sites and Bakersfield cactus likely is no longer extant (Table 1). Habitat appeared to be present on the remaining three sites, but we could not determine whether Bakersfield cactus was still present.

TABLE 1. —Status of Bakersfield cactus populations based on surveys conducted from March 2010 to May 2011.

Population	Population status		
	Confirmed extant	Confirmed or likely extirpated	Unknown
CNDDDB Element Occurrences:			
Visited ($n=33$)	25	8	
Not visited ($n=6$)		3	3
Previously undocumented	2		
Translocated	6		
Totals	33	11	3

Of the 39 EOs listed in the CNDDDB, 25 populations were confirmed extant, 11 are probably extirpated, and the status of 3 could not be determined (Table 1). Additionally, we were alerted by landowners to the presence of two previously undocumented populations, both of which were visited and assessed. Finally, we identified six Bakersfield cactus populations that were created by translocating cactus clumps or pads from other sites (Figure 1). Thus, Bakersfield cactus populations currently are present at a minimum of 33 sites (Table 1).

Ownership for lands with Bakersfield cactus populations includes both public and private entities (Table 2). Public lands are owned by the U.S. Forest Service, CDFW, California Department of Water Resources, or Kern County. Some private lands are owned

TABLE 2.—Land ownership for 33 sites with Bakersfield cactus as of May 2011. Most occurrences extend across lands with different ownership and therefore may be included in more than one category.

Ownership	Entity	Number of populations
Public	Federal	1
	State	7
	County	2
	Total Public	10
Private	Conservation organizations	2
	Corporations or individuals	23
	Multiple entities	1
	Total Private	26

by conservation organizations including the Center for Natural Lands Management, The Wildlands Conservancy, and the Kern River Corridor Endowment and Holding Company. Most of the populations occur on private lands, some of which are owned by corporations (e.g., Tejon Ranch Company, various oil companies, Pacific Gas & Electric Corporation [PG&E], development corporations) and some are owned by families or individuals. Few of the populations occur on lands with a single owner, but most populations occur on two or more parcels with different owners.

Population attributes.—Estimated population size for the 27 natural (non-translocated) populations of Bakersfield cactus assessed ranged from 2 to over 11,000 plants. Of these, 16 (59%) had 100 plants or fewer (Figure 2) while only 2 were estimated to consist of >1,000 plants. Among the assessed populations for which previous population

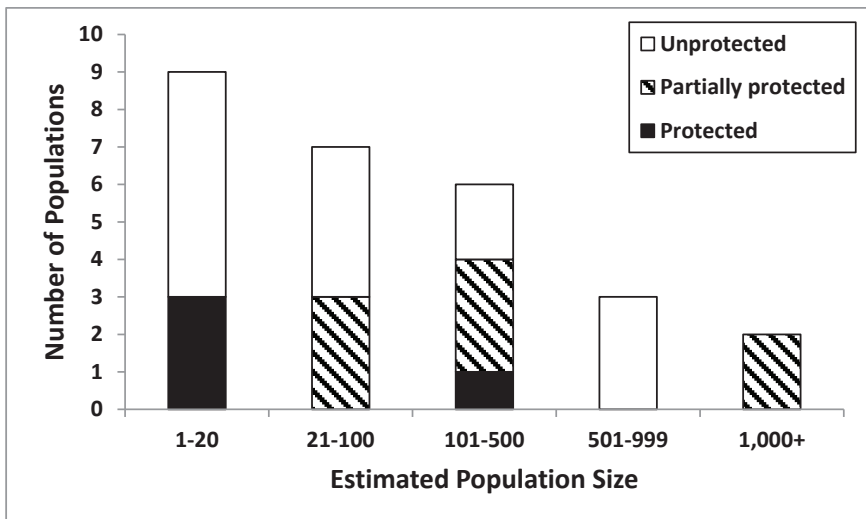


FIGURE 2.—Number of populations by size category and status of protection for extant populations of Bakersfield cactus based on surveys conducted in the San Joaquin Valley, California, March 2010 – May 2011.

size estimates were available from CNDDDB records, 10 were approximately the same size as previously reported, 2 were considerably larger, and 9 were considerably smaller. For the two populations that were larger (EO-23 and EO-24), both were on private lands where past access might have been limited, thereby resulting in incomplete counts. Also, differences in past and current population estimates could be a result of different methods of enumeration. Among the nine populations for which our estimates were smaller, some of the more notable declines included a drop from 500 down to 100 plants on EO-7; 50–75 down to 18 on EO-8; 2,000 down to 250–500 on EO-15; “several hundred” down to 50 on EO-20; 2,500–3,000 down to 500 on EO-25; and 14,000 down to 5,000 on EO-36. The apparent decline noted on EO-25, however, could be attributable to our lack of access to a large portion of this site. For some populations, declines were attributable to destruction of portions of the populations, while abundance appeared to be unchanged in the undisturbed portions. For other populations, declines appeared to have occurred within areas that did not seem to have been disturbed (e.g., EO-15 and EO-36).

Habitat conditions varied considerably among the 27 extant natural populations. In some populations, habitat disturbance was considerable, but conditions were relatively undisturbed at others. Broadly characterized, habitat was relatively undisturbed in 12 populations, low to moderately disturbed in 12 populations, and highly disturbed in 3 populations. Among partially disturbed populations the disturbance was, in some cases, distributed throughout the population. In other cases, some portions of a population evidenced disturbance, yet other portions appeared relatively undisturbed.

Internal disturbances within populations included roads (both paved and unpaved), off-highway vehicle use, human foot traffic, illegal dumping, target shooting, burning, sand mining, erosion, oil field activities, flooding, and competition from invasive non-native plants. Invasive non-native plants were present in most populations, but in some areas appeared sufficiently abundant to potentially constitute a competitive threat to Bakersfield cactus. Also, cattle grazing occurs in many populations. This activity technically constitutes a disturbance, but grazing at low to moderate intensities generally does not appear to adversely impact Bakersfield cactus and may even improve conditions by reducing the density of non-native grasses.

External threats included all of the disturbances described above, as well as habitat conversion for urban, industrial, and agricultural developments. Indeed, residential and commercial development is occurring at a rapid pace in areas northeast of Bakersfield. Industrial developments include ongoing oil field activities and expansion of sand or gravel mines. Agricultural development also is an ongoing threat: during our status survey a portion of EO-3 was converted to citrus groves.

Level of administrative protection varies considerably among the 27 natural populations, and ranges from lands being protected from development in perpetuity to a complete absence of protections. As described earlier, some populations extend across lands with different owners, and level of protection varies with ownership. Lands owned and managed by the U.S. Forest Service or CDFW are public lands on which protection of endangered species is mandated; as a result, cactus populations on these lands are very secure. Additionally, some populations occur on lands that have been placed under permanent conservation easements; among these are lands managed by the Tejon Ranch Conservancy (EO-21, 25, and 38) and private lands under easements with The Nature Conservancy (EO-23 and 24) and these populations also are very secure.

Other lands currently have protective measures in place, but those are not guaranteed in perpetuity. For example, lands owned and managed by the Center for Natural Lands Management (EO-3; Sand Ridge Preserve), The Wildlands Conservancy (EO-44; Wind Wolves Preserve), the Kern River Corridor Endowment and Holding Company (EO-18 and a previously undocumented population; Panorama Vista Preserve), and the California Department of Water Resources (EO-36 and 49) currently benefit from stringent protective measures implemented by these organizations, among which are fencing, restricted access, or avoidance measures. Cactus populations on lands owned by certain corporations receive some protection from policies or procedures (mostly avoidance), but there are no permanent protections in place. Finally, populations on some private lands, particularly grazing lands, currently receive some protection primarily because public access is proscribed or highly restricted. Among the 27 natural populations, four are entirely protected in perpetuity and portions of eight others are similarly protected. The remaining 15 populations have no permanent protection.

Among the 6 translocated Bakersfield cactus populations (Figure 1), population size ranges from 1 to 22 plants. The California Living Museum population consists of about a dozen plants and occurs among native plant gardens at a small zoo. These plants were translocated from multiple sites in the 1980s. This population is well protected, although not in perpetuity. The East Hills Mall population consists of about 10 plants in a highly disturbed landscape embankment next to a busy road and parking lot. These plants were collected and placed in this location when the mall was constructed in the 1980s. The population is in decline and probability of persistence in the current location is considered low. The California State University-Bakersfield population consists of about 10 plants, and is located on the grounds of the Facility for Animal Care and Treatment. These plants were translocated from various natural populations in the 1980s prior to the Bakersfield cactus being listed, and no formal protections are in place.

The China Grade Landfill population consists of about 10 plants in a relatively undisturbed buffer surrounding that inactive landfill. Some plants were translocated in 2008 to this site from another portion of the landfill, as were others after they were salvaged from a site in northeast Bakersfield that was about to be developed. Current zoning precludes development of that site, but no other formal protections are in place.

The Wheeler Ridge Pumping Plant population consists of one plant growing in what appears to be a small succulent garden established on the edge of a parking area, and was translocated from another location. The Bena Landfill population consists of 13 surviving plants from 10 clumps and 25 pads that were translocated from the nearby Sand Ridge Preserve (EO-3) in 2009 (Cypher et al. 2011a, Cypher et al. 2014). The plants are in a relatively undisturbed buffer area of this still active landfill. Under a Habitat Conservation Plan prepared by the Kern County Waste Management Department, the buffer area was set aside as compensation for landfill activities, and will be conserved in perpetuity.

DISCUSSION

Survey limitations.—Information collected during this survey provided substantial insights into the current status of the Bakersfield cactus, but there are some limitations to the data that must be considered. The population sizes provided are estimates; factors inhibiting precise estimates included populations dispersed over large areas, plants obscured by topography or vegetation, and lack of access to some portions of populations (e.g.,

portion of EO on private land for which access was not secured). In three instances (EO-8, EO-26, and EO-43), lack of access precluded our ability to visit any part of a population, but we were able to verify that habitat persisted on these sites. Lack of information on these three populations probably constitutes the most significant deficiency in the survey results.

Further complicating our efforts, vegetation conditions were not optimal for conducting surveys. Vegetation density was high during the survey, a result of above-average precipitation during the 2009-2010 and 2010-2011 rain years (California Department of Water Resources 2013), and obscured cacti at times. Wet conditions during winter 2010-2011 also delayed access to some sites, and also promoted regrowth of dense vegetation.

Increased density of vegetation may have affected survey results in two ways. It is possible that cacti were present but not detected on some sites, resulting in a conclusion that the population had been extirpated. It is also possible that cacti obscured by dense vegetation may have resulted in underestimates of population size. It is our opinion, however, that searches were sufficiently thorough and the likelihood of either of those potential outcomes is low.

We emphasize that the population sizes provided are estimates; factors inhibiting precise estimates included populations dispersed over large areas, plants obscured by topography or vegetation, and lack of access to some portions of populations (e.g., portion of EO on private land for which access was not secured).

Species status.—At least 27 natural populations of Bakersfield cactus currently are known to persist and 6 translocated populations have been documented, resulting in a minimum of 33 extant populations. Bakersfield cactus likely occurs in additional locations that have not yet been documented. Vast areas of potential habitat have not been surveyed, primarily because it occurs on private lands. In particular, considerable potential habitat still occurs in the Kern Front region, Caliente Creek drainage, and Comanche Point region of Tejon Ranch. Indeed, significant numbers of Bakersfield cactus plants have been found in recent opportunistic surveys in the Comanche Point region of Tejon Ranch, and the probability is high that additional plants occur in areas not yet surveyed.

In this paper, the term population has been used interchangeably with element occurrence; occurrence, however, is probably the more accurate term. A “population” generally is defined as a group of individuals that occur in a given geographic area and that have a higher probability of reproducing with individuals within the group than without (Pianka 1978). The working definition generally employed by CNDDDB for plants is that occurrences separated by at least one-quarter mile are considered separate occurrences, while those closer than this distance generally are lumped as a single occurrence. Bakersfield cactus likely was, at one time, widely distributed within San Joaquin Valley portions of the Kern River and Caliente Creek drainages (U.S. Fish and Wildlife Service 1998) and the current, highly fragmented distribution is the result of anthropogenic processes, and likely limits opportunities for gene flow or dispersal among the remaining EOs. Nevertheless, genetic partitioning within the range of Bakersfield cactus appears to be minimal (Smith 2013), and evidence of genetic isolation or other deleterious effects has not been detected. Sexual reproduction appears to be infrequent and most reproduction is vegetative through the shedding and rooting of pads (U.S. Fish and Wildlife Service 1998).

Protection status, threats, and on-going impacts.—Of the remaining occurrences of Bakersfield cactus, permanent protections are in place for four entire populations and portions of eight others. These protections include ownership and management by federal

or state natural resource agencies, and permanent conservation easements on lands managed by non-profit conservation organizations or on private lands. These occurrences may still face anthropogenic or biological threats, but are administratively secure. Furthermore, they include parts of some of the larger remaining cactus populations. Other populations, or portions thereof, currently are receiving active or passive protection, but such protections are not permanent.

Many of the Bakersfield cactus populations, including protected ones, are subject to a variety of internal and external threats. These disturbances can lead to physical damage to plants, soil contamination, fires, altered hydrologic patterns, or erosion of supporting substrate. Non-native invasive plants can compete with cactus for moisture, nutrients, and sunlight. Particularly problematic species in Bakersfield cactus populations included red brome (*Bromus madritensis* spp. *rubens*), wild oats (*Avena* spp.), Russian thistle (*Salsola* spp.), and Sahara mustard (*Brassica tournefortii*). Survival and growth of Bakersfield cactus increased substantially when non-native grasses were controlled around cactus clumps (Cypher and Fiehler 2006). Internal threats may be responsible for the marked declines in the number of Bakersfield cactus plants observed in 9 (36%) of the 25 populations assessed in previous status surveys.

Many of the remaining sites with Bakersfield cactus are subject to grazing by cattle, the effects of which likely vary with grazing intensity. Cows do not appear to feed on the cactus, but occasionally injure plants by kicking or trampling. Alternatively, cattle grazing may provide potential benefits to cactus populations. Cattle can reduce the biomass of plants that potentially compete with the cactus and also provide fuel for fires. Additionally, the detachment and movement of cactus pads might contribute to dispersal and establishment of new plants. Detached pads must come into contact with soil to become established, and this situation is facilitated when grazing reduces density of competitors.

The most substantial threat to remaining Bakersfield cactus populations is habitat conversion. Of the 11 occurrences now thought to be extirpated, at least six are attributable to agricultural or urban development. Additionally, portions of several other occurrences have been lost due to development; a striking example of continuing habitat loss was the conversion of a portion of EO-3 to a citrus grove during this survey. Other populations also are under threat from imminent development, particularly several in the northeastern portion of Bakersfield where urban development is occurring at a rapid pace.

The risk to Bakersfield cactus populations associated with the multitude of internal and external threats is enhanced by the small size of many of the known natural populations. Of the 27 populations, 16 (59%) comprise 100 plants or less. Populations with such low numbers of plants are already vulnerable to demographic or environmental stochasticity. Furthermore, the potential for natural dispersal (and subsequent establishment of new populations) appears to be low (U.S. Fish and Wildlife Service 1998). Unlike other *Opuntia* species that produce fruits that are highly attractive to potential animal dispersers, Bakersfield cactus fruits tend to be dry (Parfitt and Baker 1993) and likely are unappealing to potential dispersers. Further, Bakersfield cactus appear to readily produce seeds, but many of these seeds apparently are destroyed by insects or rodents prior to germination (E. Cypher, personal observation) and seedlings rarely are observed. The primary dispersal strategy employed by this plant appears to be the shedding of pads. However, these pads rarely move far from the parent plant unless moved by gravity, flowing water, or animals. Even where such occurs, the highly fragmented condition of remaining habitat substantially reduces the probability of a pad being transported to a suitable, unoccupied habitat patch.

A future threat to the persistence of Bakersfield cactus may be the cactus moth (*Cactoblastis cactorum*), an exotic insect native to South America. This species lays its eggs on the spines of *Opuntia* cactus and the larvae feed on the pads and kill them in the process. It was detected in the Florida Keys in 1989 and has impacted a rare cactus (*Opuntia corallicola*) that occurs there (Stiling et al. 2004). Since its initial detection in Florida, the moth has spread north and west. In 2009, it was detected in Louisiana (U.S. Department of Agriculture 2011); experts expect that it is only a matter of time before the moth reaches the southwestern United States, including California (Stiling 2002). Where it occurs, the moth has had devastating effects on *Opuntia* populations.

Conservation Needs and Strategies.—During the past decade or so, the number of Bakersfield cactus populations has declined. Additionally, declines in cactus abundance are evident in a number of the remaining populations. Furthermore, remaining populations are at risk from a number of immediate threats, and potential future threats could profoundly impact this taxon. Given the current situation and future prospects for Bakersfield cactus, aggressive conservation measures are warranted and may be necessary just to prevent its extinction.

Conservation needs for Bakersfield cactus include (1) the permanent conservation of additional populations; (2) increased protections from impacts; (3) habitat management within populations; (4) expansion within existing populations and the creation of new populations; (5) a population viability analysis to determine the optimal number of populations to maintain long-term viability of the taxon; (6) additional surveys for new populations; and (7) outreach and education programs.

Permanent conservation of additional populations could be achieved through purchase of properties by a natural resource agency or other entity willing to forfeit development rights for those lands, or through permanent conservation easements. Many remaining populations, particularly smaller ones and those located within the rapidly expanding urban landscape of Bakersfield, could benefit from increased site-specific protections, among which are fencing, road closures, signage, and the establishment of buffer areas around the populations. Additionally, vegetation management potentially could improve the health of some populations by reducing competition from non-native species or through a reduction in fuel loading. Moreover, reducing abundance of exotic plants, particularly in the immediate vicinity of cactus plants, could improve population vigor (Burger and Louda 1994, Cypher and Fiehler 2006).

Translocation to expand existing populations or create additional populations offers immense potential to improve the status of this species, and is an established methodology (Allen 1994, Given 1994, Falk et al. 1996, Stiling et al. 2000, Cypher et al. 2014). Six new Bakersfield cactus occurrences established through translocation were identified during this survey. Additionally, another population on CDFW lands was expanded significantly using translocated cactus pads recovered during a salvage effort (Cypher et al. 2014). To date, five new populations of Bakersfield cactus have been established using this technique, and in 2014 six additional populations will be created and three others expanded using this methodology (B. Cypher, unpublished data).

Completion of a population viability analysis would help determine a target number for new, translocated populations and help identify the optimum, or at least the minimum, size necessary to maintain viability of individual populations. Further surveys should be conducted to locate new Bakersfield cactus populations and, if they exist and are properly conserved, those populations would decrease the potential for extirpation of this

taxon. Finally, outreach and education programs may benefit Bakersfield cactus by raising awareness of the plight of this taxon, as well as the potential for funding or other support for conservation and recovery efforts.

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APPENDIX I

Information collected for each Bakersfield cactus location during a status survey conducted in the San Joaquin Valley, California, 2010–2011.

Variable	Description
EO #	Element Occurrence number from CNDDDB or unique label for new populations
Location	General location of EO or population
Land owner	Owner(s) of parcels within each Element Occurrence or new population
Conservation Status of Site	Whether all or portions of the site have any status that would conserve them in perpetuity, such as being owned by a federal or state conservation organization, or being covered by a conservation easement
Genetic sample collected	As part of a collaborative study of genetic variation and partitioning among Bakersfield cactus populations, 1-10 genetic samples (pads) were collected from each population and submitted to CSU-Bakersfield
Size of extant population	The number of plants (cactus clumps) was counted or estimated for each population. Clumps are defined as “groups of pads that are rooted at the same point” (U.S. Fish and Wildlife Service 1998)
Estimated overall area	General estimate of the area covered by the population.
% cover inside area	Estimate of proportion of population area actually covered by cactus
Habitat conditions	General description of dominant plant community and plant species, terrain, soil type, and any other pertinent habitat information
Internal disturbances	Any evidence of disturbance within the population including OHV use, roads, human foot traffic, dumping, shooting, grazing, burning, and invasive non-native plants
Extent of area disturbed within the population area	Percentage of area disturbed within the population
Estimated threat level from internal disturbances	Qualitative ranking of the threat to the population from internal disturbances

APPENDIX I (CONTINUED)

Variable	Description
Adjacent land uses	Uses and activities on lands immediately adjacent to the population
Estimated threat level from activities on adjacent lands	Qualitative ranking of the threat to the population from activities on lands immediately adjacent to the population
Probability of population presence in 100 years	Qualitative assessment of the probability that the population will persist and be present in 100 years, assuming that all current conditions within and around the population remain the same
Recommendations for conservation	Recommended measures for protecting, enhancing, or expanding the population
Point total	Points were assigned for protection status, population size, parcel size, internal threat level, and external threat level

Translocation as a conservation measure for endangered Bakersfield cactus

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Bakersfield cactus (*Opuntia basilaris* var. *treleasei*) is a succulent perennial in the cactus family (Cactaceae) and is endemic to Kern County, California. Due to habitat conversion and fragmentation, competition from non-native plants, and ongoing habitat degradation, Bakersfield cactus is listed as federally and state endangered. We tested a technique for establishing new populations of this taxon by translocating Bakersfield cactus pads (i.e., stem segments) and clumps (i.e., intact plants) to two sites within the historic range. Translocated clumps were more successful than pads in terms of survival, growth, and flowering. However, removal of clumps may constitute more of an impact to source populations. Cattle guards were effective in preventing damage from cows. Strategies such as supplemental water during dry summer weather and propagation of pads into small plants prior to translocation are recommended to increase the success of pads. Translocation could contribute significantly to conservation and recovery efforts for Bakersfield cactus.

Key words: Bakersfield cactus, *Opuntia*, restoration, San Joaquin Valley, translocation, transplantation

Bakersfield cactus (*Opuntia basilaris* var. *treleasei*) is unique in that it is the only cactus native to the San Joaquin Valley. An endemic plant restricted to Kern County, California, the taxon still occurs sporadically throughout its historic range from just north of the Kern River near Bakersfield southward to the southern tip of the valley and into the western foothills of the Sierra Nevada. Bakersfield cactus was once common within

its range (Twisselmann 1967), but due to habitat loss and other factors it was listed as an endangered species at both the federal and state levels in 1990 (U.S. Fish and Wildlife Service [USFWS] 1990, California Department of Fish and Game 2005). Despite publication of a recovery plan more than 15 years ago (USFWS 1998), Bakersfield cactus continues to decline; habitat loss and degradation are ongoing, and additional threats have been identified recently (Cypher et al. 2011b, 2014; USFWS 2011).

As with other prickly-pear cacti (*Opuntia*), Bakersfield cactus has flattened stem segments (cladodes) commonly known as pads. The tiny leaves of all prickly-pear cacti appear only on young growth and are soon deciduous. Bakersfield cactus differs from other varieties of beavertail cactus (*O. basilaris*) in that it has rigid spines (Baldwin et al. 2012). Some species of prickly-pear cacti have fleshy fruits known as pears or tunas, but the fruits of Bakersfield cactus are dry. Although Bakersfield cactus does produce seeds, the warm, moist conditions necessary for germination are rare within its range (Benson 1982), and predators readily consume the seeds (E. Cypher, unpublished data). Instead, reproduction is mostly vegetative by means of individual pads detaching from the standing plants and taking root (Twisselmann 1967, Benson 1982); the resulting offspring, thus, are clones of the parent plant. As a result, large masses of stems known as clumps often grow proximate to each other, and individual stems within the clumps are difficult to distinguish. Thus the terms “plant” and “clump” are used interchangeably herein, although either may range in size from a single stem to a meter or more in diameter. Longer-distance dispersal of Bakersfield cactus is mainly downhill due to gravitational movement of shed pads, although downstream movement is possible for plants that grow along watercourses (USFWS 1998). Dispersal distances for seeds and pollen are not known; a solitary bee (*Diadasia rinconis*) has been observed visiting the flowers of Bakersfield cactus (Grant and Grant 1979), but otherwise little is known about the pollination biology of the taxon.

New Bakersfield cactus populations potentially could be established in unoccupied habitat via the translocation of pads and small clumps. Using one or both of these means, small Bakersfield cactus populations have been established in several locations within the Bakersfield city limits (Fiedler 1991, Cypher et al. 2014). Thus, translocation of cactus pads and clumps may constitute a viable strategy for restoring Bakersfield cactus. Although creation of new populations was not identified as a recovery action in either the recovery plan (USFWS 1998) or the 5-year review (USFWS 2011) it is, nonetheless, a strategy that can help to ameliorate past habitat loss. Basic principles of conservation biology suggest that the probability of species extinction can be reduced and the likelihood of long-term viability can be improved by increasing the number of individuals and populations of a species (Primack 1996). Thus, successful establishment of additional populations could contribute in meaningful ways to the conservation and ultimate recovery of Bakersfield cactus. Given the highly fragmented state of remaining natural lands and inherent characteristics of the species, natural dispersal of Bakersfield cactus to unoccupied habitat is highly improbable.

We initially translocated Bakersfield cactus pads and clumps with the objectives of (1) establishing a population of Bakersfield cactus in currently unoccupied habitat, and (2) comparing two methods for conducting such population establishment. Using similar techniques, we conducted a second translocation for the primary purpose of salvaging doomed plants; however, the second translocation provided an opportunity to test additional aspects of the technique.

Although various terminology is used to describe anthropologically-assisted movement of animals and plants, we felt that “translocation” best described our effort. This term has become controversial in recent years as species have been moved outside of their natural ranges; proponents tout the benefits for preventing species extinctions and promoting biodiversity (Schlaepfer et al. 2009, Muller and Eriksson 2013), whereas opponents point out the expense and potential risks to the ecosystem (Fazey and Fischer 2009; Ricciardi and Simberloff 2009a, 2009b). We use the term “translocation” simply to mean the transfer of an individual from one location to another within its historic range. The term “transplantation” is more commonly used for plants, but did not seem appropriate for pads that were not yet rooted.

MATERIALS AND METHODS

Source population.—Bakersfield cactus pads and clumps were collected from Sand Ridge, a natural feature consisting of eolian sand deposits derived from granite (Soil Survey Staff 2013). Sand Ridge is located approximately 15 km east of the city of Bakersfield (Kern County, California), at the base of the Tehachapi Mountains (Figure 1). The Bakersfield cactus population at Sand Ridge (Element Occurrence 3) is estimated to consist of >10,000 clumps, and is the largest remaining population (Cypher et al. 2011b, 2014). Bakersfield cacti are most abundant on top of the ridge and along the eastern slope, but extend down onto the floodplain of Caliente Creek at the southeastern base of the ridge.

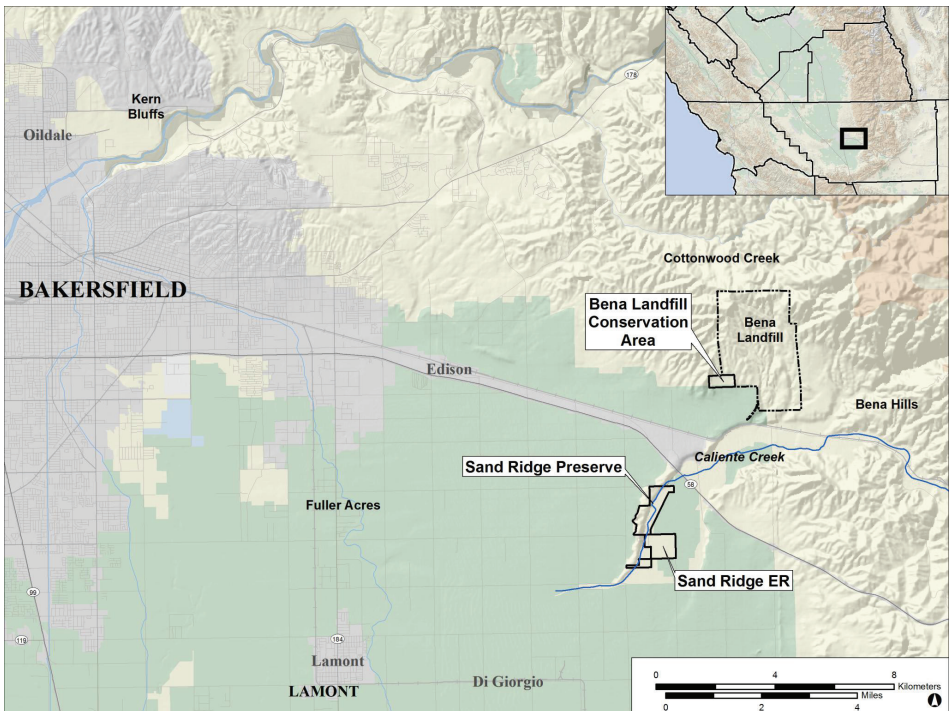


FIGURE 1.—Locations of two source populations and two recipient sites for translocations of Bakersfield cactus, Kern County, California.

The plant community at Sand Ridge is a relatively unique combination of species from the San Joaquin (Germano et al. 2011) and Mojave deserts. These species are adapted to the arid conditions, hot summers, and the Mediterranean climate, in which rain falls primarily from December through March (Twisselmann 1967, Major 1995). Dominant shrubs and subshrubs on the ridge itself are native and include bladderpod (*Peritoma arborea*), California croton (*Croton californicus*), cheesebush (*Ambrosia salsola*), and desert tea (*Ephedra californica*). Dominant herbaceous species include non-native grasses such as slender wild oats (*Avena barbata*) and ripgut brome (*Bromus diandrus*), and native forbs such as Coulter's jewelflower (*Caulanthus coulteri*), fiddleneck (*Amsinckia* spp.), Kern tarweed (*Deinandra pallida*), and several lupines (*Lupinus* spp.). All plant scientific names cited herein follow Baldwin et al. (2012).

Collection and translocation of Bakersfield cactus were conducted under a recovery permit from the USFWS (TE825573-4) and a research permit (2081(a)-09-15-RP) from the California Department of Fish and Wildlife (CDFW). Permissions were secured from landowners of the source and recipient sites.

First translocation.—Material was collected from the Sand Ridge Preserve (Preserve), a 109-ha conservation area owned by the Center for Natural Lands Management (CNLM; Figure 1). The collection area was on the upper part of the ridge ca. 250 m in elevation. The introduction site was Kern County's 390-ha Bena Landfill Conservation Area (BLCA), which is at the southwestern corner of the Bena landfill property (Figure 1) and will be protected in perpetuity as mitigation. The BLCA is approximately 5 km upstream from the Preserve along Caliente Creek and is centrally located within the historic range of Bakersfield cactus. A number of smaller, extant cactus locations occur just outside the landfill boundaries and were likely contiguous with Sand Ridge historically (Cypher et al. 2011a, 2011b, 2014). In the area of the BLCA where we introduced the cacti, the plant community is primarily annual grassland. Dominant species include non-native annual grasses such as red brome (*Bromus madritensis* ssp. *rubens*) and wild oats (*Avena* spp.), the non-native forb red-stemmed filaree (*Erodium cicutarium*), and the native forbs blue dicks (*Dichelostemma capitatum*) and fiddleneck (*Amsinckia* spp.). Topographically, the area ranges from gently rolling hills to moderately steep slopes with deep ravines. The elevation of the translocation site was ca. 350 m; the soil complex in the BLCA is derived from mixed alluvium and consists of sandy clay loam and gravelly sandy clay loam (Soil Survey Staff 2013).

Ten plants (small clumps) and 25 shed pads were collected on 19 October 2009. This timing was planned to coincide with the onset of winter rains. We followed permit requirements to collect only clumps that had five or fewer pads and that were relatively isolated from other cacti, and to collect pads that had been shed naturally. The east-facing side of each clump was marked so that plants could be planted with the same orientation at the introduction site. We dug at a minimum distance of 15 cm beyond the perimeter of each clump, then transferred it to a bucket that was lined with a large piece of fabric, along with a sufficient amount of soil to cover the roots and stabilize the plant.

By collecting pads that had been shed naturally we avoided damaging living plants. Only one pad was collected per clump to minimize impacts to reproduction at the source site. We collected shed pads that had not yet become rooted to the ground but had already formed a callus at the detachment point. Each collected pad was weighed and the maximum length and width were measured. Pads were transported to the introduction site in individual, labeled paper bags.

On 20 October 2009, we introduced the cactus pads and clumps to the BLCA. Plots were subjectively located throughout the suitable habitat to avoid ground squirrel (*Spermophilus beecheyi*) burrows and other disturbances. Within plots, spacing of cacti roughly emulated the spacing in natural populations. Each of the five plots received five pads and two clumps, which were mapped to facilitate monitoring. Before placing the pads on the ground, we removed thatch from the soil surface and loosened the top few centimeters of soil. Any pad that exhibited root growth was positioned so that the roots were in contact with the soil surface. All other pads were placed in a position that maximized contact between the pad and the soil surface (i.e., flat side down) and were secured with a wooden skewer.

For each clump, we dug a planting hole that was approximately 30 cm in both depth and width. Each clump and its soil were removed from their bucket by grasping and lifting the fabric that was lining the bucket. Clumps were transferred into the planting holes by hand and the holes were filled in with a combination of soil transported from the source population site and soil native to the introduction site. Clumps were transplanted into dry soil and watered after one week (approximately 3.5 liters each). Water was provided again on 25 November 2009. After this date, sufficient precipitation precluded the need for further supplemental watering during that growing season.

To protect the cacti from disturbance associated with cattle ranching on the site we installed 1.5-m metal T-stakes at each of the plot corners to alert ranch hands. In addition, we installed two pieces of crossed, bent rebar over each pad and clump in three of the five plots to discourage cattle from trampling the plants and to test efficacy of this protective mechanism. Thus, a total of 15 pads and 6 clumps were protected by rebar.

We visited the introduction site approximately monthly for the first six months to assess the status of the pads and clumps, then opportunistically thereafter, with the final visit occurring in November 2013. During monitoring visits we evaluated survival of each pad and clump, documented new pad growth and flowering, and removed any vegetation growing immediately next to or overhanging the translocated pads and clumps to reduce competition for light and water. Finally, at the conclusion of the project, we counted the number of live pads present for each plant and assessed damage caused by cattle.

Second translocation.—Bakersfield cactus clumps and pads were collected from an unprotected portion of Sand Ridge immediately north of the Preserve at ca. 280 m elevation that was undergoing active conversion to citrus orchards. The inadequacy of state and federal regulations to protect this species was identified as a threat at the time of federal listing (USFWS 1990) and is one of the ongoing threats to Bakersfield cactus, as summarized in the 5-year review (USFWS 2011). The recipient site for this effort was the Sand Ridge unit of the Bakersfield Cactus Ecological Reserve (Sand Ridge ER; Figure 1), which is owned and managed by CDFW.

The Sand Ridge ER is at the southeastern base of Sand Ridge, on the margin of the Caliente Creek floodplain at an elevation of ca. 195 m. Native Bakersfield cactus occurs in this area but was largely displaced by agricultural conversion prior to state ownership; only the least-disturbed southwestern parcel of the Sand Ridge ER received translocated cactus. CDFW now manages the entire Sand Ridge ER to protect the remaining Bakersfield cactus and as a buffer for the extensive population of cactus on the ridge. The dominant shrub in the Sand Ridge ER is the native scale-broom (*Lepidospartum squamatum*), but a few individuals of native allscale saltbush (*Atriplex polycarpa*) also are present. The

herbaceous species composition is similar to that of the BLCA. The topography in the Caliente Creek floodplain is level; soils are derived from alluvium and the soil texture is gravelly loamy coarse sand (Soil Survey Staff 2013).

The second translocation effort took place on 20 January 2011 after the salvaged Bakersfield cactus plants had been buried under a bulldozed mound of sand for approximately one month. The 2 large cactus clumps that we were able to locate under the mound of sand were removed in the most intact units possible, yielding 26 smaller clumps and 28 separate pads, a result of disarticulation that occurred during excavation or transport. Clumps and pads were transported in buckets as in the first effort and planted on the same day. Of the six plots established and mapped, five contained five clumps and four to five pads each; the sixth plot contained the remaining plant material consisting of a single clump and six pads. Planting sites were prepared and clumps planted as in the first effort. In an attempt to improve pad survival and to test the effect of pad orientation on survival success, we partially buried pads in the soil in one of three orientations: horizontal (i.e., flat), vertical (upright), or on edge. All pads and plants were then thoroughly watered; supplemental water was again provided on 25 July 2011. Monitoring took place on 25 July 2011, 27 February 2013, and 1 November 2013. Competing vegetation was sparse and was not removed during this period.

Statistical analyses.—The proportion surviving for each translocation was compared between pads and clumps using contingency table analyses and chi-square tests. For pads in the first translocation we compared the mean weight at initial collection and mean area (length x width) between pads that lived and pads that died using a t-test. T-tests were also used to compare mean number of pads between cacti that flowered and those that did not in both translocations. For the second translocation we used single-factor analysis of variance to compare the number of live pads present per plot in November 2013 among the three orientations of pads that had been planted.

RESULTS

First translocation.—Through May 2011, all 10 translocated cactus clumps were still alive, yielding a 100% survival rate. Among the translocated shed pads, 12 of the 25 were still alive in May 2011 for a 48% survival rate. The proportion surviving was significantly higher for the clumps ($\chi^2_1=5.33$, $P=0.021$). Although one clump died between May 2011 and November 2013, for an overall clump survival of 90%, mortality among pads was even greater with only 4 of the 25 (16%) alive at the latter time; final clump survival was highly significant when compared to pad survival ($\chi^2_1=13.73$, $P<0.001$). Among the pads, those surviving to May 2011 were significantly heavier ($t_{14}=2.11$, $P=0.05$; $\bar{x}=34.8 \pm 8.2$ g [SE]) at the time of collection from the source site than those that did not survive ($\bar{x}=16.2 \pm 3.3$ g). By November 2013 survival of pads was no longer significantly related to initial pad weights ($t_3=1.34$, $P=0.14$), although pads that survived to that point generally were heavier ($\bar{x}=45.6 \pm 17.8$ g) than those that died by that time ($\bar{x}=21.2 \pm 4.1$ g). Mean initial pad size tended to be larger among pads that survived to May 2011 ($\bar{x}=42.5 \pm 6.9$ cm²) compared to non-surviving pads ($\bar{x}=29.8 \pm 3.6$ cm²), but the difference was not significant ($t_{17}=1.64$, $P=0.12$). A similar pattern was observed as of November 2013, when pads that remained alive had an initial mean size of 46.2 ± 10.9 cm² compared to non-surviving pads ($\bar{x}=33.9 \pm 4.2$ cm²), but the difference again was not significant ($t_4=1.05$, $P=0.18$).

