Best Management Practices for Pollinators on Western Rangelands

(Completed July 2018)

XERCES SOCIETY
for Invertebrate Conservation

BEST MANAGEMENT PRACTICES FOR POLLINATORS ON WESTERN RANGELANDS

(Completed July 2018)

Stephanie McKnight
Candace Fallon
Emma Pelton
Rich Hatfield
Aimée Code
Jennifer Hopwood
Sarina Jepsen
Scott Hoffman Black

The Xerces Society for Invertebrate Conservation

www.xerces.org

XERCES
SOCIETY
for Invertebrate Conservation

The Xerces® Society for Invertebrate Conservation is a nonprofit organization that protects wildlife through the conservation of invertebrates and their habitat. Established in 1971, the Society is at the forefront of invertebrate protection, harnessing the knowledge of scientists and the enthusiasm of citizens to implement conservation programs worldwide. The Society uses advocacy, education, and applied research to promote invertebrate conservation.

The Xerces Society for Invertebrate Conservation
628 NE Broadway, Suite 200, Portland, OR 97232
Tel (855) 232-6639 Fax (503) 233-6794 www.xerces.org

Regional offices from coast to coast.

Primary Authors and Contributors

The Xerces Society for Invertebrate Conservation: Stephanie McKnight, Candace Fallon, Emma Pelton, Rich Hatfield, Aimée Code, Jennifer Hopwood, Sarina Jepsen, and Scott Hoffman Black.

Acknowledgments

Funding for this project was provided by the US Forest Service as well as the Ceres Trust, CS Fund, The Dudley Foundation, Endangered Species Chocolate, LLC., J. Crew, Justin's, Madhava Natural Sweeteners, The New-Land Foundation, Inc., Turner Foundation, Inc., White Pine Fund, Whole Systems Foundation, and the Xerces Society for Invertebrate Conservation members.

The authors wish to thank everyone who contributed to the development of these recommendations: the twelve interviewees, forty-three survey respondents, and additional researchers and practitioners with whom we had informal conversations. Xerces Society staff members Sarah Hoyle, Jessa Kay Cruz, Angela Laws, Ray Moranz, and Mace Vaughan contributed to the development and review of specific sections. Content for the case studies was developed with contributions from Dr. Sandra DeBano, Oregon State University Department of Fisheries and Wildlife, and Mary Rowland, US Forest Service Pacific Northwest Research Station—La Grande Forestry & Range Sciences Laboratory (Case Study 7), and Laura Navarette, US Forest Service (Case Study **10**). William Carromero, Tyler Johnson, Carol Spurrier, Allen Rowley, Mary Rowland of the US Forest Service, and Dr. Sandra DeBano provided input and/or review of the final document.

Editing: Matthew Shepherd, The Xerces Society

Layout: Michele Blackburn, The Xerces Society

Printing: Print Results, Portland, OR

Recommended Citation

The Xerces Society. 2018. Best Management Practices for Pollinators on Western Rangelands. **126+vii** pp. Portland, OR: The Xerces Society for Invertebrate Conservation. (Available online at https://xerces.org/best-management-practices-for-pollinators-on-western-rangelands).

Photographs & Artwork

We are grateful to the many photographers and designers for allowing us to use their wonderful photographs and artwork. The copyright for all photographs, icons, and designs is retained by the Xerces Society or the creators. None of the photographs or artwork may be reproduced without permission. For a complete list of photographers and designers, please see the Photo Credits section on page **126**. If you wish to contact a photographer or designer, please contact the Xerces Society at the address above.

Cover Photograph

Sagebrush steppe habitat in the Santa Rosa Mountains within the Humboldt-Toiyabe National Forest provides a pollinator haven on rangeland in central Nevada. by Emma Pelton..

Contents

Good pollinator habitat provides multiple kinds of blooming flowers which offer pollen and nectar to bees and butterflies.

Executive Summary

Rangelands comprise the majority of public lands in the western United States and support some of the highest diversity of bee species, as well as many butterflies, moths, and other pollinators. However, many of these pollinator species are declining and at-risk. Incorporating pollinators into rangeland management is essential to help recover pollinator populations as well as maintain healthy rangelands for plants, wildlife, livestock, and the people who rely on them.

The Xerces Society for Invertebrate Conservation partnered with the US Forest Service to identify best management practices (BMPs) for pollinators on western rangelands as recommendations for land managers. These BMPs were informed by a literature review, surveys, and interviews with pollinator experts and land managers in 2016–2018. Together, they represent the state of the knowledge about managing western rangelands for pollinators. While there is still much to learn, these BMPs provide actionable, practical recommendations that enables land managers to help conserve pollinators on public lands in the West.

An overview of major pollinator groups (bees, butterflies and moths, other invertebrates, and vertebrates), their status, and threats provide a primer on these animals and their needs. Good pollinator habitat provides food, shelter, and nest sites, is connected to other habitat patches, is safe from pesticides and high levels of pathogens. Overall, management and restoration which aims to incorporate pollinators should focus on incorporating heterogeneity into the landscape, considering interactions among management and environmental fluctuations, and using an adaptive management framework.

Management practices addressed in the BMPs include grazing, mowing, prescribed fire, and pesticide use. Incorporating pollinators into restoration projects including post-wildfire seeding and the sourcing and establishment of native plants are addressed, as are invasive nonnative and noxious plant management, managed pollinators, recreation, and climate change impacts. Each topic begins with summary of the known effects of each land use/ management practice on pollinators and their habitat, followed by recommendations on how to incorporate pollinator conservation into management decisions. Monitoring pollinator populations wraps up the document. All of the information is summarized for use by practitioners in the field, but may also be useful for decision makers.

Rangeland habitat in Nevada.

Long-tongued bee nectaring on penstemon.

Introduction

"*Pollinators are a key component of a healthy rangeland ecosystem.*" — *Black et al.* (2011)

Rangelands comprise the majority of public lands in the western United States, spanning a huge diversity of ecological regions, habitat types, and elevations—from grasslands to sagebrush steppe to pinyon–juniper woodlands to mountain meadows (Shiflet 1994). Native pollinators are an important but often overlooked group of animals that both rely upon and help maintain rangeland ecosystems. There are just over 3,600 species of native bees in the United States and Canada alone, with the highest diversity of native bees on western rangelands (see Figure **1**). Bees, as well as butterflies, moths, and other pollinators are a key component of global and local biodiversity, but pollinators are in trouble. It is estimated that 40% of invertebrate pollinator species may be at risk of extinction worldwide due to stressors including habitat loss, pesticides, disease, and effects of climate change (IPBES 2016).

A lack of pollinators can have major ecological and economic impacts on rangelands (Potts et al. 2010a, 2010b). Pollinators provide pollination services for flowering plants which are fundamental components of rangeland ecosystems; approximately 85% of flowering plant species are pollinated by animals (Ollerton et al. 2011) including threatened and endangered species (Tubbesing et al. 2014). These ecological services make pollinators keystone species in terrestrial ecosystems: pollination produces the seeds and fruits which feed everything from songbirds to grizzly bears (Ollerton et al. 2011). Pollinators are also vital for agriculture with 35% of global crop production relying on insect pollination (Klein et al. 2007). The value of crop pollination by native wild bees alone is conservatively estimated at $3 billion annually in the U.S. (Losey and Vaughan 2006; Calderone 2012), and rangelands can serve as an important reservoir of native pollinators for adjacent agricultural land (Havstad et al. 2007). Pollinators can also provide economic benefits to ranchers; they are essential for pollination and reproduction of flowering plants which provide nutrient rich forage to livestock (and native ungulates) in rangelands (Holechek 1984; Ralphs and Pfister 1992; Teague et al. 2009, 2015; Gilgert and Vaughan 2011). In some western rangelands, forbs can comprise up to 34% of cattle diets in grass-dominated rangelands, and up to 80% in forb-dominated rangelands (Pieper and Beck 1980; Ralphs and Pfister 1992). Rangelands with a high diversity of native flowering plants provide both the highest quality and most nutritious forage for livestock and native ungulates (Holechek 1984). In addition, diverse plant communities provide crucial habitat for a variety of other wildlife species of conservation concern such as anadromous fish and gallinaceous birds (Gilgert and Vaughan 2011). Managing rangelands for pollinators is also often compatible with multiple-use directives, because healthy ecosystems can increase the recreational value of public lands. In short, incorporating pollinators into rangeland management is essential to help pollinator populations recover and will also maintain healthy rangelands for plants, wildlife, livestock, and the people who rely on them (Gilgert and Vaughan 2011; Dumroese et al. 2016).

How do we start managing our rangelands with pollinators in mind? The Xerces Society for Invertebrate Conservation partnered with the US Forest Service to identify best management practices (BMPs) for pollinators on western rangelands as recommendations for land managers. These BMPs were informed by a literature review and surveys and interviews with pollinator experts and land managers. Together, they represent the state of the knowledge about managing western rangelands for pollinators. While there is still much to learn, these BMPs provide actionable, practical recommendations that enables land managers to help conserve pollinators on public lands in the West.

Sagebrush steppe habitat with juniper on rangeland in southeastern Oregon.

Report Development

Literature Review

The BMPs are based upon a thorough review of peer-reviewed literature and technical materials on the topic of rangeland management to benefit pollinators and their habitat. Relevant information on grazing, fire, mowing, invasive plant management, pesticides, recreation, and restoration were reviewed, with a particular emphasis on studies conducted in western rangelands. This body of information was reviewed using the lens of understanding how to manage rangeland ecosystems for the benefit of pollinators while considering feasibility relative to existing land use practices. Relevant findings are presented at the beginning of each management section and the complete literature review is available online (see **Appendix C** for further information).

Surveys and Interviews with Practitioners, Researchers, and Ranchers

Interviews and an online survey were conducted to better understand current rangeland management practices and the opportunities and obstacles that may limit the management of rangelands for the benefit of pollinators. Twelve in-depth interviews were conducted over the telephone with staff from the US Forest Service, the Bureau of Land Management, a state agency, and a conservation nonprofit as well as researchers from multiple universities and a rancher. The authors worked with US Forest Service staff to identify the interviewees, all of whom understood the science and practice related to management and pollinator conservation in rangeland systems.

An online survey was disseminated electronically to a wider audience with a focus on biologists and ranchers. Survey questions focused on current practices and attitudes as well as opportunities and obstacles to implementing pollinator-friendly rangeland practices. The forty-three respondents included employees of federal, tribal, state, and local governments, ranchers and consultants. Respondents came from California, Colorado, Idaho, New Mexico, Nevada, Oregon, Utah, Washington, and Wyoming. Additional responses were received from people in Alaska and South Dakota, but those states fall outside the geographic scope of this document.

Major findings from the interviews and survey responses include:

- The majority of respondents reported managed grazing is practiced on the lands they manage. Secondary management actions reported included fuels reduction/forest thinning, revegetation/restoration, and pesticide use.
- The most commonly used pesticide reported were herbicides; insecticide and rodenticide use was also reported.
- Pollinator-friendly management actions identified by respondents as the most likely to benefit pollinators include changes in grazing practices (timing, duration, stocking rates) that will increase forbs and native bunch grasses, and restoration of native ecosystems after disturbances such as invasive plant encroachment and fire.

- Pollinator-friendly management actions ranked as the most feasible were: 1) increased use of native, pollinator-attractive forbs and plants incorporated into plant materials used for management; 2) replacing broadcast herbicide use with spot spraying; and 3) protecting specific plant species (e.g., larval hosts for butterflies). Adjusting grazing practices was identified as the least feasible management action.
- The major barrier identified by respondents to managing for pollinators was strongly held attitudes towards rangeland management goals, including the perceived philosophical, economic, political, and social value of pollinators and biodiversity. Seed cost and availability and knowledge/training gaps of how to best manage for pollinators were also cited.
- The value of pollinators to rangeland productivity and ecosystem health benefits were identified as the top two reasons respondents would adopt more pollinator-friendly practices.
- Respondents identified additional benefits of managing for pollinators including creating healthier ecosystems, enhancing biodiversity, strengthening ecosystem resilience and resistance, and benefiting other wildlife species. Connections with other land management goals were identified including restoration of riparian areas and management for greater sage-grouse, desert tortoise, and other sensitive species.
- While a few individuals reported that they/their agencies were beginning to incorporate pollinators and pollinator habitat into monitoring schemes (e.g., Bureau of Land Management's Assessment, Inventory, and Monitoring (AIM) pollinator supplementary indicator), monitoring of native pollinator communities on rangelands overall is clearly minimal and inconsistent. Incorporating qualitative monitoring for pollinators and their habitat was identified by respondents as more feasible than implementing quantitative monitoring.

Report Structure

This document consists of four major sections:

- **Chapter 1: Meet the Pollinators**—an overview of pollinators, their biology, their decline, and threats to pollinators on rangelands.
- **Chapter 2: Best Management Practices—**a brief summary of pollinator habitat needs, followed by a summary of the known effects of each management practice on pollinators, with recommendations on how to incorporate the needs of pollinators into management decisions for each action. Management recommendations are provided for grazing, fire, restoration, invasive plant management, pesticides, managed pollinators, recreation, climate change, and monitoring pollinator populations.
- **Chapter 3: Monitoring—**a summary of major monitoring programs for pollinators and their habitat.
- **Appendices—**including links to the comprehensive literature review and tables detailing native bee phenology, conservation status, ecoregion associations, and habitat requirements.

These best management practices were developed to incorporate pollinator-friendly practices into management of federally managed rangelands in the eleven western United States: Arizona, California, Colorado, Idaho, Montana, New Mexico, Nevada, Utah, Oregon, Washington, and Wyoming. The practices, however, may also be applicable to state, tribal, local, nonprofit, and privately owned rangelands. All of the information is summarized for use by practitioners in the field, and may also be useful for decision makers.

1

Meet the Pollinators

All flowering plants depend on pollination. While wind and water are important pollen vectors in some plant groups (such as wind-pollinated grasses and conifers, and water-pollinated waterweed), animals provide the bulk of pollination services for 85% of flowering plants (Ollerton et al. 2011). Of animal pollinators, bees are the most important group of pollinators in temperate North America because they actively gather and transport pollen, moving it efficiently through the landscape (Michener 2007). Other important pollinators include moths and butterflies, flies, and beetles. In some regions, birds, bats, and other vertebrates are also important pollinators.

Bees

When most people think of bee pollinators, they picture the European honey bee (*Apis mellifera).* While honey bees are indeed significant pollinators of agricultural crops, they are just a single species and were introduced to North America in the 1600s. There are more than 3,600 species of bees (order Hymenoptera) native to the US and Canada with a vast array of life histories and behaviors (Michener 2007; Ascher and Pickering 2015). These native bees are essential pollinators for wildflowers and flowering shrubs including those found on rangelands, while also significantly contributing to agricultural pollination (Losey and Vaughan 2006).

All bees undergo four life stages: egg, larva, pupa, and adult. The recognizable adult bee typically flies for three to six weeks of the year, only a short period of the bee's life cycle. In the US, adult bees generally make their appearance from February to September in most areas, but in mild climates, bees

White shouldered bumble bee (*Bombus appositus*) visiting thistle.

can be active year-round. The life histories of native bees can loosely be grouped into three different guilds (while not necessarily reflecting a phylogenetic affinity): (1) solitary and semi-social ground-nesting bees; (2) above-ground tunnel-nesting bees; and (3) cavity-nesting bees (both above- and below-ground). The majority of North America's native bees are ground-nesting (~70%), followed by tunnel-nesting bees (~30%). Cavity-nesting native bees is the smallest group, limited to bumble bees, with 47 species (~1%).


Most of our native bees are ground-nesting, requiring bare or mostly bare soil to make their nests.

The vast majority of bees are solitary; a single female creates a nest, provisions the nest with food (pollen and nectar), and lays eggs. Other bees exhibit some form of social behavior such as nesting in aggregations (shared nesting area), communal nesting (shared nest entrance), or semi-social behavior (some cooperation in providing for young). Eusocial (truly social) bees, which have a solitary queen and worker caste, are limited to bumble bees and some species of sweat bee; nonnative European honey bees are also highly eusocial. Appendix **C** provides a link to a list of native bee genera by ecoregion and Appendix **B** provides a list of bumble bee species. Both lists include the US Forest Service regions that they occur in and basic life history information.

Bees have different needs depending on their life stage and social caste, but overall require the following: nectar to fuel adults, pollen and some nectar to feed young, and places to nest and overwinter. They also need habitat that is safe from pesticides and high levels of pathogens.

Bee Diet

When foraging for pollen to provision their nests, some bee species are more selective than others. Bees that visit only a few plant species or closely related species (usually within a single plant genus or family) for pollen are considered *oligolectic*; a few bee species are even *monolectic*—visiting only one plant species. Other bees are considered pollen generalists (*polylectic*—many plant species visited). Oligolecty and polylecty do not necessarily correspond with genetic lineages, though some patterns do exist (Michener 2007). There are oligolectic and polylectic bees in both ground-nesting and above-ground-nesting bees. Generally, bumble bees and most other social bees are polylectic, though some more so than others. True monolecty and oligolecty are less common than polylecty, though up to 60% of bees in some regions may be oligolectic (Minckley and Roulston 2006), including much of the West. In general, bees are less discerning when foraging for nectar (which provides energy for adult bees) and even oligoletic bees will take nectar from a wider range of flowers than those from which they gather pollen.

Ground-Nesting Bees' Nesting

Ground nesting is the norm for bees, with over 2,500 species from several different families in the US and Canada that nest this way. As such, it is difficult to generalize life history information. However, most ground nesting bees excavate nests in bare, or nearly-bare, ground on slopes that vary from

Above-ground tunnel-nesting bees, like this mason bee (*Osmia* sp). nest in small cavities which they partition into brood cells with collected mud.

horizontal to vertical; the substrate, moisture, and texture of the soil in which bees nest varies greatly by species, and can include alkaline salt flats, clay, sand, and loam, even hardened sandstone (Cane 1991; Orr et al. 2016). Exceptions include some species of *Halictus* which prefer to nest in soil covered with small rocks (Cane 2015) and some species of *Andrena* and *Colletes* which commonly nest among vegetation with only small patches of bare ground.

Nests vary greatly in depth and complexity, from a single tunnel a few inches (several centimeters) deep to a 3-dimensional network of branching tunnels up to a yard (meter) or more deep. Brood cells may be placed in clusters at the end of tunnels, excavated from the side of a tunnel along its length, or singly at the end of short branch tunnels. The tunnels are not lined, but the brood cells are carefully finished to protect the contents. Bees smooth the cell walls with their bodies and often line them with waxy secretions from glands in their bodies to help control moisture. *Colletes* secrete a cellophane-like substance with which they entirely line brood cells (O'Toole and Raw 1991; Cane 1997 and references therein; Michener 2007).

Although mostly solitary, ground-nesting bees may nest in large aggregations, at densities of up to sixty or seventy nests per square yard (O'Toole and Raw 1991). The nest entrances may be surrounded by a mound of excavated soil (tumuli) or a short turret constructed of soil pellets stuck together with a secretion.

Above-Ground Tunnel-Nesting Bees' Nesting

This group of bees also represents a large number of species in North America—over 1,000 species in the US and Canada—and includes most species of leafcutter bees, mason bees, and carpenter bees. Most above-ground tunnel-nesting bees build their nests in pre-existing cavities like bark beetle burrows, stems of pithy plants, cracks in rocks, downed or decaying wood, snail shells, or abandoned nests of other bees and wasps (O'Toole and Raw 1991). They can also utilize human-made structures such as cracks or crevices in houses, pipe openings, and artificial nests. Some species actively construct their nests on the surface of rocks, walls, or other sturdy areas. Almost all bees in this group collect materials like pebbles, chewed wood, mud, or leaves to divide the tunnel into brood cells or, in the case of leafcutter bees (*Megachile*), to entirely line the walls of their nests(O'Toole and Raw 1991). Exceptions are yellow-faced bees (*Hylaeus*), which secrete a cellophane-like substance to create the dividing partitions, and carpenter bees, large (*Xylocopa*) and small (*Ceratina*), which chew out nest tunnels and use the resulting "sawdust" to separate cells (O'Toole and Raw 1991).

Ground-Nesting and Above-Ground Tunnel-Nesting Bees' Reproduction

While the nesting biology of ground-nesting and above-ground, tunnel-nesting bees varies considerably, reproduction is similar. Once the nest is constructed, female bees collect pollen and nectar from flowers and mix the two together in a brood cell to form a food mass (often called a "pollen loaf") upon which she lays an egg. She will repeat this process of constructing and provisioning brood cells until the nest is complete. Each female will only lay a couple of dozen eggs, in some species as few as eight (Michener 2007). After the eggs hatch, the larvae eat the food mass and go through several instars before pupating. After pupation, the adult emerges, mates, and begins the life cycle anew. Whether the species overwinters as an adult, or as a prepupa, is species- and climate-dependent (Michener 2007). (Note: the nesting and reproduction of parasitic and cleptoparasitic bees is not covered here.)