The clumps consisted of one to five pads when translocated in October 2009. Both clumps and pads began producing new pads in spring 2010. By May 2011, 9 of the 10 clumps (90%) and 7 of the 12 (58%) surviving pads exhibited new growth. At that time, clump size ranged from 3 to 11 pads for a mean of 6.4 per plant, which had doubled to a mean of 12.9 (range 3 – 37) by November 2013. By May 2011, the 12 remaining plants that had originated from translocated pads consisted of an average of 2.2 living pads (range 1 – 7), but the original pad was still alive among only 3 of these. For plants originating as pads, the mean number of living pads more than doubled by November 2013 to 6.0 (range 4 – 8). Factoring in both new growth and mortality of individual pads, the net increase in pad number for clumps by May 2011 averaged 2.3 (range 0 – 7), whereas those planted as pads had a mean net increase of 1.2 (range 0 – 4). As of November 2013, the net gain in total number of pads within clumps averaged 8.8 (range -2 – 33), compared to a net gain of 5.0 pads (range 3 – 7) for plants originating as single pads.

Among the 10 translocated clumps, 2 (20%) produced flowers in spring of 2010, 4 (40%) in 2011, 2 (20%) in 2012, and 4 (40%) in 2013. As of November 2013, the 4 clumps that had produced flowers consisted of a mean 19.3 ± 6.2 pads, compared to 7.6 ± 2.2 among the 5 surviving clumps that had not flowered; this difference was significant at $\alpha=0.10$ ($t_4=1.77$, $P=0.08$). Among the 25 translocated pads, only 1 (4%) flowered; it produced a flower in spring of 2010 and again in 2011, then died before spring 2012. Initially, this had been one of the larger pads and weighed 87 g when planted.

As of May 2011, five pads had been shed from the translocated clumps; all were alive and two of the five (40%) had produced new pads. By November 2013, only one of the five (20%) shed pads seemed to be alive and no additional shed pads were observed. The translocated pads did not shed any pads during the course of the study.

In the three plots in which cattle guards were installed, no damage from cattle was observed on any of the six clumps and seven surviving pads as of May 2011. On the two plots where guards were not installed, potential damage from cattle was observed on three of the four clumps (75%) and on one of the five (20%) surviving pads. Thus, 44.4% of unprotected plants sustained possible damage from cattle in the form of broken pads and possibly an eaten flower. As of November 2013, three of the four surviving pads (75%) were protected by cattle guards; the one clump that died did not have a guard. The majority of mortality occurred during the summer months, irrespective of cattle guards (Figure 2).

Second translocation.—As of November 2013, all 26 translocated, salvaged cactus clumps were still alive (100% survival), and more than two-thirds ($19/28 = 68\%$) of translocated pads survived. As in the first effort, the proportion of clumps surviving was significantly higher than the pads ($\chi^2_1=7.85$, $P=0.0051$). As of November 2013, survival among the pads planted vertically ($8/8 = 100\%$) was significantly higher ($\chi^2_2=5.31$, $P=0.071$) at $\alpha = 0.10$ than that of pads planted on edge ($6/11 = 55\%$) or horizontally ($5/9 = 56\%$). The majority of mortality was in the first two years; during the summer of 2013 only one plant originating as a pad died, but two others that appeared dead in February were determined to be alive in November.

New growth was evident among all clumps ($26/26$; 100%) and many pads ($18/28$; 64%) by July 2011. Among plants still alive in November 2013, the mean number of pads among translocated clumps was $20.1 (\pm 2.3)$ compared with 5.4 ± 0.7 for plants originating as pads. The net gain number in total number of pads per clump cannot be determined because the exact counts were not recorded at the time of translocation; initial clump size

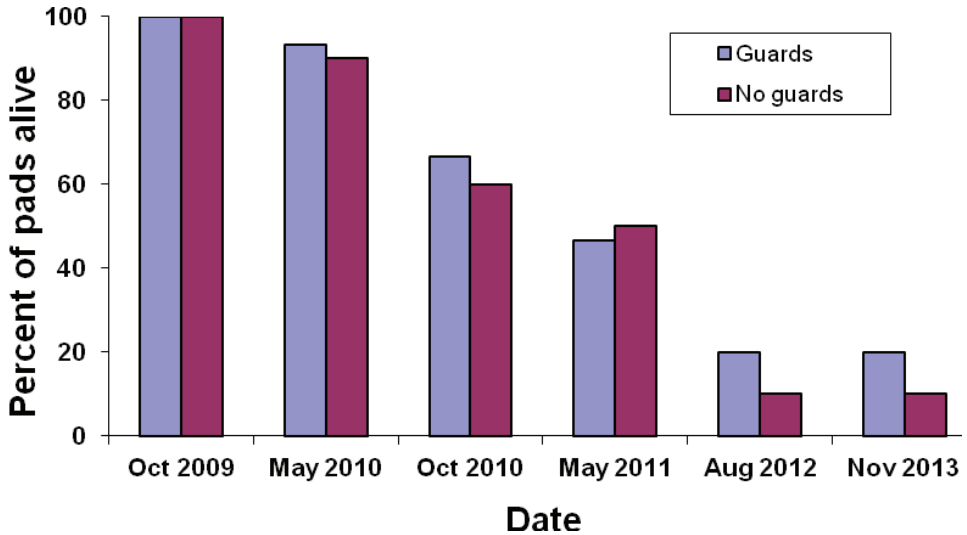


FIGURE 2.—Percentage of surviving translocated Bakersfield cactus pads by monitoring date at the Bena Landfill Conservation Area, Kern County. In October 2009, 15 pads were planted with cattle guards and 10 pads were planted without cattle guards.

was quite variable and often much larger than in the first translocation. For plants originating as single pads mean net gain in pads by November 2013 was 5.4 ± 0.7 (range 0 – 10). The three pad orientations did not differ in net pad gain ($F_{2,16} = 1.87$, $P = 0.19$), although means trended higher for pads planted on edge ($\bar{x} = 5.8 \pm 0.53$) or vertically ($\bar{x} = 4.4 \pm 1.3$) than horizontally ($\bar{x} = 2.6 \pm 1.0$).

By November 2013, the translocated cacti exhibited evidence of both flowering and vegetative reproduction. Eleven of the 26 (42%) translocated clumps produced flowers in 2013 but none of the plants originating as pads flowered. Plants that flowered contained more pads ($\bar{x} = 22.5 \pm 4.2$) in November 2013 than those that had not flowered ($\bar{x} = 10.7 \pm 1.5$; $t_{14} = 2.64$, $P = 0.019$). Three shed pads (two from clumps, one from a plant that had originated as a pad) were observed in February 2013 but did not survive the summer. Four recently-shed pads seen in November 2013 (two each from plants originating as clumps and pads) were producing roots.

DISCUSSION

Translocation and reintroduction are strategies that have been employed in conservation efforts for a number of rare plant species (Allen 1994, Given 1994, Falk et al. 1996). These strategies offer immense potential for re-establishing populations on formerly occupied sites or for establishing new populations at suitable sites. However, considerable expense and risk are always involved when moving individuals to new sites; many such efforts have failed and, as a result, all reasonable efforts should be taken to reduce this risk and also to avoid any detrimental effects to source populations. Only through long-term monitoring that documents reproduction in the founders and the presence of new generations can translocations be deemed successful in establishing self-sustaining populations (Fiedler 1991, Godefroid et al. 2011, Drayton and Primack 2012).

Although shed pads that had developed roots were observed in both translocated populations, their long-term survival is uncertain. Seedlings have not yet been observed in either of the present efforts; even if seeds are produced, recruitment for Bakersfield cactus via seedlings may occur only in years of above-average rainfall (Twisselmann 1967, Benson 1982). Ideally, the production of new plants eventually will exceed mortality of existing plants, resulting in expansion of the new populations. Sexual reproduction would expand the genetic diversity of the new populations, helping to increase their long-term viability, but we do not know if any of the flowers produced viable seed. Monitoring of both of the newly-established populations will continue periodically to determine if future generations become established through either sexual or vegetative reproduction.

Survival rates between the two efforts are not directly comparable because our techniques differed somewhat, as did the timing of collection and translocation. The pads in the second effort likely had a higher initial moisture content because they were still attached to clumps until the day of translocation and because substantially more rain had fallen in the preceding month compared to the first effort (California Irrigation Management Information System [CIMIS] 2013). Also, the second translocation effort took place more than a year after the first and, as a result, the data reflect mortality rates over only 34 months, compared to 49 months for the first effort. However, the second translocation effort had undergone two drier-than-normal water years by that time (CIMIS 2013), and the survival rate of comparably-oriented (horizontal) pads remained similar to that observed at 19 months in the first effort, which followed two wetter years.

These efforts were pilot projects, and only the first translocation was designed as a study to test the technique. However, both of these translocations incorporated several of the success factors identified in the literature (Fiedler 1991, Godefroid et al. 2011, Drayton and Primack 2012) including introduction into protected areas of similar habitat, using a stable source population, consideration of species biology, and appropriate management of the recipient site. Additional published success factors should be considered if translocations of Bakersfield cactus are attempted on a broader scale, including the use of larger transplants, outplanting a greater number of individuals, and determining the appropriate source populations based on genetic information.

Clumps exhibited a greater potential for successful translocation than pads, especially in the first effort. Clumps have more resources than pads; an existing root system allows for much faster access to soil moisture, and more stored moisture is available to help survive during dry periods. In contrast, individual pads must expend resources developing a root system, and they are more susceptible to mortality under dry conditions due to a limited reserve of stored moisture. Larger and heavier pads presumably have a greater moisture reserve than smaller pads, as suggested by the observed trends, but the low overall pad survival rates obscured any significant relationship. A similar trend was observed in Bakersfield cactus that was propagated for a restoration project, however, where 100% of large pads survived and only small pads died during the first year (Clendenen and Erickson 2013). The skewers also may have contributed to moisture loss among pads in the first translocation.

We recommend translocating clumps rather than individual pads during future translocation efforts, even though the former is more labor intensive. Because removal of established clumps from a source population constitutes more of an adverse impact than the removal of pads, we suggest collecting pads and propagating them into small clumps with established roots prior to outplanting, a technique that has been implemented

successfully at the Wind Wolves Preserve (Clendenen and Erickson 2013). Measures that could improve the success rate include choosing larger, heavier pads for propagation; collecting pads during the winter when they have a greater moisture content; conducting translocations during the fall or early winter to take full advantage of winter precipitation; and providing supplemental water to outplanted cactus clumps during the summer, especially following growing seasons with below-average precipitation. Nonetheless, we recommend that any future efforts to establish populations of Bakersfield cactus begin with much higher numbers of small clumps to compensate for potential mortality.

Genetic considerations also may influence the number of individuals used to establish a new population. Combining material from multiple source populations has been suggested as a means to increase genetic diversity during establishment of new populations (Godefroid et al. 2011). However, only one study (Smith 2013) has been conducted on the population genetics of Bakersfield cactus, and until more is known about whether genetic diversity is limiting for this species and whether local adaptation has occurred, we caution against mixing material from multiple sources. Similarly, caution should be used before establishing new Bakersfield cactus populations outside of the historical range of the taxon. Either decision would require concurrence from federal and state agencies, and should occur only after consultation with the recovery team for this species and other experts.

We recommend that guards be installed at the time of translocation if the area is grazed by livestock. The cattle guards we used at the BLCA appear to have effectively allowed the translocated cactus to become established. Although cattle may damage unprotected cactus plants, cattle grazing may provide an overall benefit to Bakersfield cactus populations by facilitating pad dispersal and reducing competition from non-native grasses as well as the threat of wildfire (USFWS 1998, Cypher et al. 2011b). In natural populations where grazing has reduced the ground cover and exposed soil, some of the pads detached by hoof contact develop roots and produce new cactus plants (Cypher et al. 2014). Cattle guards are a solution that can protect newly translocated plants while reaping other benefits of cattle grazing; once the young plants grow beyond the guards, opportunities for dispersal would become available.

A number of other elements incorporated into our efforts will be important for any future translocations. Competing vegetation must be controlled both before and after transplanting. Source populations should be relatively large with evidence of reproduction. Recipient sites should be permanently conserved through ownership by a federal or state conservation agency, or protected by a conservation easement. In addition, the sites should be appropriately managed to reduce threats to cactus populations from non-native plants, fire, or external influences. Advance planning for establishment of new populations will be essential to determine the appropriate sources of genetic material, secure permits, allow for collection of pads when they have the greatest water content, and propagate pads into well-rooted clumps before outplanting.

Despite the short-term success of these efforts, this technique is not recommended as mitigation, and translocation should not be considered a substitute for minimization of project impacts or protection of occupied habitat (Fielder 1991, Howald 1996). Translocation and introduction are best viewed as potential strategies for remediating habitat fragmentation within the range of Bakersfield cactus, which could contribute substantially to the conservation and ultimate recovery of Bakersfield cactus. Indeed, recovery of Bakersfield cactus, as defined in the recovery plan for this species (USFWS 1998), may already be precluded due to continuing habitat loss. Nevertheless, translocation

and establishment of Bakersfield cactus on permanently conserved and appropriately managed sites could substantially advance conservation and recovery of this species.

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Identifying practical, small-scale disturbance to restore habitat for an endangered annual forb

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We sought to identify appropriate treatments to restore a small, urban patch of habitat for the endangered *Clarkia franciscana* (Presidio clarkia) in serpentine grassland. Goals included identifying effective and pragmatic treatments for introducing disturbance to the site and determining whether treatments used to establish Presidio clarkia would be appropriate in areas already occupied by this endangered serpentine endemic. This experiment tested fall burning, fall flaming, fall mowing with thatch reduction, fall scraping, fall tarping, spring burning, and spring mowing with and without thatch reduction. Half of all treated plots were seeded with clarkia. Clarkia density and vegetation composition were measured one and two years after treatment. Fall scraping, fall tarping, and fall flaming stood out as the most effective methods for increasing density. Fall scraping and fall flaming enhanced clarkia populations in unseeded plots where clarkia was initially present. In Year 1, these three treatments were also most successful in reducing annual grass cover and decreasing nonnative plant cover. Although other studies have shown spring treatments to be useful for reducing annual grass and thatch, and increasing native forbs, this study found that treating in late fall, after annual grasses had germinated, was critical for this site – and not, as was previously presumed, harmful to the clarkia.

Keywords: *Clarkia franciscana*, disturbance, endangered species, restoration ecology, serpentine grasslands, urban preserve

Lands set aside for conservation are still threatened by habitat fragmentation (Quinn and Hastings 1987, Bolger et al. 1991), invasive species (Mooney and Hobbs 2000), and climate change (Vitousek et al. 1997), and require active management to conserve biodiversity

(Hobbs and Huenneke 1992, Meffe and Carroll 1997). In many cases, management seeks to restore or mimic historic disturbance regimes, such as grazing or fire. These landscape-scale techniques can be difficult to implement in small, urban settings. Managing for rare species may increase the complexity of introducing disturbance, because of the potential risk of harming individuals (Pendergrass et al. 1999, Marcot and Sieg 2007).

This project sought to identify appropriate treatments to enhance habitat for the endangered *Clarkia franciscana* Harlan Lewis & Raven (Presidio clarkia) at a 6-ha remnant serpentine grassland in the Presidio of San Francisco. Goals included identifying effective and pragmatic treatments for the small, urban site; quantifying the value of seeding treated areas; and determining whether treatments used to establish Presidio clarkia would be appropriate in areas already occupied by this endangered serpentine endemic.

Serpentine soils are generally characterized by low levels of macronutrients such as nitrogen and phosphorus, low calcium to magnesium ratios, and high concentrations of heavy metals such as nickel, chromium, and cobalt. This unusual chemistry of serpentine soils creates a harsh environment for plant growth, and many of the species adapted to serpentine soils are rare (Kruckeberg 1984, 2006).

California grasslands are dominated by invasive plants, primarily annual grasses of Mediterranean origin (Bartolome et al. 2007). In many cases these invasives are less competitive in serpentine soils due to the unique composition of those soils (Huenneke 1989, Huenneke et al. 1990); as a result, California serpentine grasslands act as refuges for many native grasses and forbs (Murphy and Ehrlich 1989).

Invasives are still a threat in serpentine grasslands, particularly where nutrient addition alters the chemically harsh serpentine soil. Both dry nitrogen deposition from air pollution (Weiss 1999) and organic matter from tree plantings are adding nutrients to serpentine soils in the Presidio (NPS 2001).

Presidio clarkia is a slender, erect, herbaceous annual of the evening-primrose family (Onagraceae) that grows up to 40 cm tall with few, very small and narrow leaves. It has unlobed and unclawed petals, which are pink with a red blotch at the base. Generally restricted to serpentine soils, Presidio clarkia blooms from May to July. In a study of reproductive success at the Presidio habitat, each Presidio clarkia plant produced an average 872 ± 84 seeds (Bode 2000). These seeds are presumed, but not confirmed, to be “long-lived” (Roof 1972).

Until the 1980s, Presidio clarkia was believed to be restricted to the Presidio of San Francisco, but several small populations are known to exist in the Oakland Hills in Alameda County (USFWS 1998, EBRPD 2008). Presidio clarkia was listed as an endangered California taxon in 1989 (Sanguamphai 1989) and as federally endangered in 1998 (USFWS 1998). It is known from only five sites, two of which are in the Presidio.

In large habitat areas, various management techniques such as grazing (Heady 1988, Huenneke et al. 1990, Weiss 1999, Safford and Harrison 2001, Weiss et al. 2010), burning (Harrison et al. 2003, Weiss et al. 2010), and mowing (Maron and Jefferies 2001, Weiss 2002, EBRPD 2008) have effectively reduced invasives and increased biodiversity on serpentine soils. However, these techniques are challenging in small habitat fragments, particularly those in urban settings such as that of the Presidio. Burning poses the risk of an escaped fire, and air quality concerns are high in this urban matrix. Grazing is difficult at this small site, which includes high numbers of recreational users, including dog walkers. However, managers agreed to introduce these treatments if they proved more effective than alternative

simpler treatments at improving habitat for the clarkia. Stakeholders agreed to test a variety of treatments concurrently to compare efficacy and pragmatism on the site.

Because the clarkia is listed as endangered, all treatments must minimize take. It was assumed (though not previously tested) that management during the growing season would harm established clarkia. It was believed that treatments in habitat occupied by clarkia should take place only in the period between set seed and germination. This study tested treatments throughout the season, both in potential clarkia habitat and in areas where clarkia was already present. The suitability of each treatment was measured by comparing species composition in each plot pre- and post-treatment.

METHODS

Study area.—This study was conducted in the Presidio of San Francisco, at a 6-hectare remnant serpentine grassland named Inspiration Point (37° 47' 32.80" N, 122° 27' 26.75" W, WGS 1984). Inspiration Point is the type locality for Presidio clarkia, which was first collected in 1956 (Lewis and Raven 1958). Average October–May precipitation from 1960 to 2013 at the Presidio is 56.5 cm. Precipitation during the study period was lower than average (Table 1) (PRISM 2014). The average annual minimum temperature is 9.9° C; the average maximum is 17.6° C (WestMap 2009). Summers are relatively cool and foggy.

Growing season	Mean precipitation
	Oct–May (cm)
2006-2007	37.7
2007-2008	46.5
2008-2009	47.9
Mean 1960–2013	56.5

TABLE 1.—Precipitation during the study period (2006–2009) compared with long-term mean precipitation (1960–2013) at the Presidio of San Francisco, California.

The 600-hectare Presidio, part of the Golden Gate National Recreation Area, lies at the northern tip of the San Francisco Peninsula in a densely populated urban setting. It receives heavy recreational use from walkers, joggers, cyclists, and dog owners. A former military post, the Presidio is a cultural landmark with a number of historic buildings. In 1994, the region was transferred to the National Park Service, which now manages the coastal areas; the Presidio Trust now manages the interior 80% of lands. Remaining wild lands at this site are small and fragmented, and provide habitat to more than 350 native plant species, five of which are federally listed as threatened or endangered, including the Presidio clarkia.

During the late 19th and early 20th centuries, tree planting efforts throughout the Presidio converted the original coastal prairie into stands of Monterey pine (*Pinus radiata*), Monterey cypress (*Hesperocyparis macrocarpa*), coast redwood (*Sequoia sempervirens*), and blue gum eucalyptus (*Eucalyptus globulus*). The resulting litter and duff created thick organic layers that enriched the nutrient-poor serpentine soil, and nonnative annual species invaded

remaining grassland patches (NPS 2001). Annual grasses and thatch have been shown to have negative effects on forbs (Huenneke et al. 1990, Coleman and Levine 2007).

Although Presidio clarkia population estimates are unavailable prior to 1970, it is reasonable to assume based on observations that populations of this and other serpentine-restricted species dropped in the newly shaded, nutrient-rich, invaded environment (USFWS 1998, NPS 2001). The invasive grasses on site — including *Avena barbata* Link (wild oats), *Festuca perennis* Lam. (Italian ryegrass), and *Bromus hordeaceus* L. (soft chess) — also threaten other native species, both common and rare (USFWS 1998, Strathmann 2001). Controlling invasive grass to enhance clarkia habitat should also benefit these other native plants.

Experimental Design.—The experiment uses a Before-After Control-Impact (BACI) randomized block design, with three blocks placed in Presidio clarkia habitat. The blocks were placed in three areas that captured a range of clarkia habitat variability on site: a southeast-facing slope west of the main Presidio clarkia population, a gentle east-facing slope with high bunchgrass cover, and a northeast-facing slope. Each block has nine treatments in 1x1 m plots spaced 0.5 m apart, with nine replicates of each treatment, for a total of 243 treatment plots. Within each large block, 3x3 sub-blocks were defined to account for fine-scale variability and to disperse the treatments throughout the large block, with treatments randomly assigned within the sub-blocks.

We randomly assigned nine treatments a plot within each sub-block: control (no treatment), fall burning, fall flaming, fall mowing with thatch reduction, fall scraping, fall tarping, spring burning, spring mowing with thatch reduction, or spring mowing without thatch reduction. Treatments were selected because they have improved serpentine habitats at other sites, or because they seemed logical tools for reducing the annual grasses and associated thatch that are likely the primary threats to the clarkia in the open grasslands. Baseline percent cover of all species was recorded before treatments in April 2007, and plot treatments took place spring and fall 2007.

Following treatment, the left half of each plot was broadcast with 100 Presidio clarkia seeds (collected throughout the site in summer 2007) in late November 2007 before major rains. Density of Presidio clarkia was measured in spring 2008 by counting individuals on each half of the 1-m² plots. To reduce edge effects, species composition data were collected from a 0.25-m² quadrat at the center of each 1-m² plot. Plots were re-examined in late April or early May 2008 to determine treatment effects on clarkia density and community composition. Plots were read a final time in early May 2009 to determine which effects persisted two years after treatment.

Statistical analyses.—All analyses were conducted in JMP 10.0 (SAS Institute). Clarkia counts were log(x+1) transformed and analyzed with ANOVA, with treatment and block effects. Plots were divided into those that had clarkia in 2007 and those with none, and the seeded and unseeded halves of the plot were analyzed separately. Individual treatments were compared pairwise by Tukey HSD post-hoc-tests.

Vegetation composition data were analyzed as a BACI design, within a randomized block ANOVA, with Tukey HSD post-hoc testing. For each species and abiotic metric, data were log(x+1)-transformed. For Year 1 results, log-transformed 2007 cover was subtracted from transformed 2008 cover to determine the change in cover at each treatment plot. For Year 2 results, transformed 2007 cover was subtracted from transformed 2009 cover. We set $\alpha = 0.05$ as the threshold for significance for all statistical tests.

Treatment methods and timing.—Each plot was treated once in the baseline year (2007). Plots were not treated in Year 1 (2008) or Year 2 (2009).

Spring burning was conducted in July, as soon as thatch was dry enough to carry a fire, but before annual grass seeds had shattered. The fire carried well, consuming all thatch, but it is unknown whether it was hot enough to destroy seeds. Burns were conducted in burn boxes, constructed of four $\sim 1 \times 1.5$ -m metal sheets bolted together, to contain the fire treatment in the desired plot and enhance safety.

Spring mowing (with and without thatch reduction) was conducted in early May when the *Festuca* was at a soft to medium dough stage (seeds had formed, but were still soft). Plots were mowed with a stringcutter. In plots with thatch reduction treatment, biomass was raked from the plot.

Fall burning was conducted in October, during dry conditions after seeds had fallen. Burn boxes again were used. The burn thoroughly consumed thatch and standing dead biomass.

Fall flaming took place in November on a cold, dewy day just before the season's first significant rain; burn boxes were not required. Nonnative annual grasses were about 5-cm tall. A propane torch was passed over live plants and held in place long enough to burn litter, a more intense treatment than the typical method of passing the torch quickly over the plant until it wilts.

Fall mowing with thatch reduction was conducted in late November, using the method described for spring mowing with thatch reduction. The purpose of fall mowing is largely to break up the thatch. Nonnative annual grasses were about 5-cm tall, and no major rains had fallen.

Fall scraping took place in November; annual grasses were about 5 cm tall. A McLeod was used to remove 8–10 cm of topsoil. Fall tarping also occurred in November. Plots were securely tarped using Lumite® weed barriers. Annual grasses had germinated and grown to about 5-cm tall. Tarps were left on for about six weeks, with the aim of killing annuals but not perennials.

With the exception of the tarped plots, each plot (including the control) was seeded with *Presidio clarkia* in late November 2007, following completion of all 2007 treatments. Tarped plots were seeded upon tarp removal, in January 2008. A total of 100 seeds were broadcast on half of each plot. Seeds were mixed with clean sand for better dispersion, and then broadcast on the left half of each square meter plot, facing uphill.

When considered in their entirety, the control plots do not represent natural recruitment (defined as plants establishing without active management); only the unseeded, right half of the control plots do. Effects of seeding were analyzed only in Year 1. By Year 2 it was assumed plants that reached reproductive maturity on each half of the 1-m² plots would naturally drop seed into the adjacent half, obscuring the original seeding effects. To examine persistence, we counted the number of *clarkia* in the entire plot in Year 2.

RESULTS

Presidio clarkia density.—In unseeded areas, fall scraping significantly increased *clarkia* (Figures 1 and 2; $F_{8,232}=4.34$, $P<0.0001$). The seeded half of the plots showed a slightly different pattern. Fall scraping, fall flaming, and fall tarping significantly increased *clarkia* density (Figures 3 and 4; $F_{8,232}=34.2$, $P<0.0001$).

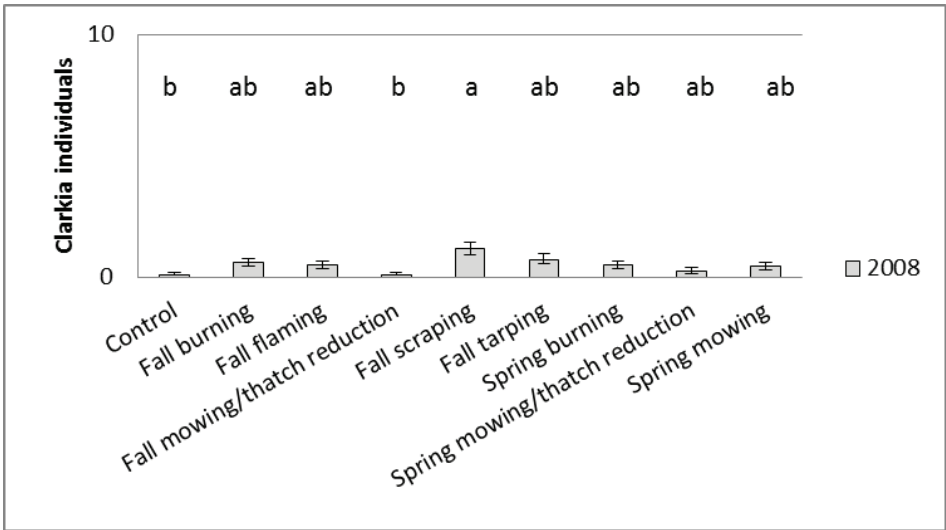


FIGURE 1.—Presidio clarkia density in unseeded plots, initially absent, \pm 1 SE. Levels not connected by the same letter are significantly different.

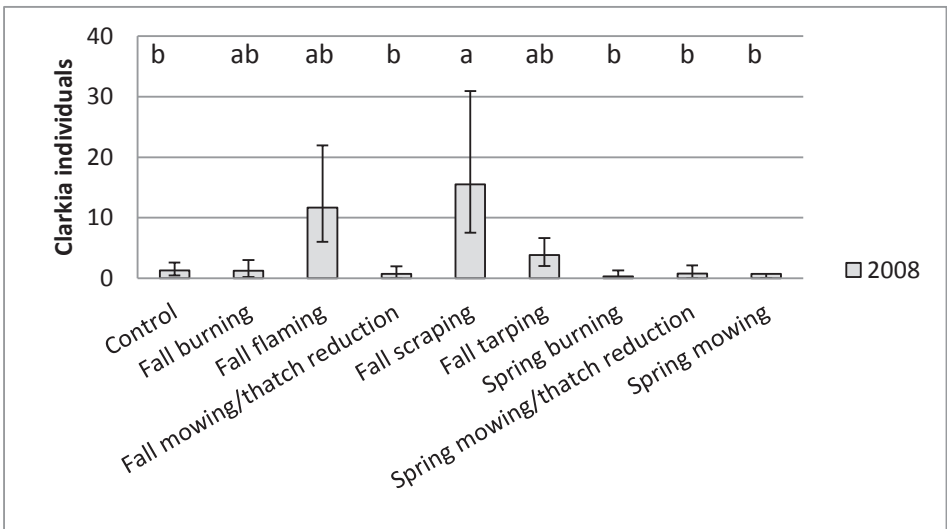


FIGURE 2.—Presidio clarkia density in unseeded plots, initially present, \pm 1 SE. Levels not connected by the same letter are significantly different.

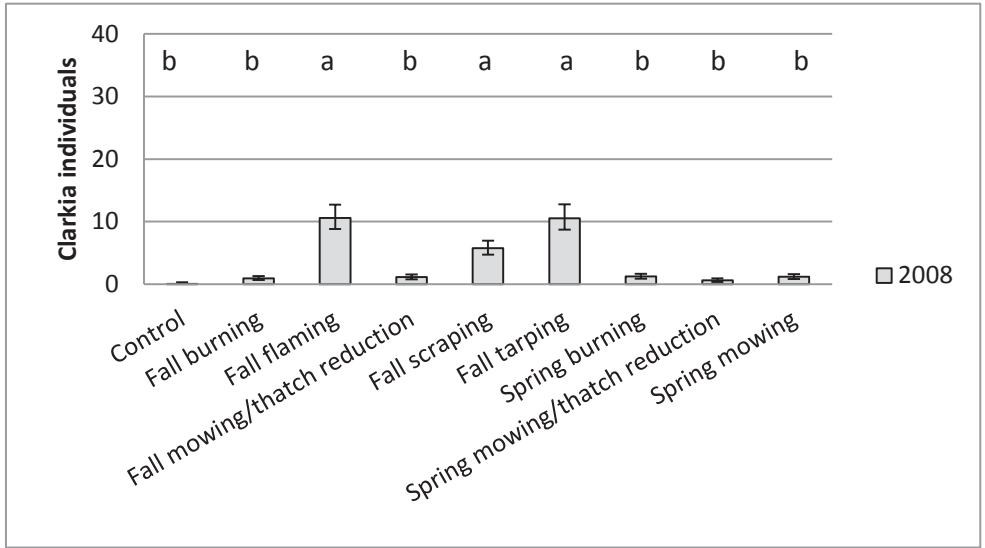


FIGURE 3.—Presidio clarkia density in seeded plots, initially absent, \pm 1 SE. Levels not connected by the same letter are significantly different.

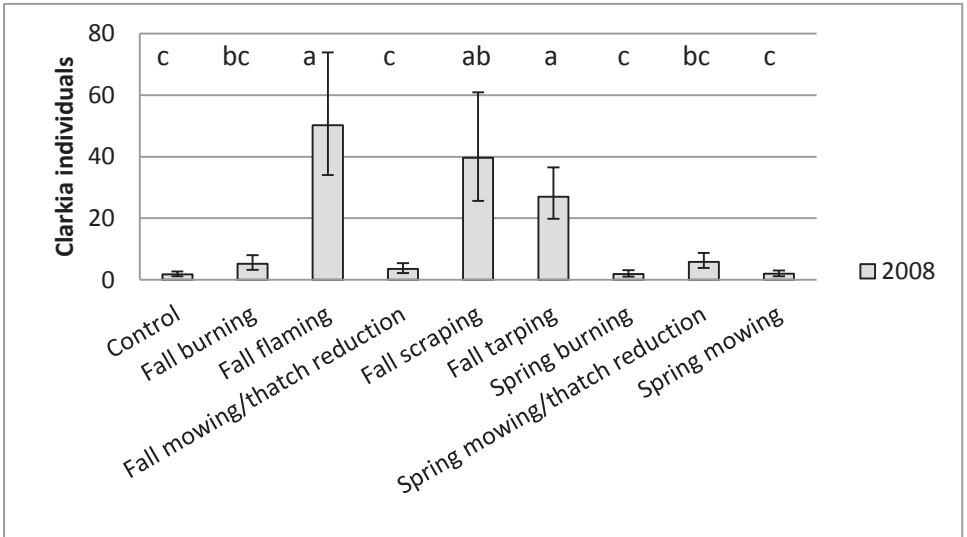


FIGURE 4.—Presidio clarkia density in seeded plots, initially present, \pm 1 SE. Levels not connected by the same letter are significantly different.

Overall clarkia density showed similar patterns. Combining the seeded and unseeded halves of plots, fall flaming, fall scraping, and fall tarping showed the largest increases in clarkia numbers in Year 1 ($F_{8,232}=29.83, P<0.0001$). By Year 2, only fall flaming and fall tarping remained significant (Figure 5; $F_{8,230}=5.43, P<0.0001$). Density throughout the site was greatly increased as clarkia persisted throughout areas where it was initially absent (Table 2).

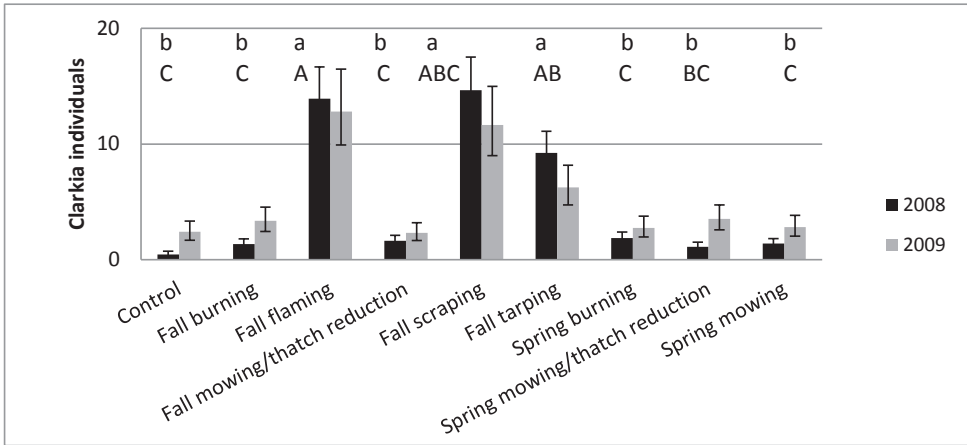


FIGURE 5.—Presidio clarkia density per treatment, seeded and unseeded plot halves combined, ± 1 SE. Levels not connected by the same letter are significantly different.

TABLE 2.—Number of study plots containing clarkia by Year 2, compared to initial presence or absence at Inspiration Point, Presidio of San Francisco, California, 2007–2009.

Block	Clarkia status	2007	2009
1	absent	38	33
1	present	43	48
2	absent	78	13
2	present	3	68
3	absent	81	30
3	present	0	51

Cover by guild.—Guilds are defined as functional groups based on life form, further divided between native and non-native species. We present data on key guilds of this system: nonnative annual grass, native annual/biennial forbs, and native perennial grasses. Total native cover, total nonnative cover, bare ground, and thatch are also examined. There was a significant difference in non-native annual grass cover among treatments ($F_{8,230}=26.5$, $P<0.001$). In Year 1, the control increased 42.33% in absolute cover, with the wild oats reaching more than 1.7 m high in many plots. The most effective treatments in reducing annual grass cover were fall tarping, fall scraping, and fall flaming. The spring mow with thatch reduction also had significantly less annual grass than the control (Tukey HSD $P<0.05$). By Year 2, treatment effects persisted (Figure 6; $F_{8,232}=3.54$, $P<0.001$), but only plots treated with fall scraping were significantly different from the control (Tukey HSD $P<0.05$).