Cavity-Nesting Bees' Nests and Reproduction

The indiscriminate cuckoo bumble bee (*Bombus insularis*), is a nest parasite which does not build nests or provision resources for its offspring.

North America is home to 47 species of bumble bees which generally nest in pre-existing, large cavities. In natural environments, these can be abandoned rodent burrows, abandoned bird nests, or on the surface of the ground, usually under thatched grasses or moss (Williams et al. 2014). The only exceptions are the six bumble bee species which do not build nests, but are cleptoparasites of other bumble bees' stores. Unlike the majority of bee species, bumble bees are social, with colonies formed by a solitary queen that raises female worker brood(s) before producing gynes (new queens) and males, the reproductive members of the colony. In these nests, queens build waxen pots in which they provision food (pollen and nectar) and lay eggs. Development in cavity-nesting bees is similar to ground-nesting and above-ground tunnel-nesting bees, except that in bumble bees, all larvae proceed through pupation (they do not overwinter as prepupae) and emerge as winged adults. After the worker caste emerges, the queen stays in the nest tending to developing larvae and laying eggs; the workers forage for pollen and nectar to bring back to the nest. At the end of the flight season the new queens and males mate, and the new queens find a new place—usually a small cavity— to overwinter. The remainder of the colony, including the foundress queen, dies (Goulson 2010). See Appendix **A** for bumble bee species distribution by state and US Forest Service region, and floral associations.

Bee Diversity in Western Rangelands

Western rangelands support a high diversity of native bees, with the lowest diversity found in northern temperate regions and the highest diversity found in areas of the intermountain West and Desert Southwest. A large proportion of western rangelands are in arid desert regions that are both floristically diverse and support an incredible diversity of native bees (Griswold et al. 1997; Messinger 2006; Colla et al. 2012). For example, the original boundary of Grand Staircase Escalante National Monument in Utah supports over 650 species of native bees, which is almost as many as the 750 native bees found in all of eastern North America (Messinger 2006; Colla et al. 2012). Arid desert regions are also at greater risk of losing biodiversity as a result of anthropomorphic influences such as livestock grazing, invasive species, development, and the interaction with ecological stressors such as climate change (Brussard et al. 1998; Lovich and Bainbridge 1999; Donahue 2000; Brooks et al. 2001; Gasparatos et al. 2017). While current knowledge suggests that temperate northern regions of the West have lower bee diversity than arid regions of the Southwest, there have been very few inventories of bees in these areas. One native bee inventory in the mountains of northern Washington found a relatively rich bee community with a total of 140 species of native bees that visited 57 plant species (Wilson et al. 2010). In a study in a Pacific Northwest bunchgrass prairie in Oregon, 211 species or morphospecies were discovered (Kimoto 2012a). Our understanding of regional bee diversity may improve as more native bee inventories are conducted across the West; Figure **1** shows our current understanding of bee diversity.

Butterflies and Moths

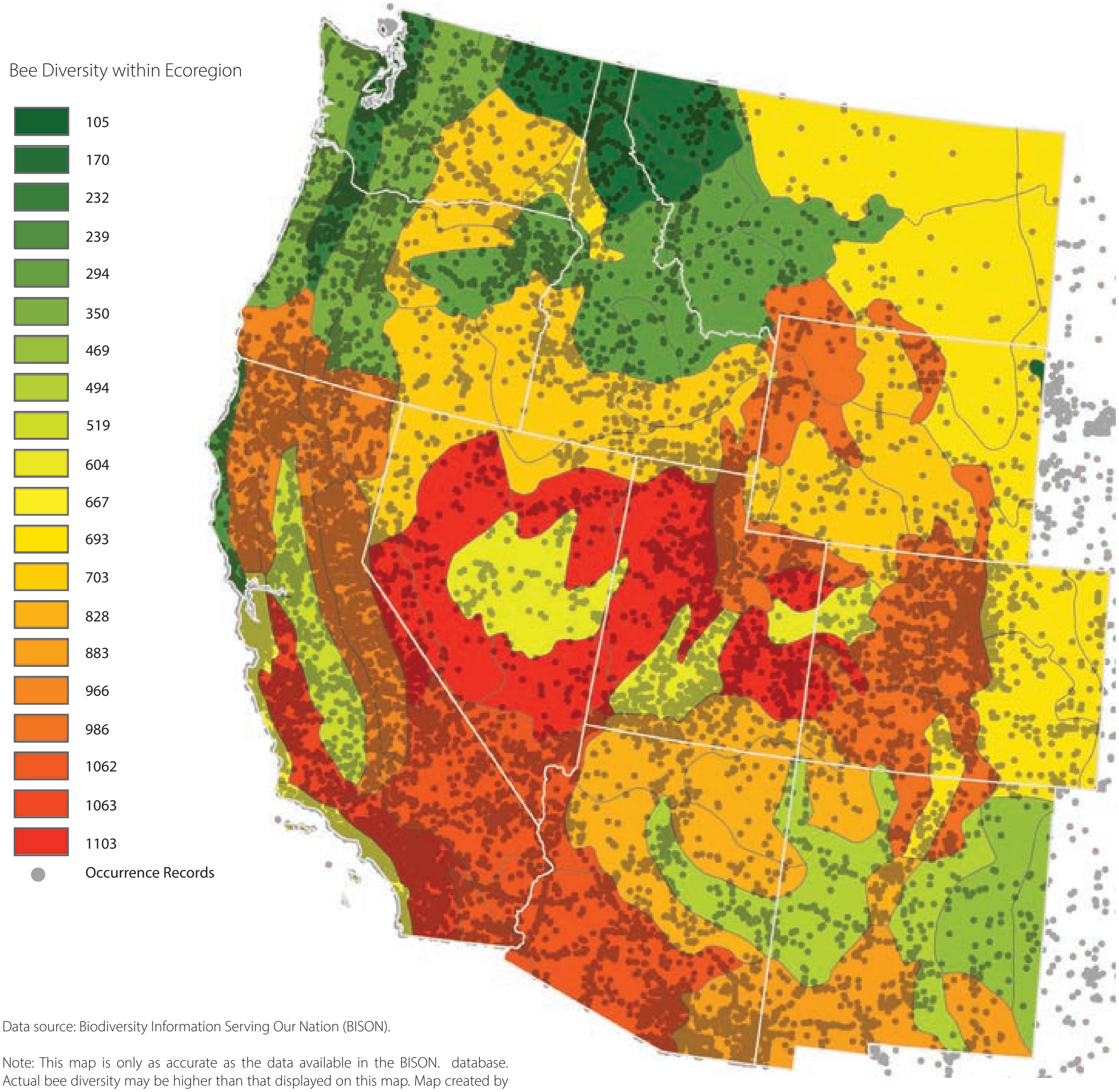
Beyond bees, butterflies and moths are among the most well-known group of pollinators, making up the incredibly widespread and diverse order Lepidoptera. Nearly 800 species of butterflies and 11,000 species of moths are found in North America alone (Black et al 2016). The vast majority of butterflies in the US provide pollination services, as do many moth species, although some moths do not feed as adults and thus are not pollinators. Butterflies and moths undergo complete metamorphosis which includes four distinct life stages: egg, caterpillar (or larva), chrysalis (or pupa), and adult. Depending on the climate and location, they may have two or more generations per year. Generations that occur in the spring or summer will typically complete their life cycle within three to eight weeks, whereas generations that overwinter (diapause) may live from six to nine months. Species that produce only one generation per year (univoltine) generally take twelve months to complete their life cycle but many lepidoptera species have multiple generations per year (multivoltine). Butterflies and moths have different needs depending on their life stages, but overall require the following: nectar to fuel adults, food plants for caterpillars, and places to pupate and overwinter. They also need habitat that is safe from pesticides and high levels of pathogens. See Appendix **C** for a list of at-risk butterflies and moths in the West, along with host plant information.

Adult Butterfly and Moth Diet

Most adult butterflies and moths depend on flower nectar for energy; they feed using long, tubular mouthparts (the proboscis) that they unfurl to reach nectar in flowers. Some butterflies and moths also seek out sugars from tree sap, rotting fruit, or aphid honeydew. Additional nutrients, salts, and minerals are obtained from animal waste and carcasses, puddles, and moist soil. Pollen transfer by butterflies and moths is usually incidental and tends to occur when adults land on a flower to nectar. However, there are important exceptions such as the yucca moth (*Tegeticula* spp.) which is the sole pollinator of Joshua trees (*Yucca* spp.). The moth actively collects pollen from one Joshua tree flower and then moves to a different flower to lay their eggs and deposit the pollen into that flower's stigma (Pellmyr 2003).

FIGURE 1: Bee Diversity in the West.

Displayed as the total number of bee species collected within each ecoregion



Data source: Biodiversity Information Serving Our Nation (BISON).

Note: This map is only as accurate as the data available in the BISON. database. Actual bee diversity may be higher than that displayed on this map. Map created by Stephanie McKnight of the Xerces Society for Invertebrate Conservation, 2018.

Caterpillar Diet

Many species of grasses, wildflowers, shrubs, and trees are hosts for butterflies and moths. In fact, the same native plants that provide high-quality nectar for adults can also be host plants for their caterpillars. Host plants are a requirement for all caterpillar species in the West, and while some will eat a wide range of plants from multiple families, others will eat only a single species or several very closely related plant species. Unsurprisingly, butterflies and moths with strict host plant needs are restricted to areas where their host plants are found; if the host plant itself is restricted, the butterfly or moth will be as well. For example, the federally endangered Laguna Mountains skipper (*Pyrgus ruralis lagunae*) requires Cleveland's horkelia (*Horkelia clevelandii*), a rare forb species found only in a few montane meadows in southern California (Black and Vaughan 2005). Caterpillars are voracious eaters, quickly growing and moving through several instars, or molts, before reaching pupation.

Monarch butterflies form a light green chrysalis adorned with gold. ~10-14 dayes later they emerge as adult butterflies.

Pupation and Overwintering

All caterpillars must pupate before emerging as the winged adults we recognize as butterflies and moths. Also referred to as chrysalids (butterflies) or cocoons (moths), pupae are immobile and highly vulnerable to predation. Because of this, safe places for refuge are critical. Caterpillars often pupate on or near their host plants or they will seek protected places under duff or leaf litter, in shrubs, or in the base of grass tussocks; they may also attach to nearby fences, buildings, or other structures.

Protected places are important during diapause, a state of suspended development that typically occurs during periods of inclement weather. Most often this occurs over the winter, but for some species it happens during dry summer seasons when floral and host resources are scarce or unpalatable. During this time, no growth or development occurs, and mating usually ceases. Most butterflies undergo this period of relative inactivity as caterpillars or chrysalides, but some species overwinter as eggs or adults. Typically, this occurs in or near the area where a butterfly is born, but some species will migrate elsewhere to overwinter. Monarchs (*Danaus plexippus plexippus*) are a classic example of this, traveling from the northern and inland US and southern Canada to overwintering sites along the California coast and in central Mexico. Protected places are also important for spending the night (or day, in the case of nocturnal moths) or escaping storms, and species may seek out the same habitat features that are used during pupation: tree crevices, leaf litter, and bunch grasses. Some species like monarchs will sometimes form small aggregations in tree canopies to rest overnight during the migration before moving on the next day.

Beyond bees, moths, and butterflies, many other insects can also be pollinators such as this bee-mimicing syrphid fly (top), fly (lower left), and beetle (lower right).

Other Invertebrate Pollinators

Bees, butterflies, and moths are not our only invertebrate pollinators. Some flies (order Diptera) can be incredibly important pollinators. They are found in almost every type of terrestrial environment and often at high abundances; at high elevations, flies—not bees—are the principle pollinators (Arroyo et al. 1982; Primack 1983; Inouye and Pyke 1988; Orford et al. 2015). Nearly half of the ~150 know Diptera families include flies that feed on flowers as adults, and more than 550 species of flowering plants have been documented as being regularly visited by flies (Larson et al. 2001). With over 160,000 species, flies form an extremely large and diverse group. Some have specialized relationships with flowers, while others are generalists. Flies visit flowers to eat nectar and pollen, or to lay eggs so the larvae can feed on flower heads and seeds. Many flies have few hairs compared to bees, making them less effective pollinators. However, others have very hairy bodies that are adept at picking up pollen when they visit flowers for nectar. These include some of our bee mimics, the bee flies (family Bombyliidae). Other flies that mimic bees or wasps may have few hairs but are still important pollinators; some of the best known are the syrphid flies, or hover flies. Even some unlikely fly family members—mosquitoes and midges—can pollinate certain flowers.

Beetles (order Coleoptera) are thought to be among the first insects which evolved to visit flowers, and they remain important pollinators in some of the more ancient lineages such as magnolias. More than 184 flowering plant species are pollinated almost exclusively by beetles (Bernhardt 2000), although the majority of these species are found in the tropics or areas with Mediterranean climates. Estimates of pollinating beetle species in western North America are difficult to determine given general lack of research and taxonomic uncertainty in many groups, but beetles are known to be important pollinators for species like mariposa lilies (*Calochortus* spp.) and beargrass (*Xerophyllum* spp.) (National Research Council 2007).

Other invertebrate groups play less of a role in pollination, although some species of ants and wasps (order Hymenoptera) are known to pollinate plants.

Birds, Bats, and other Vertebrate Pollinators

While invertebrate species pollinate the vast majority of the world's flowers, vertebrate pollinators such as hummingbirds, bats, small mammals, and even lizards also play an important role in the pollination of some plants. For example, birds pollinate approximately 5% of cultivated plant species with known pollinators (Nabhan and Buchmann 1997). Over 920 species of birds worldwide are known to pollinate plants. In North America, the most common pollinating birds are hummingbirds (Whelan et al. 2008). Other less widespread species include the white-winged dove (*Zenaida asiatica*) which pollinate saguaros (*Saguaro* spp.), and verdin (*Auriparus flaviceps*).

Many hummingbird species are long distance migrants traveling between the US, Canada, and Mexico. Hummingbirds are specialized nectar feeders with morphological, ecological, and physiological adaptations for this diet, and serve as pollinators for a wide array of native plants (Stiles 1981; Brown and Bowers 1985; Temeles and John Kress 2003; Gegear and Burns 2007). They depend almost entirely on floral nectar for their energy supply, and their survival hinges upon a reliable supply of this fuel. In the West, about 130 plant species are pollinated by hummingbirds (Johnsgard 2016). The US Forest Service and Pollinator Partnership have created recommendations for *Maintaining and Improving Habitat for Hummingbirds in the Western U.S.* which is available on the US Forest Service website (www.fs.fed.us/wildflowers/pollinators).

Bats are also important pollinators in tropical and desert climates. While the diet of the majority of bats is composed of insects, some bats also seek nectar from flowers, acting as efficient pollinators thanks to their furry heads that can capture pollen for long-distance transfer. (In the Neotropics alone, an estimated 500 species of plants depend on bats for pollination [Lobova et al. 2009].) In the West, agaves (*Agave* spp.), saguaros (*Saguaro* spp.), and Cardón (*Pachycereus pringlei*) rely on bats for pollination, and bats visit other succulent species like the organ pipe cactus (*Stenocereus thurberi*). Both the lesser long-nosed bat (*Leptonycteris curasaoae yerbabuenae*)—a federally endangered species (USFWS 1988)—and the Mexican long-tongued bat (*Choeronycteris mexicana*) are key pollinators in the Southwest that depend on nectar to help fuel long migrations.

The best management practices presented in this document provide recommendations primarily for insect pollinators such as bees, butterflies, moths, and flies, but considering the needs of birds, bats, and other pollinators in management decisions is highly encouraged.

Hummingbird on Rocky Mountain bee plant.

The western bumble bee (*Bombus occidentalis*), historically broadly distributed in western North America, has experienced serious declines in relative abundance and range in recent years.

Threats to Pollinators and Population Declines

Little baseline data exists for the majority of wild pollinator taxa in most regions across the world, which makes determinations on the conservation status of pollinators difficult to ascertain (National Research Council 2007; Goulson et al. 2015). However, some data does exist and Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) estimates that 40% of invertebrate pollinator species may be at risk of extinction worldwide (IPBES 2016). Given their economic importance, the best available estimates for decline exist for managed European honey bees (Potts et al. 2010a; Goulson et al. 2015; Kulhanek et al. 2017), which have been monitored for years in both North America and Europe. The annual nationwide survey for the 2016–2017 season showed a 33% loss of managed honey bee colonies in the US (Steinhauer et al. 2017). However, honey bees may be faring better than many species of wild, native pollinators. Honey bees differ from most native bees in that they are relatively large bodied, form large colonies, and are highly managed by humans. Among wild bees, bumble bees are the best studied, and it is known that many North American species have undergone severe declines in recent decades (Colla and Packer 2008; Cameron et al. 2011; Colla et al. 2012; Hatfield et al. 2014; Dolan et al. 2017). Range losses have also been documented for several species, including the western bumble bee (*Bombus occidentalis*), and 27% of bumble bee species in the US and Canada are listed in an extinction risk category by the International Union for Conservation of Nature (IUCN) (Hatfield et al. 2014). This includes 11 species found in western rangelands (see Appendix **A** for details on western bumble bee species). Two other bee groups have been assessed by NatureServe and are also in decline: 50% of leafcutter bee species and 27% of mason bee species are at-risk. See Appendix **B** for a list of at-risk pollinators in the West.

Butterfly populations are also declining in parts of the world. Butterfly Conservation in the U.K. recently released a report documenting declines in occurrence and/or abundance for 76% of the country's butterfly species over the last 40 years (Fox et al. 2015). Other countries are reporting similar declines (Habel et al. 2016). Approximately 19% of the 800 described butterfly species in North America have been placed in some risk category (NatureServe 2018). In the US and Canada, 47 butterflies are listed as federally threatened or endangered under the US Endangered Species Act (ESA) or Canada's Species at Risk Act (SARA). Another three butterflies are listed as extirpated under SARA and at least seven species

Monarchs in the West have experienced dramatic population declines since the 1980s.

are thought to be extirpated in the US. Perhaps most alarming is that declines are being seen not just in rare species but also in once-common and widespread species such as the monarch butterfly. In North America, the eastern monarch population, which overwinters in Mexico, has declined by more than 80% since the 1990s (Semmens et al. 2016). Similarly, data from the Xerces Society's Western Monarch Thanksgiving Count and comparable historical data show a population decline of 74% since the 1990s (Pelton et al. 2016) and over 95% since the 1980s, with a high risk of quasi-extinction (Schultz et al. 2017). In the 1980s, millions of monarchs overwintered in California annually (Schultz et al. 2017); in 2017, fewer than 200,000 monarchs were observed (WMTC 2018). Declines have also been documented in monarchs during the spring and summer in California (Espeset et al. 2016). Increasingly, researchers are documenting regional declines of many butterfly species. A long-term monitoring program in northern California has revealed declines in numerous butterfly species at sites in the Sacramento Valley, Coast Ranges, and Sierra Nevada (Forister et al. 2011; Casner et al. 2014).

Data for other invertebrate pollinator groups are scant, as the majority of these species lack coordinated monitoring programs and many are still relatively understudied and/or are in need of taxonomic clarification. Evidence of decline has been documented for several vertebrate pollinator species as well, including the lesser long-nosed bat (*Leptonycteris yerbabuenae)*, rufous hummingbird *(Selasphorus rufus*), and Allen's hummingbird (S. sasin; reviewed in National Research Council 2007).

In the West, numerous pollinators are now designated by the US Forest Service or Bureau of Land Management as sensitive species or Species of Conservation Concern, and state agencies have designated some pollinator species as Species of Greatest Conservation Need. Twenty species found in the West are currently listed as threatened or endangered under the federal Endangered Species Act. In California alone, 75 species of bees, butterflies, and moths are considered to be at-risk. See Appendix **B f**or a list of at-risk pollinator species by state and US Forest Service region, and Appendix **C** for a list of at-risk butterfly and moth species and their host plants.

The underlying reasons for most pollinator declines are still under investigation, and the drivers can be difficult to pinpoint given the sheer number of species involved, general lack of baseline data, and limited taxonomic and geographic coverage of available data (Vanbergen and Insect Pollinators Initiative 2013; Gill et al. 2016). Compounding these issues is the fact that there are many potential drivers of pollinator declines, and these rarely act in isolation. Increasingly, studies are showing that interactive, non-additive effects are leading to ongoing declines in wild and managed pollinators.

Primary drivers implicated for pollinator decline include habitat loss (e.g., from development or wildfire) and degradation (e.g., due to improper or heavy grazing), pesticide use, introduction of managed pollinators, invasive species, pathogens and parasites, climate change, land use change, and the interplay of these threats (Kearns et al. 1998; National Research Council 2007; Brown et al. 2016; Gill et al. 2016; Everaars et al. 2018; Kopit and Pitts-Singer 2018; Vanbergen et al. 2018). Pollinators of western rangelands face all of the above threats, but many of them can be minimized through changes in management—which this document aims to help land managers address—and policy.