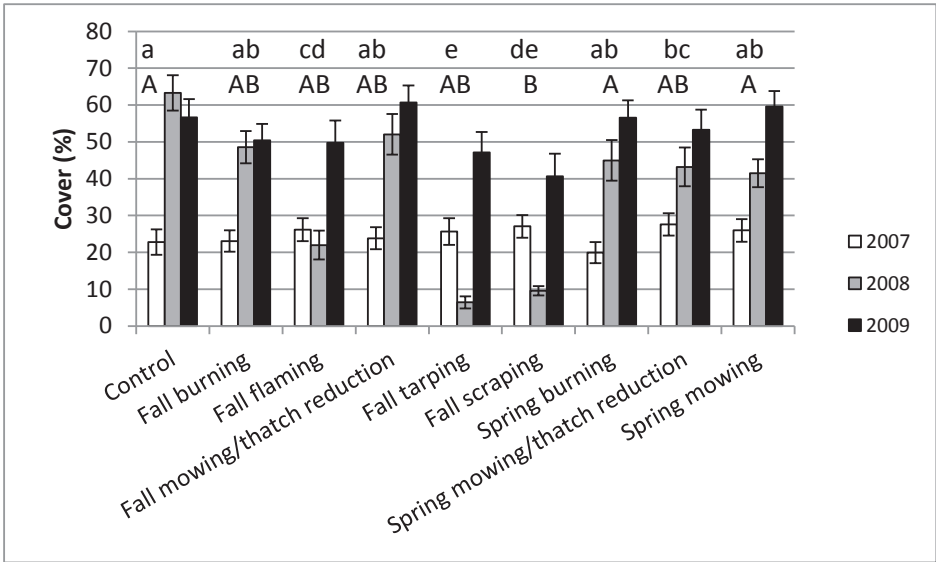


FIGURE 6.—Non-native annual grass, $P<0.0001$ (2008); $P=0.0008$ (2009). Error bars show ± 1 SE. Levels not connected by the same letter are significantly different. Lower case letters denote changes from baseline to Year 1 (2008); capital letters denote changes from baseline to Year 2 (2009).

There was a significant difference in bare cover among plot treatments. In Year 1, fall scraping yielded the greatest bare ground. Fall flaming, fall tarping, spring burning, and spring mowing also resulted in significant increases in bare cover (Figure 7; $F_{8,232}=10.3$, $P<0.0001$). By Year 2, treatments were significant ($F_{8,232}=3.08$, $P=0.0025$), only fall scraping was significantly different from the control (Tukey HSD $P<0.05$).

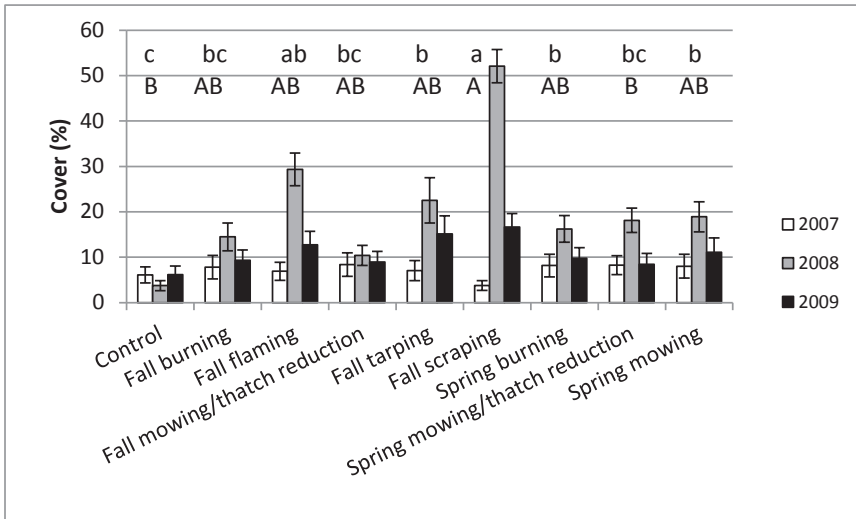


FIGURE 7.—Bare cover, $P < 0.0001$ (2008); $P = 0.0025$ (2009). Error bars show ± 1 SE. Levels not connected by the same letter are significantly different. Lower case letters denote changes from baseline to Year 1 (2008); capital letters denote changes from baseline to Year 2 (2009).

In Year 1, fall tarping showed significantly more thatch than the other treatments (Figure 8; $F_{8,232} = 12.1$, $P < 0.0001$). However, measurements of cover did not account for variation in biomass; while tarped plots appeared to have high levels of thatch and less live cover, the tarps also appeared to have broken down the thatch biomass. Measuring only absolute cover rather than biomass may have misrepresented these results. In Year 2, the overall treatment effect was significant ($F_{8,232} = 2.5$, $P = 0.0125$; Tukey HSD $0.05 < P < 0.10$).

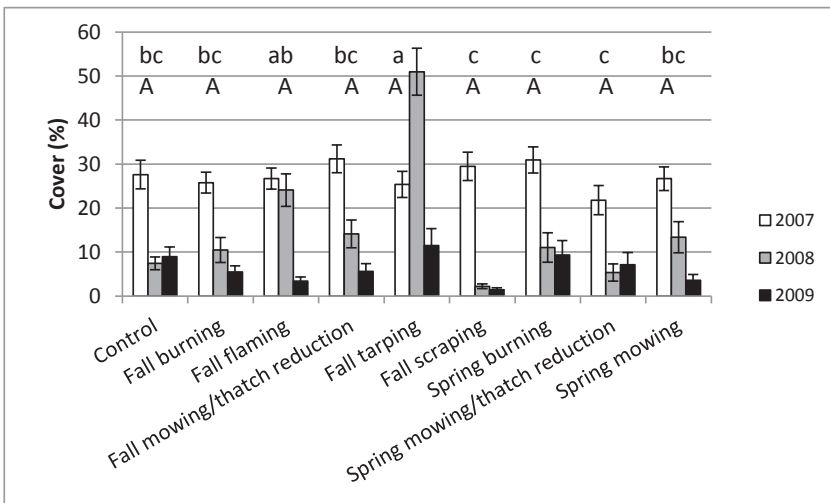


FIGURE 8.—Thatch cover, $P < 0.0001$ (2008); $P = 0.0125$ (2009). Error bars show ± 1 SE. Levels not connected by the same letter are significantly different. Lower case letters denote changes from baseline to Year 1 (2008); capital letters denote changes from baseline to Year 2 (2009).

Total native cover declined across all treatments and the control, likely due to below-average precipitation in 2007–2009 (Table 1). There was no difference in total native cover among treatments in 2008 ($F_{8,232}=0.75, P=0.65$) or in 2009 year ($F_{8,232}=0.79, P=0.61$).

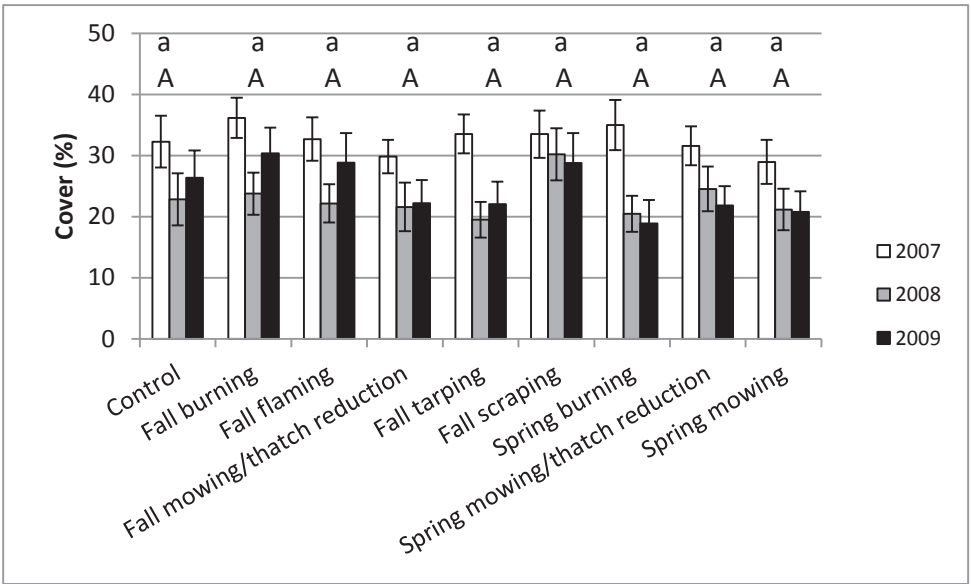


FIGURE 9.—Total native cover, $P=0.65$ (2008); $P=0.6116$ (2009). Error bars show ± 1 SE. Levels not connected by the same letter are significantly different. Lower case letters denote changes from baseline to Year 1 (2008); capital letters denote changes from baseline to Year 2 (2009).

Changes in annual grass cover drove changes in total nonnative cover (Figure 10). In Year 1, nonnative cover was significantly lower with the fall flaming, fall tarping, and fall scraping ($F_{8,232}=24.6, P<0.0001$). By Year 2, treatment effects were significant ($F_{8,232}=3.05, P=0.0027$) but no treatment plots differed significantly from the control (Tukey HSD $P>0.05$). In Year 1, plots treated with spring mowing with thatch reduction, fall flaming, fall tarping, and fall scraping showed significantly more native annual/biennial forbs than the control ($F_{8,232}=6.63, P<0.0001$). However, by Year 2, treatment effects were no longer significant ($F_{8,232}=1.86, P=0.68$). Additionally, even the control plots showed decreases from the baseline. The dry years of 2008 and 2009 appear to have greatly decreased native forb cover (Figure 11).

In Year 1 treatment effects led to significantly different native perennial grass cover (Figure 12; $F_{8,232}=3.33, P=0.0012$), but no treatment was significantly different from the control (Tukey HSD $P>0.05$). In Year 2, treatment effects persisted ($F_{8,232}=3.21, P=0.0018$), but no treatment was significantly different from the control (Tukey HSD $P>0.05$). Species richness was also investigated for each guild, with no effects lasting into Year 2 (data not shown).

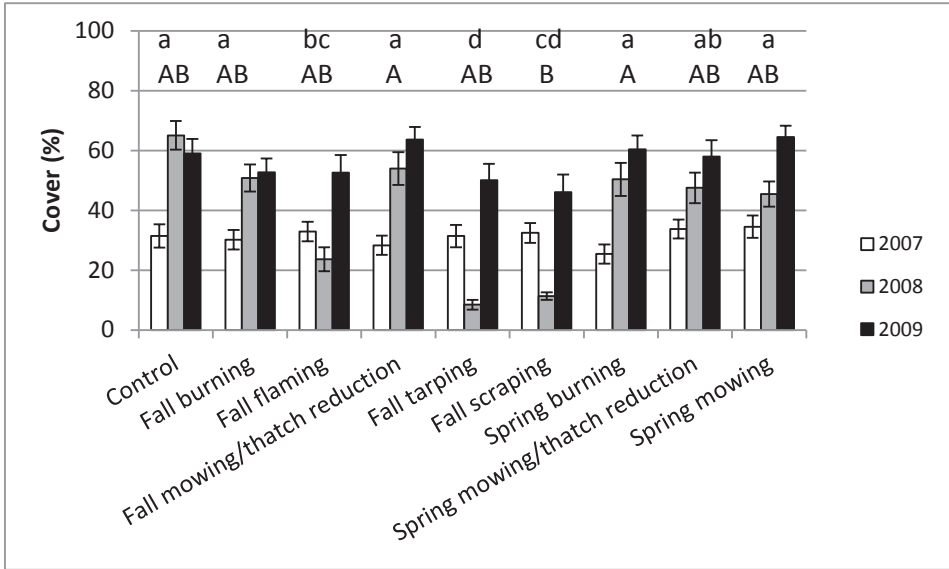


FIGURE 10.—Total non-native cover, $P < 0.0001$ (2008); $P = 0.003$ (2009). Error bars show ± 1 SE. Levels not connected by the same letter are significantly different. Lower case letters denote changes from baseline to Year 1 (2008); capital letters denote changes from baseline to Year 2 (2009).

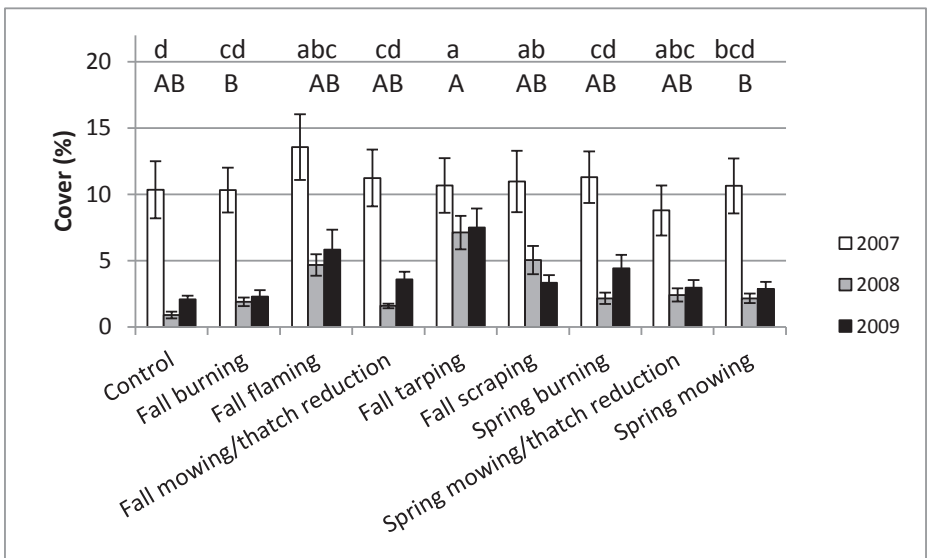


FIGURE 11.—Native annual and biennial forb cover, $P < 0.0001$ (2008); $P = 0.068$ (2009). Error bars show ± 1 SE. Levels not connected by the same letter are significantly different. Lower case letters denote changes from baseline to Year 1 (2008); capital letters denote changes from baseline to Year 2 (2009).

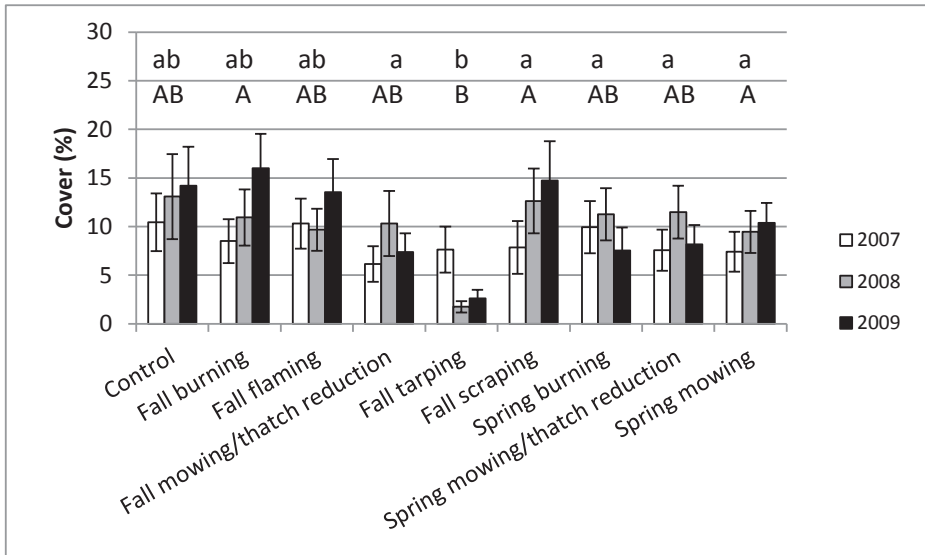


FIGURE 12.—Native perennial grass cover, $P=0.0012$ (2008); $P=0.0018$ (2009). Error bars show ± 1 SE. Levels not connected by the same letter are significantly different. Lower case letters denote changes from baseline to Year 1 (2008); capital letters denote changes from baseline to Year 2 (2009).

DISCUSSION

By all measures, fall scraping, fall tarping, and fall flaming stood out as the most effective methods for increasing the number of Presidio clarkia at Inspiration Point. These three treatments were most successful in increasing one-year density in seeded clarkia plots. Fall scraping enhanced clarkia populations in the unseeded plots, although at much lower total numbers compared with seeded plots. In Year 1, these three treatments were also most successful in reducing annual grass cover and decreasing nonnative plant cover — standard measures of habitat suitability for a native forb such as the clarkia. Along with other treatments, they also significantly increased bare ground in Year 1. Plots treated with the three most successful treatments were visually identifiable in both years by their low cover of annual grasses.

Two years after treatment, some effects were muted. Only fall scraping had significantly more bare ground and less annual grass in all blocks. Effects persisted in the critical parameter of clarkia density; fall flaming and fall tarping were significantly different from the control, with fall scraping also showing an increase, albeit not significant.

Perhaps most important, clarkia persisted in newly introduced areas. While disturbance was key to establishing this ruderal species, clarkia persisted after other parameters returned to their original state. After only two years, it is not yet clear how long clarkia will persist before requiring additional disturbance; this is being studied further.

Post-germination treatments most successful.—Although other studies have shown spring treatments to be useful for reducing annual grass and thatch and increasing native forbs (Weiss 1999, 2002; Naumovich et al. 2009), that was not the case in this study. With spring treatments, annual grasses recovered too quickly to allow clarkia recruitment. Treating in late fall, after the annual grasses had germinated, appears critical for this site.

Fall treatments that did not address the flush of newly germinated annual grasses were less successful. The fall fire was conducted before the rainy season and subsequent grass germination. The fall mow with thatch reduction took place after germination, but it was designed to reduce thatch, not to affect germinated plants. These results are a significant departure from other studies, and reflect the critical importance of timing for weed control treatment. These findings all demonstrate the importance of assessing the suitability of treatments for any given location.

Seeding effectiveness.—Where active seeding did not occur, a low but statistically significant number of clarkia was present in fall scraped plots where the plant had initially been absent, indicating the treatment may have stimulated a dormant seedbank. It was not surprising to find the clarkia had difficulty recruiting on its own, since California native forbs are often seed-limited (Seabloom et al. 2003).

Where active seeding did occur, the three successful treatments increased clarkia density regardless of whether it had been present initially. These results suggest that efforts to introduce clarkia into unoccupied areas should include both habitat treatment and active seeding. Given a limited number of seeds available for planting, increasing the distribution of clarkia into areas with very low or no numbers should be a higher priority than seeding densely occupied areas.

Active management of an endangered species.—At Inspiration Point, clarkia will likely require ongoing management to address the continuous threat of invasive species. Gains from management treatments are expected to be short-term as annual grasses reinvade treated areas.

This experiment addressed the concern that some habitat treatments might be inappropriate in areas occupied by clarkia because of the threat to existing individuals of that species. Despite the fact that the most successful treatments took place during the growing season, our results show that the species responded positively to fall scraping, fall flaming, and fall tarping, when it was thought that the forb might be vulnerable to treatment. While some clarkia individuals may have been harmed by the treatments, overall habitat and density were both improved.

Site-specific studies.—Regional experiences and the literature suggested that all of the experimental treatments would reduce thatch and increase bare ground (Weiss 1999, 2002; Naumovich et al. 2009), and many were also expected to decrease annual grass (Brown and Smith 2000, Tu et al. 2001, Moore 2004, DiTomaso and Johnson 2006). For example, a single spring mow with thatch reduction in the serpentine grasslands at Edgewood Natural Preserve (about 45 km south of the Presidio) significantly reduced annual grass and thatch for three years, leading to an increase in native forb cover (Weiss 2002). A spring burn on Tulare Hill (about 100 km south of the Presidio) had similar results, although the effect lasted only two years (Weiss et al. 2010).

Many of these treatments did not prove effective at this study site. The relatively cool and wet weather at Inspiration Point lengthens the grass-growing season, making it more difficult to exert control with a single treatment. Indeed, other researchers have found burning and grazing to be less successful in coastal grasslands as compared with valley grasslands (Hatch et al. 1999). While the less successful treatments could likely be adjusted to be more effective on the site (i.e., mowing several times per season), the discovery of three techniques that work so well to establish clarkia and its preferred habitat eliminates the need to find other solutions.

Grasslands are known for spatial and temporal variability (Hobbs et al. 2007). It is possible that responses to treatments could be specific to the phenology and weather of the years in which they were performed. Precipitation was below average during the study period (37.7 to 47.9 cm compared with an average of 56.7 cm; Table 1). A similar study of *Clarkia franciscana* at Redwood Regional Park in Oakland, about 30 km east of the Presidio, showed that clarkia numbers were closely related to total precipitation from 2008–2013 ($r^2 = 0.90$) (Naumovich et al. 2014). While enough data are available to yield informed management decisions, it is important to recognize the role of annual variability and climate change in affecting plant communities. It is possible that treatment results would be different in a year of greater rainfall. Observations of treatments conducted in wetter years after this experiment was completed have shown similar, but unmeasured, results.

Pragmatism.—As with all active management programs, pragmatism is as important as efficacy in the selection of appropriate habitat treatments. Land managers at the Presidio consider the three most effective treatments in this study to be practical for use in this small, urban habitat fragment.

Experimental comparisons of alternative management techniques allow managers to make the best decisions. For MacDougall and Turkington (2007), the discovery that mechanical treatments resulted in cover increases to native flora and decreases to exotics that mimicked the responses to burning led them to recommend limiting the use of fire, which was physically and politically risky in their rural savannah ecosystem. In this study, the burn treatments were logistically difficult, even when confined to the burn boxes. On this site, however, burning proved less effective than other treatments and, thus, could be abandoned. Testing alternative grassland disturbance regimes can be useful in small or urban areas where classic methods such as grazing and prescribed fire may be neither pragmatic nor effective.

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Seed banking California's rare plants

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In addition to the protection and management of native habitats, preservation of biodiversity in *ex situ* storage facilities is an important component of plant conservation. In this article we provide an overview of the field of *ex situ* seed storage. We describe the basic guidelines of seed storage and how seed banks operate and manage genetic diversity. Focusing on the rare, wild plants of California, we examine the progress that has been made in long-term genetic conservation. We report data showing the numbers of rare taxa currently in seed bank storage, and show which institutions currently house seeds of Californian rare plant taxa. Using accession data from these institutions, we show that rare taxa with legal protection status under federal or state law have been relatively well covered in *ex situ* storage facilities (59% of taxa), while rare plants not afforded state or federal listing status have lower coverage (17% of taxa).

Key words: California, *ex situ*, rare plants, seed banks, conservation

California is an important region for plant conservation; it is home to more than 6,500 native taxa — more than any other U.S. state (Jepson Interchange 2013). Further, the California Floristic Province (the region of California experiencing a Mediterranean climate) is considered one of 25 global biodiversity hotspots based on high rates of endemism and levels of threat to native taxa (Myers et al. 2000, Pavlik 1995). More than one third of the plant species native to California are endemic (Baldwin et al. 2012). Many plants endemic to California are naturally rare with narrow geographic ranges or habitat parameters (Pavlik 1995); however, their rarity is further compounded by the threats that they face. This is reflected in the more than 2,300 taxa or 35% of the total flora that are included by the California Native Plant Society (CNPS) in the Inventory of Rare and Endangered Plants of California (CNPS 2013). Plants in the CNPS Inventory are assigned ranks (known as California Rare Plant Ranks [CRPR]) in an effort to categorize their degree of rarity and concerns for threats or endangerment (CNPS 2013). These rare taxa continue to be a special focus for conservation efforts because they are thought to have an increased risk of extinction compared to taxa that are more common or wide-ranging (Pimm and Raven 2000). With

ever increasing threats such as development, the spread of non-native species, and climate change, time is of the essence to advance conservation strategies to protect native species (Pavlik 1995).

Conservation of plant species can be achieved by protection of habitats and populations in nature (*in situ*), or by the preservation of genetic diversity outside of an organism's natural habitat (*ex situ*), often in gene banks, botanic gardens, zoos, or translocation sites (Cohen et al. 1991). *Ex situ* conservation provides a back up for biological diversity that might otherwise be lost in nature due to human induced environmental change or natural extinction. Seed banks that specialize in maintaining samples from wild populations are increasingly seen as a central component of *ex situ* plant conservation (Maunder et al. 2004). In comparison with living plant collections, seed banks can house much larger quantities of genetically distinct individuals using much less space. Further, seeds of many taxa can be kept alive for decades or longer with relatively inexpensive equipment (Walters et al. 2005; Rancho Santa Ana Botanic Garden [RSABG], unpublished data).

Seed banking efforts for rare plant taxa have been of regional and international focus for nearly 30 years (Center for Plant Conservation [CPC] 2013). Networks such as the CPC and Botanic Gardens Conservation International (BGCI) promote seed banking as an effective conservation tool to enhance recovery and restoration of rare plant species (CPC 2013, BGCI 2013). This on-the-ground strategy has global significance and is featured as a target in the Global Strategy for Plant Conservation, which sets the goal of securing *ex situ* conservation collections of at least 75% of the world's threatened plant species by the year 2020 (Convention on Biological Diversity [CBD] 2013). Here we examine *ex situ* conservation efforts in California, focusing on the progress of seed banking as a mechanism to conserve rare plants.

SEED BANKING OF WILD NATIVE PLANTS IN CALIFORNIA

Seed banks house viable seed samples (accessions) of diverse plant taxa (Figure 1). These facilities serve two major purposes: to acquire and maintain living seeds, and to distribute seeds to approved parties for various uses. Methods that are employed to maintain high viability in stored seed collections include drying seeds to a low moisture content and storing them in airtight packaging in freezers maintained at $\leq -18^{\circ}\text{C}$ (Linington 2003). Seeds that are long-lived and survive this drying and freezing process are classified as 'orthodox,' and generally maintain high viability for decades or more under these storage conditions (Walters et al. 2005; RSABG, unpublished data). Notable and diverse genera in California with orthodox seeds include *Clarkia*, *Ceanothus*, and *Arctostaphylos* (RSABG, unpublished data). Certain taxa, especially those with large, high lipid content seeds or those that are adapted to hydric environments do not survive the drying and freezing process and are classified as 'recalcitrant'; these cannot be stored using traditional methods (Roberts 1973). Well known examples of recalcitrant genera with taxa in California include *Quercus* and *Aesculus* (Bonner 1990). Emerging technologies such as cryopreservation may be employed for long-term storage of recalcitrant seeds (Walters et al. 2013).

California's diverse and unique flora includes many species that are adapted to seasonal aridity or cold temperatures. In addition, many taxa are known to maintain persistent soil seed banks (Keeley 1991). Assessments of storage behavior of wild collected seeds indicate that a wide range of taxa native to California can be considered orthodox located



FIGURE 1.—Seeds of *Xylorhiza cognata*; California Rare Plant Rank 1B.2 (rare, threatened or endangered in California and elsewhere), with no legal listing status. Photo credit: John Macdonald.

and long-lived under traditional storage regimes (RSABG, unpublished data). A number of institutions maintain seed banks that contain wild collected seeds of California native plants (Table 1). The most significant repository for seeds of taxa native to California is

TABLE 1.—Seed bank holdings of California Rare Plant Rank (CRPR) taxa at various institutions*.

Institution	Taxa Banked	Number of Accessions
Rancho Santa Ana Botanic Garden	404	1,143
Santa Barbara Botanic Garden	50	223
United States Department of Agriculture (various sites)	35	343
Kew Millennium Seed Bank	24	24
Rae Selling Berry Seed Bank	21	53
University of California Botanical Garden at Berkeley	20	36
Total (some taxa banked by multiple institutions)	514	1,822

* Additional seed collections may exist which are not accounted for in this analysis.

at RSABG in Claremont, California, where wild, field collected seeds comprise 3,994 accessions, representing more than 1,700 total taxa (RSABG, unpublished data). An ongoing focus of the RSABG seed bank is to secure rare plant seed collections for conservation purposes (Tables 1 and 2). Many taxa native to California are currently not maintained in seed banks (RSABG, unpublished data), and therefore are not secured for future research or conservation use.

TABLE 2.—Collections per California Rare Plant Rank (CRPR) and Federal Endangered Species Act (FESA) or California State Endangered Species Act (CESA) listing status. Analysis excludes extinct and extirpated taxa (CRPR 1A and 2A).

Rarity or Legal Status	Taxa Banked	Total Number of Taxa	Percentage Banked
All California Rare Plant Rank (CRPR) Taxa (excluding 1A and 2A)	514	2283	23%
CRPR 1B Taxa	345	1143	30%
CRPR 2B Taxa	41	495	8%
CRPR 3 Taxa	5	67	7%
CRPR 4 Taxa	123	578	21%
FESA and/or CESA Listed Taxa	167	284	59%
CESA Listed Taxa	124	218	57%
FESA Listed Taxa	138	195	71%

An analysis of current seed accession holdings of rare taxa native to California shows significant progress in meeting global goals for *ex situ* conservation (CBD 2013); however, there is still much work to be done. To date 167 (59% of the total) taxa with legal protection under the Federal Endangered Species Act (FESA) or the California Endangered Species Act (CESA) have been deposited at regional or national seed banks (Table 2). In comparison, seeds of only 17% of taxa that are considered rare, but do not have formal legal status, have been banked. Emphasis on collection of CESA- and FESA-listed taxa is likely due to funding that becomes available to secure seed collections through mitigation measures or strategic conservation efforts.

UTILIZATION AND STORAGE OF SEED COLLECTIONS

It is important to consider the intended use of a seed collection before it is made (Guerrant et al. 2004). For *ex situ* conservation collections, capturing high levels of representative genetic diversity in each accession is important for maximizing potential use. Various studies (Richards et al. 2007, Namoff et al. 2010) have focused on quantifying the amount of genetic diversity captured in *ex situ* plant populations. In both studies, allelic diversity of the collection was higher with increased sampling of individuals. Dolan et al. (2008) observed levels of allelic diversity in remnant wild populations, nurseries, and restoration sites to vary considerably depending on the taxon that was being sampled.

Guerrant et al. (2004) recommend collecting from ≥ 50 individual plants if possible in order to establish a minimum baseline of genetic diversity. For rare plants, the separate collection and storage of seeds from individual plants (known as maternal line collections) is recommended. While processing and curation costs are considerably greater for maternal line collections (versus bulked collections), this practice allows for the maximum control and management of genetic diversity. A target quantity of 2,500 to 10,000 seeds per accession is thought to provide sufficient material needed to meet the diverse uses of seed collections, including long-term genetic preservation and active use of the collection that may include germination testing, research, and restoration (Wall 2009).

Seed banks are, in essence, an ongoing seed longevity experiment. Many seed banks test seeds prior to being placed in storage and then test periodically throughout their storage life. Information on viability and germination is recorded and allows for easy longitudinal comparison of data throughout the storage history of each accession. These data not only provide information on seed storage behavior, but also provide dormancy and germination information for a large group of rarely cultivated plants. The data also guide germplasm management practices and advance horticultural and life history knowledge of these plants. The long-term storage and conservation of representative genetic samples, especially of threatened taxa, is another important facet of seed banks. Many seed banks maintain collections that are intended to stay in storage indefinitely to serve as propagules that can be utilized for reintroduction in the case of extinction or extirpation of populations in the wild.

DISCUSSION

A large proportion of rare plant taxa native to California in seed banks have FESA or CESA status (Table 2). Funding opportunities and mitigation requirements for these taxa have created mechanisms to secure these valuable collections; however, this has also created a bias toward the collection of taxa that have formal legal status over other taxa of conservation concern. The listing of endangered species by the state and federal governments is an important aspect of plant conservation, but is subject to political pressures and a legislative process, which does not always reflect the most current knowledge of plant rarity and threats. Other ranking systems of rarity and threats, such as NatureServe Explorer and the CNPS Inventory of Rare and Endangered Plants are useful in assessing the conservation status of the entire flora (NatureServe 2013, CNPS 2013). These indices, as well as online diversity databases such as the California Natural Diversity Database (CNDDB) and the Consortium of California Herbaria (CCH) provide valuable assessment tools for seed collection prioritization (CNDDB 2014, CCH 2014).

Unique funding mechanisms need to be employed to target rare or threatened taxa that do not have FESA or CESA status, and to broaden *ex situ* conservation efforts in California. Rancho Santa Ana Botanic Garden seeks to collaborate with a wide variety of partners to advance this important endeavor, and staff currently are working on a conservation strategy and needs assessment for seed banking that will identify targets for collection based on presence or absence in seeds banks, storage behavior, rarity, threat levels, and other factors. We hope that this analysis will advance ongoing efforts to obtain high quality *ex situ* germplasm collections of the rare plants of California and to secure them for long-term preservation.

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Utilizing the California Natural Diversity Database to aid in plant conservation: a case study from the California desert

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The California Natural Diversity Database (CNDDDB) inventories the status and locations of rare taxa and natural communities within California. Due to the fast-tracking of renewable energy projects within the desert regions of California, there has been concern over how best to balance the need for renewable energy development with the conservation of desert biological resources. With this in mind, the CNDDDB botany program, in conjunction with the California Native Plant Society (CNPS), has made an effort since 2008 to determine the status of many plant taxa in the desert regions of California. CNDDDB staff have also made an effort to enter all available information for rare plant taxa from the desert regions into the CNDDDB. This has resulted in an approximate 50% increase in the number of CNDDDB rare plant occurrences known from desert regions, and updating the CNDDDB has resulted in a more complete picture of the status of rare taxa in the desert regions of California. This information can be used to aid in plant conservation and management, especially within the area covered by the Desert Renewable Energy Conservation Plan.

Key words: California Native Plant Society, California Natural Diversity Database, conservation, Desert Renewable Energy Conservation Plan, rare plants, status

The California Natural Diversity Database (CNDDDB) is a program that inventories the status and locations of rare plants, animals, and natural communities throughout California. The CNDDDB is part of an international network of natural heritage programs that were originally developed by The Nature Conservancy in the 1970s (Bittman 2001). The CNDDDB was created in 1979 and has been housed within the California Department of Fish and Wildlife (CDFW) since 1981 (York et al. 1982). The CNDDDB is currently part of the Biogeographic Data Branch of the CDFW.

The CNDDDB compiles data-rich occurrence records and spatial features for the rarest taxa in California. Data are submitted by a wide variety of people and organizations including environmental consultants, state agencies, federal agencies, and non-profit organizations. While the CNDDDB has been working to inventory rare plant taxa for over 30 years, plant taxa present in the desert regions of California have been somewhat neglected in order to concentrate both survey and data entry efforts in areas of the state experiencing the greatest development pressures. However, in November 2008, the Governor of California issued Executive Order S-14-08 requiring an increase in the amount of electricity generated by renewable resources (Office of the Governor 2008). As a result, the Desert Renewable Energy Conservation Plan (DRECP) was established to help streamline the regulatory process for renewable energy development, while at the same time providing for protection and conservation of desert ecosystems and species (Dudek and ICF International 2011).

One of the main goals of the DRECP is to provide for the long-term conservation and management of covered species (Dudek and ICF International 2011). Covered plant species within the DRECP could potentially include those that are officially listed as rare, threatened, or endangered (under the Native Plant Protection Act, California Endangered Species Act, or the Federal Endangered Species Act), as well as species that are in the California Native Plant Society's (CNPS) *Inventory of Rare and Endangered Plants of California* (CNPS 2014a) or in the CNDDDB's *Special Vascular Plants, Bryophytes, and Lichens List* (CNDDDB 2014, Dudek and ICF International 2011).

As a result of the DRECP, CNDDDB staff have concentrated efforts on updating the rare plant information for the desert regions so that the highest quality data would be available. One aspect of this effort involves reviewing the status of plant taxa from the desert regions in conjunction with CNPS staff. In addition, CNDDDB staff have made an effort to enter all of the rare plant data that had been in the CNDDDB backlog for the DRECP area, and to enter incoming data for the same region in a timely manner. By inventorying and documenting rare plant taxa present in a particular area, better conservation and management decisions can be made to avoid or mitigate impacts to those taxa. The intent of this article is to describe the progress that has been made since 2008 in assigning a rarity status to desert plant taxa and in updating CNDDDB rare plant data for the DRECP area.

MATERIALS AND METHODS

Study area.—The boundary of the DRECP area (Figure 1) was designed to encompass the majority of the Mojave and Sonoran deserts in California and includes portions of seven California counties (Imperial, Inyo, Kern, Los Angeles, Riverside, San Bernardino, and San Diego) (Dudek and ICF International 2011). The DRECP covers an area of about 9 million ha and includes regions in California where renewable energy development could potentially occur (Dudek and ICF International 2011).

CNDDDB Data.—The CNDDDB tracks all taxa that have a CNPS California Rare Plant Rank (CRPR) but only maps data for CRPR 1 and 2 taxa into its Geographic Information System (GIS) due to staffing limitations (see Table 1 for CRPR categories). CNDDDB plant data reviewed in this article are restricted to CRPR 1 and 2 taxa so that GIS data would be available to use in queries. All CNDDDB data used in this article are from CNDDDB GIS data made available to CNDDDB users in January 2008 and January 2013. January 2008 was used as a baseline for all queries since it was not until later in 2008 that CNDDDB staff began a

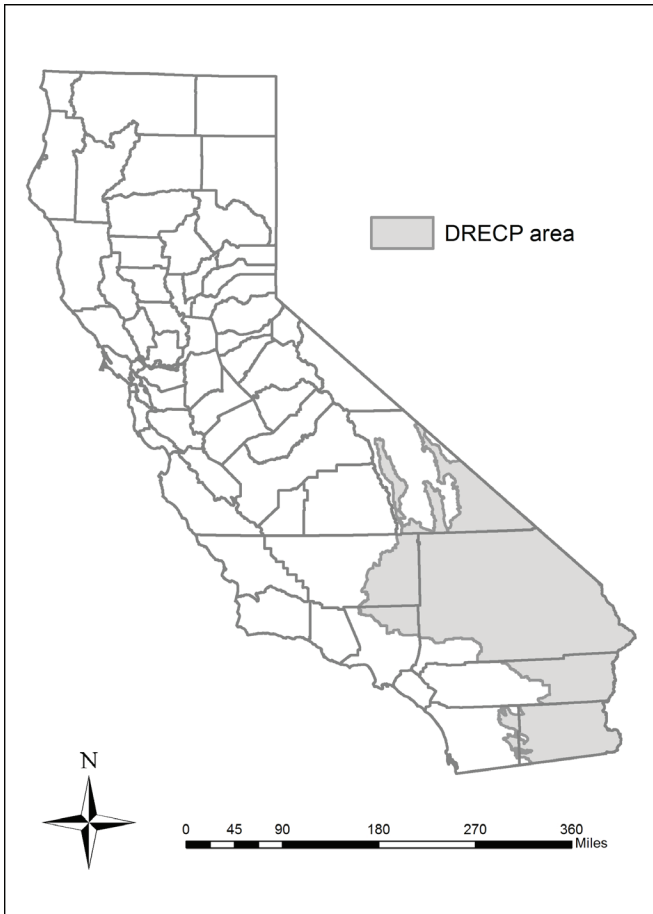


FIGURE 1.—Location of the area covered by the Desert Renewable Energy Conservation Plan (DRECP; Dudek and ICF International 2011). The DRECP boundary was used when querying for rare plant taxa and occurrences within the California Natural Diversity Database.