2

Best Management Practices for Pollinators on Western Rangelands

The best management practices (BMPs) below provide a brief summary of known effects on pollinators from each management practices, followed by recommendations on how to incorporate the needs of pollinators into management decisions. The intent of this document is not to suggest that rangelands should be managed exclusively for pollinators, but instead is provided to help land managers incorporate pollinators into considerations of other goals of rangeland management such as biodiversity, livestock productivity, native plant communities, sensitive species, and water-quality. Adopting these BMPs will help support a diverse and abundant pollinator community, as well as benefit other wildlife species, and, ultimately, the livestock and humans that depend on healthy rangelands.

What is Good Pollinator Habitat?

Good pollinator habitat provides

1) Provides food, in the form of nectar, pollen, and host plants;
2) Offers shelter and nest sites; and
3) Is safe from pesticides and high levels of pathogens.

At the landscape–level, pollinators also need this habitat to be connected to other habitat at distances they can disperse to.

Food Sources

Flowering plants provide nectar and pollen, the primary food source for most bees and butterflies. Adult bees and butterflies require nectar for energy; most bee larvae require pollen. In addition, butterflies and moths require larval hosts (plants their caterpillars eat) such as forbs, grasses, shrubs, or trees.

Providing a diverse, abundant, and season-long supply of food sources is an important component of good pollinator habitat.

- Aim to have a minimum of three different species blooming at any point during the growing season to provide nectar and pollen.

Big Summit Prairie on the Ochoco National Forest in Oregon in full bloom.

This is only a general consideration—the more species the better! Aim for a wide range of flower structure, shape, color, and size as certain flowers are more attractive to some pollinator species than others (e.g, long, tubular flowers are often more attractive to butterflies than bees). Early- and late- season flowering resources can be especially important for bumble bees, which are often active in the "shoulder seasons," as well as migrating monarch butterflies (late August through November).

- Provide a diversity of host plant food sources: native wildflowers, succulents, thistles, perennial bunch grasses, sedges, trees such as oaks, willows, and wild cherries, and shrubs such as rabbitbrush and coyote brush.

Globe mallow (*Spaeralcea*), is a genera that supports many specialist and generalist bees.

Shelter and Nest Sites

At some point in a pollinator's life, it will need to take shelter to survive a storm, form a chrysalis, build a nest, and/or overwinter. Some bees and butterflies crawl down into the bases of bunch grasses; others seek shelter under tree leaves, rock crevices, litter, woody material, or abandoned rodent nests.

- Leave some woody, hollow, or pithy-stemmed vegetation and ground litter intact and in-place permanently. These materials can often be used by native bees, syrphid flies, and soldier beetles to overwinter.
- Maintain or leave some undisturbed naturally occurring bare ground characteristic for a given habitat type, or provide some bare ground in restoration. Naturally occurring bare ground and nest sites can include abandoned rodent nests, naturally occurring microtopography such as grass tussocks. Many native bees nest below-ground and require bare ground or existing cavities to nest in.
- Avoid mowing, burning, or grazing an entire area down to the ground. Overwintering pollinators, even adults, are generally immobile at low temperatures and unable to escape blades, flames, or livestock.

Pesticides and Disease

Pollinators need habitat that is protected from pesticides and high levels of disease or pathogens. See "Pesticides" section on page **70** to learn more about the importance of creating and maintaining habitat that is safe for pollinators. Pollinators also need habitat that is safe from high levels of pathogens, such as those that can spread from managed to native pollinators. See the "Managed Pollinators" section on page **78** to learn more about disease spread from managed to native pollinators.

Connectivity

As is the case with most wildlife, habitat which is closer and more connected to other habitat patches is better for pollinators. This minimizes the flying distance pollinators must cover to find food, build nests, and find mates. However, the impact of connectivity greatly varies by pollinator species: monarch butterflies and painted ladies routinely travel hundreds or even thousands of miles in their lifetime, while smaller-bodied butterflies and some bees can be restricted to just a few hundred feet from where they hatched.

- Larger, continuous patches of habitat are typically more valuable for wildlife than smaller, more fragmented ones.
- Provide corridors, even narrow ones, between habitat patches wherever possible. Pollinator habitat along roadsides, fencerows, or utility lines can offer important linear connectivity in fragmented landscapes.
    - Refer to the Federal Highway Administration handbook (https://bit.ly/2y90bU9), *Roadside Best Management Practices that Benefit Pollinators* (developed by the Xerces Society and ICF International), for more guidance on managing roadsides (Hopwood et al. 2015).

Connectivity is of particular importance for less mobile, at-risk species. For example, the Fender's blue butterfly's (*Icaricia icarioides fenderi)* is a specialist that feeds almost exclusively on the federally threatened Kincaid's lupine (*Lupinus oreganus*) – both species are near endemics to upland prairies of the Willamette Valley of Oregon. This habitat is now so fragmented and rare that throughout most of its range, it is unlikely that a butterfly could locate a new host plant patch even if it left its natal patch (Schultz 1998).

General Considerations in Management and Restoration Projects

There are thousands of native pollinator species in the West, each with its own unique phenology, range, life history strategy, and floral and nest habitat requirements. Many species, including some bumble bees, have broad geographic ranges with varying phenologies— emerging as early as January and remaining active until as late as December —and they visit a variety of flowering plants (Hatfield et al. 2012). A few species of bees are extreme specialists with narrow geographic ranges, diet breadth, or phenologies that are timed with the emergence of a single plant species they visit exclusively (Minckley et al. 2013, Wilson and Messinger Carril 2015). In addition, native bee and butterfly communities and their phenology can vary widely across the landscape, such that sites even within a few miles of one another can be quite distinct (Fleishman et al. 1999; McIver and Macke 2014; Kimoto et al. 2012a, 2012b; DeBano et al. 2016). This combination of dizzying diversity and limited information in many areas, makes it impossible to prescribe a single management plan that is ideal for all pollinators in all places. Management for pollinators must be implemented on a site-specific basis, with varying strategies which focuses on maintaining healthy functioning ecosystems.

However, there are certainly general considerations that will benefit pollinators in many situations. Habitat management tools—grazing, fire, mowing, and herbicide applications—can be used to benefit pollinators and their habitat, but can also cause damage, especially in the short-term. To minimize potential damage, the following recommendations apply to most management.

Create heterogeneity in the plant community and provide refuge for pollinators.
In general, diversity in vegetation, structure, and management practices can maximize biodiversity,

TABLE 1: Pollinator Flower Preferences, Food Resources, and Shelter Needs.

	Bats	Hummingbirds	Flies	Beetles	Bees	Moths	Butterflies
Flower Color	Dull white, green or purple	Scarlet, orange, red or white	Pale and dull to dark brown or purple; flecked with translucent patches	Dull white or green	Bright white, yellow, blue, or UV	Pale and dull red, purple, pink or white	Bright, including red and purple
Flower Shape	Regular; bowl shaped – closed during day	Large funnel like; cups, strong perch support	Shallow; funnel like or complex and trap-like	Large bowl-shaped flowers	Shallow; have landing platform; tubular	Regular; tubular without a lip	Narrow tube with spur; wide landing pad
Food	Nectar, pollen, fruit	Nectar, insects, tree sap	Nectar, pollen, and insects (aphids, mites, thrips)	Pollen, nectar for adults, vegetation and insect prey for larvae	Nectar for adults, nectar, pollen to provision larvae	Larval host plants, nectar, minerals in soil (mud puddles), tree sap, animal dung and urine, salts from rotting fruit	
Shelter/ Nest	Caves, tree snags, mines	Trees, shrubs, and vines as nesting, perching sites	Soil or leaf litter, plants for larvae	Loose soil, leaf litter for overwintering larvae; rocks, brush, and logs as shelter for adults	Nest sites and material: cavities, rodent or insect burrows, pithy stemmed plants, bare ground, dead standing trees, mud, plant resins and oils	Host plants, protected sites for roosting/perching, habitat undisturbed for overwintering larvae, adults, and chrysalis	
Specific Habitat Goals	Maintain large columnar cacti	Maintain nectar sources of funnel shaped flowers	Diverse plant communities	Diverse plant communities with areas of leaf litter, brush, rocks, and logs	Diverse plant communities with a diversity of flower colors, shapes, sizes that flower throughout the active flight period from frost to frost. living and dead plants with pithy stems, dead wood, standing snags, and native bunchgrasses provide nest sites.	Diverse plant communities with host and nectar plants, perching sites, water	
General Habitat Goals	Areas of refugia from burning, grazing, mowing, or other disturbances to habitat.						

including the diversity of pollinators (Gilgert and Vaughan 2011). Mowing, burning, or intensively grazing an entire habitat area at once or in the same year, for example, can severely impact local pollinator populations and slow recolonization. Historically, rangeland landscapes contained sufficient areas where vegetation was in various stages of succession to support a wide range of pollinators with differing habitat needs. Today, some rangeland habitat is often reduced to fragments in intensively managed or disturbed landscapes, and managers have to consider the distance and connection potential between pollinator populations (USFS 2012).

For these reasons it is better to treat separate areas of a site in a multi-year cycle, retaining undisturbed refuges from which pollinators can disperse. A general consideration is to treat no more than one-third of continuous habitat or site (e.g., a meadow, riparian area) with a single management action (such as prescribed burning) in a single year. Even within treatment areas, leaving small untreated patches (e.g., areas skipped by mowing, fire, or grazing) provides micro-refuges and greater heterogeneity in the landscape, which can support a wider range of pollinators. When achieving the one-third goal is not feasible—such as in areas with season-long grazing try to keep at least some area free from disturbance. For example, exclude, limit, or carefully time when livestock utilize highly sensitive areas such as riparian zones or spring-fed ecosystems which are often biodiversity hotspots for pollinators in rangelands.

Consider how management interacts with natural stressors to affect pollinators.
For example, if a drought severely suppresses wildflower blooms one season, grazing it heavily may further stress pollinators' ability to find sufficient nectar and pollen. To help minimize the effects of interacting stressors, you may need to adjust grazing pressure in years of drought. Focus efforts on conserving existing habitat which is of high value to pollinators and strive to establish plant communities which are both resilient and resistant to disturbance. The US Forest Service fact sheet Putting Resilience and Resistance Concepts into Practice (Chambers et al 2015) defines *resilience* as "the capacity of an ecosystem to regain its fundamental structure, processes and functioning when altered by stresses and disturbances" and *resistance* as "the capacity of an ecosystem to retain its fundamental structure, processes, and functioning ... despite stressors, disturbances, or invasive species."