TABLE 1.—Number of California Native Plant Society (CNPS) rare plant taxa known from within the Desert Renewable Energy Conservation Plan area in California, January 2008 and January 2013; only CNPS California Rare Plant Rank 1 and 2 taxa are included.

California Rare Plant Rank ^a	Number of CNPS rare plant taxa known from within the DRECP area	
	January 2008	January 2013
1 (including 1A and 1B)	125	135
2 (including 2A and 2B)	127	144
Total	252	279

^a CRPR 1A: Plants presumed extirpated in California and either rare or extinct elsewhere
 CRPR 1B: Plants rare, threatened, or endangered in California and elsewhere
 CRPR 2A: Plants presumed extirpated in California, but more common elsewhere
 CRPR 2B: Plants rare, threatened, or endangered in California, but more common elsewhere

concerted effort to update rare plant taxa within the area delineated by the DRECP. GIS data distributed in January 2013 were used as a “post-DRECP” dataset. Work to update rare plant taxa within the DRECP area is an ongoing process, and CNDDDB plant GIS data distributed in January 2013 mark five years of data entry effort.

Reviewing rarity status.—CNPS and CNDDDB staff regularly review taxa for inclusion in the *Inventory of Rare and Endangered Plants of California* (CNPS 2014) and *Special Vascular Plants, Bryophytes, and Lichens List* (CNDDDB 2014a). In order to reduce duplication, staff from both organizations work collaboratively to review the status of rare plant taxa. CNPS staff take the lead on preparing status review documents, which are detailed summaries of plant taxa under review. These documents are sent out to groups of local botanical experts and taxonomists for comments and additional information. CNPS and CNDDDB staff then review all of the information gathered regarding rarity of the plant and assign it to a CRPR (CNPS’s rarity ranking system) and a Natural Heritage Conservation Rank (CNDDDB’s rarity ranking system), if appropriate.

In 2008, CNPS and CNDDDB staff began emphasizing status reviews for desert plant taxa due to growing concerns about the impacts future solar energy projects could have. While both CRPR and Natural Heritage Conservation Ranks are valuable assessments of rarity, CRPRs are used in this article since there are only four main categories to consider; Natural Heritage Conservation Ranks include more rarity categories that make them more descriptive, but also more complicated for analyses.

Data were used from CNDDDB shapefiles and comparisons made between the number of CRPR 1 and 2 desert plant taxa known from the area delineated by the DRECP in January 2008 with the number of CRPR 1 and 2 desert plant taxa known from the same geographic area in January 2013. Those CNDDDB plant taxa that had spatial features that intersected the DRECP boundary area were then exported from ArcMap into a Microsoft Access database. A summary query was subsequently run to determine the number of plant taxa within each CRPR category.

Entering plant occurrence records and spatial data.—The CNDDDB gathers data from a wide variety of sources and synthesizes the information into distinct occurrence records, with accompanying spatial features. CNDDDB occurrence records contain descriptive text information (e.g., site location, habitat, threats, population size, site quality, and the date that the taxon was last seen at the site) for each occurrence. Every time a plant is updated, CNDDDB staff search for and incorporate data from herbarium specimens, field survey forms, shapefiles, reports, and personal communications into these occurrence records and spatial features. Updating CNDDDB occurrence records for a single species can take anywhere from one day to over a month, depending on the complexity and volume of the data.

Between January 2008 and January 2013, all CRPR 1 and 2 plant taxa known from within the DRECP area were updated ≥ 1 time, but could have been updated more than once. All available information was incorporated into distinct occurrence records with accompanying spatial features, and comparisons made between the number of plant occurrences added to the CNDDDB in 2008 and 2013. Plant spatial features were selected from the CNDDDB shapefile that intersected with a shapefile of the DRECP area for each taxon. Those taxa that had spatial features that intersected the DRECP boundary area were then exported, and a spatial selection was performed for each county of interest. The number of selected plant features was counted for each county within the DRECP area.

RESULTS

Within the DRECP area, a total of 27 CRPR 1 and 2 plant taxa were added to the list of CNPS- and CNDDDB-tracked taxa between January 2008 and January 2013, reflecting the increased emphasis on reviewing the status of desert taxa. Ten of those taxa were assigned to CRPR 1 and 17 of those taxa were assigned to CRPR 2 (Table 1). In January 2008, before the CNDDDB began a concerted effort to update plant taxa in the desert, there were 1,706 CNDDDB plant occurrence records present in the DRECP area (Table 2). The CNDDDB began utilizing GIS to map plant occurrences around 1990, so these 1,706 CNDDDB plant occurrence records reflect the number of occurrence records entered between 1990 and January 2008, an average of approximately 95 per year. Between January 2008 and January 2013, there were 1,728 additional CNDDDB plant occurrences added to the DRECP area, an average of approximately 346 per year.

TABLE 2.—Number of California Natural Diversity Database rare plant occurrences known from within the Desert Renewable Energy Conservation Plan area in California, January 2008 and January 2013.

County	January 2008	January 2013	Percent increase
Imperial	159	239	33
Inyo	271	514	47
Kern	240	303	21
Los Angeles	83	103	19
Riverside	133	318	58
San Bernardino	751	1,838	59
San Diego	69	119	42
Total	1,706	3,434	50

Between 2008 and 2013, there was an approximate 50% increase in the overall number of CNDDDB plant occurrences within the DRECP area (Table 2). The largest increases occurred in San Bernardino (+59%) and Riverside (+58%) counties. While these large increases may be partly because these counties make up a large portion of the DRECP area, they also reflect the plant survey data submitted as a result of pre-project surveys for solar energy projects proposed within those counties. All of the of the solar energy project data received for CRPR 1 and 2 taxa were entered by CNDDDB staff between 2008 and 2013.

DISCUSSION

One of the main goals of the DRECP is to provide for the long-term conservation and management of covered species within the plan area (Dudek and ICF International 2011). However, in order for the DRECP to address this goal, high quality and current information on which taxa should be considered rare, and therefore potentially covered by the DRECP, is essential. Such data were largely lacking before 2008 due to low development pressures and resultant low survey efforts in desert areas. With the DRECP covering an area of about

9 million ha, it can be problematic to determine which areas and taxa are of the highest conservation concern. CNDDDB and CNPS staff have made a concerted effort since 2008 to ensure that the most up-to-date information on rare plant taxa in the desert regions is available for use by those involved with the DRECP. Reviewing rarity status of plant taxa and updating CNDDDB rare plant occurrence information for the DRECP area are two key components of providing a more complete picture of the sensitive resources that could be impacted by renewable energy projects.

The majority of the 27 taxa added to CRPR 1 or 2 between January 2008 and January 2013 were assigned to CRPR 2. Although CRPR 1 taxa are high priority for conservation due to their limited global distribution, many desert taxa just cross into California from adjacent states where they may be more common. By assigning these taxa a rarity status of CRPR 2, we can be more confident that a full assemblage of desert plant taxa of conservation concern will be accounted for within the DRECP process. The increase in number of CRPR 2 plant taxa was a big step forward in preparing for the DRECP by making sure that not only is biodiversity within California conserved, but also by ensuring that those peripheral populations are conserved for the long-term benefit of the species, as discussed by Leppig and White (2006).

The large increase in number of CNDDDB rare plant occurrences entered within the DRECP area is a reflection of the priority that CNDDDB staff have placed on data entry in this area, and also reflects the type of data the CNDDDB has received. Pre-project surveys for proposed solar energy projects have had a large impact on the number of CNDDDB occurrences for rare plant taxa. For example, the Ivanpah Solar Electric Generating System (ISEGS) consists of three large solar thermal power plants that cover a total of about 1,330 ha in northeastern San Bernardino County (Figure 2; CH2M Hill 2009). There were no CNDDDB plant data in January 2008 from the vicinity of the ISEGS. However, almost entirely as a result of pre-project surveys for the ISEGS, January 2013 CNDDDB data included seven CRPR 1 and 2 plant taxa in that area, with some of those taxa having extensive occurrences within the project boundary. Of these seven plant taxa, six were CRPR 2 taxa and one was a CRPR 1 taxon.

While the CRPR 2 taxa documented within the project area were not new CRPR 2 taxa (i.e., they were not assigned to CRPR 2 between January 2008 and January 2013), the high number of CRPR 2 taxa within the project area highlights the importance of determining which taxa merit ranking of CRPR 2 so that a full assemblage of rare taxa are properly accounted for within the DRECP. If CRPR 2 plants were ignored, the ISEGS project would only have identified a single plant taxon of conservation concern, and the remaining six taxa would not have received attention.

In conclusion, joint efforts by CNPS and CNDDDB staff since 2008 to evaluate the status of desert plant taxa, and efforts made by CNDDDB staff to update rare plant occurrence information, have resulted in many additional taxa being tracked by CNPS and the CNDDDB, and in a large increase in the number of CNDDDB rare plant occurrences. The CNDDDB is an essential tool for assessing the rarity of California taxa and determining which are in the greatest need of conservation, and is used by a wide range of organizations for planning, research, and conservation purposes. While the CNDDDB can not be used as proof that a taxon is absent from a particular area, as more surveys are performed in the desert and those data are incorporated, the database will provide a more complete picture of the distribution and status of sensitive desert taxa. By doing so, the CNDDDB data can be used to help the DRECP move toward its stated goal of contributing to the conservation of covered species, as well as other unique and valuable desert resources.

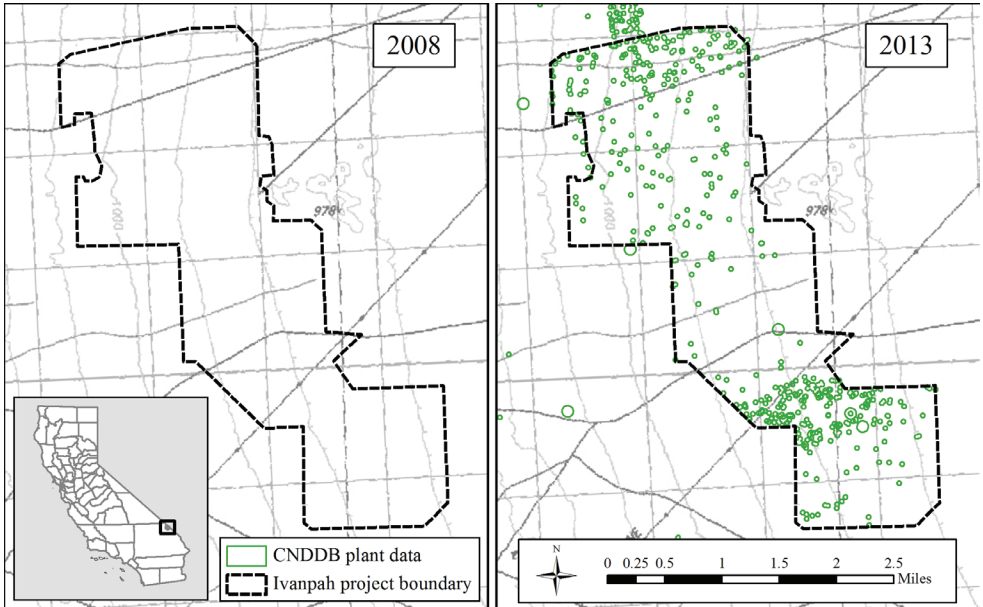


FIGURE 2.—The two maps in this figure show the boundary (dashed black lines) of the Ivanpah Solar Electric Generating System (CH2M Hill 2009). The map on the left shows California Natural Diversity Database (CNDDDB) rare plant data available in January 2008 (when no CNDDDB rare plant populations were present in this area), and that on the right shows CNDDDB rare plant data available in January 2013; green polygons represent CNDDDB rare plant populations in this area of San Bernardino County, California.

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Tanoak conservation: a role for the California Department of Fish and Wildlife

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Tanoak trees and forests are ecologically, culturally, and economically important, providing valued wildlife habitat and forest products. Since the horticultural trade accidentally introduced the sudden oak death pathogen (*Phytophthora ramorum*) to North America, well over a million tanoaks (*Notholithocarpus densiflorus*) have died, and an unknown number are infected. In roughly twenty years, the lethal disease has spread extensively south and north of San Francisco with disjunct outbreaks as far as southwestern Oregon, despite efforts to contain it. Currently no cure exists for infected trees, and thus far tanoak exhibits little genetic resistance to the exotic water mold that causes the disease. Fortunately large areas remain uninfected, but computer models rank uninfected areas on the north coast of California as high risk for infection. The current sudden oak death epidemic warrants concern because tanoak provides food and habitat for many wildlife species. People also value this evergreen, flowering tree as a source of nuts, edible fungi, and hardwood.

Key words: tanoak, *Notholithocarpus densiflorus*, *Lithocarpus densiflorus*, sudden oak death, *Phytophthora ramorum*

The dramatic decline of American chestnut (*Castanea dentata*), reminds us that even common plants can rapidly become threatened. A century ago in North America's eastern deciduous forests the exotic plant disease chestnut blight began to spread after its inadvertent introduction on an infected, imported Chinese chestnut. Within thirty years after horticulturalists accidentally introduced the causal pathogen from Asia to North America, American chestnuts were virtually destroyed "through most of their natural range" (Brasier 2008). Today computer models indicate that tanoak (*Notholithocarpus densiflorus*) may experience a similar, massive die off on the west coast of North America due to the introduction of *Phytophthora ramorum*, a plant pathogen that causes sudden oak disease (Meentemeyer et al. 2011). The non-native water mold (or oomycete) was first detected in

North America in the mid-1990s and has been spreading in wildlands of central and northern California via garden plants, firewood, wind-blown rain, and moving waterbodies.

Ecosystem change can occur rapidly after the introduction of novel pathogens (Anderson et al. 2004, Brasier 2008, Desprez-Loustau et al. 2007, Loo 2009). Although it is unlikely that *P. ramorum* will cause tanoak extinction, it will likely cause “the rapid and extensive loss of overstory trees ... within 30 years of pathogen establishment in many forests” (Figure 1; Cobb et al. 2012). Our current understanding of tanoak resistance to *P. ramorum* is incomplete but, given observed levels of susceptibility, a “risk of extirpation”



FIGURE 1.—Tanoak killed by *Phytophthora ramorum* and photographed circa 2001 at Joe Hall Creek in Curry County, Oregon, one of the first confirmed sudden oak death sites in that state. Photograph by and courtesy of Everett Hansen.

exists (Hayden et al. 2011). Along the Big Sur coast in central California, some sites have already experienced 100% tanoak mortality after infestation (Davis et al. 2010). “In the absence of extensive control,” a team of seven university scientists predicted “a ten-fold increase in disease spread between 2010 and 2030 with most infection concentrated along the north coast between San Francisco and Oregon” (Meentemeyer et al. 2011). As a result, “substantial tree mortality, particularly of tanoak, is likely to follow.” Based on their computer model, they predicted “explosive growth in [*P. ramorum*] infection and disease ... to occur around 2016.” For more discussion of the threat sudden oak death poses to tanoak, see Bowcutt (2013), Cobb et al. (2013), and Dillon et al. (2013).

Given its mission “to manage California’s diverse fish, wildlife, and plant resources, and the habitats upon which they depend, for their ecological values and for their use and enjoyment by the public,” the California Department of Fish and Wildlife (CDFW) is the logical and appropriate agency to become a leader in tanoak conservation (CDFW 2014). In addition to providing food and habitat for numerous native and naturalized animals including important game species, tanoak produces delectable acorns, edible fungi, and beautiful hardwood. Efforts are already underway to develop conservation strategies using science-based management practices that foster tanoak wellness and minimize *P. ramorum* infection risks (Cobb et al. 2013).

DISTRIBUTION

As California's most abundant hardwood or flowering tree, tanoak serves as a foundational species in a variety of ecosystems, from mixed forests to those dominated by coast redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*), to prairie balds with scattered trees. Tanoak trees grow from southwestern Oregon through the California Coast Range to near Santa Barbara, with inland populations occurring through the Siskiyou Mountains and from the southern tip of the Cascade Range along the western slopes of the Sierra Nevada to Yosemite National Park (Figure 2; Munz 1973, Baldwin et al. 2012). Much of its coastal distribution overlaps with that of coast redwood, but due to

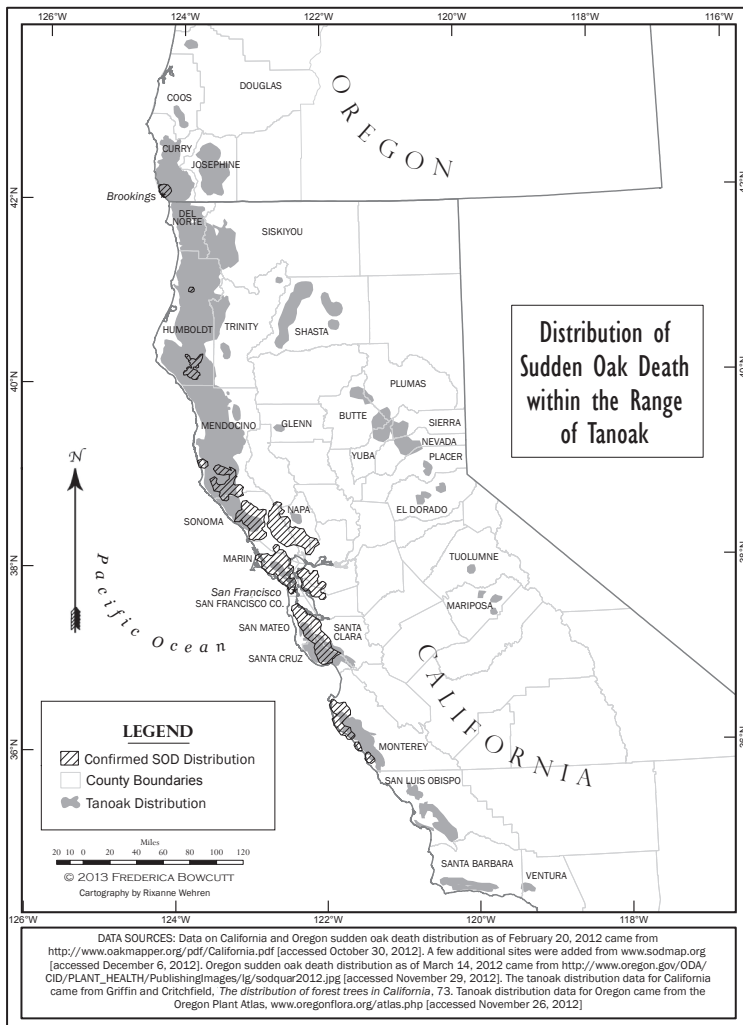


FIGURE 2.—Distribution of sudden oak death, caused by *Phytophthora ramorum*, in relationship to tanoak distribution. Data sources: California and Oregon distribution as of February 20, 2012 came from Geospatial Innovation Facility (2012); a few additional sites were added from U.C. Berkeley Forest Pathology and Mycology Laboratory (2012). Oregon sudden oak death distribution as of March 14, 2012 came from the Oregon Department of Agriculture (2012). The tanoak distribution data for California came from Griffin and Critchfield (1976). Tanoak distribution data for Oregon came from the Oregon Flora Project (2012).

its greater tolerance of drought, tanoak extends further inland. The shrub variety, *Notholithocarpus densiflorus* var. *echinoides*, occurs from southwestern Oregon to parts of northern California's Klamath Range, Cascade Range, and Sierra Nevada. It tolerates poorer soils and extends tanoak distribution to higher elevations (McDonald and Huber 1995). A mutant shrub-like form grows within Yuba County in the northern Sierra Nevada (*N. densiflorus* forma 'attenuato-dentatus') (Tucker et al. 1969). This mutant is used in horticulture due, in part, to its rarity and its unusual leaves, which are deeply toothed and taper to a very narrow apical tip. Despite being abundant in much of its range, tanoak's global distribution is limited.

EVOLUTIONARY RELATIONSHIP TO CHESTNUTS

For over a century, botanists viewed tanoak as an evolutionary link between oak (*Quercus*) and chestnut (*Castanea*) based on morphological features. Tanoak acorns resemble those of *Quercus*, but its upright male catkins echo those of *Castanea*. In 1840, two British botanists, W. J. Hooker and G. A. W. Arnott, wrote the original description for tanoak and assigned it to *Quercus* but described tanoak as a "remarkable plant [that] has very much the appearance of a *Castanea*" (Hooker and Arnott 1840). W. L. Jepson (1909) adopted the revised tanoak name that placed it in the southeast Asian genus *Pasania*, claiming it to be "equally related to" oaks and chestnuts. Currently, *Pasania* is included within *Lithocarpus*. C. S. Sargent's 1922 manual of North American trees called tanoak an oak-chestnut "intermediate" and favored its inclusion in the genus *Lithocarpus* (Sargent 1965). Modern molecular genetic research indicated that oaks, chestnuts, and Asian chinquapin (*Castanopsis*) are more closely related to tanoak than to the southeast Asian genus *Lithocarpus* and its sister taxon, the North American chinquapin, in the genus *Chrysolepis* (Manos et al. 2008). Consequently, a new monospecific genus was established for the North American tanoak, *Notholithocarpus*.

CLIMATE CHANGE AND EVOLUTIONARY HISTORY

Climate change is projected to impact tanoak resilience to disturbance. To make predictions about possible impacts, paleobotanical and post-glacial research is used to learn how related species have responded to past changes in climate. Unfortunately, the fossil record of tanoak remains unresolved. When tanoak was moved to *Notholithocarpus*, the North American paleospecies assigned to the genus *Lithocarpus* were not automatically moved to the new genus. To date, paleobotanists have not determined whether the fossils ascribed to the genus *Lithocarpus* in North America require reassignment. In addition, multiple paleospecies are disputed. Because of the extreme range in leaf variation in *Lithocarpus*, macrofossils are difficult to identify with certainty when preserved fruits do not occur with fossilized leaves. For this reason, *L. klamathensis* and *L. weidei* are disputed species (D. Erwin, University of California Museum of Paleontology, personal communication). Based on leaf shape, venation, and acorn cupule characteristics preserved in macrofossils, *Lithocarpus nevadensis* did grow in Nevada 10–15 million years ago at elevations $\geq 1,830$ m (>6,000 ft) under a much warmer and wetter climate than exists at that elevation today (D. Erwin, University of California Museum of Paleontology, personal communication). Based on macrofossil specimens also housed at the University of California Museum of Paleontology that include an acorn cap, *L. coatsi* dated to the Eocene also appears to be a

defendable species, and grew in present day Nevada (D. Erwin, University of California Museum of Paleontology, personal communication).

Although helpful in distinguishing other members of the Fagaceae from one another, pollen microfossils are unlikely to further illuminate the current understanding of tanoak evolution. Researchers studying Quaternary vegetation in southwestern Oregon found fossilized pollen of tanoak to resemble North American chinquapin pollen (Briles et al. 2005). This was corroborated based on light microscopic study of extant tanoak and North American chinquapin pollen (E. Leopold, University of Washington, Seattle, personal communication). Two Swedish paleobotanists found pollen ornamentation highly useful in delineating evolutionary lineages within the genus *Quercus* when examined using a scanning electron microscope (Denk and Grimm 2009). However, Denk doubts pollen can be used to distinguish *Notholithocarpus* from *Lithocarpus* (T. Denk, Swedish Museum of Natural History, personal communication). Although pollen micromorphology is “a character of known diagnostic significance in the family,” within the chestnut subfamily Castaneioideae it is “relatively uniform” (Crepet 1989).

The beech family (Fagaceae), to which tanoak belongs, originated in the northern hemisphere. Although widely considered a natural group derived from a shared ancestor, evolutionary relationships among taxa within the family remain “far from resolved” (T. Denk, Swedish Museum of Natural History, personal communication; see also Nixon 1989). Bidirectional migration reputedly occurred between Eurasia and North America via the North Atlantic and Bering land bridges (Manos and Stanford 2001). However, two evergreen taxa, *Castanopsis* and *Lithocarpus*, appear to have migrated only over the Bering Land Bridge; based on the fossil record, this occurred “by at least the mid-Eocene” (Manos and Stanford 2001).

Later isolation allowed for the evolution of novel species including tanoak and North American chinquapin. Uplift of mountain ranges (or down-drop of adjacent land) due to tectonic activity in western North America resulted in a rainshadow effect that probably caused tanoak’s range to shrink to areas that still received moisture from storms moving east from the Pacific Ocean. Beginning roughly 4.5 million years ago, the rising elevation of the Sierra Nevada and the Cascade Range resulted in drier summer conditions east of these mountains (Graham 1999). By the late Pliocene and Pleistocene, a Mediterranean climate resembling today’s dry-summer, wet-winter regime developed (Graham 1999).

Recent genetic research has reinforced the notion that tanoak is a paleoendemic, a relict of “an ancient and formerly widespread broadleaf evergreen flora, which persists today in the Indochinese tropics” where summer rainfall is the norm and killing frosts are not (Manos et al. 2008). As a climatic relict of a wetter, more temperate period in North America’s past, tanoak may be vulnerable to periods of increased drought and erratic frost events, both of which are predicted to occur more frequently with global climate change. Frost can compromise sexual reproduction, and drought stress can reduce its resistance to pathogens and insect pests. Multiple disturbances linked to climate change affect tanoak’s resilience. Periodic wetter and warmer conditions will radically increase tanoak’s vulnerability to sudden oak death by favoring *P. ramorum* spore production (Meentemeyer et al. 2011). Although snags do not substantially elevate fire risk, areas with many recently killed tanoaks still standing with dead leaves can increase wildfire severity (Metz et al. 2011).

VALUE TO WILDLIFE

If the predicted massive tree die-off of tanoak occurs due to sudden oak death, many species of vertebrates will be impacted, as will many insects that warrant more study given their significant influence on ecosystem function. For example, filbert weevils (*Curculio uniformis*), filbertworm moth larvae (*Cydia latiferreana*), and other insects can destroy over half of the acorn crop in the absence of frequent fires (Roy 1957a). These nut-bearing trees feed numerous animal species. The relatively large acorns typically exceed the size of a hazelnut. It is one of the more reliable acorn producers in California and southwestern Oregon, rarely failing completely and bearing bumper crops more frequently than species of *Quercus*. Tanoaks “are heavily laden almost every alternate year and complete seed crop failures are rare,” helping to give it the reputation of being the heaviest acorn producer of all Pacific Coast oak species (Roy 1962). Tanoak trees typically begin to bear an abundance of acorns when they have reached between 30 and 40 years old, “although 5-year-old [root] sprouts also have produced fairly heavy crops” (Roy 1962). The shorter, often conical shaped mature trees in full sun tend to produce more than full-grown shaded trees (Figure 3). A mature tanoak tree bears more than 90 kg (200 pounds) of nuts on average in a good year, with estimates as high as 454 kg (1,000 pounds) annually for large mature trees (Baumhoff 1963, Radtke 1937).



FIGURE 3.—Tanoak in open prairie with robust canopy, a legacy of frequent, low intensity fires set by Native people. Ukiah, California circa 1903. Photograph by A. O. Carpenter (also Plate 7 in Jepson, *The silva of California*, 1910). Image courtesy of the University and Jepson Herbaria Archives, University of California, Berkeley.

Many wildlife species cache tanoak acorns for later consumption, including acorn woodpeckers (*Melanerpes formicivorus*), Stellar's jays (*Cyanocitta stelleri*) and at least four species of squirrels (Fryer 2008, Roy 1957a). One tanoak nut hoarder, the dusky-footed woodrat (*Neotoma fuscipes*), is an important prey of the northern spotted owl (*Strix occidentalis caurina*). Other predators of tanoak herbivores include coyote (*Canis latrans*), cougar (*Puma concolor*), and fisher (*Martes pennanti*) (Raphael 1987). Because tanoaks produce their abundant nut crop in the fall, they provide a critically important food source for deer (*Odocoileus* spp.) and black bear (*Ursus americanus*). The now extinct grizzly bear (*Ursus arctos*) likely fed on tanoak acorns given its former distribution (Storer and Usinger 1963). Other important game species benefit from tanoak mast, such as band-tailed pigeon (*Patagioena fasciata*), wild turkey (*Meleagris gallopavo*), and feral pigs (*Sus scrofa*). Various species of native mice (*Peromyscus* spp.) also consume tanoak acorns (Fryer 2008). The abundant nuts are a "vital" food source for many wildlife species (McDonald and Huber 1995).

Tanoak provides more than just an abundance of acorns as food for wildlife. Mule deer (*O. hemionus*) browse its leaves (Fryer 2008). Northern flying squirrels (*Glaucomys sabrinus*) consume ectomycorrhizal fungi that grow on tanoak roots (Fryer 2008). Various salamanders and rodents use tanoak for cover or nesting (Raphael 1987). Because tanoaks often grow in the shade of taller coast redwood and Douglas-fir, they help to create forests with multi-layered tree canopies favorable to northern spotted owls and other animals (North et al. 1999). A variety of birds forage for insects on tanoak, including chickadees (*Poecile* spp.) (Fryer 2008).

Although botanists, foresters, and plant pathologists have completed much research on tanoak, a full understanding of the organisms and ecological processes affected by tanoak remains incomplete. As is typical of members of the beech family, tanoak is a monoecious species and produces separate female and male flowers on the same plant. Each small, simple flower lacks petals and typically appears in summer (Roy 1957b), with acorns maturing two years after pollination. Until recently, it was widely believed that tanoak was wind pollinated like true oaks in the genus *Quercus*. Although self-fertilization does occur and some wind pollination is likely, most female tanoak flowers appear to be insect pollinated (Wright and Dodd 2013). However, the insect species involved remain to be systematically identified. Further research is recommended to study the significance of tanoak pollen as a food source in pollinator communities (Wright and Dodd 2013).

Tanoaks host a variety of fungi that grow on its roots (mycorrhizae) that are known to play important roles in ecosystems including as sources of wildlife food. Bergemann and Garbelotto (2006) found 119 taxa of ectomycorrhizal fungi growing on tanoak roots in northern California, which they believed to be an underestimate given their sampling method. Their estimated species richness of root associated fungal taxa was 265. Researchers predict that *P. ramorum* will cause a decline in ectomycorrhizal fungi, which is troubling given their significance in "ecosystem function through their control over decomposition, nutrient acquisition, and mobilization and regulation of succession in plant communities" and their decline "will likely disrupt the function and structure of these forests" (Bergemann et al. 2013). In coast redwood forests, tanoak is the dominant ectomycorrhizal host (Bergemann and Garbelotto 2006).

NATIVE AMERICANS

Many Native Americans are deeply committed to continued use of tanoak acorns as a traditional food, and seek partnerships to address the *P. ramorum* threat (Ortiz 2008). Human use of tanoak acorns for food extends over at least 5,000 to 7,000 years. Most, if not all, tribes within the range of tanoak (Figure 4) consumed its nutritious nuts. Gathering and processing of tanoak acorns for human use continues today, particularly in northern California among Native Americans, and tanoak acorn-based foods are important to cultural identity.

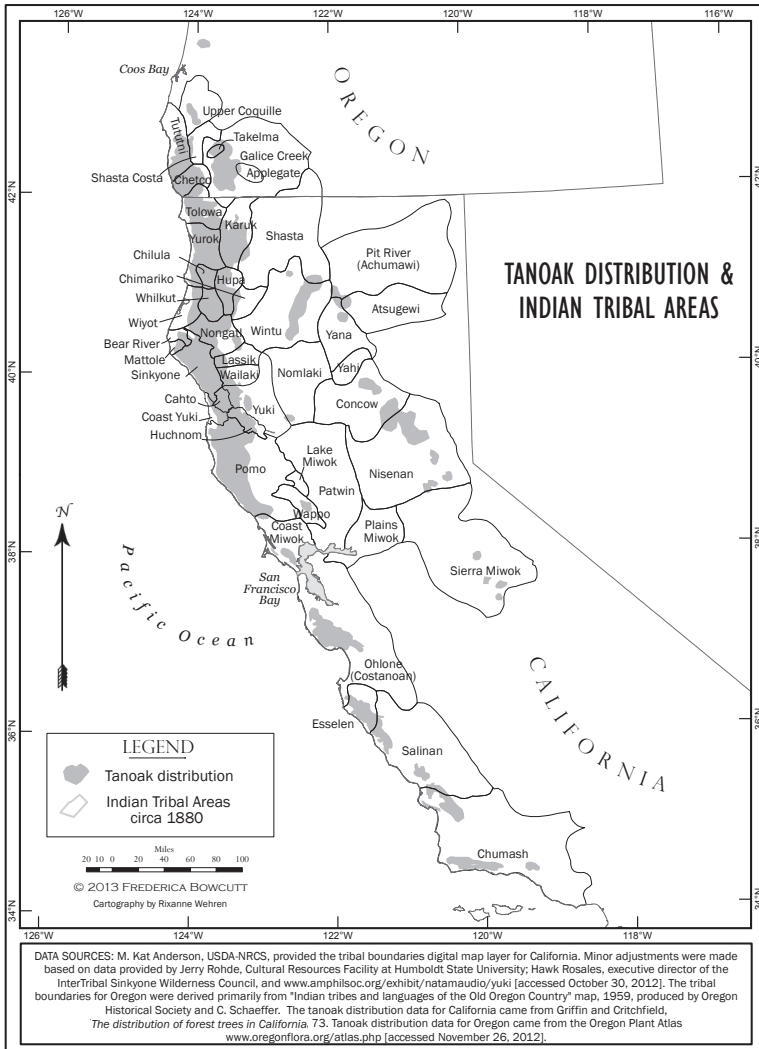


FIGURE 4.—Tribal territorial map and tanoak distribution. Data sources: M. Kat Anderson, USDA-NRCS, provided the tribal boundaries digital map layer for California. Minor adjustments were made based on data provided by Jerry Rohde, Cultural Resources Facility at Humboldt State University; Hawk Rosales, executive director of the InterTribal Sinkyone Wilderness Council, and the American Philosophical Society (2012). The tribal boundaries for Oregon were derived primarily from Schaeffer (1959). The tanoak distribution data for California are from Griffin and Critchfield (1976). Tanoak distribution data for Oregon are from the Oregon Flora Project (2012).

The anthropological literature documents Native American use of burning to foster tanoak health on a landscape scale; indeed, it was noted by a Karuk woman that annual burning protected tanoak best from infection and insects (Schenck and Gifford 1952). Burning reduces insect populations because trees abort weevil and moth larvae infested acorns during development; thus, a surface fire set after initial acorn drop kills the larvae inside and those already in the leaf litter. The Pomo Indians of Redwood Valley burned annually to maintain widely spaced oaks with a grassy understory; in their “beautiful park landscape,” burning controlled the brush while leaving “the larger trees ... uninjured” (Kniffen 1939). By decreasing fuel loads, regular burning by tribal peoples reduced the risk of catastrophic wildfire that would destroy mature tanoak trees (Anderson 2005). For a more extensive treatment of tanoak ethnobotany, including traditional ecological knowledge, see Bowcutt (2013).

Tanoak vulnerability to sudden oak death increased with fire exclusion according to a 2005 study using GIS (Moritz and Odion 2005). However, these results have been challenged given the limitations of *P. ramorum* distribution and fire-history maps, which make studying the relationship between “pathogen invasion and persistence” and burning difficult (Lee 2009). Fires do not appear to immunize forests, nor do prescribed burning or catastrophic wildfires eliminate *P. ramorum* from a site, though they can reduce its spread (Lee 2009). Preliminary results from experimental treatments in southwestern Oregon and northern California forests suggest “that burning can be a valuable tool in cleaning up small infectious material in infested sites,” even when it does not eliminate the pest (Lee 2009). Thus far, frequent, low-intensity fires that mimic traditional ecological practices of indigenous peoples, have not yet been tested as a prophylactic measure or to treat an infected site.

Traditional burning practices may provide insights into adaptive responses to current climate change, which will likely impact the spread of sudden oak death. Current trends in global climate change indicate that weather patterns are growing “increasingly erratic and extreme” which “could have consequences for ecosystem stability and the control of pests and diseases” (Kelly 2011, Medvigy and Beaulieu 2012). Tanoak acorns ripen in their second autumn, thus increasing their vulnerability to late frost, which can destroy reproductive organs and radically reduce acorn productivity. By clearing underbrush, Native Americans maintained good airflow around harvested tanoaks, which reduced loss of flowers and developing acorns to cold temperatures. Unfavorable climatic conditions also provoked the southwestern Pomo to pray for acorns “when hail comes from the north” (Gifford 1967).

EDIBLE FUNGI

In addition to producing edible nuts, tanoak logs, snags, and forests produce a variety of edible fungi. One of the most treasured mushrooms hunted in tanoak stands is the American matsutake (*Tricholoma magnivelare*), also known as tanoak mushroom. It is harvested for local consumption and commercially for export. Multiple northwestern California tribes particularly value the American matsutake, including the Hupa, Karuk, Wailaki, and Yurok (Anderson and Lake 2013). They typically combine autumn mushroom hunting with tanoak acorn and huckleberry harvesting (Anderson and Lake 2013). Native people from northwestern California still consume several other species of fungi associated with tanoak including oyster mushrooms (*Pleurotus cornucopiae*), black trumpet (*Craterellus*

cornucopioide), and lion's mane (*Hericium erinaceus*) (Anderson and Lake 2013). Choice "oyster mushrooms will repeatedly fruit from rotting ... tanoak ... snags and logs until the decay is too advanced" (Anderson and Lake 2013). Shiitake mushrooms (*Lentinula edodes*) can be cultivated on wood chips from tanoak logs (Donoghue and Denison 1996).

HARDWOOD

Tanoak wood is used for heating, flooring, cabinets, furniture, tool handles, wood chips, paper pulp, and biofuel, and it has the potential to become more widely used. The misperception that the wood is inferior to eastern hardwoods persists, in part, due to unskilled producers using milling practices and drying schedules suited for easier to process softwoods (conifers). Consumers developed a negative attitude about tanoak wood and other California hardwoods because poorly manufactured products were of inferior quality (Huber and McDonald 1992). According to the authors of the Hoopa Valley Reservation Hardwood Study Report released in 1968, "A major reason for failure to harvest and manufacture western hardwoods profitably has been a general reluctance to recognize fundamental differences between softwoods and hardwoods requiring the use of different equipment and techniques" (Economic Development Administration 1968).

Leading foresters and others advocated for using tanoak wood beginning in the 1800s. "No other oak begins to vie with it for beauty of grain" according to one booster who claimed that "it will stay exactly where the workman puts it and will stand the roughest knocks without flinching" (Armstrong 1891). A founder of the Society of American Foresters and chief dendrologist for the Bureau of Forestry (later to become the U.S. Forest Service [USFS]), described tanoak in 1908 as "a tree of the greatest importance in Pacific forest, both for its valuable tanbark and for the promise it gives of furnishing good commercial timber in a region particularly lacking in hardwoods" (Sudworth 1967). H. S. Betts conducted timber tests for the USFS and concluded in 1911 "there seems to be no good reason why tanbark oak should not take its place in the Pacific coast hardwood market for many if not all the purposes for which eastern hardwoods are now imported" (Betts 1911). "All things considered," Betts continued, "the seasoning of tanbark oak seems to offer little, if any, more difficulty than is experienced with eastern oaks" (Betts 1911). The wood is particularly well suited for flooring because of its "pleasing grain and color, and the necessary hardness" (Betts 1911). In fact, the Union Lumber Company in Fort Bragg, California had successfully milled tanoak for flooring by 1910 (Huber and McDonald 1992).

The technical ability existed to mill tanoak with no more difficulty than experienced with eastern oaks. Pfeiffer (1956) claimed, "western hardwoods are equally satisfactory as comparable eastern species ... and we need not apologize for any of them where care is exercised in their manufacture." In 1977, it was noted that "native California hardwoods, and specifically tanoak, which could provide a major opportunity for increased wood and fiber production, are scarcely utilized" (McDonald 1977). McDonald (1977) also noted that "reliable techniques are available now and are described extensively in the literature." Tanoak ranked among the densest and stiffest of North American woods (Shelly and Quarles 2013).

Inventories of the tanoak resource indicated that mid-century sawtimber volume was approximately 4.8 million cubic meters or "2,036 million board feet in California" (Roy 1957b). Another estimate from the mid-1980s put the volume of tanoak sawlogs at over

8.6 million cubic meters (3,660 million board feet) in just “the California counties of Del Norte, Humboldt, Mendocino and Sonoma” (Sullivan *circa* 1986). Daniel Oswald (1972) noted that “768,000 acres [310,798 hectares] or 49 percent of the commercial forest land in Mendocino-Sonoma” Counties supported hardwoods, much of which is tanoak. Statewide tanoak dominated over 348,000 hectares (861,000 acres) of California timberlands in 1988, 87% of which was held in private ownership. The same study found that tanoak occurred on over 981,000 hectares (2,425,000 acres) in the state, not including national forests and parks (Bolsinger 1988). The “non-industrial private forestlands of the northern California coast region” alone could potentially sustain extraction of nearly 118 thousand cubic meters (50 million board feet) of tanoak wood annually (Shelly 2001).

Dean Huber and Philip McDonald (1992) asserted that, “Now is the time to develop a philosophy for managing California hardwoods for wildlife, wood, water, and esthetics.” According to those authors, California’s hardwood resource is significant but “poorly managed and scarcely utilized for lumber and wood products” (McDonald and Huber 1994), and they concluded that in the future tanoaks and other California hardwoods “will contribute significantly to the state’s economy ... The art of hardwood silviculture in California should enjoy its finest hour” (McDonald and Huber 1994). Quarantines to limit the spread of *P. ramorum* are already limiting commerce in tanoak hardwood; however, Shelly and Quarles (2013) claim the tree remains worthy of use.

SUDDEN OAK DEATH THREAT

Combined with forestry, catastrophic wildfire, and other disturbances, sudden oak death threatens tanoak “with functional extinction ... throughout large portions of its range” (Dillon et al. 2013). Forest management within the range of tanoak focuses almost exclusively on favoring conifers (softwoods) at the expense of hardwood trees. Since the 1950s, use of herbicides has become common practice to weed industrial western forests of these competing species (Bowcutt 2011). Tanoak has demonstrated “substantial resilience under these adverse conditions, but the introduction of *P. ramorum* into tanoak ecosystems presents a new and significant threat to this species” (Dillon et al. 2013). Much like American chestnut today, tanoak could become reduced primarily to populations of asexually reproduced juveniles that never reach sexual maturity because they are killed by resident *P. ramorum* before the trees can begin to bear acorns. Technically they would not be extirpated, but they would no longer function ecologically or culturally as a key acorn producer in a significant portion of its natural distribution.

Diseased and dying tanoak trees were first noticed in the mid-1990s north of San Francisco in Marin County in the vicinity of Mount Tamalpais (McPherson et al. 2005). Plant pathologists ultimately concluded that a previously undescribed species caused the observed bleeding stem cankers. The new lethal tanoak pathogen, *P. ramorum*, probably originated from eastern Asia, but when it arrived in North America remains uncertain. *Phytophthora* means plant destroyer, aptly named given the devastating impact species in this genus have had historically, such as *Phytophthora infestans*, which caused the Irish Potato Famine. *P. ramorum* obstructs xylem cells and reduces water supply to individual branches or the entire crown, which can ultimately kill the host particularly during drought (Parke et al. 2007). By 2002, sudden oak death had “reached epidemic proportions in coastal California” from the Big Sur Coast to Sonoma County (Rizzo et al. 2002a). The water mold has spread through commerce in garden plants (Mascheretti et al. 2008, Rizzo et al. 2005). Although tanoak

has proved to be the most susceptible, many native California species and common nursery and landscape plants serve as carriers that help spread the pathogen (Rizzo et al. 2002b). Of the ornamental hosts most prone to spread the disease, plant pathologists list *Rhododendron*, *Camellia*, *Viburnum*, *Pieris*, and *Kalmia* (mountain laurel) (Frankel 2008). While fatal to tanoak and some other related tree species, most of its hosts suffer only shoot die back or leaf spots and blotches. An official list of host plants is maintained by the United States Department of Agriculture (USDA 2012).

Plant pathologists, foresters, and others have developed extensive recommendations for land managers working with tanoak threatened by sudden oak death (California Oak Mortality Task Force 2014). According to plant pathologists actively researching the disease, “landscape management strategies for *P. ramorum* must incorporate prevention, treatment, restoration, and conservation into an overall program” (Rizzo et al. 2005). Unfortunately, the exotic disease poses a serious threat even in preserved public lands. Numerous local parks, roughly thirty state park units, and nine federal land holdings are already infested and many more are at risk (Bowcutt 2013). Given the inability to effectively treat sudden oak death, prevention is the first best response to the current tanoak crisis. Plant pathologists recommend further limiting trade in nursery plants to reduce the risk of spreading *P. ramorum* and other devastating plant diseases (Brasier 2008, Rizzo et al. 2005). Other strategies merit consideration including establishment of refuges or reserves, seed banks, living collections of plantings, and educational outreach.

Existing public lands still provide opportunities for safeguarding tanoak. In anticipation of *P. ramorum* range expansion in North America, some land managers are creating tanoak refuges where infection risks can be reduced. For example, Redwood National and State Parks (RNSP) natural resource managers are preparing for “the inevitable arrival of *P. ramorum* to the parks” by adopting preventive measures to slow its spread once it arrives. Park managers recognize tanoak as a valuable ecological component of the coast redwood forests in the park. “RNSP also has an important cultural legacy of large stands of old tanoak trees that have been managed by Native American families for many generations” (Bueno et al. 2010). Park managers are considering “creating tanoak refuges (defined as tanoak groves that are least likely to become infected due to spatial or temporal factors) and protecting them through the creation of no-host buffers.” Grasslands could function as no-host buffers if wide enough around islands of vulnerable tanoaks. Based on epidemiological modeling, widely spaced tanoaks associated with plants that are immune to *P. ramorum* infection “resulted in slow-enough transmission to retain overstorey tanoak” (Cobb et al. 2012). Further, “Recent work identifying heritable disease resistance traits, ameliorative treatments that reduce pathogen populations, and silvicultural treatments that shift stand composition hold promise for increasing the resiliency of tanoak populations” (Cobb et al. 2013). Sudden oak death will probably not get established in the southern end of tanoak’s range because suitable hosts for *P. ramorum* are too scattered in the landscape. Also the climatic conditions are less hospitable to the water mold. Sierra Nevada populations of tanoak may also be safe due to climatic conditions. Based on the computer models, however, the area between Mendocino County and southwestern Oregon is at high risk (Meentemeyer et al. 2011).

CONSERVATION OF GENETIC DIVERSITY

Retaining large tracts of undeveloped land in northern California where extensive tanoak die off without intervention can occur may be important so disease resistance might develop and/or have a chance to express itself. According to Loo (2009), “[m]aintaining large, relatively natural populations of all native tree species will allow natural selection to operate with sufficient intensity to ensure different mechanisms and levels of resistance and tolerance can develop over time, without catastrophic losses of genetic diversity.” Liquidating American chestnut trees for lumber, firewood, and tanbark during the chestnut blight crisis potentially contributed to their loss by not allowing the populations to express or develop disease resistance (Freinkel 2007).

Seed saving may be a successful strategy as “reintroduction of material stored *ex situ* has made the difference between extinction in the wild and continued survival” for some plant species (Guerrant 2012). Unfortunately, tanoak seed saving beyond a year is currently not a viable option because the embryo inside acorns is short lived. Viability plummets with desiccation of the nuts making them resistant “to standard drying and frozen storage, used on species with orthodox seed storage behavior” (E. Guerrant, Portland State University, personal communication). Cryogenic storage of recalcitrant seeds (desiccation resistant) like tanoak may offer an alternative. However, this seed saving approach is labor intensive and more expensive, requiring seed storage at liquid nitrogen temperatures (E. Guerrant, Portland State University, personal communication).

The creation of living collections through plantings could safeguard genetic diversity. American chestnut breeding programs to create chestnut blight resistant individuals relied in part on small-scale plantings in North America that survived outside its natural range (Freinkel 2007). Suitable planting areas outside the natural range of tanoak will likely experience summer drought. For the purpose of conserving genetic diversity and reintroducing tanoak into infested areas, efforts to identify “suitable seed sources will be critical” (Dodd et al. 2013). Seed exchanges could be used as a way to distribute acorns as long as safeguards are in place to ensure that the acorns are disease free. If infected acorns are distributed, this could worsen the current problem. Use of sucker tip layering to reproduce vegetative offshoots of resistant individuals of *P. ramorum* by forcing them to root may be useful in the future (F. Lake, USFS, personal communication).

EDUCATIONAL OUTREACH

One of the biggest challenges to rallying concern for tanoak is the widespread perception that it is a nuisance species with little value and that it competes with economically important species, like coast redwood and Douglas-fir (Bowcutt 2011). Wildlife biologists, ethnobotanists, environmental historians, and others could contribute to an educational campaign designed to counter this misperception. In addition to its substantial value to wildlife and Native Americans, tanoak has a history of being used to tan leather, feed livestock, and make various wooden products including furniture and cabinets (Bowcutt 2011). Botanic gardens, arboreta, parks, natural history museums, and societies dedicated to conserving California’s native plants could provide venues for educational outreach about this indigenous nut tree. Better interpretation about tanoaks is recommended, especially in *P. ramorum* infected parks with high visitation. Efforts to raise awareness of the value of tanoak might include the creation of commemorative U.S. Postal Service stamps modeled

on their vanishing wildlife species program. Possibly non-timber forest product collecting permits could be developed modeled on the Federal Migratory Bird Hunting and Conservation Stamps issued by the U.S. Fish and Wildlife Service which have functioned as hunting licenses and a source of revenue for the creation of wildlife refuges. Duck stamps also helped to educate the public about the plight of waterfowl and the importance of defending their wetland habitats and flyways.

CONCLUSIONS

In 2000, Dr. Steve Zack with the Wildlife Conservation Society said, “The cascading effect of losing these trees is going to be awesome. We’re just waiting for the other shoe to drop” (Yoon 2000). Current efforts to limit the spread of *P. ramorum* are not working adequately. The demise of well over a million tanoaks in less than twenty years suggest it is time more wildlife biologists joined with plant pathologists, botanists, foresters, horticulturalists, landowners, environmental organizations, and tribes in calling for policy changes to accommodate tanoak’s needs to thrive. CDFW could work with the California Wildlife Conservation Board to buy conservation easements and land in northern California as mature tanoak ecological reserves. By leading efforts to defend tanoak, CDFW can demonstrate a commitment to ecosystem management and embrace a shift in its priorities to include plants.

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Plant community characterization and ranking of fens in the Lake Tahoe Basin, California and Nevada

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The California Native Plant Society (CNPS) Vegetation Program worked collaboratively with the USDA Forest Service (USFS) and California Department of Parks and Recreation to produce a vegetation classification, map, and quantitative ranking of sites with fens and wet meadows in the Lake Tahoe Basin. Project goals included surveying and classifying fen sites for their vegetation type, vegetation diversity, and presence of rare species, and ranking sites for their ecological integrity and quality. CNPS staff visited 15 sites in 2010, and completed 57 vegetation stand surveys. We recorded field data using standard CNPS vegetation plot protocols in combination with an expanded USFS Region 5 Fen survey protocol developed for this project. We analyzed the field data from CNPS in 2010 with pre-existing USFS data from 2009-2010. The analysis resulted in a classification of 26 alliances and 38 associations, which are floristically and environmentally defined plant communities per the National Vegetation Classification System. We also established a system for ranking fen sites to assist land managers in recognizing high priority sites and in making long-term management decisions.

Key words: California, classification, fens, Lake Tahoe Basin, meadows, Nevada, ranking, vegetation, wetlands

Fens are peat-forming wetlands, supported by nearly constant groundwater inflow (Bedford and Godwin 2003). Perennial saturation creates oxygen-deprived soils with very low rates of decomposition that allow the accumulation of organic matter produced by wetland plants. They differ from other wet meadows because the deep organic layer in fens means that plants rooting in the peat derive all, or almost all, of their water and nutrients from the peat body, rather than the underlying mineral layer.

Fen peat bodies accumulate very slowly and persist for thousands of years (Wood 1975). Fens also are hotspots of biological diversity. In California, fens have formed in many mountainous and north coastal areas that vary in botanical, ecological, geochemical, and hydrologic characteristics. The perennial supply of water provides refugia for plant and animal species that persist only in fens. Many of these species have their main ranges of distribution far to the north in Alaska and Canada (Chadde et al. 1998), with their southernmost ranges in California or Rocky Mountain fens. The presence of water in fens makes them an important component of surrounding forest ecosystems, providing moisture and forage for animals in drought situations (Cooper and Wolf 2006).

Most fens in California are less than a hectare in size (Sawyer et al. 2009). All peatlands in the Sierra Nevada are fens supported by groundwater flow (Benedict and Major 1982). Fens in the Sierra Nevada often occur in meadow complexes, along with areas of dry meadow or wet meadow, or both, which can be categorized by the depth and persistence of the localized water table (Allen-Diaz 1991, Cooper and Wolf 2006). Most meadows and fens are dominated by herbaceous plants, though they may also have high cover of woody vegetation or mosses (Figure 1).



FIGURE 1.—Variation in vegetation at Ginny Lake Fen, Washoe County, Nevada, July 2010. Photograph by Kendra Sikes.

A main criterion for fen determination is the requirement of at least 40 cm of organic soil in the upper 80 cm of the soil profile (per the U.S. Forest Service [USFS])

Region 5 Fen Protocols; USFS 2010), which is the definition of a Histosol (Soil Survey Staff 1999). Another primary criterion is soil saturation for most of the year. To measure this characteristic, surveyors try to determine whether the water table is within 20 cm of the soil surface during July and August of a normal precipitation year. This saturation criterion is based on fen studies in the Southern Rocky Mountains (Cooper 1990, Chimner and Cooper 2003) and Sweden (Silvola et al. 1996), which found that only those areas where water tables are within 20 cm of the soil surface through July and August accumulated peat (Weixelman and Cooper 2009).

California fens are rare natural communities (CDFG 2010, Sawyer et al. 2009) having unique ecological characteristics and limited range. Recent detailed surveys indicate that each fen or meadow complex may contain few to many vegetation types that are not necessarily rare. In addition, fens have been identified as one of the most sensitive habitat types in the Sierra Nevada (Sierra Nevada Ecosystem Project 1996, USFS 2004). Fens can be classified by their vegetation type, rarity, and diversity as well as by their soils, geomorphology, and hydrologic factors. By identifying vegetation of fens, we are able to better understand the patterns of plant species assemblages, as well as environmental factors that are associated with this rare wetland habitat.

Fen vegetation in the Sierra Nevada has not been well studied or inventoried except in the last decade (Cooper and Wolf 2006, Sikes et al. 2010). Fens and meadows have already been identified in the Tahoe Science Plan (Manley et al. 2009) as special communities that are small in area but have great functional importance. Specific fen sites, including Grass Lake and Hell Hole (Figure 2), have been designated by the Tahoe Regional Planning Agency



FIGURE 2.—Hell Hole Fen, El Dorado County, California, July 2010. Photograph by Kendra Sikes.

as unique and uncommon plant communities for which they have established standards to assure non-degradation of the natural characteristics of the community (TRPA 2011). These sites support a high diversity of species that are often restricted to these communities.

The Lake Tahoe Basin Management Unit (LTBMU) of the USFS carried out a reconnaissance of meadow sites containing fens on their lands, with 10 sites identified prior to 2009, and >35 sites identified during 2009 (S. Gross, USFS, personal communication). Sites were identified as containing fens using the USFS Region 5 criteria for peat accumulation and water table depth (USFS 2010). However, further research was needed to determine the vegetation diversity, complexity, and quality of these fens. Our project addresses current knowledge gaps by providing maps of fens and associated meadows and describing their ecological characteristics, vegetation types, and site conditions.

MATERIALS AND METHODS

Study area.—The Lake Tahoe Basin (LTB) is comprised of Lake Tahoe and the lands that drain into the lake. The LTB spans three counties in California and two in Nevada, and is approximately 70 km in distance lengthwise. It ranges in elevation from 1900 m at lake level to 3300 m at Freel Peak. The LTBMU, managed by the USFS, covers over 75% of the LTB land area across >62,000 hectares (150,000 acres) (USFS 2014).

Existing fen data were provided by the LTBMU staff for our analysis. Like other Region 5 Forests, the LTBMU has been identifying and surveying their fen resources using the Region 5 (R5) fen survey protocol (USFS 2010). LTBMU staff completed an aerial imagery assessment in 2007 to identify potential fen sites across the LTB. During 2009–2010, they visited potential fen sites to collect vegetation data and soil samples to determine whether the sites contain fens. In addition to USFS data, we obtained fen plot data collected on state parks and USFS lands by Stanton et al. (2002). While previous efforts used differing protocols, they contained vegetation data adequate for our analysis.

We selected sites as a diverse subset of known, confirmed fen sites in the LTB region (see Figure 3). Sites visited in August and September 2010 by CNPS were located within three watersheds (hydrological units at the 10-digit level, HU-10; NRCS 2007), and five subwatersheds (HU-12). These sites were selected from five regions in the LTB, which we identified geographically during a larger fen conservation assessment effort for the USFS throughout the Sierra Nevada and adjacent areas, including West Basin, Incline Village, East Basin, South Basin and Meiss Country (Sikes et al. 2010).

Sampling methods.—Sampling was implemented using an Expanded Draft Protocol for USFS R5 Fen Surveys, version August 2010, which incorporated methods from previous versions, the USFS Colorado peatland protocol, and the CNPS vegetation sampling methods. This expanded survey protocol includes two parts. The first part focused on the meadow complex or site and is completed once per location (or per sampling year, if return visits are made). The second part was plot-based and focused on visibly ‘homogenous’ stands of vegetation within the fen-meadow complex. These surveys included ocular estimates of percent cover for all species present within 20-m² plots.

Soil samples were collected from a 40-cm soil column to confirm organic carbon (OC) content in some cases. When the soil column showed distinct horizons, multiple samples were taken, and the width of the portion recorded. We calculated average total carbon (TC) in columns with multiple samples according to the portion of the column that

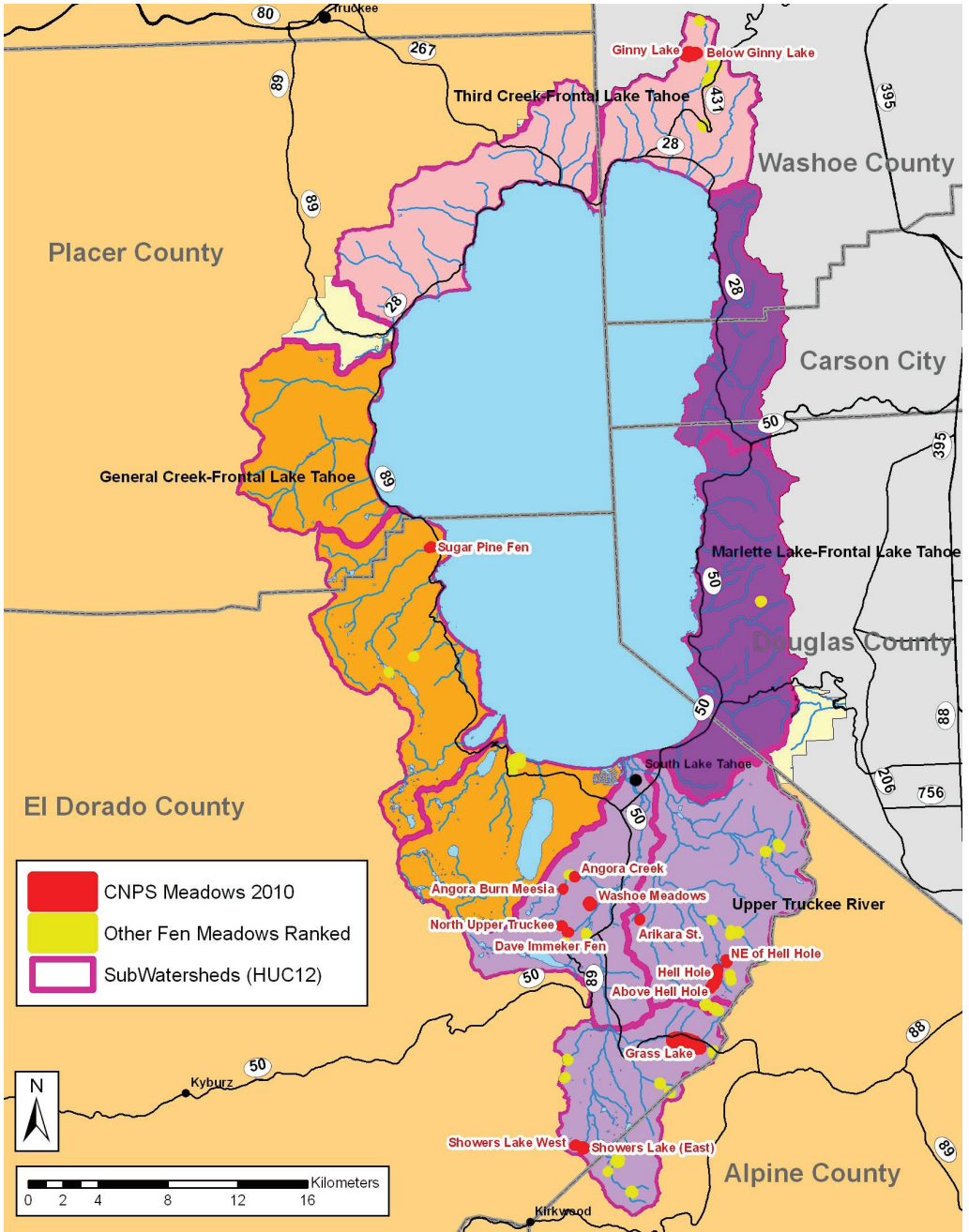


FIGURE 3.—Lake Tahoe Basin study area displaying HU-10 watersheds by color blocks, HU-12 subwatersheds, and ranked sites with confirmed fens including labeled sites visited by CNPS in 2010 (see Appendix I).

each sample represented. Because most sites were previously visited and fen status had already been confirmed with soil analysis, we typically took soil samples only when our vegetation stands were outside of the area previously recognized as a fen. Soils in the LTB are low in clay content (S. Gross, USFS, personal communication), so particle size was not analyzed. We stored the field data in a version of the R5 Fen Geodatabase with modifications, including additional data fields and domains. The original User's Guide (Fischer et al. 2006), our modifications, and some instructions for updating fields were provided to the LTBMU along with the geodatabase.

Vegetation classification analysis.—The vegetation classification in this report is based upon the U.S. National Vegetation Classification (Grossman et al. 1998, NatureServe 2013a). In California, the classification has been developed by the State Natural Heritage Program of the California Department of Fish and Wildlife (CDFW) and CNPS in partnership with NatureServe. In addition to sites visited by CNPS, meadows previously confirmed to have fens in the LTB were included in our analysis. Data from the LTB were combined with other novel information that had not previously been analyzed within the entire Sierra Nevada region, including fen and wet meadow surveys from Sequoia, Shasta-Trinity, Stanislaus, and Tahoe National Forests. Thus, we assembled and analyzed a total of 280 surveys, which included 3,470 plant records, integrating new data with the preliminary classification (Sikes et al. 2010) to begin describing the local variation of LTB fens.

Data quality control procedures prior to analysis included checking plant names for synonymy, reviewing consistency in the taxa at the subspecific or generic level (such that a single name was used for each taxon), lumping infrequently cited taxa to the next highest level, and removing uncommon species that occurred in less than three plots, which reduced the number of taxa from 270 to 191 and avoided potential noise associated with species that were rare in the dataset. Three plots that were statistical outliers and greater than three standard deviations away from the other plots, using Euclidean distance of species composition and abundance, were also deleted. Uncommon species and outlier plots were removed only for the cluster analysis, and are included in the species list and other descriptive information (see Sikes et al. 2011).

Cluster analysis was performed on the 277 surveys described above, using PC-ORD Version 5.05 (McCune and Mefford 2006). The Flexible Beta linkage method of Hierarchical Cluster Analysis (with a value of -0.25) was used along with the Sorensen distance measure, which is a distance measure recommended for species composition data (McCune and Grace 2002). Indicator Species Analysis (Dufrene and Legendre 1997) was used to determine the number of groups that had both a low average P -value and a high number of significant indicator species. The result was 25 groupings of samples based on their species and abundances. These groupings assist in detecting and describing the value of different species for indicating specific environmental conditions. Once samples were assigned to groups, we reviewed each sample individually to identify those that matched current vegetation alliance descriptions and keys and to determine if descriptions of new alliances were warranted.

We had difficulty separating some of the groups that were dominated by *Sphagnum*, *Vaccinium*, *Kalmia*, and *Pinus contorta* ssp. *murrayana*; thus, we performed a second analysis on those groups. We assembled data containing *Sphagnum* as a dominant component from any location in the entire region, including surveys that had been assigned to an association with *Sphagnum* in our preliminary classification (Sikes et al. 2010). This subset included 160

surveys with 184 plant taxa. All *Sphagnum* records, which were the common denominator for the selection, were removed for the cluster analysis, and uncommon species that occurred only once were deleted, with a reduction from 184 to 128 taxa. Using the same analytical process of clustering followed by indicator species analysis, the result was 10 groups.

Mapping methods.—Using a combination of field data collection and aerial photo-interpretation, we altered existing delineations of meadow and fen (or potential fen) extent provided by the USFS, and we created some polygons of new fen sites. In the field, we estimated the extent of each fen using a soil probe (identifying boundaries or areas of at least 40 cm of peat depth), drew the outline on printed aerial imagery, or used a GPS to mark the boundary. In the office, we used the field data and aerial imagery (FSA 2009) to allow computer digitizing of the information recorded in field sketches and GPS data. We also mapped the surveyed vegetation stands using plot photographs and other field data.

For each site, we created a separate map that displays vegetation stands, a fen delineation, and a meadow opening outline (see Figure 4). The meadow outlines provide general indicators for the size and extent of the fen meadow complexes, but they are not based on a specific scientific definition of a meadow. Values for the meadow areas based on those outlines were used as one of the viability factors in assessing that criterion (see below).

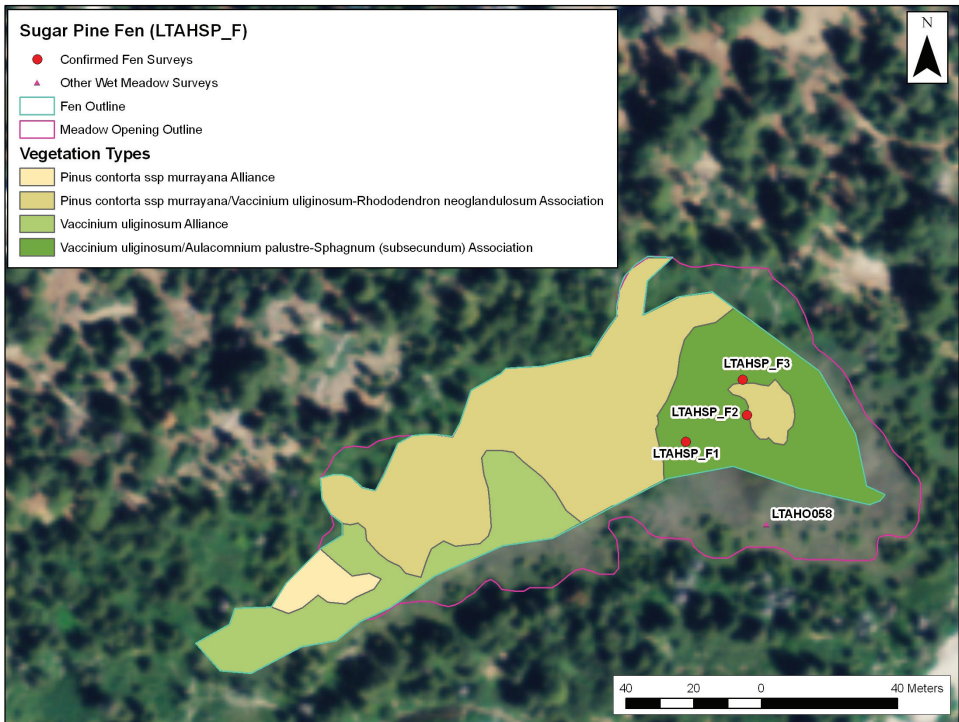


FIGURE 4.—Vegetation map and surveys at Sugar Pine Fen, General Creek – Frontal Lake Tahoe Subwatershed and Watershed, Ed Z'Berg Sugar Pine State Park, El Dorado County, California.

Ranking criteria and methods.—We ranked all sites within the LTB where wet meadows have been confirmed as containing fens, which consisted of 49 confirmed fen meadows (out of 125 individual meadows with data available) in four watersheds and seven subwatersheds in four counties (Alpine and El Dorado counties in California, and Washoe and Douglas counties in Nevada). These do not, however, represent a complete inventory of fens within the LTB. Additional fens likely exist on private lands, or on public land, that were not detected in the initial aerial photo interpretation and follow-up field surveys.

We adapted the ranking system developed by Chadde et al. (1998) for the Rocky Mountains, which subjectively assesses each peatland using seven criteria on a 3-point scale. We modified the Chadde et al. (1998) system by including two types of within-meadow diversity, physical/topographic diversity and biological diversity (or biodiversity), making 8 criteria. We have attempted to more objectively rate each criterion, by assembling and combining various factors with quantifiable characteristics to determine ratings. Since we were able to quantify a fairly large range of variation for some criteria assessed, we have chosen a 5-point scale to score each site for each criterion. An overall “conservation significance” ranking was the sum of the eight criteria, as defined below: uniqueness, quality, rarity (of plant species and vegetation types), biological diversity (or biodiversity of plants), physical / topographical diversity, viability, defensibility, and scientific and educational value. Lower ranking values represented lower conservation significance based on comparisons across the current data using this quantitative system.

Uniqueness was represented by three environmental conditions (elevation, geology, and pH), and whether these aspects were within the normal range for the group of sites or outside of the primary distribution. Quality was represented by minimal impacts or disturbance noted, distance to the closest road, past range-allotment status, and designation within an urban defense zone. Rarity was scored according to the presence of rare plant species and rare vegetation types. Biodiversity was represented by species richness, presence of woody-dominated fen types, and number of vegetation types per site. Physical diversity was scored according to the presence of five topographical features, general topographic complexity, and the number of water sources recorded. Viability was rated according to size of the fen complex, presence of other fens within the subwatershed, and distance to nearest fens. Defensibility was based on our knowledge of each site’s state of protection, and whether they were within specially designated zones. Lastly, scientific and educational value were determined by accessibility of the sites and whether they had been used in the past for research or educational purposes. Further information on these criteria can be found in Sikes et al. (2011).

For 13 of the 49 ranked meadow locations, we had complete meadow diversity data obtained after revising and expanding the USFS R5 protocol. For the other 36 fen meadows with confirmed fens, a previous version of the R5 fen meadow protocol was used; therefore, we were not able to fully assess their diversity criteria and their rank. These sites received automatic scores of 3 (intermediate) for biodiversity and physical diversity.

RESULTS

Species and vegetation data.—We visited 15 different sites within the LTB, with 13 in the USFS-managed lands and two on lands managed by California State Parks (Figure 3). USFS staff conducted other surveys during 2009–2010 in LTB fen meadows (see Appendix

I for locations). Multiple stand or plot surveys were often conducted at each site to capture information on distinct stands of vegetation and to define the plant communities in each fen (Appendix I). In all, 243 vascular plant taxa and 44 nonvascular species were identified in the combined surveys.

CNPS staff recorded five rare plant species in the 2010 fieldwork, along with three species of *Sphagnum* which are on the Special Interest List for the LTBMU. Six additional rare species and a fourth species of *Sphagnum* have been recorded in fens of the LTB, according to existing USFS and California Natural Diversity Database (CNDDDB 2011) records (Appendix II). Eight of these species are CRPR 2B plants, which are rare in California but more common elsewhere. Two species are CRPR 4 plants, taxa to watch because they have such limited distribution. The last moss on the list, *Tomentypnum nitens*, was recently found in California for the first time; it is not currently ranked, but review from the CNDDDB has been requested.

Soil analysis.—Thirty-five soil samples from 40-cm soil columns were analyzed for organic carbon (OC) and total nitrogen. The 35 samples came from 18 soil pits at 12 sites. One of the 18 soil pits did not meet criteria for organic soils and, thus, was not classified as a fen. Samples from three pits returned inconclusive results, since their total percentage carbon was within the histosol range (between 12 and 18%) that depends on clay content, which we did not assess. To be defined as organic, soil OC must be greater than 18% if the soil is greater than 60% clay, and it must be greater than 12% OC if the soil is without clay (Soil Survey Staff 1999).

Vegetation classification and mapping.—Our classification includes 26 alliances and 38 associations assigned to 177 stand samples from the Lake Tahoe Basin (Appendix III). Thirty-five plots were not classified to the association level due to unusual species composition or generic-level plant identifications, though most fit into definitions of existing alliances. Sixty-five additional stand samples were analyzed from other locations beyond LTB.

We categorized 10 associations that were not previously present in our 2010 classification of fens in the Sierra Nevada (Sikes et al. 2010). Three of the ten associations have not been previously described: *Carex simulata*–*Carex scopulorum* (Provisional), *Oreostemma alpigenum*, and *Sphagnum*–graminoid (Provisional) (Figure 5). The other seven associations have been described by other authors, including four listed by Sawyer et al. (2009). One of the ten associations, *Carex aquatilis*–*Carex utriculata*, has not been previously ascribed to California, but is known from Colorado and Montana (NatureServe 2013b). Appendix III provides the State (S) ranks for alliances and a designation for association rarity (T. Keeler-Wolf, California Department of Fish and Wildlife, personal communication).

Upon visiting 15 sites and conducting 57 stand surveys, CNPS staff updated a geodatabase of point data and polygon boundaries for meadows and fens in the LTB. Based on the new data, other data from the USFS, and aerial imagery, we created detailed maps showing 39 fen vegetation types in 109 polygons representing stands within 14 sites. Fen site maps are provided by Sikes et al. (2011; Figure 4).

Site ranking.—Conservation significance ratings for the known fens of the Lake Tahoe Basin are presented in Appendix IV. A sum of scores for the eight individual criteria led to total scores ranging from 18 to 30 (out of a possible score of 40) and resulted in 13 levels or unique values for the 49 sites (Figure 6).



FIGURE 5.—An example of the newly described *Oreostemma alpigenum* Association, Washoe Meadows Fen, El Dorado County, California. Photograph by Kendra Sikes.

The fens rating highest for conservation significance are Dave Immecker Fen and Grass Lake East (Figure 7). Several others of the South Basin region rated higher than those in any other fen region of the LTB. The three subwatersheds (HU-12) of the Truckee River watershed had the three highest average conservation ranks, in addition to being the subwatersheds with the most fens recorded. More specifically, the average conservation significance rating for fens of the Angora Creek subwatershed was the highest at 25.0 ($n=10$). The lowest average rating was 21.0 for both Incline Lake subwatershed in the Incline Village fen region ($n=8$) and Fallen Leaf Lake subwatershed in the south basin fen region ($n=2$).

DISCUSSION

We recorded new occurrences for three rare mosses (*Bruchia bolanderi*, *Meesia triquetra*, and *Tomentypnum nitens*) and two rare vascular plants (*Carex limosa* and *Eriophorum gracile*; Figure 8), adding to the resource assessment of the region. We also have documented a richness of vegetation from woody to herbaceous types in LTB fens. The LTB contains approximately half the number of alliances currently identified across all fen habitats in the Sierra Nevada and southern Cascade Ranges (Sikes et al. 2010). This vegetation alliance richness can be attributed to the geologic, hydrologic, and topographic complexity in the LTB.