Use an adaptive management framework.
The response of pollinators to livestock grazing and many other management practices in the West has been largely unstudied, and more research is needed to further refine rangeland management for pollinators. Given imperfect and incomplete knowledge, adaptive management using the best currently available science is necessary. Adaptive management is an iterative process of decision-making which acknowledges uncertainties and encourages balancing short-term benefits with learning about the system in order to improve management in the long-term. Experiment on small areas, keep records, and share what works and what fails with others. Monitor vegetation and, when possible, the pollinators themselves to see how they respond to management. (See the "Monitoring" section on page **88**.)


Monitoring vegetation and pollinators to see how they respond to management can help inform strategies for adaptive management.

Regularly revisit your management plan and adapt it as needed based on what you learn and to address future, changing conditions.

Time management to minimize negative impacts on pollinators.
Figure **2** offers broad information on when native bees are less likely to be affected by management such as burning, grazing, or mowing. Figure **3** offers recommended management windows for monarch butterfly breeding habitat by ecoregion. However, note that above-ground nesting bees (including some bumble bees) may be sensitive to management year-round.

FIGURE 2: Recommended Management Timing for Native Bees in Western North America.
(*Based on above- and below-ground nesting bee activity by genera and bumble bee activity by species*)

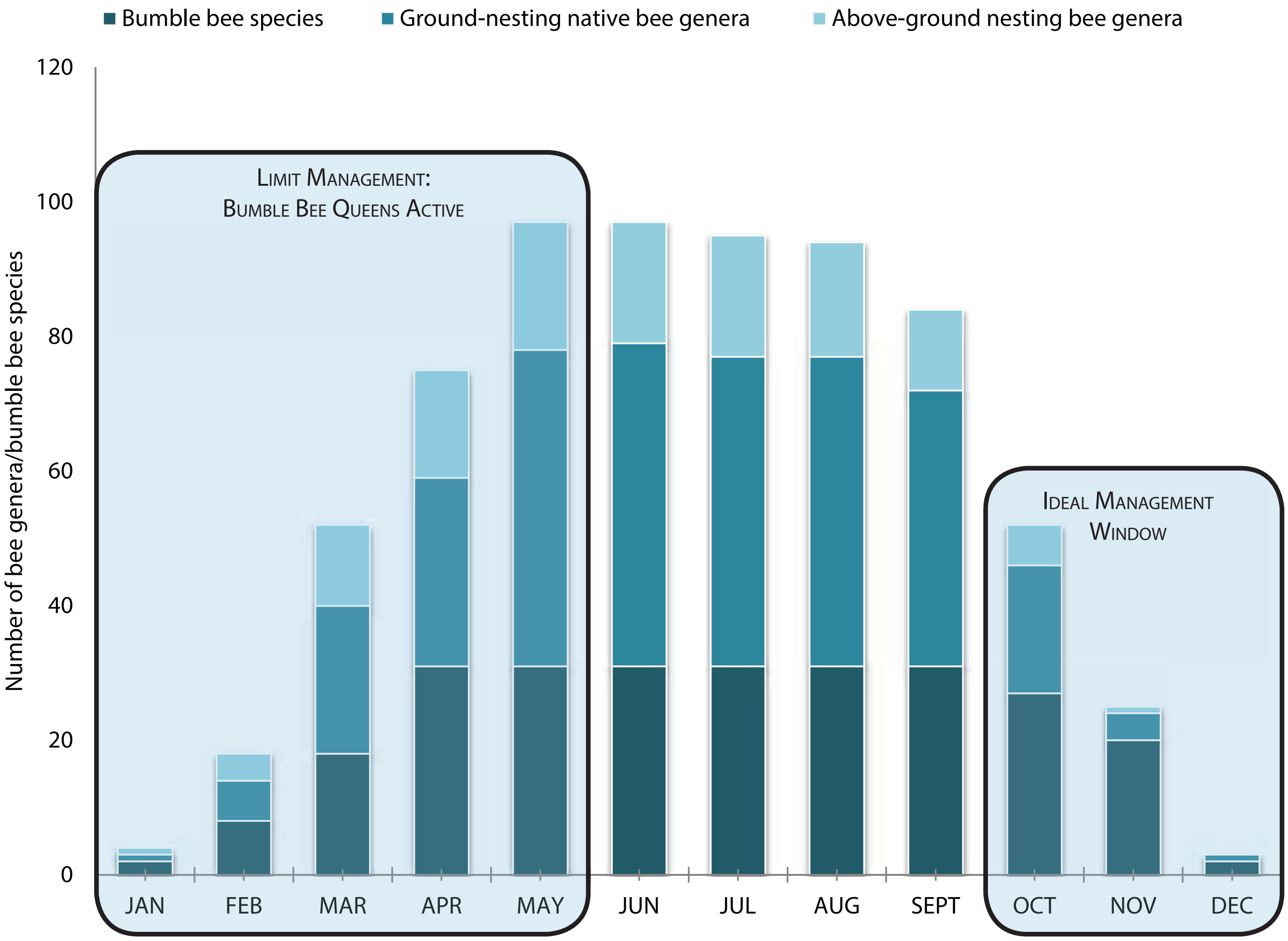
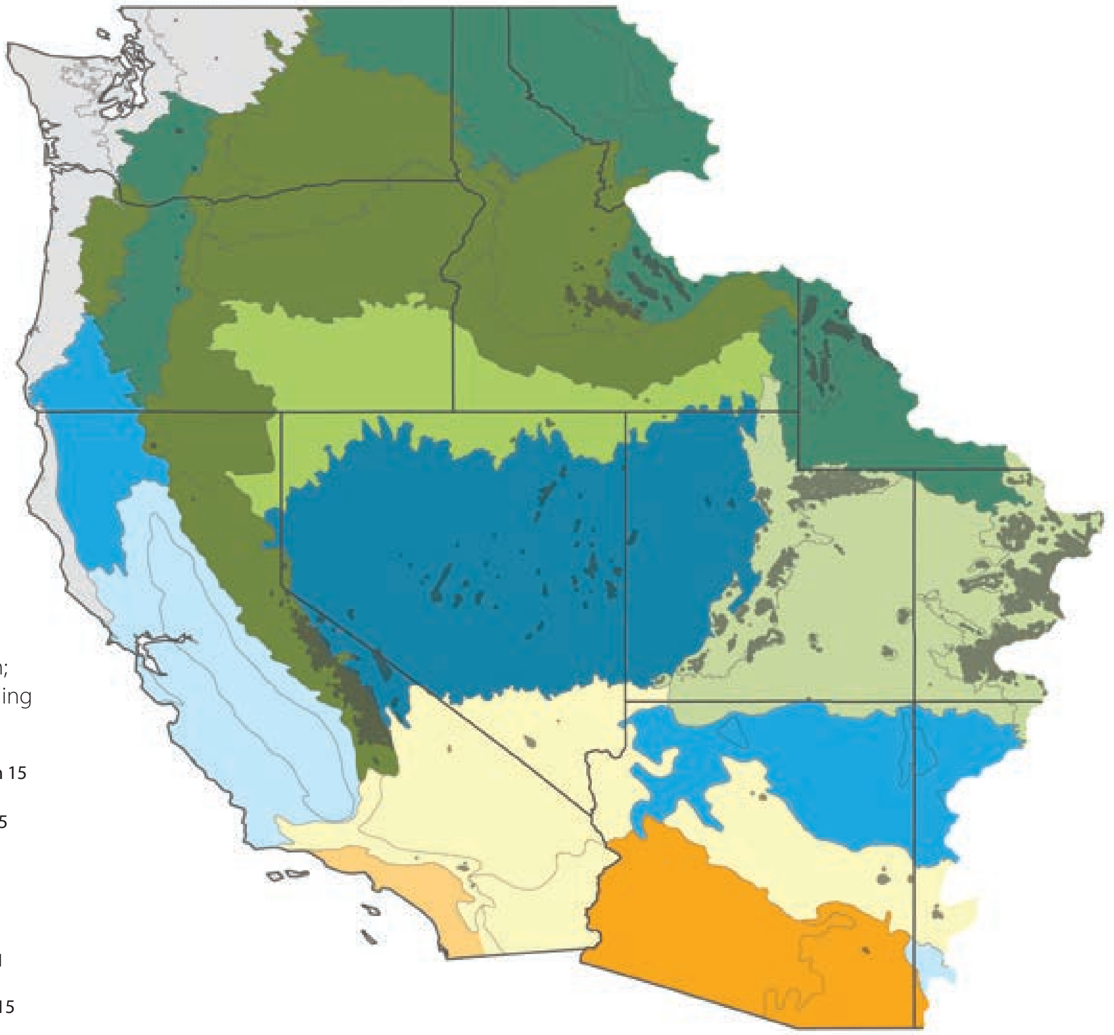


FIGURE 3: Recommended Management Timing for Monarch Breeding Habitat.



Date range to manage within; monarchs typically not breeding during this time period

🟨	November 30 – March 15
🟦	October 31 – March 15
🟦	October 31 – April 1
🟦	October 31 – May 1
🟩	September 30 – May 1
🟩	September 30 – May 15
🟩	September 30 – June 1
🟩	August 31 – June 1
🟧	Potential year-round breeding [November 30 – March 15]
🟧	Potential year-round breeding [Summer: June 20 – August 10, Winter: November 30 – March 15]
⬜	No breeding/milkweed
━	Above 9,000 feet (no breeding)
☐	EPA Level III Ecoregions

Options listed in [ ] are recommended only if necessary. These summer and winter management windows may still cause some mortality.

Data source: EPA Level III Ecoregions, Western Milkweed Mapper, Journey North, Southwset Monarch Study, Department of Defense Legacy Fund Research, Dingle et al. 2005.

A Few Notes

- Milkweed (and therefore monarch breeding) does not typically occur above 9,000 feet.
- In southern California, monarchs are known to breed year-round on tropical milkweed (*Asclepias curassavica*), a nonnative species commonly planted in gardens.
- In southern Arizona, monarch have been documented breeding year-round on native evergreen milkweed species such as rush milkweed (*A. subulata*).
- For more details on monarch habitat, see **Box 6**: Where Should Monarch Habitat Be Restored?

Grazing

"…managing rangeland for pollinators provides a useful framework for overall biodiversity conservation, restoration, and management for public and private land managers alike." (Gilgert & Vaughan 2011)

Grazing on US Forest Service Lands

Livestock grazing by cattle, sheep, and horses has been widespread in the West since the mid-1800s, and there are now millions of head of cattle and sheep grazing public lands. In addition, many rangelands are also grazed by populations of native ungulates such as elk and deer, as well as feral and wild horses and burros.

Grazing on US Forest Service land occurs under 10-year term grazing permits that specify the allotment, stocking rate, and timing of use. Grazing allotment management plans are developed for each permit and are based on a National Environmental Policy Act (NEPA) review process (generally renewed every 10 years). Management plans specify management objectives that are in line with a forest plan. Operating procedures for each management plan are reviewed annually, with adaptive guidelines on livestock rotations and timing. The National Recisions Act of 1995 guides the national NEPA review process for grazing allotments and the current schedule covers 2017-2028.

Stocking rates are defined as Animal Unit Months (AUMs): the amount of monthly forage required by a cow-calf pair which consists of one mature cow approximately 1,000 pounds and a calf six months or younger. Stocking rates are assigned by the US Forest Service at the allotment scale and vary within and among regions, based on rangeland health and specific management objectives outlined in the forest plan. Of the 9 million livestock AUMs permitted annually for public rangelands in the West, 5.7 million of those AUMs are authorized to graze on land managed by the US Forest Service (USFS 2015a).

Rangeland sagebrush steppe in southeastern Oregon.

Table 2: Summary of Authorized Livestock AUMs on US Forest Service Land in Western States.

State	Total Authorized Livestock AUMs on USFS System Land	Total AUMs on National Forests	Total AUMs on National Grasslands	Total Number of USFS Acres
Arizona	828,945	828,945	None	11,264,377
Colorado	772,423	628,803	143,620	14,519,030
New Mexico	667,114	644,017	23,135	9,413,211
Idaho	631,604	605,385	26,219	20,466,617
Montana	600,710	600,710	None	16,962,737
Utah	590,714	590,714	None	8,200,161
Wyoming	536,437	330,443	205,994	9,241,187
Oregon	451,732	437,492	14,240	15,667,657
California	303,193	301,234	2,005	20,802,641
Nevada	230,821	230,821	None	5,853,963
Washington	78,894	78,894	None	9,282,376
Total	**5,692,587**	**5,277,458**	**415,213**	**141,673,957**

Effects of Grazing on Pollinators and Their Habitat

If carefully managed, grazing can be a useful management tool for maintaining the open early seral landscapes important for many butterflies and other pollinators (Kobernus 2011). Well-managed or targeted grazing can improve habitat for pollinators by maintaining heterogenous and open herbaceous forb-dominated plant communities, allowing growth of spring and summer flowering plants (Murphy and Weiss 1988; Elligsen et al. 1997; Smallidge and Leopold 1997; Weiss 1999; WallisDeVries and Raemakers 2001; Pöyry et al. 2004, 2005; Saarinen et al. 2005; Nilsson et al. 2008; Konvicka et al. 2008; Potts et al. 2009; Kobernus 2011; Vanbergen et al. 2014), and in some special circumstances, targeted grazing can suppress noxious or invasive plants (Olson 1999; Weiss 1999; Schmelzer et al. 2014).

If managed inappropriately, grazing can severely degrade ecosystems (Bilotta et al. 2007) by substantially altering the structure, diversity, and growth habits of a plant community and the associated insect community (Debano 2006a, 2006b; Kruess and Tscharntke 2002a; Zhu et al. 2012). Livestock grazing can alter plant communities by reducing biomass, selecting for or against plant species, changing the plant community structure (physical and species composition), and by affecting the reproductive capacity of plants (e.g., seed production, dispersal). Grazing systems that remove a high level of forage, and have livestock in a given pasture for extended periods of time and do not provide long rest periods can cause plant community shifts towards invasive plants, that are both less palatable to ungulates and less suitable habitat for native pollinators (Vavra et al. 2007; Knight et al. 2009; Kobernus 2011; Hanula et al. 2016). Grazing can also alter hydrology and soils, increasing bare ground, erosion, and compaction

The impact of cattle grazing on pollinators and their habitats vary by many factors including stocking rate and timing.

(DeBano 2006b; Schmalz et al. 2013). The effects of grazing on native plant and pollinator communities are compounded by environmental stressors such as drought, climate change, large-scale wildfires, and invasive species.

Generally, as the intensity of livestock grazing increases, pollinators, including butterflies, moths, and other insects, decline in abundance and/or diversity (Morris 1967; Hutchinson and King 1980; Sugden 1985; Dana 1997; Balmer and Erhardt 2000; Cagnolo et al. 2002; Carvell 2002; Kruess and Tscharntke 2002a, 2002b; Vulliamy et al. 2006; Pöyry et al. 2006; Kuussaari et al. 2007; Sjödin 2007; Yoshihara et al. 2008; Littlewood 2008; Börschig et al. 2013; Jerrentrup et al. 2014; Elwell et al. 2016; van Klink et al. 2016). Low intensity grazing is generally defined as utilization that allows palatable species to reproduce, and moderate intensity grazing is generally defined as utilization that maintains palatable species, but limits their reproduction (Holechek et al. 1999). Low to moderate grazing intensities have less severe, but often still negative effects on native pollinators (Carvell 2002; Hatfield and Lebuhn 2007; Sjödin 2007; Xie et al. 2008; Kearns and Oliveras 2009; Kimoto 2011; Roulston and Goodell 2011; Kimoto et al. 2012b; Minckley 2014). Grazing can also cause shifts in pollinator and insect communities (Cagnolo et al. 2002; Yoshihara et al. 2008; Kimoto 2011). Moreover, grazing can cause direct mortality to pollinators through destruction of bee nest sites or trampling of immobile stages of butterflies and moths (Sugden 1985; Kearns and Inouye 1997; Gess and Gess 1999; Bonte 2005).

The majority of grazing on public lands is of moderate intensity, in some kind of rotational pattern, but frequently there is grazing pressure during some or all of the most active season for most pollinators (generally May to September). This can be problematic for pollinators because it tends to homogenize the landscape, may not allow vegetation and pollinators refuge or sufficient time to recover, and reduces the availability of floral resources for pollinators. Season-long summer grazing is especially problematic as it occurs at the same time every season, which eventually limits plants' ability to set seed, and limits the recovery of plant communities from disturbance. More pollinator-friendly grazing regimes reduce grazing pressure on pollinator resources by reducing stocking rates or, if feasible, changing the season of grazing to fall or winter. Examples of grazing schemes that may achieve this include high-density short-duration (HDSD), or rotational grazing such as the Santa Rita grazing schemes used in the desert Southwest (Howery 2016). For example, in one study, short-term spring cattle grazing of less than 1 AUM/ha did not significantly impact flowering plant or pollinator abundance, richness, or diversity in sagebrush steppe habitat (Elwell et al. 2016). Other examples of grazing schemes that may benefit pollinators include high-density short-duration (HDSD), or rotational grazing such as the Santa Rita grazing schemes used in the desert Southwest (Howery 2016). Generally grazing management plans should maintain forb diversity and abundance for pollinators from frost to frost.

Pollinators often exhibit species-specific responses to grazing depending upon their diet, as well as foraging and nesting behavior and requirements (Roulston and Goodell 2011). Generally, specialist bees and bumble bees are more sensitive than other generalist native bees to changes in the plant community and grazing (Hatfield and Lebuhn 2007; Yoshihara et al. 2008; Kimoto 2011; Kimoto et al. 2012a, 2012b). Pollinators are especially sensitive to grazing during times of scarce floral resources (early and late in the season), which can result in insufficient forage for pollinators (Carvell 2002; Hatfield and Lebuhn 2007; Xie et al. 2008).

Different livestock species may affect native plant communities and pollinators in different ways. The botanical composition of livestock browse varies with the availability or proportion of forbs to grass available in a rangeland. Both sheep and cattle can consume between 55–80% forbs in a forb-dominated rangeland (Bryant et al. 1979; Pieper and Beck 1980; Ralphs and Pfister 1992). Generally though, sheep tend to prefer forbs and often graze in concentrated herds which are more likely deplete pollinator resources compared to cattle; cattle tend to prefer grasses and graze in a more dispersed fashion. Bumble bees, for example, appear to be more sensitive to sheep grazing than cattle grazing during the spring and summer in forb-dominated habitats, as sheep tend to prefer foraging on flowering plants (Hatfield and Lebuhn 2007). Native ungulates (deer, elk), cattle, and pollinators can also have overlapping and competing foraging preferences (DeBano et al. 2016). As such, their presence on the landscape may need to be considered in a grazing management plan where large herds of native ungulates occur.

Managing livestock for pollinators can also be consistent with managing for other wildlife species such as greater sage-grouse and other gallinaceous birds, anadromous fish, trout, upland birds, and upland game. In addition, grazing management for pollinators (low AUMs, HDSD) is also compatible with livestock production on rangelands, as low stocking rates reduce the chance of poisoning from toxic plants.

Some pollinator plants are considered toxic to livestock, but with careful management, these pollinator plants can remain in a rangeland to benefit pollinators. See Box **1 f**or information on how toxic milkweeds (the larval host of the monarch butterfly) and livestock can coexist on rangelands.

Grazing Best Management Practices

Grazing Management Plan Objectives for Pollinators

When developing grazing management plans, include pollinator resources as management objectives, with a goal of maintaining a minimum of three (and ideally, more) flowering plant species in an allotment throughout the season. This is especially important if a grazing allotment has at-risk pollinators, sensitive habitats important for pollinators, or areas of high pollinator abundance or diversity; see sensitive plant and bee genera lists in Appendix **B**. However, it is important to note that pollinators that overwinter above ground will always be sensitive to grazing. Therefore, one single grazing management system will never suit all pollinators, site conditions, or management objectives of an allotment.

The following sections include specific recommendations on how to manage grazing impacts on pollinators.

Intensity and Duration

- High-density short-duration (HDSD), low AUMs, and/or rest-rotation are recommended for maintaining habitat for pollinators.
- Use low intensity (low AUMs for site or allotment) for season-long grazing or rotations that exceed 45 days in any single pasture.
- In general, keep grazing periods short, with recovery periods for at least one-third of the habitat area relatively long (e.g., months to years depending on the habitat type).

Box 1: Milkweed Toxicity: How Livestock and Milkweed can Coexist on Rangelands

Many plants are classified as toxic to livestock. Their common name frequently includes the term "weed," and they are lumped together as being equally problematic. Milkweeds are one example of this, plants that are important for wildlife but because they contain chemical compounds called cardenolides, which are toxic to many animals, they are disliked and removed. Cardenolide levels, however, vary by milkweed species and local conditions, causing plants to range from relatively nontoxic to very toxic to livestock, including sheep, cattle, horses, goats, turkeys, and chickens (FDA Poisonous Plant Database; Panter et al. 2011). These plants play an important role in the ecosystem, providing nectar for butterflies and bees and supporting a wide range of specialist and generalist beetles, true bugs, flies, and aphids—and only become a "weed" when livestock is present. A large percentage of milkweed species native to North America have also been documented as host plants of the monarch butterfly, which the caterpillars need to complete their life cycle.