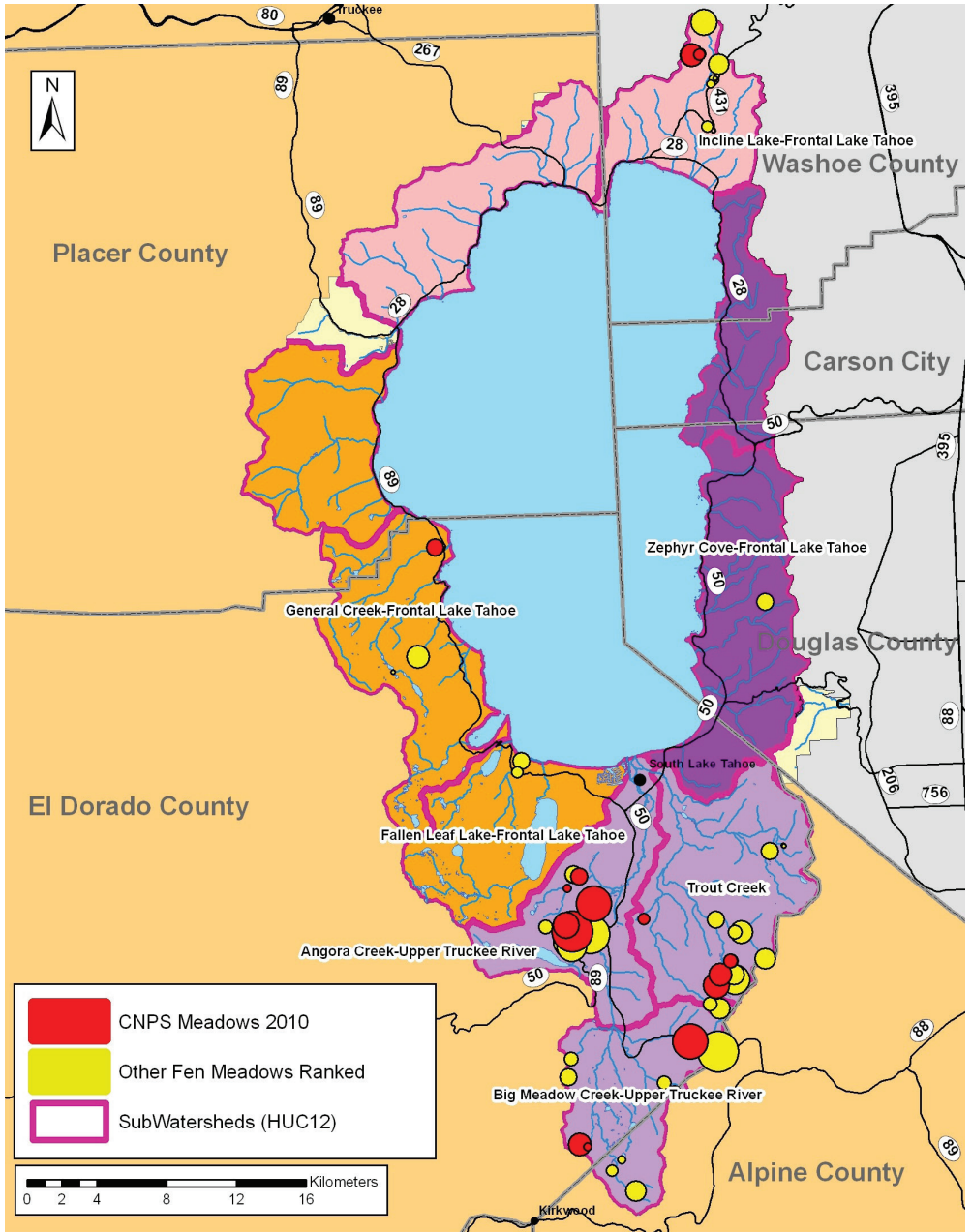


FIGURE 6.—Conservation Rankings of fen sites symbolized with graduated circles, and names of subwatersheds displayed; the largest circles are the highest ratings. Lake Tahoe Basin, California and Nevada.



FIGURE 7.—Grass Lake Fen, El Dorado County, California is a vast fen meadow complex containing a diversity of vegetation types. Photograph by Julie Evens.

By analyzing existing data, we have ranked 49 confirmed fen sites in the LTB based upon eight conservation significance criteria, including inherent diversity considerations and management-related criteria. One value of the ranking process lies in the recognition of vegetation diversity along with other important botanical, site history, and environmental characteristics in some of the smaller and less well-known fens. The application of and expansion of the Chadde et al. (1998) rankings (from seven to eight criteria by splitting within-meadow diversity into the biological diversity and physical-topographic diversity measures) enables managers to consider biological factors separate from environmental factors in evaluating sites. For example, managers can evaluate fen sites for biodiversity and/



FIGURE 8.—*Carex limosa* and *Eriophorum gracile* are rare plants that also form rare vegetation assemblages at Grass Lake Fen, El Dorado County, California. Photograph by Julie Evens.

or rarity (of plant species/vegetation types) as important values for conservation, which may or may not have strong correlations with other environmental or geographical factors. More than five fens that have high rankings (of 4 or 5) for rarity also have high rankings for site quality, while only two fen sites that have high rankings (of 4 or 5) for rarity (of vegetation and species) have high physical diversity. Many fens that have low rankings (1 or 2) for rarity (of species and vegetation) also have low rankings for uniqueness (taking elevation, geology and pH into consideration). On the other hand, no clear correlation between quality and topographic diversity was observed. Thus, a manager could choose between sites with higher quality or uniqueness to maintain their rarity components.

Additionally, the combined conservation significance ranks can assist land managers in making restoration and other management decisions by providing a means for direct comparison between sites. Depending on their purpose, managers can consider only one or more ranking criteria when comparing fens and setting priorities for management, and they should consider high levels of protection for fens receiving high scores (4 or 5) in any criterion. Regarding the regional variation in the rankings, managers can focus on restoration plans for the Incline Village sites with low average ranks (e.g., to restore hydrologic functions), while drafting long-term protective measures to maintain the uniqueness, quality, diversity, or rarity of South Basin sites.

This project presents quantitative and repeatable procedures based upon a comparison of the entire set of sites, with ranks for each criterion based on the position of other sites in a continuum (i.e., rank order was used to assign scores for the 8 different criteria). The additional data collected at each site using the revised protocol added negligible time to the surveys and provided further information to help distinguish features of fen sites; we recommend including these in future surveys (whether ranking of fens is intended or not). We postulate that at least 75% of the fens in the LTB have now been identified and inventoried. Information for remaining unidentified fen locales could be inventoried and incorporated to evaluate the full extent and significance of fen resources of the LTB. However, new sites or new information would require a full analysis of the entire set of fen sites to be incorporated into the ranking.

Our results demonstrate new techniques for ranking the significance of fen sites in the LTB that could be utilized across other lands in California, and beyond, to assist in long-term conservation and management. Our study also contributes to the knowledge of vegetation within fens, including the identification of 10 new associations. Twenty out of the 30 associations in our classification are considered rare. While the vegetation of fens in the LTB may not be particularly unique in comparison to other sites within the Sierra Nevada, the LTB exhibits a high degree of vegetation richness and rarity. Taken together, the fens of the LTB provide an excellent representation of a habitat that is scarce in California.

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APPENDIX I: CONFIRMED FEN SITES WITH STAND SURVEYS ANALYZED IN THE LAKE TAHOE BASIN, CALIFORNIA AND NEVADA

The stands surveyed by the California Native Plant Society are displayed in bold font, and other existing United States Forest Service surveys are in regular font. For subwatersheds and watersheds, Frontal Lake Tahoe is abbreviated as (FLT), and site names are general descriptors rather than official placenames.

Subwatershed	Watershed	Site Name	Owner	No. of Surveys	County	Fen Region
Angora Creek–Upper Truckee River	Upper Truckee River	Angora Burn Meesia	USFS	1, 3	El Dorado, CA	South Basin
		Angora Creek	USFS	3	El Dorado, CA	South Basin
		Dave Immeker Fen	USFS	4	El Dorado, CA	South Basin
		Near Angora Fen	USFS	2	El Dorado, CA	South Basin
		North Upper Truckee	USFS	2, 2	El Dorado, CA	South Basin
		Osgood Swamp South	USFS	3	El Dorado, CA	South Basin
		Osgood Swamp West	USFS	2	El Dorado, CA	South Basin
		SE of Angora Lakes	USFS	1	El Dorado, CA	South Basin
		Upper Truckee River	USFS	4	El Dorado, CA	South Basin
		Washoe Meadows	State Park	5, 2	El Dorado, CA	South Basin
Big Meadow Creek–Upper Truckee River	Upper Truckee River	Meiss Lake	USFS	2	Alpine, CA	Meiss Country
		South of Meiss Lake	USFS	2	Alpine, CA	Meiss Country
		SW of Meiss Lake	USFS	2	Alpine, CA	Meiss Country
		Big Meadow Creek	USFS	2	El Dorado, CA	Meiss Country
		NW of Elbert Lake	USFS	1	El Dorado, CA	Meiss Country
		Showers Lake	USFS	3, 1	El Dorado, CA	Meiss Country
		Showers Lake West	USFS	2, 1	El Dorado, CA	Meiss Country
		W of Elbert Lake	USFS	1	El Dorado, CA	Meiss Country
		Freel Meadows	USFS	1	El Dorado, CA	South Basin
		Grass Lake East	USFS	2	El Dorado, CA	South Basin
Fallen Leaf Lake–FLT	General Creek–FLT	Big Meadow	USFS	1	Alpine, CA	Meiss Country
		Grass Lake West	USFS	9, 15	El Dorado, CA	South Basin
		Tallac Creek Lower	USFS	1	El Dorado, CA	South Basin
Upper Truckee River	Upper Truckee River	Tallac Creek Upper	USFS	1	El Dorado, CA	South Basin

APPENDIX I (CONTINUED)

Subwatershed	Watershed	Site Name	Owner	No. of Surveys	County	Fen Region
General Creek-FLT	General Creek-FLT	Crag Lake	USFS	1	El Dorado, CA	West Basin
		Shay Fen, Lonely Gulch	USFS	1	El Dorado, CA	West Basin
		Sugar Pine Fen	State Park	3, 1	El Dorado, CA	West Basin
Incline Lake-FLT	Third Creek-FLT	Below Ginny Lake	USFS	5, 1	Washoe, NV	Incline Village
		Below Incline Lake, E	USFS	1	Washoe, NV	Incline Village
		Below Incline Lake, W	USFS	1	Washoe, NV	Incline Village
		Ginny Lake	USFS	6, 1	Washoe, NV	Incline Village
		Incline Lake	USFS	3	Washoe, NV	Incline Village
		Liz Fen	USFS	1	Washoe, NV	Incline Village
		Mount Rose	USFS	3	Washoe, NV	Incline Village
		S. of Incline Lake	USFS	1	Washoe, NV	Incline Village
		Trout Creek	Upper Truckee River	Above Hell Hole	USFS	3
Arikara St.	USFS			1, 4	El Dorado, CA	South Basin
Armstrong Fountain Place E	USFS			1	El Dorado, CA	South Basin
Armstrong Pass	USFS			3	El Dorado, CA	South Basin
Bear Glade, High Mdws	USFS			2	El Dorado, CA	South Basin
Fountain Place	USFS			3	El Dorado, CA	South Basin
Fountain Place Road	USFS			2	El Dorado, CA	South Basin
Freel Meadows W, top of Saxon	USFS			1	El Dorado, CA	South Basin
Hell Hole	USFS			9, 10	El Dorado, CA	South Basin
High Meadow	USFS			2	El Dorado, CA	South Basin
Lower East of Hell Hole	USFS			2	El Dorado, CA	South Basin
NE of Hell Hole	USFS			4, 1	El Dorado, CA	South Basin
Upper East of Hell Hole	USFS			1	El Dorado, CA	South Basin
Zephyr Cove-FLT	Marlette Lake-FLT	W. of Genoa Peak	USFS	1	Douglas, NV	East Basin

APPENDIX II: SPECIAL STATUS PLANTS FOUND IN MEADOWS WITH CONFIRMED FENS IN THE LAKE TAHOE BASIN, CALIFORNIA AND NEVADA

The number following the decimal point in the CRPR is the threat rank, where 0.2 indicates a moderate degree of threat, and 0.3 indicates a low degree in California (CNPS 2014).

Scientific name	Common name	Family	Global rank ^a	State rank ^b	USFS status ^c	CRPR ^d	No. of fen sites
Vascular Plants							
<i>Carex limosa</i>	mud sedge	Cyperaceae	G5	S3	SI	2B.2	8
<i>Epilobium palustre</i>	marsh willowherb	Onagraceae	G5	S2	SI	2B.3	1
<i>Eriophorum gracile</i>	slender cottongrass	Cyperaceae	G5	S3.3		4.3	6
<i>Schoenoplectus subterminalis</i>	water bulrush	Cyperaceae	G4 G5	S3	SI	2B.3	1
<i>Scutellaria galericulata</i>	marsh skullcap	Lamiaceae	G5	S2	SI	2B.2	1
<i>Utricularia ochroleuca</i>	cream-flowered bladderwort	Lentibulariaceae	G4?	S1		2B.2	1
Non Vascular Plants							
<i>Bruchia bolanderi</i>	Bolander's bruchia	Bruchiaceae	G3	S3?	R5S	2B.2	3
<i>Helodium blandowii</i>	Blandow's helodium	Helodiaceae	G5	S1	R5S	2B.3	2
<i>Meesia triquetra</i>	three-ranked hump moss	Meesiaceae	G5	S4	R5S	4.2	15
<i>Meesia uliginosa</i>	broad-nerved hump moss	Meesiaceae	G4	S3	R5S	2B.2	1
<i>Sphagnum russowii</i>	Russow's peat moss	Sphagnaceae	G5	NR ^e	SI	–	3
<i>Sphagnum squarrosum</i>	spreadleaf peat moss	Sphagnaceae	G5	NR	SI	–	3
<i>Sphagnum subsecundum</i>	sphagnum	Sphagnaceae	G5	NR	SI	–	3
<i>Sphagnum teres</i>	sphagnum	Sphagnaceae	G5	NR	SI	–	1
<i>Tomentypnum nitens</i>	tomentypnum moss	Brachytheciaceae			SI		2

^a Global Rank is assigned to each species according to its global range, with G3 defined as vulnerable [at moderate risk of extinction], G4 as apparently secure [uncommon but not rare], and G5 as globally common (CNDDB 2011, CNPS 2014).

^b State Rank is assigned to represent the taxa's status within the state, where S1 is critically imperiled, S2 is imperiled, S3 is vulnerable, and S4 is apparently secure (CNDDB 2011, CNPS 2014).

^c Special status designations by USFS are represented as R5S for taxa listed on the USFS Pacific Southwest Regional Forester's Sensitive Species List (USFS 2006) and SI for plants on the LTBMU Special Interest List.

^d CRPR= California Rare Plant Rank = CRPR (CNPS 2014); see *Species and Vegetation Data* section within the Results for more details.

^e NR = not ranked.

APPENDIX III: VEGETATION CLASSIFICATION OF ALLIANCES AND ASSOCIATIONS WITH SAMPLE SIZE FOR FENS (AND OTHER RELATED WET MEADOWS) IN THE LAKE TAHOE BASIN, CALIFORNIA AND NEVADA

The state rarity status of each alliance is provided in bold font after its name, where S2 = Imperiled (6–20 viable occurrences statewide), S3 = Vulnerable (21–100 viable occurrences statewide), and S4 = Secure (>100 viable occurrences statewide). Rare associations are marked with an asterisk (*). The number of confirmed fen stands is given under No. of surveys, and in parentheses is the number of additional surveys not confirmed as fens.

Alliance	Association	No. surveys of fens (No. other)
WOODLAND		
<i>Pinus contorta</i> ssp. <i>murrayana</i> S4		3 (2)
<i>Pinus contorta</i> ssp. <i>murrayana</i> / <i>Vaccinium uliginosum</i> – <i>Rhododendron columbianum</i>		1 (1)
<i>Pinus contorta</i> ssp. <i>murrayana</i> / <i>Carex</i> spp.		
SHRUBLAND		
<i>Alnus incana</i> S4		4 (3)
<i>Alnus incana</i> ssp. <i>tenuifolia</i> *		
<i>Kalmia microphylla</i> S3		9 (2)
<i>Kalmia microphylla</i> / <i>Sphagnum fuscum</i> , <i>subsecundum</i> *		
<i>Rhododendron columbianum</i> S3		(1)
<i>Rhododendron columbianum</i> / <i>Pinus contorta</i> ssp. <i>murrayana</i> *		
<i>Salix eastwoodiae</i> S3		1
<i>Salix eastwoodiae</i> *		
<i>Salix lemmonii</i> S3		1
<i>Salix lemmonii</i> / <i>Carex</i> spp. *		2 (4)
<i>Salix lemmonii</i> /mesic forb *		
<i>Salix orestera</i> S4		3
<i>Salix orestera</i> / <i>Carex (scopulorum)</i>		3
<i>Salix orestera</i> /moss (Provisional)		
<i>Vaccinium uliginosum</i> S3		2
<i>Vaccinium uliginosum</i> / <i>Sphagnum teres</i> (Provisional) *		5 (4)
<i>Vaccinium uliginosum</i> / <i>Aulacomnium palustre</i> – <i>Sphagnum (subsecundum)</i> *		

APPENDIX III (CONTINUED)

Alliance	Association	No. surveys of fens (No. other)
HERBACEOUS		
	<i>Bistorta bistortoides</i> – <i>Mimulus primuloides</i> S4	
	<i>Bistorta bistortoides</i> – <i>Mimulus primuloides</i>	(1)
	<i>Mimulus primuloides</i>	1 (1)
	<i>Carex (aquatilis, lenticularis)</i> S3-S4	
	<i>Carex aquatilis (lenticularis)</i>	9 (7)
	<i>Carex aquatilis</i> – <i>Carex utriculata</i> *	3 (3)
	<i>Carex (illota, luzulina)</i> /Bryum pseudotriquetrum S2?	
	<i>Carex capitata</i> (Provisional) *	(1)
	<i>Carex luzulina</i> /Bryum pseudotriquetrum *	1
	<i>Carex (utriculata, vesicaria)</i> S4	
	<i>Carex vesicaria</i>	5 (5)
	<i>Carex utriculata</i>	13 (5)
	<i>Carex canescens</i> (Provisional) S2?	
	<i>Carex canescens</i> (Provisional) *	1
	<i>Carex limosa</i> S3?	
	<i>Carex limosa</i> – <i>Menyanthes trifoliata</i> *	13 (1)
	<i>Carex nebrascensis</i> S4	
	<i>Carex nebrascensis</i>	1 (4)
	<i>Carex scopulorum</i> S3-S4	
	<i>Carex scopulorum</i>	3 (5)
	<i>Carex simulata</i> S3?	
	<i>Carex simulata</i> *	3 (1)
	<i>Carex simulata</i> – <i>Carex utriculata</i> *	8 (1)
	<i>Deschampsia cespitosa</i> S4?	
	<i>Deschampsia cespitosa</i> – <i>Perideridia parishii</i>	(1)
	<i>Deschampsia cespitosa</i> – <i>Carex nebrascensis</i>	1 (4)
	<i>Eleocharis quinqueflora</i> S4	
	<i>Eleocharis quinqueflora</i> /Philonotis fontana–Bryum pseudotriquetrum *	1
	<i>Eleocharis quinqueflora</i>	(4)
	<i>Eleocharis quinqueflora</i> /Drepanocladus (aduncus, sordidus) *	5
	<i>Juncus arcticus</i> S4	
	<i>Juncus arcticus</i> var. <i>balticus</i>	(1)
	<i>Muhlenbergia filiformis</i> (Provisional) S4?	
	<i>Muhlenbergia filiformis</i> (Provisional)	4
	<i>Nuphar lutea</i> (Provisional) S3	
	<i>Nuphar lutea</i> ssp. <i>polysepala</i> (Provisional) *	1 (2)
	<i>Oreostemma alpigenum</i> –(<i>Gentiana newberryi</i>) S4?	
	<i>Oreostemma alpigenum</i>	2
	<i>Phalacroseris bolanderi</i> – <i>Juncus oxymyris</i> S3	
	<i>Juncus oxymyris</i> /Philonotis fontana *	1
	<i>Sphagnum</i> spp. S3	
	<i>Sphagnum</i> –graminoid *	1 (1)
	<i>Veratrum californicum</i> S4	
	<i>Veratrum californicum</i>	(1)

APPENDIX IV: SUMMARY OF CONSERVATION SIGNIFICANCE RANKINGS FOR CONFIRMED FEN SITES IN THE LAKE TAHOE BASIN, CALIFORNIA AND NEVADA

Sites are arranged by Fen Region and then by significance rankings (highest to lowest, with higher values indicating more noteworthy fens). Sites in bold font were visited by CNPS and were afforded full assessment, whereas other sites received automatic scores of 3 for biodiversity and physical diversity. Refer to Figure 6 for a graphical representation of these scores.

Site Name	Uniqueness	Quality	Rarity	Biodiversity	Physical Diversity	Viability	Defensibility	Value	Total Score
East Basin, Douglas Co., Nevada									
W of Genoa Peak	1	5	5	3	3	1	3	1	22
Incline Village, Washoe Co., NV									
Mount Rose	3	5	3	3	3	2	3	3	25
Ginny Lake	3	4	2	2	4	4	3	2	24
Incline Lake	1	3	2	3	3	5	2	4	23
Below Ginny Lake	3	3	2	3	2	3	2	2	20
Liz Fen	3	2	1	3	3	2	2	4	20
Below Incline Lake, W	1	3	1	3	3	3	2	3	19
S of Incline Lake	1	5	1	3	3	2	2	2	19
Below Incline Lake, E	1	3	1	3	3	2	2	3	18
Meiss Country, Alpine and El Dorado Cos., CA									
Showers Lake West	5	3	4	3	3	3	3	1	25
South of Meiss Lake	3	4	2	3	3	4	3	1	23
W of Elbert Lake	3	4	1	3	3	4	3	1	22
NW of Elbert Lake	1	5	1	3	3	4	3	1	21
Big Meadow Creek	3	4	1	3	3	3	3	1	21
SW of Meiss Lake	3	4	1	3	3	2	3	1	20
Big Meadow	1	4	3	3	3	2	3	1	20
Meiss Lake	1	4	2	3	3	2	3	1	19
Showers Lake	3	3	2	1	2	4	3	1	19
West Basin, El Dorado Co., CA									
Shay Fen, Lonely Gulch	3	5	5	3	3	1	3	1	24
Sugar Pine Fen	3	3	4	4	1	1	3	3	22
Crag Lake	1	5	1	3	3	1	3	1	18
South Basin, El Dorado Co., CA									
Grass Lake East	1	4	5	3	3	4	5	5	30
Dave Immecker Fen	5	4	5	4	3	3	3	3	30
Upper Truckee River	3	5	5	3	3	4	3	3	29
Grass Lake West	1	3	4	3	3	4	5	5	28
Washoe Meadows	3	5	4	3	3	3	4	3	28
Osgood Swamp West	5	4	5	3	3	3	3	3	27
Upper E of Hell Hole	5	3	3	3	3	4	4	1	26

APPENDIX IV (CONTINUED)

Site Name	Uniqueness	Quality	Rarity	Biodiversity	Physical Diversity	Viability	Defensibility	Value	Total Score
South Basin, CA (cont.)									
Osgood Swamp South	3	3	4	3	3	3	3	4	26
Above Hell Hole	1	3	5	2	4	5	4	1	25
Freel Meadows	3	3	3	3	3	5	4	1	25
North Upper Truckee	3	4	3	4	3	3	3	2	25
Armstrong Fountain Pl. E	1	3	3	3	3	5	3	3	24
Hell Hole	1	2	4	2	5	5	4	2	24
Lower East of Hell Hole	1	3	4	3	3	4	4	1	23
Armstrong Pass	3	3	3	3	3	4	3	1	23
Angora Creek	3	1	3	4	1	3	3	5	23
Fountain Place Road	1	3	3	3	3	4	2	3	22
Near Angora Fen	3	1	3	3	3	2	3	4	22
Tallac Creek Lower	3	2	3	3	3	2	2	4	22
Bear Glade, High Mdws	1	4	3	3	3	3	2	3	22
NE of Hell Hole	1	3	2	3	2	5	4	1	21
Freel Meadows West, top of Saxon	3	3	1	3	3	4	3	1	21
SE of Angora Lakes	1	5	1	3	3	2	4	2	21
Fountain Place	1	3	1	3	3	5	2	3	21
Tallac Creek Upper	3	4	1	3	3	1	2	3	20
Arikara St.	3	1	2	5	2	2	2	3	20
Angora Burn Meesia	3	1	3	4	1	2	3	3	19
High Meadow	1	4	1	3	3	2	2	2	18

Recognizing a new species of *Silene* (Caryophyllaceae) from California: a splitter's game?

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Silene krantzii T.R. Stoughton is a new species endemic to higher elevation, alpine habitats in the San Bernardino Mountains, San Bernardino County, California. The new species appears to be abundant in locations where it has been observed, but it is narrowly restricted in overall distribution, presumably due to the limited availability of suitable habitat. *Silene krantzii* sp. nov. is morphologically similar to a closely related species in the San Bernardino Mountains, *Silene verecunda* S. Watson, but differs from this taxon in the San Bernardino Mountains by the presence of glandular trichomes on the basal leaves and proximal stems, smaller leaf size, a red calyx that is inflated at flowering, and a short-statured, spreading habit. *Silene krantzii* also occurs in a unique ecological setting. Qualitative evidence of these differences is summarized for the new species and relevant information regarding ongoing study of genetic diversity within the *S. verecunda* complex is discussed.

Key words: alpine, catchfly, morphology, natural history, San Bernardino Mountains, San Gorgonio Mountain, *Silene krantzii*, *Silene verecunda*, Southern California, species concepts

Over the course of multiple botanical forays to the alpine zone of San Gorgonio Mountain, located in the San Bernardino Mountains, California, several plants resembling *Silene verecunda* S. Watson (1875) were collected by the first author. All known specimens previously identified as *Silene verecunda* subsp. *platyota* (S. Watson) C. L. Hitchcock & Maguire (1947) from subalpine (<3,000 meters), mixed conifer forests of the San Bernardino Mountains held at RSA/POM and UC/JEPS herbaria were examined. We observed several distinct differences between *S. verecunda* subsp. *platyota* and the unidentified *Silene* collections from the alpine zone of San Gorgonio Mountain. We now propose that these differences are sufficient to recognize the populations on San Gorgonio Mountain as a

new species of *Silene* in the San Bernardino Mountains, utilizing a taxonomic concept proposed by Cronquist (1978) in which taxa are circumscribed based on discontinuity of morphological features. Natural history information included here lends additional support for the recognition of a new species of *Silene* endemic to the San Gorgonio summit region of the San Bernardino Mountains drawing on aspects of more contemporary species discourse recently articulated by Baum (2009).

TAXONOMY

Silene krantzii T.R. Stoughton, sp. nov. (Figures 1–4).—TYPE: USA, California, San Bernardino Mountains, ‘The Tarn’ just below (south of) the San Gorgonio summit region (Figure 1A), alpine gravel fell-field with *Festuca saximontana*, *Raillardella argentea*, *Calyptridium umbellatum*, *Hulsea vestita* subsp. *pygmaea* and *Elymus elymoides*, granitic substrates, 3,272 m elev., 34° 5’ 34.37” N, 116° 49’ 23.30” W, 23 July 2011, Thomas Stoughton 1391, with R. Shores, M. Seccombe, and P. Boyd (Holotype: RSA806300; Isotypes: GB, JEPS, CAS, NY, UCR).

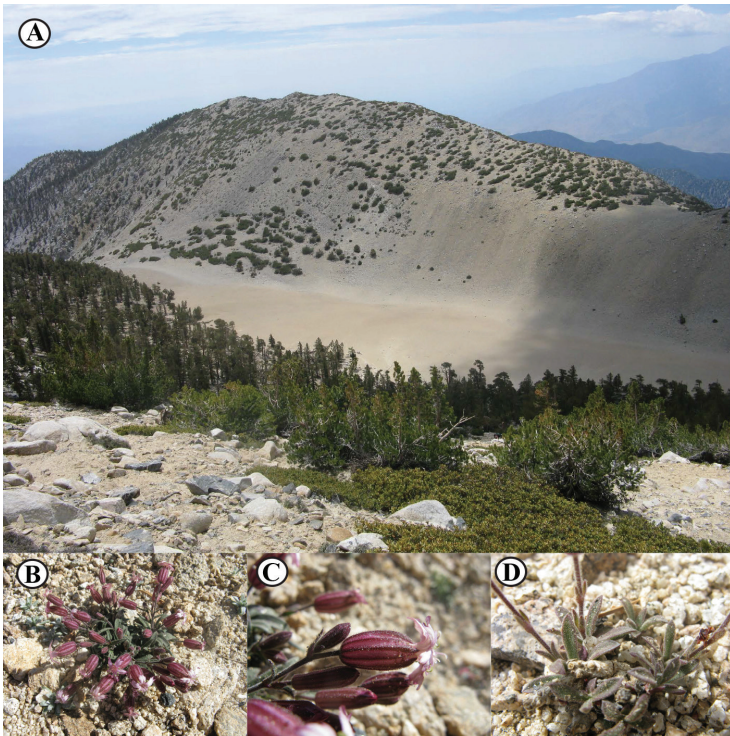


FIGURE 1.—*Silene krantzii* in native habitat. A. Alpine fell-field habitat at the type locality, “The Tarn” just below the south face of San Gorgonio Mountain in the San Bernardino Mountains, San Bernardino County, California. B. Overall plant habit demonstrating prostrate nature of stems and racemose inflorescences. C. Flower displayed in side-view, showing inflated and ribbed condition of the mature calyx in flowering. D. Basal leaves and stems, showing oblanceolate shape and presence of glandular trichomes. Photographs by Thomas R. Stoughton.

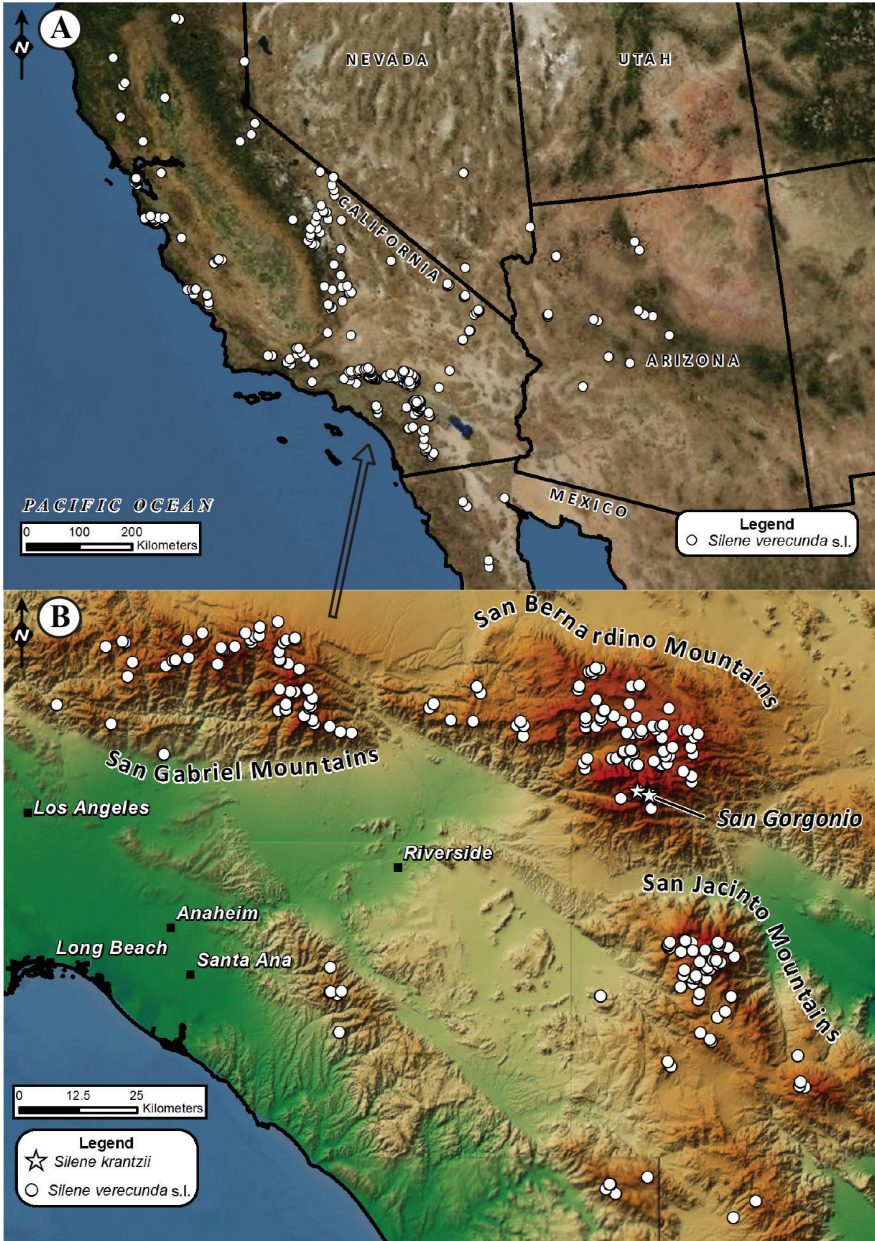


FIGURE 2.—Distribution of the *Silene verecunda* complex. A-B. Locality information downloaded from the Consortium of California Herbaria and Intermountain Regional Herbarium Network. A. The easternmost taxon in the *S. verecunda* complex (circles) is subsp. *andersonii* (Clokey) C. L. Hitchcock & Maguire, whereas the majority of the morphological variation exhibited by this species is attributed to subsp. *platyota* (S. Watson) C. L. Hitchcock & Maguire in the Sierra Nevada, Transverse Ranges and Peninsular Ranges of California (including Baja, MEX). Plants from the Coast Ranges of California represent the 'typical' variety, with the type specimen for *S. verecunda* S. Watson reported from serpentine substrates on Mt. Davidson in the San Francisco Bay Area. B. Inset showing the Transverse and Peninsular Ranges of southern California, with the location of *S. krantzii* (stars) on San Gorgonio Mountain indicated in the southern portion of the San Bernardino Mountains, San Bernardino County, California.



FIGURE 3.—*Silene krantzii*. Illustration by Diana D. Jolles.

Caespitose perennial herb, caudex branches few to many below ground, many fertile shoots at the base 2–10(14) cm long, spreading to ascending, generally unbranched, glandular-viscid throughout, hairs generally less than 0.5 mm, glands stipitate. Basal leaves linear-oblongate, (5)8–30 × 1–3 mm, apex acute, base attenuate into short petiole, both surfaces glandular-viscid, 1-veined or venation obscure, in opposite pairs of 2. Cauline leaves similar to basal leaves, but gradually reduced apically, linear-lanceolate, few, sessile, often with fertile axillary branches, opposite. Inflorescence generally an open cyme, monochasial (generally not a compound dichasium), 1–4 flowered with ascending branches. Bracts linear-lanceolate, margins membranaceous. Peduncles 0.5–2.5(3.5) cm long. Bracteoles similar to cauline leaves, but lanceolate-ovate and gradually smaller than bracts. Pedicels up to 2.5(3.5) cm long, generally longer than calyx. Calyx 10-nerved, 8–15 mm long, cylindrical-campanulate to campanulate-clavate, narrowed proximally around carpophore, reddish, densely glandular-viscid, generally somewhat inflated at flowering with pale commissures between prominent parallel veins, not clearly papery in fruit; teeth keeled, broadly ovate to triangular, *ca.* 1–3 × 1–1.5 mm, obtuse, with membranaceous and ciliate margins. Carpophore *ca.* 0.5–1(1.5) mm long, hairy. Petals pink, (7)9–17 mm long; limb broadly obovate, *ca.* 2.5–4 mm long, distinctly divided to *ca.* 2/5 total length; lobes broadly oblong, *ca.* 0.5–1.5 × 0.5 mm, margins lacinate at base; claw elliptic-oblong to obovate, slightly exserted beyond calyx, *ca.* (5)6–9(10) × 0.3–1.2 mm, ciliate near base; coronal scales 2, oblong, 0.5–1.5(2) mm long, rounded at tip or toothed, sometimes with lacinate margins. Stamens exserted from the calyx but ± equal to corolla, styles slightly more exserted; filaments usually sparsely pilose proximally. Styles 5.5–8 mm long. Capsules narrowly elliptic-obovate to clavate, (5.5)7–10 × 4–5.5 mm, included in the calyx. Seeds brownish, usually orbicular-reniform to oblong, 1–1.5 mm long, flattened, papillate-tuberculate with papillae developed into a low crest along margin.



FIGURE 4.—*Silene krantzii*. Scan of the type specimen, Stoughton *et al.* 1391 (RSA806300), collected from "the Tarn" in the San Gorgonio Wilderness, San Bernardino National Forest, San Bernardino County, California.

Silene krantzii is morphologically similar to the broad interpretation of *S. verecunda* S. Watson by Morton (2005) in some respects, but differs in its ecological setting and the presence of glandular trichomes on the basal leaves and proximal stems, smaller leaf size, a red calyx that is inflated at flowering, and a low-stature, spreading habit.

Silene krantzii is typically found on sand or gravel substrates of primarily igneous origin. Most records describe it in open, exposed areas above tree line with mixed shrubs and herbs (including *Festuca saximontana*, *Raillardella argentea*, *Hulsea vestita* subsp. *pygmaea*, *Calyptridium umbellatum*, and *Elymus elymoides*), although some records describe the habitat as rocky rather than sandy or gravelly. The species is cited as being scarce to common in different localities and habitats around the summit region of San Gorgonio Mountain, but personal observations by the first author suggest it is densely abundant at the type locality and in other areas around the summit region of San Gorgonio Mountain. That said, *S. krantzii* is thus far known only from high elevation (3000–3500 m) locations in the San Bernardino Mountains of southern California. Plants flower from as early as mid-May to as late as early September and develop mature fruits from June to October.

The specific epithet, *krantzii*, refers to the first author's botanical mentor and friend, Timothy Krantz, who has greatly enriched our knowledge of the flora of the San Bernardino Mountains (Krantz 1994). The suggested common name for the species is Krantz's catchfly.

Paratypes: USA, California, Transverse Ranges, San Bernardino Mountains District, San Gorgonio summit region, San Bernardino County: RSA793545, *Bell 3757*, 6 Jul 2012; RSA498876, *Cooper 2994*, 2 Sep 1948; RSA436977, *Dunkle 3787*, 3 Sep 1933; CAS35901, *Howell 23678*, 25 Jul 1947; POM17581, *Munz 9596*, 24 Aug 1923; RSA39583, *Munz 12081*, 25 Jul 1947; RSA65065, *Peirson 615*, 24 Apr 1923; CAS408760, *Raven 11145*, 11 Aug 1957; RSA806301, *Stoughton et al. 1407*, 7 Aug 2011; RSA806302, *Stoughton et al. 1432*, 20 Aug 2011; RSA281181, *Thorne 49871*, 20 Jul 1977; RSA438861, *Wallace k-III*, 17 Aug 1938.

TAXONOMIC RELATIONSHIPS

Silene krantzii appears to be a close relative of *Silene verecunda* S. Watson (1875), a staggeringly variable species of *Silene* that is widespread in southwest North America. *Silene verecunda* (Figure 5) occurs in California, Nevada, Utah, Arizona, and in Baja California, Mexico (Morton 2005), with unsubstantiated reports of the species from Oregon. Disregarding all subspecies in the most recent treatment, Morton (2005) suggested that separation of taxa would be arbitrary and that the species complex is in need of in-depth study. Despite this conclusion, Morton provides numerous morphological characters and disparate ecological settings for distinguishing taxa associated with the group. Morton's taxonomic concept of *Silene* is in direct contrast to later findings of Popp and Oxelman (2007), who used base pair substitutions from both nuclear and plastid markers to assess relationships among many members of *Silene* in North America. Popp and Oxelman (2007) found *Silene andersonii* Clokey [= *S. verecunda* subsp. *andersonii* (Clokey) C. L. Hitchcock & Maguire] to be resolved in a separate clade than that of two other accessions of *S. verecunda* they included in their analyses. A more recent study (Petri and Oxelman 2011) further substantiates paraphyly of *S. verecunda* as circumscribed by Morton (2005). This suggests that some of the morphological variation observed by previous authors (Watson 1875, 1882, 1888; Jepson 1914; Clokey 1939; Hitchcock and Maguire 1947) may



FIGURE 5.—*Silene verecunda* subsp. *verecunda* in native habitat near the type locality of *S. verecunda* S. Watson on Mt. Davidson, in the vicinity of San Francisco Bay, California. A. Overall plant habit demonstrating decumbent to erect nature of the stems of these considerably larger plants. *Silene verecunda* subsp. *verecunda* occupies coastal, lower elevation habitats in California, which is in stark contrast to the inland, alpine habitat of *S. krantzii*. B. Basal leaves and stems, showing oblanceolate shape with much more pronounced petioles compared to *S. krantzii*. *Silene verecunda* subsp. *verecunda* does exhibit glandularity on the proximal stems and leaves like *S. krantzii*, but these trichomes are larger, less-dense, and of a different structural arrangement. C. Flower displayed in side-view, showing somewhat inflated condition of the mature calyx in flowering similar to *S. krantzii*. Note that calyces are considerably less red in color than *S. krantzii* and a noticeable keel on the calyx teeth is lacking. Photographs used by permission from Scott Simono, San Francisco State University, California.

have a genetic basis and that the associated ecological affinities may represent locally adapted genotypes (with corresponding morphotypes) rather than phenotypic plasticity in a widespread, generalist species.

QUALITATIVE ANALYSES AND CLASSIFICATION

Specimens of all members of the *S. verecunda* complex (including *S. krantzii*) from herbaria at RSA/POM and UC/JEPS were examined (Appendix I) as part of a larger project (T. Stoughton *et al.*, Rancho Santa Ana Botanic Garden, unpublished data) evaluating the taxonomic validity of the previously recognized subspecies of *S. verecunda* using molecular phylogenetics. A total of 130 specimens of *S. verecunda* s.l. (Morton 2005) collected from California (including Baja) were examined, including 82 specimens of *S. verecunda* subsp. *platyota* and 11 specimens of *S. krantzii* collected from the San Bernardino Mountains. A limited number of representative specimens for the other subspecies of *S. verecunda* outside of the San Bernardino Mountains are listed in Appendix I, including 9 specimens of subsp. *andersonii*, 20 specimens of subsp. *platyota*, and 8 specimens of subsp. *verecunda*.

We considered treating the new taxon as a variety or subspecies of *S. verecunda* because, like *S. verecunda* s.l., it is ciliate on the petal claw with two petal limb lobes that have lacinate margins. *Silene krantzii* overlaps morphologically with *S. verecunda* s.l. in portions of its geographic range in several other features. The branching pattern of *S. krantzii* is similar to that of *S. verecunda* subsp. *platyota*, but the primary axes of *S. krantzii* are generally more prostrate to spreading in comparison and *S. verecunda* subsp. *platyota* tends to have inflorescences that are compound dichasia. Plants of *S. krantzii* are smaller than *S. verecunda* subsp. *platyota*, which occurs at lower elevations in the San Bernardino Mountains (below 3,000 m). The range of corolla size among *S. krantzii* plants is similar to that of *S. verecunda* s.l. Seeds of *S. krantzii* appear to be smaller than that of *S. verecunda* s.l., but are similar to the latter species in being flattened, papillate-tuberculate, and orbicular-reniform to oblong in shape with papillae that are developed into a low crest along the margin. The principal distinction between *S. krantzii* and the subspecies of *S. verecunda sensu* Hitchcock and Maguire (1947) is the unique combination of: (1) glandular trichomes on the basal leaves and proximal stems; (2) leaves of reduced size; (3) a red calyx that is inflated at flowering; and (4) a low-stature, spreading habit (see Appendix II, Key to *Silene verecunda* species complex). These distinctive morphological features of *S. krantzii*, along with its isolated geographic distribution in the alpine zone of the San Bernardino Mountains having no known intergradation with *S. verecunda* s.l., warrant its species status by a taxonomic (morphologic) species concept (Cronquist 1978) that incorporates information regarding natural history of the species.

Physical isolation of *S. krantzii* populations at high elevation in the San Bernardino Mountains region and the evolution of unique morphological and ecological features suggest that this species has either a restricted niche, limited dispersal ability, or both. In general, we know very little about the phylogenetic relationships within the *S. verecunda* complex (Popp and Oxelman 2007, Petri and Oxelman 2011), so gaining a better understanding of these relationships should be the goal of future molecular studies of California *Silene* species. Although no explicit attempts have been made to characterize genetic diversity in the group, recent research (Popp and Oxelman 2007, Petri and Oxelman 2011) indicates that the *S. verecunda* complex, like the rest of the genus, is relatively slow to accumulate genetic synapomorphies at loci traditionally used for phylogenetic inference. Adopting a metapopulation view of monophyletic species circumscription (De Queiroz 2007) has recently increased in popularity because it can accommodate a greater number of natural processes, including ones causing reticulate patterns of evolution like hybridization and introgression. However, as Baum (2009) points out, monophyly is a feature of taxa that

arises after metapopulation lineages have been isolated for sufficiently long enough to accumulate fixed mutations. Given the lack of resolution in *Silene* phylogeny estimates to date (e.g., Oxelman *et al.* 1997; Popp and Oxelman 2004, 2007; Popp *et al.* 2005; Petri and Oxelman 2011), a ‘long time’ in *Silene* may be relatively much longer than in other Angiosperm lineages. It is not possible for us to evaluate Baum’s (2009) concepts of predictive power or robustness for *S. krantzii* at this time due to a paucity of genetic data, but recognition of this new species does have biological significance, utility, and precedent (Baum 2009). *Silene krantzii* is ecologically distinct from *S. verecunda*, occurs sympatrically only with another phylogenetically distant *Silene* species (*S. parishii* S. Watson), and can be distinguished phenotypically from putative close relatives. Additionally, we argue that there is taxonomic precedence for species recognition of *S. krantzii* despite the fact that putative close relatives have been previously treated both as separate species (Watson 1875, 1882, 1888; Jepson 1914; Clokey 1939) and subspecies (Jepson 1914, Hitchcock and Maguire 1947) of *S. verecunda*. All of the subspecies of *S. verecunda* treated by Hitchcock and Maguire (1947) were described as unique species in previous works (Watson 1875, 1882; Clokey 1939). Preliminary genetic evidence from nuclear and chloroplast loci suggest that numerous monophyletic taxa exist in the *S. verecunda* complex (T. Stoughton *et al.*, Rancho Santa Ana Botanic Garden, unpublished data), including but not limited to all of the subspecies previously recognized by Hitchcock and Maguire (1947) included in our key (Appendix II). Additionally, botanists who are familiar with the *S. verecunda* complex have confirmed that this ‘species’ has served as a taxonomic dumping ground for numerous entities possessing some, but not all, of the diagnostic morphology of *S. verecunda* S. Watson (D. Taylor, Jepson Herbarium, personal communication). These taxa may be the products of recent, rapid radiation following hybridization and allopolyploidization (Petri and Oxelman 2011), but the signature is phenotypic diversification with minimal corresponding genetic change. Lest this phenotypic polymorphism be mistaken for plasticity, it should be noted that *S. krantzii* is supported by genetic synapomorphies, albeit extremely few at present, and a pilot common garden study in which phenotypic differences among *S. krantzii* and *S. verecunda* subsp. *platyota* from the San Bernardino Mountains were retained in flowering individuals (T. Stoughton, Rancho Santa Ana Botanic Garden, unpublished data).

Does the circumscription of *S. krantzii* represent one more slice in the *ad nauseum* splitting process of micro-taxonomists? Perhaps! Before elucidating the reasons why we think it is not only beneficial but our responsibility to recognize *S. krantzii* as distinct from *S. verecunda* s.l., we briefly discuss the negative aspects of so-called ‘taxonomic splitting’. Philosophically, it is undesirable to reduce species phylogenetically to their elements (i.e., genotypes). Species are meant to represent groups of individuals that share synapomorphies (shared, derived characters) rather than subgroups sharing particular alleles (i.e., genotypes). Recognizing genotypes as species predictably leads to species breakdown as a result of sexual reproduction and gene flow, and underestimation of intraspecific genetic diversity. From a conservation standpoint, taxonomic splitting may lead to increased, positive efforts to conserve particular species assemblages based on species richness. However, overestimates of richness due to taxonomic splitting may result in the conservation of certain genotypes rather than the species containing them (Zachos 2013). Conservation efforts based on inaccurate species circumscription resulting from overzealous taxonomic splitting could create a genetic bottleneck, which would then artificially put the species at very high risk. That said, a survey by Morrison *et al.* (2009) indicated that taxonomic splitting, relative to ‘lumping’ or making no taxonomic change, has produced only positive effects, increasing

awareness of particular groups and increasing their protection as well.

Disintegration of *S. verecunda* s.l. seems eminent and arguably represents taxonomic splitting. However, as our ability to understand phylogenetic relationships among taxonomic entities increases, our ability to understand morphological evolution and geographic distributions increases in turn. We think that it is our responsibility as taxonomists not only to circumscribe *S. krantzii* as distinct based on the evidence laid out in this study, but to learn everything we can about its natural history, testing its taxonomic status with new data when it is available. To this end, we hope that recognition of the narrowly endemic *S. krantzii* at the species level will also catalyze a re-evaluation of conservation needs for the entire *S. verecunda* complex. Although synonymized by Hartman et al. (2012) in the current treatment of *The Jepson Manual*, the infraspecific taxon *Silene verecunda* subsp. *verecunda* is still recognized as a rare plant (Rank 1B.2 — plants rare, threatened, or endangered in California and elsewhere; moderately threatened in California) by the California Native Plant Society (CNPS 2013).

BIOGEOGRAPHY

The San Bernardino Mountains make up a floristically and geologically diverse region of significant botanical importance (Krantz 1994). San Gorgonio Mountain was the southernmost glaciated peak in western North America (Sharp et al. 1959, Owen et al. 2003) and consequently represents the southernmost limit of several widespread alpine plant taxa (Krantz 1994). A vast majority of high elevation (i.e., >3,000 m) areas in the San Bernardino Mountains are designated by the San Bernardino National Forest as the San Gorgonio Wilderness, the only place that *Silene krantzii* is currently known to occur. Krantz (1994) reviewed the vascular plant flora of the San Bernardino Mountains, citing nearly 1,600 taxa native to this region. Nineteen flowering plant species that grow in the alpine zone of San Gorgonio Mountain (ca. 40% of the total number of species surveyed by Krantz) are not found elsewhere in the San Bernardino Mountains but occur more widely to the north (Krantz 1994). With this study we add one additional species, increasing our knowledge of the endemic flora (ca. 8%) of the very rich and interesting summit region of San Gorgonio Mountain.

The geographic range of *S. krantzii* is intriguing given the distribution of other taxa in the *S. verecunda* complex (Figure 2). Other subspecies of *Silene verecunda* s.l. in the southwest U.S. include *S. verecunda* subsp. *platyota*, a common understory component in pine forests at lower elevations (<3,000 m) in the San Bernardino Mountains, and *S. verecunda* subsp. *andersonii*, a more xeric-adapted taxon that grows in the desert mountains, primarily to the northeast and not overlapping in distribution with *S. krantzii*. As mentioned previously, preliminary molecular assays suggest that *S. krantzii* is distantly related to *S. verecunda* subsp. *andersonii* and that other subspecies in the *S. verecunda* complex appear to be unique evolutionary lineages (Popp and Oxelman 2007; Petri and Oxelman 2011; T. Stoughton et al., Rancho Santa Ana Botanic Garden, unpublished data). *Silene krantzii* has been found only around the summit region of San Gorgonio Mountain thus far, but we recommend looking for this species on adjacent high peaks in southern California, particularly in the White Mountains of Inyo County, and the nearby San Gabriel Mountains (San Bernardino and Los Angeles counties) to the west. In the higher elevations of the eastern San Gabriel Mountains, Philip Munz collected individuals from “little baldy” that resemble *S. krantzii* (Munz 6119, POM13373 and UC218196) but differ in that they do not have leaves of reduced

size or a low-statured, spreading habit. The Munz collection also has inflorescences that are strict dichasia, more similar to those typical of *S. verecunda* subsp. *platyota*. This and other collections from the higher elevation habitats in the San Gabriel Mountains did not escape the attention of Hitchcock and Maguire (1947) during preparation of their monograph on North American *Silene*, but ultimately they did not offer a name.

Silene krantzii may represent just one example of many plant lineages in southern California that have become widely disjunct from close congeners in the southern Rocky Mountains during the Pleistocene, as explained by the hypothesis of Major and Bamberg (1967) and others (e.g., Morefield 1992, Krantz 1994). The hypothesis of Cordilleran disjunction, sensu Major and Bamberg (1967), has been rejected by some authors (e.g., Chabot and Billings 1972, Raven and Axelrod 1978) who have instead favored more direct north to south dispersal along the Sierran-Cascadian axis. Although *S. krantzii* is currently known only from the San Bernardino Mountains, the preponderance of examples of other species on San Gorgonio Mountain would lend support to either of the two competing hypotheses. Many widespread species that inhabit the alpine zone on San Gorgonio Mountain occur also in the alpine habitats of the eastern Sierra Nevada and White Mountains of California (Morefield 1992, Krantz 1994). Discontinuous populations of these plants are spread across the Basin and Range Province linking populations in the western mountain ranges to populations in the southern Rocky Mountains (Major and Bamberg 1967, Morefield 1992, Krantz 1994). Therefore, locating additional populations of *S. krantzii* and an understanding of the phylogenetic relationship of this new species will be essential for developing hypotheses regarding its origin and evolution over time.

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**APPENDIX I: LIST OF VOUCHER SPECIMENS OF *SILENE KRANTZII* AND *S. VERECUNDA*
EXAMINED FOR THIS STUDY**

USA, California, Transverse Ranges, San Bernardino Mountains District, San Bernardino County: ***S. krantzii* specimens.**—RSA793545, *Bell 3757*, 6 Jul 2012; RSA498876, *Cooper 2994*, 2 Sep 1948; RSA436977, *Dunkle 3787*, 3 Sep 1933; POM17581, *Munz 9596*, 24 Aug 1923; RSA39583, *Munz 12081*, 25 Jul 1947; RSA65065, *Peirson 615*, 24 Apr 1923; RSA806300, *Stoughton 1391*, 23 Jul 2011; RSA806301, *Stoughton et al. 1407*, 7 Aug 2011; RSA806302, *Stoughton et al. 1432*, 20 Aug 2011; RSA281181, *Thorne 49871*, 20 Jul 1977; RSA438861, *Wallace k-111*, 17 Aug 1938. ***S. verecunda* subsp. *platyota* specimens.**—POM283, *Abrams 2039*, 30 Jul 1901; POM158085, *Abrams 2911*, 5 Aug 1902; RSA743190, *Adams s.n.*, 18 Aug 2001; RSA752761, *Bell 457*, 26 Sep 2009; RSA767348, *Bell 1349*, 21 Jun 2010; RSA767029, *Bell 1370*, 23 Jun 2010; RSA767021, *Bell 1445*, 29 Jun 2010; RSA766995, *Bell 1547*, 7 Jul 2010; RSA769000, *Bell 1703*, 28 Jul 2010; RSA779190, *Bell 2687*, 15 Jun 2011; RSA779746, *Bell 2747*, 5 Jul 2011; RSA781783, *Bell 2857*, 3 Aug 2011; RSA615913, *Boyd 10272*, 10 Aug 1998; RSA477824, *Cooper 2910*, 24 Aug 1948; RSA438874, *Davidson 2145*, Jul 1905; RSA438873, *Davidson 2322*, Jul 1905; RSA438860, *Davidson 3094*, 16 Aug 1975; RSA438871, *Davidson 3147*, 16 Aug 1975; RSA438872, *Davidson 4599*, 14 Aug 1976; RSA753631, *De Groot 6057*, 8 Jun 2009; RSA438878, *Detmers s.n.*, 15 Jun 1929; RSA438875, *Fosberg 55871*, 3 Aug 1931; RSA753792, *Fraga 2870*, 3 Jun 2009; RSA753874, *Fraga 2878*, 4 Jun 2009; RSA753399, *Fraga 2997*, 16 Jun 2009; RSA780520, *Fraga 3544*, 12 Jul 2010; RSA616039, *Gross 40*, 2 Aug 1998; RSA749476, *Gross 4077*, 15 Jun 2009; POM11670, *Harwood 4316*, 4 Jul 1920; POM9373, *Harwood 4327*, 4 Jul 1930; RSA761653, *Honer 3537*, 28 Jun 2010; RSA760901, *Honer 3604*, 16 Jul 2010; RSA41849, *Howe s.n.*, 24 Jun 1947; RSA287, *Howell 349*, 10 Jul 1927; POM123149, UC311416, *Jones 6183*, 19 Jul 1900; RSA741121, *Mistretta 3414*, 17 Jul 2008; RSA740480, *Mistretta 3448*, 18 Jul 2008; RSA752705, *Mistretta 3823*, 9 Jun 2009; RSA753115, *Mistretta 3917*, 22 Jun 2009; RSA767691, *Mistretta 5377*, 12 Jul 2010; RSA767424, *Mistretta 5612*, 9 Aug 2010; RSA768252, *Mistretta 5623*, 11 Aug 2010; RSA768946, *Mistretta 5668*, 27 Aug 2010; POM12832, *Munz 6246*, 25 Aug 1922; POM48822, *Munz 8673*, 17 Jul 1924; POM96546, UC310485, *Munz 10544*, 29 Jun 1926; POM96621, *Munz 10710*, 14 Jun 1926; RSA38752, *Munz 11992*, 22 Jul 1947; POM184179, *Munz 12787*, 9 Jun 1932; RSA102380, *Parish 930*, May 1881; RSA102382, UC136296, *Parish 931*, May 1881; RSA102381, *Parish 3064*, 24 June 1894; UC8498, *Parish 3728*, 25 Jun 1895; RSA65062, *Peirson 3270*, 25 Aug 1922; RSA65050, *Peirson s.n.*, 24 Jun 1941; POM306370, *Ramsey 1463*, 2 Jul 1939; RSA692071, *Sanders 14940*, 26 Jun 1994; RSA438879, *Spalding s.n.*, 6 Aug 1931; RSA438896, *Templeton 76H371B*, 18 Jul 1932; RSA369513, *Thorne 47341*, 11 Aug 1975; RSA370810, UC1536094, *Thorne 47704*, 15 Jul 1976; RSA371147, *Thorne 47729*, 15 Jul 1976; RSA337855, *Thorne 53246*, 15 Jun 1979; RSA302740, *Thorne 53358*, 3 Jul 1979; RSA611493, *Wheeler 1097*, 24 Jul 1932; RSA554044, *White 91-199*, 20 Jul 1991; RSA554483, *White 715*, 29 Aug 1992; RSA632097, *White 6924*, 27 Jul 1998; RSA627588, *White 7072*, 13 Aug 1998; RSA674800, *White 8623*, 24 Jun 2001; RSA674819, *White 8689*, 2 Jul 2001; RSA682583, *White 9656*, 29 Jul 2003; RSA715743, *White 11530*, 20 Jun 2006; RSA752960, *White 13322*, 24 Jul 2009; RSA752333, *Wood 942*, 18 Jun 2009; RSA752331, *Wood 947*, 18 Jun 2009; RSA764869, *Wood 2105*, 1 Jul 2010; RSA766694, *Wood 2127*, 2 Jul 2010; UC1137880, *Yates 6616*, 24 Jun 1937.

USA, California, Desert Mountains District (including White and Inyo Mountains), Inyo and San Bernardino Counties: ***S. verecunda* subsp. *andersonii* specimens.**—RSA296490, *Castagnoli 198*, 6 Jun 1980; RSA624136, *DeDecker 3310*, 27 Jul 1973; RSA270672, *Henrickson 13986*, 13 May 1974; UC1549855, *Morefield 4148*, 18 Jul 1986; RSA486336, *Morefield 4804*, 19 Jul 1988; RSA352210, *Peterson 601*, 15 Jun 1982; RSA552077, *Romsper 425*, 11 Aug 1977; RSA290816, *Thorne 44874*, 10 Jul 1974; RSA334384, *Thorne 54802*, 21 Sep 1980.

USA and MEX, California (including Baja), Sierra Nevada, Transverse Ranges, and Peninsular Ranges Districts, Fresno, Kern, Los Angeles, Orange, Riverside, San Bernardino, and Tulare Counties: ***S. verecunda* subsp. *platyota* specimens.**—RSA507942, *Boyd 2260*, 28 May 1988; RSA507251, *Boyd 2373*, 4 Jun 1988; RSA519440, *Boyd 2762*, 20 Jul 1988; RSA438862, *Davidson 2978*, 3 Jul 1975; UC1618234, *Ertter 6311*, 9 Jun 1986; RSA727590, *Fraga 735*, 19 May 2003; RSA727455, *Fraga 1265*, 18 Jun 2004; RSA680009, *Gross 1020*, 10 Jun 2003; RSA795882, *Gross 5573*, 11 Jul 2012; RSA225227, *Moran 14479*, 21 Aug 1967; RSA659142, *Moran 28862*, 21 Jun 1980; RSA679167, *Roberts 5936*, 25 Jun 2003; RSA546449, *Ross 3063*, 2 Jul 1990; RSA637501, *Soza 514*, 10 Jun 1999; RSA682367, *Soza 1701*, 25 Jun 2003; RSA599719, *Swinney 3942*, 29 Jun 1995; RSA596614, *Swinney 3974*, 6 Jul 1995; RSA733088, *Swinney 7994*, 20 Jun 2001; RSA774867, *Swinney 11253*, 24 Jun 2009; RSA673473, *White 7712*, 20 Apr 2000.

USA, California, Coast Ranges and San Francisco Bay Area Districts, Monterey, San Benito, San Francisco, San Louis Obispo, and Santa Cruz Counties: ***S. verecunda* subsp. *verecunda* specimens.**—JEPS102187, *Douglas HL369b*, 20 May 1994; UC1583623, *Keil 20704*, 11 Jun 1988; RSA330449, *Norris 4829*, 14 Mar 1985; JEPS81534, *Stone 462*, 13 May 1982; RSA502348, *Taylor 9617*, 29 Apr 1988; JEPS90790, *Taylor 11942*, 30 Jun 1991; JEPS82771, *West 43*, 11 Apr 1983; JEPS82967, *West 80*, 11 May 1983.

APPENDIX II: KEY TO THE *S. VERECUNDA* SPECIES COMPLEX

1. Flowering calyx somewhat inflated (slightly at times), generally reddish in color, lobes (teeth) keeled or not; basal leaves generally (5)8-60 mm long
 2. Alpine (high elevation) habitats, trichomes glandular on proximal-most leaves; flowering calyx lobes generally strongly keeled; inflorescence a monochasium, branches spreading to ascending *S. krantzii*
 - 2' Coastal (low elevation) habitats, at least some non-glandular trichomes on petioles of proximal-most leaves; flowering calyx teeth generally not keeled; inflorescence a compound dichasium (occasionally simple, sometimes monochasial below), branches ascending to erect *S. verecunda* subsp. *verecunda*
 - 1' Flowering calyx tubular, generally greenish in color (rarely reddish), lobes (teeth) not keeled; basal leaves generally 60-100 mm long (sometimes withering early)
 3. Proximal-most leaves sparsely minute-glandular with noticeably thickened midribs and marcescent bases, generally withering early; transmontane pine and oak woodlands; inflorescence a monochasium (rarely dichasial)
..... *S. verecunda* subsp. *andersonii*
 - 3' Proximal-most leaves generally non-glandular (if glandular, then not sparsely so) and lacking noticeably thickened midribs or marcescent bases, not withering early; generally cismontane pine and oak woodlands; inflorescence a dichasium (sometimes monochasial below) *S. verecunda* subsp. *platyota*

Doing adaptive management: improving the application of science to the restoration of a rare Lake Tahoe plant

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Adaptive management is probably the best available structure for linking science with decision-making when conserving biological resources. We have found that implementation of adaptive management requires: (1) modification of the conceptual model to include benefits to biological resources *in situ*; (2) upfront participation of all stakeholders in the conservation strategy and design of the adaptive management program with clear structuring of information flow and the sequence of project stages to facilitate stakeholder responses within a reasonable timeframe; and (3) use of key management questions to focus data collection and identify beneficial management actions. These guidelines are illustrated using our experience with Tahoe yellow cress (*Rorippa subumbellata* Rollins, Brassicaceae), a plant endemic to the shores of Lake Tahoe in California and Nevada and a candidate for protection under the Endangered Species Act. The project provides an operative example of science-driven decision-making that has been ongoing for over ten years. Several corollary ingredients are identified that have improved the chances of project success and helped to sustain the long-term effort.

Key words: adaptive management, Lake Tahoe, monitoring, plant conservation, *Rorippa subumbellata*, species restoration, Tahoe yellow cress,

“The gap between theory and practice remains surprisingly wide in conservation biology.”

Nature

8 November 2007

One of the greatest challenges in biological resource management is the development of an institutional structure that allows available scientific information to contribute to resource management and land-use decision-making. The structure must be strong enough to counteract political and economic influences that frequently lead to resource degradation. It must provide a transparent mechanism that directly links empirical information with alternative outcomes, and to implementation of the outcome that appears most beneficial to the target resource. And, most importantly, uncertainties and inevitable setbacks must be countermanded by long-term stakeholder commitment, attentive oversight, and stable financial support. Only then can a synergy between conservation theory and practice be realized, leading to significant improvement in the condition of species or ecosystems.

Adaptive management is probably the best available structure for linking science and decision-making and most government agencies have to some extent adopted its basic conceptual framework and rhetoric (Holling 1978, Walters 1986, Stankey et al. 2005, Gregory et al. 2006). It has been applied to efforts to conserve a wide variety of biological resources, including butterflies (Wisconsin Department of Natural Resources 1999, Lentz and Nogami 2007), fish (Walters 1992, Kareiva et al. 2000, Cinner et al. 2009), birds (Schmiegelow and Hannon 1993, Williams and Johnson 1995, Johnson et al. 1997, Dimond and Armstrong 2007), marsupials (Rout et al. 2009), terrestrial vegetation (Taylor et al. 1997, Gray 2000, Murray and Marmorek 2003, Haynes et al. 2006, Ascoli et al. 2009) and the biota of rivers (Walters et al. 2000, Levine 2004, Lovich and Melis 2007, King et al. 2010). Yet, despite its obvious strengths and intuitive simplicity, there are few examples of successful implementation of adaptive management (Stankey 2003, Stankey et al. 2005, Gregory et al. 2006), especially if the criterion for defining success is a demonstrated improvement in the condition of species or ecosystems.

Some argue that the failings of adaptive management are institutional, resulting from a conflict between the need for legal and political certainty within government agencies and the complexity and variability inherent to biological systems (Doremus 2001, Doremus and Tarlock 2005, Gregory et al. 2006). The necessary flexibility to experiment and freely adopt any one of several possible outcomes is at odds with typical regulatory procedures and administrative law (Ruhl 2005, Haynes et al. 2006). Others argue that adaptive management is itself complex and intimidating, requiring simplification in order to build programmatic momentum and participant enthusiasm (Morghan et al. 2006). Adding more scientific expertise, along with researcher-management dialogue to ensure better, more relevant studies is also recommended (Morghan et al. 2006).

We have found that implementation of adaptive management can be successful if: (1) the conceptual model of the process is modified to include direct benefits to target resources; (2) stakeholders are included early in the development of the conservation strategy and design of the adaptive management framework and information flow and the sequence of project stages are clearly structured to facilitate stakeholder responses within a reasonable timeframe; and (3) key management questions (KMQs) are used to focus science and realize a management vision. These facets of our adaptive management approach will be illustrated with the conservation of a single species, Tahoe yellow cress (*Rorippa subumbellata* Rollins, Brassicaceae), a rare plant endemic to the sandy shoreline of Lake Tahoe, California and Nevada.

CONCEPTUAL MODELS MUST INCLUDE TARGET RESOURCE IMPROVEMENT

Resources and land management agencies have widely embraced adaptive management as their approach to improving the condition of species and ecosystems (Walters 1986, Nudds 1999, Smit 2003, Pavlik and Espeland 2005). Adaptive management recognizes inherent complexity and uncertainty by using “learning by doing” as its operational definition (Taylor et al. 1997, Stankey et al. 2005, Gregory et al. 2006). The process is iterative, usually portrayed as a cycle of strategy, design, implementation, monitoring, evaluation, and adjusting management (Figure 1; based on Sit and Taylor 1998). Decisions or actions are evaluated using carefully designed monitoring, and modifications to management actions are in turn tested with updated monitoring protocols. With each turn of the cycle, active learning through monitoring and evaluation reduces management uncertainties by developing tools that prove beneficial to the resource. Adaptive management is logical, can deal with uncertainty and data gaps, and is similar to the scientific process of hypothesis testing (Haynes et al. 2006).

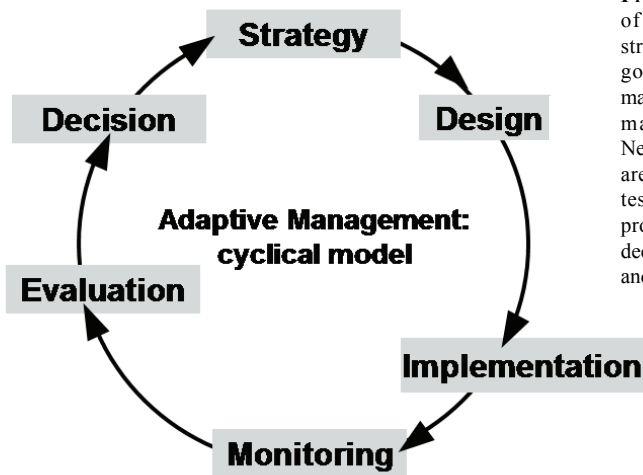


FIGURE 1.—The cyclical model of adaptive management. The strategy includes the assembly of goals, objectives, tools and key management questions by an adaptive management working group. Necessary research and monitoring are designed and implemented to test a novel management action, providing data for evaluation and decision-making. Modified from Sit and Taylor (1998).

A weakness in the classical model of adaptive management is that benefits to the target resources are not always readily apparent. Although learning and communication are key outputs of the process (Stankey et al. 2005), there must be a strong connection to decision-making that leads to resource improvement. File cabinets across the country are filled with monitoring data that have never been used to make a critical decision, much less make a difference in the condition of a targeted resource. Ultimate success is not found in the turning of the cycle — that is, the endless accumulation of data or continual amendments to monitoring design (Walters 1997). Strategic elements, such as developing objectives or key management questions, should not necessarily be constantly revised as the cycle implies. Instead, monitoring should be explicitly linked to tests of specific management actions (Macnab 1983, Pavlik 1996, Morghan et al. 2006). Success can only be found in using monitoring data to improve stakeholder understanding and management effectiveness to improve the condition of a target resource.

We propose a small but significant modification of the *de rigueur* conceptual model of adaptive management. We envision adaptive management not as a circle, but as a helix composed of cycles linked by prudent, “best-available” actions sustained over time (Figure 2). The incline of the helix is determined by resource response, which is the real measure of worth for any conservation action (Palmer et al. 2005). The response metric is any measurable attribute deemed critical to the quality of the target resource. It is best defined by the strategy of goals and objectives developed by stakeholder participants on an Adaptive Management Working Group (AMWG). The time scale of the metric depends on the targeted resource; twists of the adaptive management helix could take years for single species and decades for ecosystems (see Haynes et al. 2006, Lovich and Melis 2007).

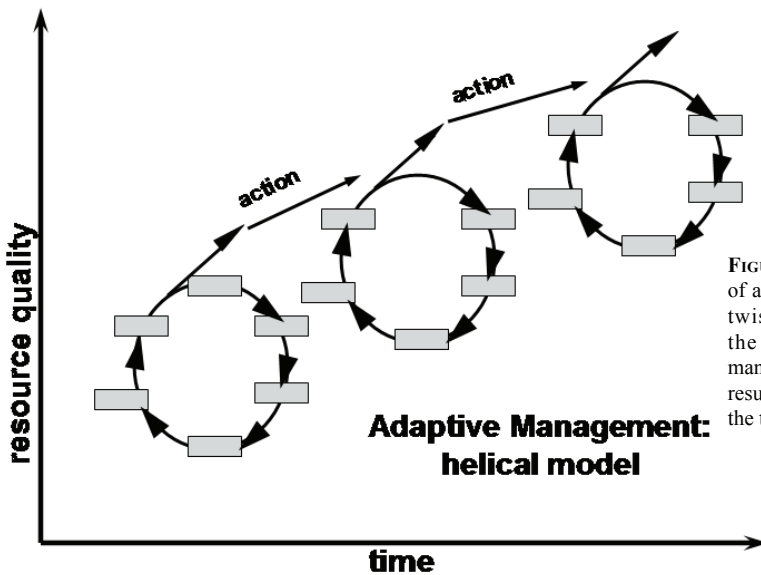


FIGURE 2.—The helical model of adaptive management. Each twist of the helix results in the application of a proven management action, in this case resulting in improved quality of the target resource.

It is important to note that not all resource management requires a rigorous, science-driven adaptive management framework (Lee 1993, Gregory et al. 2006). Where there is minimal uncertainty as to the outcome of an action, and the overall effect on resource quality of existing tools is well known, common practices management can be applied with a high probability of success. Circumstances with greater uncertainty, but combined with reliable, previously developed tools, justify use of a less intense and presumably less costly form of adaptive management, referred to here as adaptive management with best available technology. In the past, this type was unfortunately labeled “passive” (Walters and Holling 1990) even though it requires a formal, structured approach, including strategic planning, design, monitoring, etc., along with stakeholder participation on an AMWG. When using best available technology, the focus is on implementation, and monitoring data are used to confirm that actions are producing the desired trajectory for the resource. But, when there is little or no available technology with known effects and therefore a high degree of uncertainty regarding the outcome of an action, a fully developed program of adaptive management with hypothesis testing should be employed. This is the most costly form of

management (“active”), in which each action is treated as a test of an individual management-oriented hypothesis (Walters and Holling 1990, Pavlik 1996, Gregory et al. 2006). Adaptive management with hypothesis testing requires an experimental design with randomization, replication, and adequate statistical power to develop reliable tools that can be applied to the target resource. Correctly assessing the degree of uncertainty and choosing the appropriate form of management for a target resource can greatly simplify the institutional requirements and greatly lower the costs associated with a given project.

STAKEHOLDERS PARTICIPATE IN THE DEVELOPMENT OF THE CONSERVATION STRATEGY AND DESIGN OF THE ADAPTIVE MANAGEMENT FRAMEWORK

Initiating adaptive management requires a structured, cooperative approach to developing a conservation strategy. The strategy sets the vision for the species or ecosystem and articulates the goals and objectives for the target resource within a defined, realistic timeline. The objectives for each resource element need to be measurable so that they can be used as yardsticks for measuring success (Walters 1986; Pavlik 1994, 1996; Elzinga et al. 2001). The conservation strategy is not simply a laundry list of recommended studies or knowledge gaps that should be filled before any action is taken or decision is made — a major failing of most recovery plans (Schemske et al. 1994).

The vision as conveyed in the goals and objectives of the conservation strategy is best achieved through the consensus of affected stakeholders. Typically, a panel of land managers, government regulators, and scientists forms an AMWG, but representatives of private landowners, affected industries and the public at large should also be encouraged to bring their concerns or objections to the table. In exchange for access, AMWG members must be cooperative and committed to the conservation of the target resource; each stakeholder brings a distinct perspective to the process, but all must focus on improving resource quality by cooperating in an open, non-adversarial forum. Utilizing a structured, cooperative approach to developing the conservation strategy allows the AMWG to address all members’ concerns or objections directly and build trust through straightforward communication (Fule 2003, Stankey et al. 2005). Without broad stakeholder support, opposition or apathy can halt both the development and implementation of the conservation strategy.

Once the conservation strategy is agreed upon and finalized, the integrity and effectiveness of the decision-making process in an adaptive management framework depends on a structured and timely flow of data so that stakeholders will be able to anticipate and respond to their own, institutional constraints (e.g., permits, public notice, funding, hiring) with minimal difficulty. Within this framework, the decision-making entities have clearly defined positions in the flow of information (Figure 3).

The AMWG is the workhorse of the process as it provides the direct communication conduit for all affected agencies, local governments, and private entities. It is through the AMWG that adaptive management becomes a community learning process, imbedded within a regulatory and bureaucratic environment with its logistical, economic, and political constraints (Haynes et al. 2006). It is a major responsibility of the AMWG to address these constraints as it prioritizes research and monitoring tasks and carries out the duties of budgeting and long-term planning. The AMWG may solicit outside scientific review and public comment and brings forth funding needs to an executive committee, comprised of agency decision makers, and executive directors. It is the role of the executive officers to

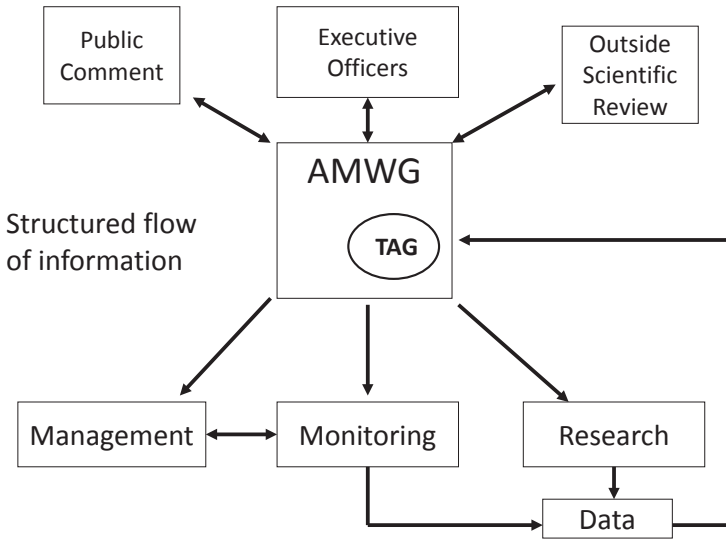


FIGURE 3.—The structured flow of information between participants and activities of the adaptive management program for Tahoe yellow cross. Members of the Technical Advisory Group also sit on the Adaptive Management Working Group, but convene separately to solve technical problems and oversee management actions, monitoring, and research. Modified from Pavlik et al. (2002).

identify and generate funding opportunities and integrate the resource-specific focus of the particular conservation strategy into other local or regional planning.

The more technical aspects of implementing research are best addressed by a subset of members that form a Technical Advisory Group (TAG). The TAG is given direction by the AMWG, but they are insulated from the politics of the AMWG because they are only charged with providing a robust mechanism for evaluating management actions and adding to the knowledge base. If there are 12 members of the AMWG, then two to four with research experience serve on the TAG. The TAG translates management objectives into monitoring objectives with precise definitions of what will be measured, and with what degree of statistical certainty (Elzinga et al. 2001). The TAG then selects study sites, determines experimental design, and analyzes the data in order to make technical recommendations that inform the decision-making process within the AMWG.

KEY MANAGEMENT QUESTIONS FOCUS SCIENCE TO REALIZE A MANAGEMENT VISION

The function of Key Management Questions (KMQs) is to focus science on the specific management issues and data gaps that, once resolved, will assist in realizing the goals and objectives set forth in the management vision (Figure 4). A well-constructed KMQ narrows an otherwise broad base of scientific inquiry (represented by the lower triangle in Figure 4) to a more finely resolved endeavor directly pertinent to future management. Similarly, the broad base of management vision (the upper triangle in Figure 4) is narrowed to another fine point by the same KMQ. Thus, a good KMQ directly links the management vision to the science and all research is then designed to inform the specific goals and objectives of the conservation strategy. In this way, a monitoring program is directly linked with an objective, and there is no *post hoc* as to the utility and application of monitoring data that are generated (Lee 1993, Pavlik 1996, Gregory et al. 2006).

The ultimate test of a good KMQ, however, is that its answer provides concrete guidance to the AMWG. In evaluating a particular question, each stakeholder should be

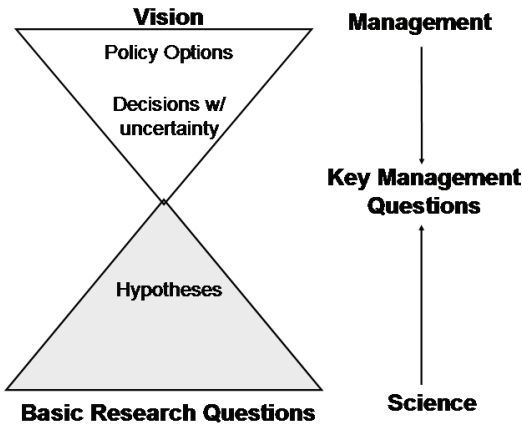


FIGURE 4.—Key management questions narrow the focus of basic research by limiting hypotheses to those that test a resource management action. Such actions, when implemented, narrow the alternative policies that affect a management vision for the target resource. Modified from Pavlik and Espeland (2005).

able to work backwards from a major decision (e.g., “Do we need to protect every patch of occupied habitat currently supporting a population?”) and deduce the KMQ (e.g., “Are all suitable habitat patches occupied by an existing population?”). From there it is not difficult to envision the basic design of an appropriate study that provides the answer (e.g., through an experimental reintroduction) and to reject others that do not link into the conservation strategy. Key management questions also have the effect of focusing agency effort and leadership. The process of developing good KMQs helps agency leadership and staff to understand and support needed research as a critical part of conservation and shrinks the domain of possible issues and concerns to a manageable number.

PRACTICING ADAPTIVE MANAGEMENT ON TAHOE YELLOW CRESS

Although simplification of the adaptive management process has been urged (Morghan et al. 2006), the lack of concrete, operating examples often is a significant impediment to its success (Doremus 2001). The conservation of Tahoe yellow cress (*Rorippa subumbellata* Rollins, Brassicaceae) provides an ongoing, operative application of science-driven decision-making to the conservation and restoration of an imperiled biological resource.

Tahoe yellow cress (TYC) is a rare plant endemic to the sandy shoreline of Lake Tahoe in California and Nevada. This low-growing perennial mustard has small yellow flowers, fleshy leaves, and exhibits vigorous clonal growth by spreading rootstocks. Since first described in 1941, TYC has been collected or observed at over 60 locations around Lake Tahoe. The total number of TYC occurrences and the locations of those occupied sites have fluctuated through time, largely in response to the level of Lake Tahoe. Lake level is regulated through the operation of the Truckee River dam, which adds an additional six feet storage capacity above the natural rim of Lake Tahoe. Lower lake levels expose a greater amount of sandy habitat and TYC has been documented at as many as 48 locations in one survey period. When the lake is near its legal capacity, as few as nine sites have been occupied.

In response to ongoing threats from recreation, development, and lake-level management, the species was listed as endangered by the State of California in 1982 and

state-listed as critically endangered in Nevada the following year. In 1986, the U.S. Fish and Wildlife Service identified Tahoe yellow cress as a candidate species for protection under the Endangered Species Act. Near extinction of the species between 1995 and 1999 prompted an upgrade in the priority status for federal listing in 1999.

The threat of listing TYC under the federal Endangered Species Act brought together a myriad of interests among personnel representing issues related to lake-level management, habitat preservation, recreational development, and private property rights. Federal protection would immediately affect about 70% of the shoreline around the Lake, inhibiting dam operations and reducing recreational access for millions of beach visitors a year. Virtually every pier renovation, storm drain replacement, and erosion control project that required a federal permit would become much more complicated and costly. In 2000, the affected stakeholders formed an AMWG to develop and implement a conservation strategy for the species.

The overarching goal of the AMWG was to produce a voluntary conservation strategy that would preclude listing of TYC under the federal Endangered Species Act and eventually provide grounds for down-listing under state laws in California and Nevada. The development of the Conservation Strategy for Tahoe yellow cress (Conservation Strategy) (Pavlik et al. 2002) was the first stage of the Tahoe yellow cress AM program (Figure 5). One year was required to synthesize 22-years of survey data collected by various agency personnel and to clarify stressors to the species and to the Lake Tahoe nearshore ecosystem.

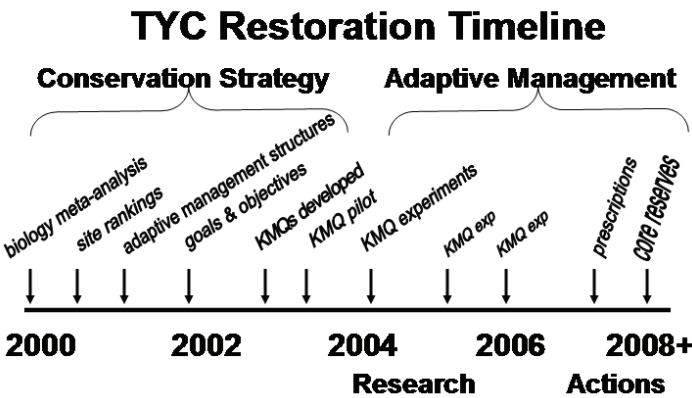


FIGURE 5.—Timeline of major events for restoring Tahoe yellow cress, including the Conservation Strategy and adaptive management.

As a result of this biological meta-analysis, the Conservation Strategy proposed a mainland-island metapopulation model for TYC. This model of metapopulation dynamics refers to spatio-temporal changes in distribution and abundance where “mainland” subpopulations persist over long periods of time while other “island” subpopulations come and go through the processes of local colonization and extirpation. Thus, the species can persist in sandy beach habitat around Lake Tahoe despite periodic high water levels and human-related impacts (Pavlik et al. 2002). Consequently, restoration and maintenance of the metapopulation dynamic became the major focus for devising KMQs and testing management actions

Existing populations were then prioritized for conservation with a quantitative ranking system based on the abundance, persistence, and variability of each subpopulation. The site rankings formed the foundation for the specific measurable objectives of the Conservation Strategy. Next, the AMWG came to an agreement about an initial adaptive management framework for structuring information flow and making management decisions (see Figure 3). An “Imminent Extinction Contingency Plan” was devised defining the types and degree of actions to be taken when the number of populations or the sizes of populations become critically low. This kind of pre-planning for future action is necessary because: (1) there may be insufficient time between the identification of an imperiled population and need to take action; (2) the description of possible actions to be taken to save the species will be known to all stakeholders in advance; and (3) the level of effort and resource commitment is acknowledged by all agencies and stakeholders. Addition of an “Imminent Extinction Contingency Plan” to a conservation strategy strengthens the often weak link, where monitoring and research fail to lead to any change in management.

Another year was spent reviewing, discussing and revising the draft strategy by all potential stakeholders, including the general public. After external review, the Conservation Strategy (Pavlik et al. 2002) was formally adopted by federal, state, and local governments with lakeshore management responsibilities, as well as the primary lakefront homeowner’s association.

While the development of the Conservation Strategy did not result in an immediate direct benefit to TYC populations, the process of identifying and ranking external stressors that degraded the population and the Lake Tahoe system made it apparent that the best available technology to mount a restoration effort was quite limited, and adaptive management with hypothesis testing and a KMQ framework would be required.

The Conservation Strategy for TYC identified two main stressors: (1) artificially high lake levels imposed by dam operations, and (2) trampling from recreational beach use. A strong, negative correlation ($r^2=0.71$, $P<0.001$) was established between lake levels and the number of populations found around the lake in a given year. This stressor is compounded by the fact that annual visitor density increases exponentially as rising waters submerge available beach habitat. While TYC response to both external stressors is simple and obvious — population distribution is restricted in wet years with high lake levels, and trampling reduces local abundance — the resulting management actions are not. Given the political realities of water and power, those actions must compensate for artificial fluctuations in the lake without requiring changes in the operation of the dam.

To focus the research phase of the Tahoe yellow cress adaptive management program, five KMQs were derived (Pavlik and O’Leary 2002) that addressed knowledge gaps for decision-making (Table 1). KMQs were shaped first by a written survey of AMWG members, who identified more than 60 variables they believed were relevant to TYC conservation. Many of these variables and the questions they evoked were academic, lacking a direct connection to realistic management options (e.g., pollen flow, pollinator availability) or they were components of larger questions that could be subsumed and thus simplified. Having fewer, more general KMQs helped AMWG members: (1) fully envision the range of relevant research that would be done; (2) see linkages between specific research projects and specific decisions they would be facing; (3) decide which research to fund and which to reject or forestall; and (4) understand that the costs and timeframe for research would be finite.

TABLE 1.—Key management questions for focusing on science and management of Tahoe yellow cress.

- (1) Can TYC populations occupy any site around the lake margin that has sandy beach habitat?
 - (2) Are there ecosystem factors that can affect TYC performance within an occupied site or microhabitat?
 - (3) Can TYC populations be created or enlarged in order to restore the self-sustaining dynamics of the species?
 - (4) Can any TYC genotype or gene pool perform equally well at any appropriate site?
 - (5) Can TYC microhabitats or places be found or created that are less likely to be adversely disturbed despite high visitor use or intense shoreline activity?
-

The first twist of the hypothesis-driven AM helix for TYC began with a pilot study in 2003. The one-year pilot demonstrated that TYC was a “cooperative” species: easily grown in a greenhouse, amenable to *in situ* experimentation, and responsive to critical variables that could be manipulated by management actions (e.g., planting distance from the shore, recreational impacts). Replicated experiments using over 10,000 container-grown plants were subsequently designed and installed at multiple locations around Lake Tahoe from 2004 to 2010 (Pavlik and Stanton 2005, 2007; Stanton and Pavlik 2010). During this period, the level of Lake Tahoe fluctuated from the natural rim to the highest level allowed by federal regulations. Experimental reintroductions in different microhabitats and in years with different lake levels allowed us to evaluate the role of source population genetics, planting distance above the water table, and inundation in the growth and persistence of experimental populations of container-grown plants. Importantly, we learned that the clonal growth form and prolific seed production of TYC make it amenable to effective translocation within or among beaches around Lake Tahoe (Stanton and Pavlik 2010). Experimental plants in suitable habitats produced more than 1.5 million seeds and nearly 10,000 asexual plantlets. Such tangible benefit to the species prompted the U.S. Fish and Wildlife Service to downgrade the priority status for federal listing of TYC in 2005 because of “continued commitments to conservation demonstrated by regulatory and land management agencies participating in the Conservation Strategy.” This first twist of the helix took eight years.

While the first twist of the AM helix focused on implementing the research agenda developing propagation, outplanting, and restoration tools, the second twist of the AM helix represents a transition to implementing appropriate management actions at a wide diversity of sites around Lake Tahoe. Newly developed available actions are composed of complimentary, research-vetted actions formulated into two ends of a management spectrum: (1) protecting habitat quality at core “mainland” and high-priority “island” sites; and (2) enhancing the size and extent of core and high-priority populations with outplanting of container-grown TYC.

Implementation of these management actions on public lands has already led to intra-agency conflicts between resource and recreation interests, as well as regulatory issues. These need to be acknowledged and addressed in identifying new planning strategies for establishing core population reserves. On private properties, the AMWG is in the process of developing innovative community engagement strategies that increase the role of landowners in Tahoe yellow cress protection and restoration through a Stewardship Program. The

resolution of such institutional and public outreach matters may rely on the science developed during the first helical twist, but will also require new approaches that evolve out of personal communications and commitment. It is this translation of knowledge into beneficial actions that is the crux of the second helical twist. We expect the next phase to take several years, after which modifications and improvements can be made and the degree of project success (i.e., the achievement of pre-defined objectives) can be ascertained.

INGREDIENTS FOR SUCCESSFUL ADAPTIVE MANAGEMENT

Not all biological resources are equally amenable to conservation through adaptive management. Adaptive management projects that involve large, complex systems with many target species and a very large number of stakeholders pose many challenges. Although these are the exact situations that seem to warrant a highly structured approach to applying science in a decision-making framework, they may not be the best situations for teaching us how to do it. Institutions that want to apply adaptive management, that is, learn to alter their policies and procedures using focused hypothesis testing, should start by evaluating a pool of fewer, more “cooperative” targets in order to select a species or ecosystem that is amenable to actions and monitoring, and that is likely to respond (positively or negatively) over short periods of time (e.g., from one to five years). In that way, the AMWG, the TAG, and the stakeholders have an opportunity to turn the helix and become engaged in the learning process. Intractable political situations are not the best place to start, but the right amount of conflict, consequence, and stakeholder commitment are necessary elements that can make success a near-term possibility.

Our project with TYC provides an example of the helix of adaptive management that has been operating for 12 years. During that time we have identified several ingredients that have helped generate a near-term sense of success among stakeholders.

The right amount of conflict and consequence.—Conflict is often a motivating force in convincing stakeholders to participate in a conservation strategy (Haynes et al. 2006). The threat of federal listing of TYC was the original consequence that led to the development of the Conservation Strategy and formation of the AMWG, and it continues to bring stakeholders to quarterly AMWG meetings. This looming consequence has moved the adaptive management process forward and reduced the amount of conflict among stakeholders. Once the AMWG was united in this cause, the first twist of the AM helix produced a wealth of knowledge useful to managers, and it also directly benefitted the species with the release of new seeds and plantlets into appropriate habitats around Lake Tahoe. Such tangible benefit to the species prompted the Fish and Wildlife Service to downgrade the priority status of the species under the ESA, highlighting how the continued commitments to conservation demonstrated by regulatory and land management agencies participating in the Conservation Strategy can lead to positive regulatory outcomes. The second twist of the helix, involving the translation of acquired knowledge into management prescriptions and restoration actions, is establishing new reserves for core populations and enhancing each to exceed an empirically derived minimum viable population size. Such improvements to the resource, along with a systematic approach to learning and cooperation, are the principal benefits that can be achieved from adaptive management.

Cooperative species and ecosystems with strong identifiable stressors.—Many rare species tend to be idiosyncratic (Fiedler et al. 1997). Genetic aberrations (Nickrent

and Weins 1989, Korbecka et al. 2002), complex breeding systems (DeMauro 1993, 1994; Scobie and Wilcock 2009), susceptibility to microbial and insect interactions (Ledig 1996, Klironomos 2002) and other, less-than-robust life history traits, offer significant impediments to both research and restoration (Guerrant and Pavlik 1997). Small and few populations also constrain efforts to manipulate and expend this kind of biological material. Therefore, the rarest species under the most urgent circumstances might not be the best candidate for learning to do adaptive management. Although there may not be a choice as threats build, regulations are invoked, and politics drive conflicting agendas, choosing a “cooperative” species, with few internal or cryptic constraints, would vastly increase the chances of sustaining a productive and potentially successful program. And the axiom follows, that “success breeds success.”

Tahoe yellow cress is a cooperative species that responds primarily to fluctuations in the level of Lake Tahoe and to recreational pressures. It is a short-lived herbaceous perennial that produces copious seed and is capable of robust vegetative growth. Flowering and fruiting occur during the earliest stages of establishment, and self-compatibility reduces the importance of pollinator availability. It is readily propagated under greenhouse or lab conditions from seed or rootstock with ordinary potting mix, and seed viability is high and germination exceeds 80%. Compared to other plants that have been inventoried by starch gel electrophoresis, TYC has very low levels of isozyme variation (Bair 1997; Saich and Hipkins 2000; DeWoody and Hipkins 2004, 2006). We were able to confirm this lack of genetic differentiation using common garden techniques. This minimizes the need for mixing plants from different source populations during restoration, although the existence of locally unique alleles still justifies a broadly stratified approach.

Are there such things as cooperative ecosystems that would be as amenable to learning AM with hypothesis testing? This we cannot say for sure. Tidal marshes readily form wherever restoration establishes the requisite regime of inundation, sedimentation, and propagule arrival (Zedler et al. 1982, Breaux et al. 2005). Mid-elevation ponderosa pine forests respond well to low-intensity groundfires (Korb and Springer 2003, Zimmerman 2003). And dammed riverine systems can be manipulated to affect fisheries and sand bar deposition (Kareiva et al. 2000, GCDAMP 2007), but biological benefits have been mixed. Perennial grasslands, however, are very difficult to wrestle from the clutches of invasive species (Carlsen et al. 2000), and desert scrub is subject to the long-term vagaries of climatic stochasticity (Lovich and Bainbridge 1999, Bainbridge 2007). Again, the choice may be an early determinant of the prospects of successful adaptive management.

Potential for long-term funding.—Long-term funding is usually a major factor that limits effective adaptive management (Levine 2004, Haynes et al. 2006). The obvious advantage of having so many dedicated stakeholders on the AMWG is that the probability of obtaining long-term funding is increased. Of the 13 signatories to the TYC Conservation Strategy, six agencies have provided money during the first six years, and six others have provided in-kind contributions of labor and materials. In addition, we have had support from two outside sources that have been administered through local agencies. The budget for adaptive management has averaged \$72,000 per year, including the costs of the Conservation Strategy, running the AMWG and the TAG, conducting pilot studies, surveys and research projects, and production of all reports (two per year). The total amount of contracted grant money for running the AMWG and conducting research has been over \$500,000.

Continuity and communication to counteract turnover.—During long-term projects, it is inevitable that representatives serving on the AMWG will come and go. New members

join, bringing with them a set of experiences and philosophies that differ from those who began the process. They will not have the benefit of knowing exactly what decisions were made and why, nor will they have the time to read and digest the volumes of minutes, progress reports, and annual summaries that rapidly accumulate after a few years. They may start to question why certain designs or analyses were employed, as well as the collective wisdom of what colleagues have previously decided to do. A certain amount of such scrutiny and re-evaluation is absolutely necessary to ensure quality, but too much can bring forward progress to a grinding halt. Therefore, it is essential that complete turnover of AMWG personnel be avoided by designating an “anchor” agency or consulting firm that remains committed to the project for its duration. The anchor does not have to lead the AMWG, but it does have to serve as an archive, communications hub, and steady presence to insure continuity. When necessary, it must also provide workshops for new AMWG members to help them understand the backlog of decisions and information generated by the project and where the uncertainties, gaps, and conflicts now stand. Ultimately, the contributions of these new members should come from focusing on current problems that affect their stakeholder constituency.

DISCUSSION

Despite its obvious strengths and intuitive simplicity, examples of successful implementation of adaptive management are lacking, especially if the criterion for defining success is a demonstrated improvement in the condition of species or ecosystems. Our project with TYC provides an example of an adaptive management program that has been operating successfully for 12 years. From its inception in 2002, stakeholders have been united in the common cause to prevent the federal listing of the species and kept the program focused on improving the TYC population as a direct outcome of the adaptive management process. The science-driven approach to recovery, guided by a KMQ framework, has led to direct benefits to the population through experimental outplantings and subsequent seed production. This success and the continued threat of federal listing have propelled the adaptive management process forward. We have learned that having an anchoring entity for the AMWG is critically important to maintaining continuity and keeping momentum. Long-term funding made it possible to keep independent consultants as part of the AMWG to conduct research, facilitate the group, and be an anchoring entity. Tahoe yellow cress proved to be a very cooperative species that exists in a system with clearly identifiable stressors, and we recommend taking great care to select an amenable target species or system for those who want to learn to do adaptive management.

One of the key lessons learned from our project is about how to better apply science in an adaptive management program. Recently, Murphy and Weiland (2014) outlined a framework that identifies five essential points where science guides adaptive management: (1) developing conceptual models; (2) confronting management prescriptions with available data; (3) building quantitative models; (4) designing monitoring schemes; and (5) interpreting returns from monitoring. The process of developing the Conservation Strategy for TYC brought science into the adaptive management program at the beginning and provided the initial point of engagement. It brought 22 years of survey data out of the darkness of file cabinets and resulted in a biological conceptual model for recovery and a quantitative model for prioritizing sites. The annual monitoring scheme subsequently has been revised several times until the AMWG came to the conclusion that the monitoring

could not tell them anything new and a more adaptive, less intensive approach was adopted to simply meet regulatory requirements. The next turn of our adaptive management helix will focus on confronting management prescriptions with the knowledge we have gained through the intensive research program. Even with the many successes of the program, it has been difficult to get managers to change their on-the-ground operations and integrate newly developed management tools into the complex regulatory environment at Lake Tahoe. The AMWG continues to struggle with this, and the ultimate confirmation of our project success will be removal of TYC from the candidate list under the ESA.

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BOOK REVIEW

A manual of California vegetation. Second Edition.

John O. Sawyer, Todd Keeler-Wolf, and Julie M. Evens. 2009. California Native Plant Society, Sacramento, California, USA. 1,300 pages. \$82.00 paperback. ISBN 978-0-943460-49-9

Available from: California Native Plant Society, 2707 K Street, Suite 1, Sacramento, CA 95816-5113 (www.cnps.org)

In 1995 the California Native Plant Society (CNPS) published *A Manual of California Vegetation* (Sawyer and Keeler-Wolf 1995) (hereafter, the MCV 1st Edition) with the goals of developing common language and clear definitions to further the inventory and understanding of all plant communities within California and developing quantitative, defensible definitions of rare and threatened communities. After 14 years of rigorous surveys, analyses, descriptions, and mapping of vegetation, original authors John Sawyer of Humboldt State University and Todd Keeler-Wolf of the California Department of Fish and Game (California Department of Fish and Wildlife [CDFW] as of 2013) were joined by Julie Evens, California Native Plant Society Vegetation Program Director, to produce the second edition of the manual.

This updated publication contains >450 vegetation descriptions that include a brief summary of the vegetation type, its habitats, rarity ranking, the names applied to similar types in other classifications, quantitative membership rules, detailed life histories, fire characteristics, regional variation, management considerations, and a list of all associations for the type found in supporting literature. Distribution maps are included for 352 vegetation types, and >2,400 references were consulted.

The book's initial chapters provide an introduction to vegetation classification and mapping necessary for readers to fully understand the detailed chapters that follow: How to Read the Alliance Descriptions and the Key and Descriptions for California Vegetation. The key and the descriptions are arranged by vegetation strata — i.e., types dominated by trees, by shrubs, or by herbs. Five very useful appendices are presented in the MCV 2nd Edition: (1) Comprehensive Life History of Species Table; (2) Comprehensive Fire Regime Table; (3) National Classification Hierarchy of Vegetation Types; (4) List of Unranked Vegetation Types; and (5) Reclassified Vegetation Types.

What is the significance of this volume? Recognizing the growing utility of vegetation maps for land use planning and natural resource conservation, the California Legislature codified state standards for classification and mapping of vegetation in 2007. Map products that meet the California Vegetation Classification and Mapping Standards (CVCMS) integrate two principal standardized datasets: (1) classified field samples representing the full array of vegetation types in an area, and (2) a map layer displaying the classified vegetation types along with other information, among which are cover, structural data, and site quality information. The state standards were developed and endorsed by a committee comprised of representatives from several federal and state agencies. An expanded discussion of the process by which the standards were developed from the California Department of Fish

and Wildlife's perspective is available at <<https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=54411&inline=1>>.

The endorsed California Vegetation Classification is presented in MCV 2nd Edition, and this classification meets United States National Vegetation Classification (USNVC) standards. These standards, developed by the Federal Geographic Data Committee and peer-reviewed by the Ecological Society of America's Vegetation Panel, facilitate the analysis of biological and geographical data and the management of lands across different agencies on a nation-wide basis. All federal agencies are encouraged to meet the standards, as are institutions that receive federal funding; at a minimum, classifications must crosswalk to the USNVC. Therefore, map classifications adhering to the standards presented in the MCV 2nd Edition may be crosswalked with any recently produced map, or map under production for federal lands or using federal funds; nearly 48% of the land in California is owned by federal agencies (Gorte et al. 2012).

Approximately one third of the state's vegetation has been mapped using the classification set forth in this volume through integrated classification and mapping projects. The use of the standards will help these maps to be unified into a seamless statewide map. When incorporated with other data layers in geographic information systems, a detailed, accurate vegetation map will allow planners to avoid or reduce unnecessary habitat destruction and costly litigation when siting developments or large infrastructure projects. Planning at the local scale can take place with a statewide or regional vision of the cumulative impacts of a given project on an important habitat, or with an eye toward connecting wildlife corridors to other regions. Required compensatory mitigation can be directed to valuable habitat adjacent to existing conservation lands to help create large reserves and maximize benefits to plants and wildlife.

Wildland managers should find the fire regime information particularly valuable. The fire information was developed in consultation with members of the Association of Fire Ecologists and other scientists, and comprises an extensive literature and decades of experience. Fire characteristics are discussed for each alliance and also presented in tabular form in Appendix 2.

Federal and state agency staff charged with planning the conservation and wise use of, or preventing the extinction of, natural communities under the California Environmental Quality Act, the National Environmental Policy Act, and enabling legislation or operating policies of a variety of agencies should find that the objective analysis behind each defined alliance or association, description, and rarity ranking strengthens the definition of the vegetation community, thereby making it easier to discern, describe, and defend.

How has the second edition rectified criticisms of the MCV 1st Edition? (1) The classification presented earlier was criticized for over-emphasis of dominance in the tallest vegetation stratum. Schwartz (1997) stated this approach would not satisfy phytosociologists who base their classifications on detailed samples of all vegetation layers. MCV 2nd Edition advocates data-driven vegetation classification. Vegetation classifications are often more detailed than vegetation map units because of differences in scale and the use of remotely sensed imagery, and therefore only layers that are visible, for mapping. This is explained in the chapter entitled *The Difference between Vegetation Classification and Vegetation Mapping*. (2) Keil (1997) advocated inclusion of more associations based on samples. In response, the authors of MCV 2nd Edition were able to include many more associations based on the published results of sampling in the numerous studies that followed publication of MCV 1st

Edition. Each of the associations listed in the second edition is based on a full complement of published independent samples, typically 10 or more. (3) Lacey (1996) expressed a need for more discussion of regional variation to help horticulturists plan successful native gardens. MCV 2nd Edition includes regional variation, extensive life history notes, and habitats in which vegetation types have been documented. This information should be of use to horticulturists and restoration ecologists.

The MCV 2nd Edition has been in circulation long enough for users' comments to be based on practical experience, not on first impressions. Some users find the book too complicated; they want to quickly learn which vegetation types occur in an area, and conversely, in which areas particular vegetation types occur. Some have complained that all types are not included. The book only includes types that had been defined in works published prior to the publication of MCV 2nd Edition.

Although the MCV 2nd Edition is large, it is not truly a "stand-alone" volume, as it does not explain the California Mapping Standard it discusses — these standards are state standards, not CNPS standards, and may be found at <<http://biodiversity.ca.gov/vegmou.html>>. MCV 2nd Edition does not include the protocols (repeatable methods) necessary to achieve the sampling standards it advocates. Those protocols have been developed and refined cooperatively by CNPS and CDFW over the past 15 years. Minor revisions are made as field work reveals omissions, and may be found at <<http://www.cnps.org/cnps/vegetation/pdf/protocol-combined.pdf>>.

Even with these limitations, the comprehensive MCV 2nd Edition presents an enormous body of information in an organized way and, as a result, readers can develop a better understanding of California vegetation. The print volume does not provide color photographs of the vegetation types and is not intended to serve as a general overview of California vegetation for a casual audience. It will be of interest to students of vegetation, botanists, ecologists, environmental scientists, and natural history enthusiasts, and is a must-have for land-use managers and conservation planners.

Is *A Manual of California Vegetation* 2nd Edition the definitive work on California vegetation? No; it is a step in the process of developing an understanding of the complete variation and distribution of California vegetation. This understanding will evolve as more data are collected.

The California Native Plant Society is working diligently on a searchable, online version of MCV 2nd Edition, which currently is being beta tested. When it becomes available, many of the criticisms listed above will be resolved by using the "search" feature of the online version. CNPS also plans to include new vegetation types identified in studies published since the publication of MCV 2nd Edition, and photographs of many vegetation types in the online version.

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Allium abramsii CDFW photo by Kristi Lazar



Lupinus stiversii photo by Jeb Bjerke



Lilium rubescens CNPS photo by Aaron E. Sims



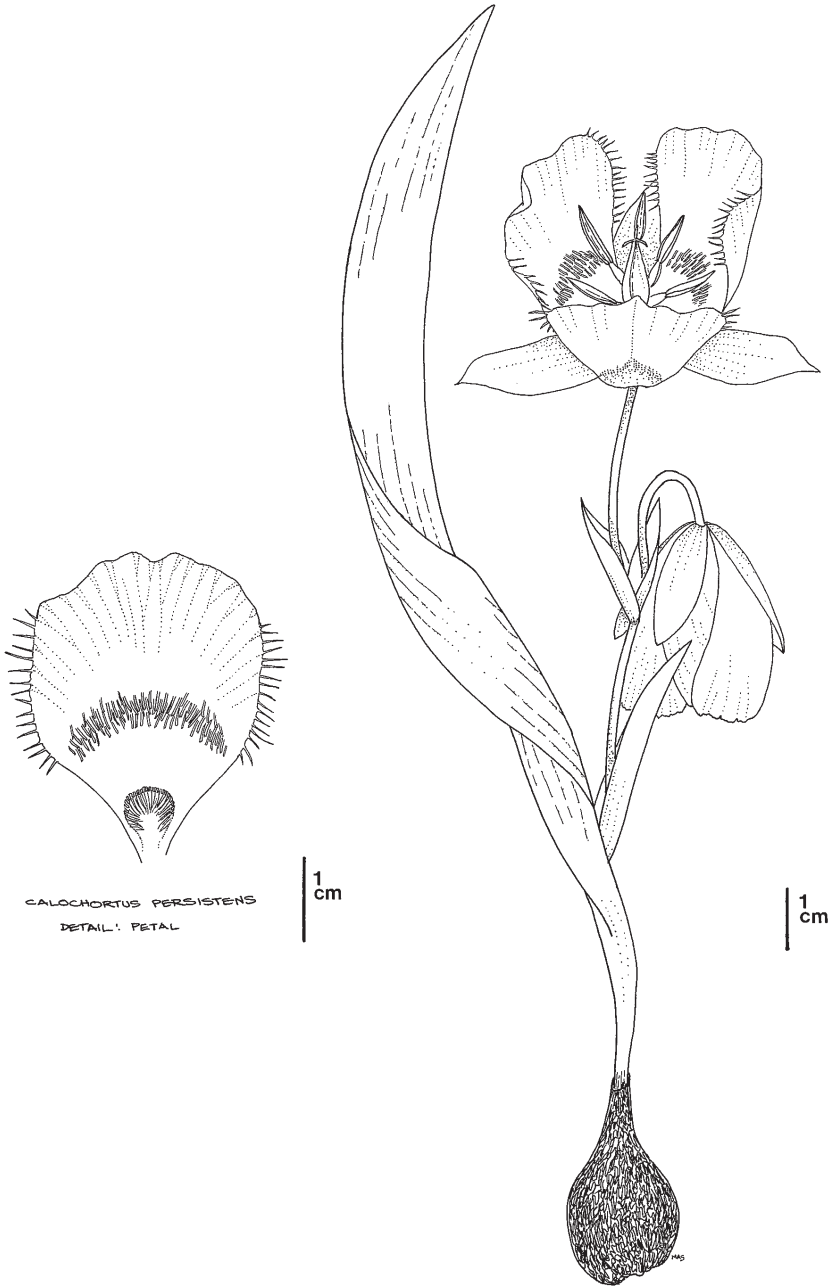
Astragalus claranus CNPS photo by Roxanne Bittman



Tuctoria mucronata CDFW Photo by Cheryl Burton



Amsinckia grandiflora CDFW Photo by Susan Cochrane Levitsky



Calochortus persistens CDFW Illustration by Mary Ann Showers

BOOKS RECEIVED AND AVAILABLE FOR REVIEW

Copies of the following books have been received, and are available for review by interested parties. Individuals interested in preparing a formal review that will be published in *California Fish and Game* should contact the editor (Vern.Bleich@wildlife.ca.gov) with their request to do so.

GOTSHALL, D. W. 2012. Pacific Coast inshore fishes. Fifth edition. Sea Challengers, Monterey, California, USA. 363 pages. \$9.99 (E-Book).

KIRKWOOD, S., AND E. MEYERS. 2012. America's national parks: an insider's guide to unforgettable places and experiences. Time Home Entertainment, Inc., New York, New York, USA. 208 pages. \$24.95 (hard cover).

LOVE, M. S. 2011. Certainly more than you want to know about the fishes of the Pacific coast: a postmodern experience. Really Big Press, Santa Barbara, California, USA. 650 pages. \$29.95 (soft cover).

INFORMATION FOR CONTRIBUTORS

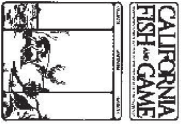
California Fish and Game is a peer-reviewed, scientific journal focused on the biology, ecology, and conservation of the flora and fauna of California or the surrounding area, and the northeastern Pacific Ocean. Authors may submit papers for consideration as an article, note, review, or comment. The most recent instructions for authors are published in Volume 97(1) of this journal (Bleich et al. 2011), and are accessible through the California Department of Fish and Wildlife web site (www.dfg.ca.gov/publications).

Planning is in progress to provide an avenue for authors to submit manuscripts directly through the web site, and to enable restricted and confidential access for reviewers. In the meantime, manuscripts should be submitted by e-mail following directions provided by Bleich et al. (2011). The journal standard for style is consistent with the Council of Science Editors (CSE) Style Manual (CSE 2006). Instructions in Bleich et al. (2011) supersede the CSE Style Manual where differences exist between formats.

Authors of manuscripts that are accepted for publication will be invoiced for charges at the rate of \$50 per printed page at the time page proofs are distributed. Authors should state acceptance of page charges in their submittal letters. The corresponding author will receive a PDF file of his or her publication without additional fees, and may distribute those copies without restriction. Plans are underway to make the complete series of *California Fish and Game* available as PDF documents on the California Department of Fish and Wildlife web site.

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