In the last thirty years, the monarch population that overwinters along the California coast and migrates through the west has declined by over 95% (Schultz et al. 2017). Part of this decline may be attributable to the loss of milkweed and nectar plants due to herbicide use, urban and agricultural development, and long-term drought (linked to climate change) over much of the monarch's breeding and migratory range. Thus, the conservation of milkweed in rangelands of the West will ultimately contribute to monarch and pollinator conservation.

While there have been instances of livestock poisoning from milkweed, the record is sparse and mostly associated with hungry animals being released into milkweed patches (Fleming 1920) or confined to an area without sufficient alternate forage. Milkweed plants are toxic to livestock year-round during all growth stages, but can be of particular concern when dried—such as in hay—because palatability to livestock increases (Fleming 1920; DiTomaso and Healy 2007; Schultz 2003). Although toxicity varies, all milkweed plants should be considered toxic to livestock (Malcolm 1991; Agrawal et al. 2015). However, two species, western whorled milkweed (*Asclepias subverticillata*) and narrowleaf milkweed (*A. fascicularis*), have been reported as especially problematic species for cattle and sheep, likely because of their growth

While milkweed can be toxic to livestock in some situations, poisonings are rare and can be prevented.

Narrowleaf milkweed growing along a fence line on rangeland in Nevada.

forms. Their thin stems and leaves are easily tangled in grasses and difficult for grazing animals to separate out.

Livestock graze in areas with milkweed all over North America and there are anecdotal reports of cattle and sheep eating milkweed even when other forage is available (Stephanie McKnight, personal observation). Despite this, poisoning events are rare, possibly because livestock must consume a large amount of milkweed to become sick or die. A cow weighing roughly 1,200 lbs will need to eat 12 lbs or more (or 1–2% of their body weight) of dried milkweed on average to die of poisoning (Kingsbury 1964; Burrows and Tyrl 2007). In a recent survey of forty-three land managers and ranchers (see Report Development on page **2**), poisoning events from milkweed were not reported as a major concern, and no one reported first-hand knowledge of a poisoning event.

Given this, conserving milkweed is compatible with livestock grazing, if you take some basic precautions:

- Maintain an appropriate stocking rate and ensure livestock have sufficient forage.
- Closely monitor animals that are new to an area where milkweed occurs.
- Keep livestock driveways and small paddocks free from milkweed because confined animals may be more likely to eat it.
- Avoid planting western whorled milkweed and narrowleaf milkweed in grazing allotments.
- Keep fields that will be used for hay free from milkweed.

- Stocking rates should be appropriate for the characteristics of a site, livestock species, and management objectives.
- Ensure rangelands are provided sufficient rest times to allow native vegetation to meet management objectives. Rest in some circumstances, such as in heavily grazed sagebrush rangelands, may not achieve management objectives or improve habitat alone (Davies et al. 2014b) and other actions may need to be considered.

Utilization

- Managers should aim for utilization rates up to but not exceeding 40% of the current season's growth to reduce impacts to pollinators (Kimoto et al. 2012b) and to the native forb component of plant communities. Forty percent is a rule-of-thumb, but land managers should work closely with local wildlife biologists and botanists to determine site-specific percent utilization and stubble heights that will maintain forb diversity and abundance for pollinators from frost to frost.
    - In the sagebrush biome, land managers can use the current grazing utilization rates or stubble height recommendations for their region for greater sage-grouse conservation as a best management practice for pollinators (USFS 2015b).
- Utilization rates should be determined on an annual basis for mesic meadows, springs, riparian areas, and in times of drought because drought, grazing history, and native ungulate use all affect utilization rates. Utilization rates in these habitat types should generally be less than the surrounding xeric landscape (<40%).

Timing

- Fall and winter grazing have the least impact on pollinators because most plants and pollinators are least active in November and December; however, soils must be able to withstand late-season or winter grazing. If feasible, adjust grazing time to fall or winter when most flowering plants are dormant and pollinators are least active (see Figure **2**).
- Avoid grazing the same location at the same time every year (e.g., alternate the timing of grazing within an allotment).
- Sheep grazing should occur in the fall and winter after flowering plants have senesced. If sheep grazing must occur during peak pollinator activity (May–September), the sheep should be introduced at low stocking rates and continuously moved to avoid depleting floral resources in any single location.
- In arid regions such as the Desert Southwest, there may be a large flush of annual flowers after a high precipitation event or flood. Similar concentrated bloom events can occur after spring snowmelt in high elevation meadows. In both habitats, adjusting the timing of grazing can ensure ephemeral flowering plants have time to set seed and pollinators can use the plants as nectar resources if they are present in the area.

Livestock Movement

- Aim to maintain even grazing utilization across an allotment to prevent concentrated hoof damage and utilization. Excessive hoof damage to soil may cause direct mortality to ground-nesting bees, or change the topography of soil in mesic habitats. Ground-nesting bees may nest alone, or in aggregations of hundreds or thousands of nests. These nesting areas range in size from less than an inch to several hundred square meters.
    - Some areas within allotments may not get grazed by livestock in a given year due to

geographical barriers or how livestock move across the landscape. These ungrazed areas may be left annually as refugia for pollinators.

- Establish exclosures or moveable fencing so that livestock can be rotated through grazing allotments to allow recovery of the vegetation community and keep livestock out of any overutilized or sensitive areas. If fencing is not an option, then geography, water structures, salt blocks, or nutritional supplements might be useful in keeping livestock in a desired area (Stephenson et al. 2017). The placement of supplements in combination with low-stress herding shows promise for keeping livestock in desired areas, and away from areas deemed important for pollinators, sensitive wildlife, or plants.
- Consider implementing a rotational grazing scheme for allotments. In public land management allotments where continuous season-long grazing is the norm, rotational grazing is possible with some ingenuity, including close collaboration with grazing permittees. Rotational grazing could be achieved by using natural barriers (topography that limits livestock movement), herders, water, or fencing to keep livestock in desired areas and out of an area designated to be rested or excluded from livestock for the year.
    - In a rotational grazing scheme, the area excluded from grazing would change every year to maintain habitat heterogeneity, avoid overutilization of any given area, and to maintain floral resources for pollinators (Scohier et al. 2012).
- Sheep should be herded regularly and through different routes each year with a 3- to 5-year rotation of routes used. Sheep should not be allowed to graze one location longer than one to two days, and floral resources should be closely monitored to avoid depleting an area of flowering plants during peak bee abundance (May–September). See Box **2** for a Case Study on the effects of summer sheep grazing on bumble bees in montane meadows.

CASE STUDY

Sheep tend to graze preferentially on forbs in montane meadows, which can deplete floral resources and impact bumble bee communities in these habitats.

Box 2: Summer Sheep Grazing in Montane Meadows is Detrimental to Bumble Bees

Sheep are generalist grazers that tend to prefer forbs. This means they are more likely to deplete floral resources compared to cattle, which may lead to greater impacts on bumble bees, because they are especially sensitive to reduction in floral resources. Hatfield and LeBuhn (2007) looked at summer grazing of both cattle and sheep and found that cattle grazing had no detectable effect on bumble bees, whereas sheep grazing was severely detrimental to the bumble bee community—eliminating bumble bees from study sites. The following year, however, bumble bees were able to recolonize the grazed areas from adjacent intact and ungrazed refugia meadows (Hatfield and Lebuhn 2007). The degree to which these grazed habitats were serving as long-term sink populations for bumble bees was not assessed.

While this is only one study, the results suggest that summer sheep grazing can have severe negative effects on bumble bee populations. Land managers should implement summer sheep grazing in montane meadows with caution, and carefully monitor the effects of grazing on both bumble bees and floral resources.

Grazing exclosure showing grazed and ungrazed vegetation in the Great Basin during spring.

Adaptive Management and Flexible Grazing Management Plans

Flexible adaptive management is key to maintain long-term forage for grazing animals and habitat for wildlife, including pollinators. Grazing management plans should be site specific and flexible in order to adapt grazing stocking rates, timing, and duration to changing environmental conditions, which include but are not limited to a depletion of pollinator resources (flowering or nesting plants), overutilization, drought, fire, and invasive species. Flexible management plans should allow adjustments of stocking rates and timing to prevent depletion of important floral resources for target pollinators. This will vary annually as well as by region, elevation, habitat type, and season. The following are some special circumstances that will require adaptive management.

Grazing Post-Fire

- Allow a minimum of 2–3 years of rest after a fire before grazing again in order for the plant community to recover, especially if the area was seeded with native plants. This interval will vary depending on site conditions and rate of plant recovery and establishment.
- Perennial grasses need to resume reproduction, and the cover of perennial and annual flowering plants, biological soil crusts, and accumulation of litter need to be sufficient to stabilize soils (Veblen et al. 2015).
- Carefully managed winter grazing post-fire can be a useful tool to suppress some invasive annual grasses (Davies et al. 2016).

Overutilization

After unplanned high utilization occurs (in excess of 40% utilization), livestock should be excluded from the area for at least a year or more—based on monitoring of the vegetation—to allow the habitat time to recover. The length of the rest period needed will vary by region and site conditions.

Drought

Grazing during drought can deplete already scarce floral and host plant resources for pollinators. For example, drought has been documented as the cause of extinction for three populations of the Bay checkerspot butterfly (*Euphydryas editha bayensis*). However, grazing under non-drought conditions is important in maintaining the forb-dominated grasslands essential for this species, and for reducing the cover of invasive annual grasses (Murphy and Weiss 1988).

- Adjust grazing intensity and duration to account for drought conditions, and avoid depleting already scarce floral resources (Howery 2016). See Finch et al. 2016 for additional information on adjusting rangeland grazing during drought conditions.
- Avoid grazing areas with milkweed during times of drought because this may make livestock more likely to consume toxic plants (McDougald et al. 2001).

Box 3: Native Bees and Other Invertebrates' Response to Grazing in the Desert Southwest

Arid shrub-dominated regions of the Desert Southwest support a high diversity of native bees, including many specialists. These regions have been historically heavily and extensively grazed since the 1800s. How grazing has affected insects in this region has been the focus of two studies which came to similar conclusions: native bee abundance is negatively impacted by grazing.

Minckley (2014) conducted sampling of a species-rich bee community in the San Bernardino Valley in the northwestern Chihuahuan Desert in Arizona. Research was done at sites in five habitat types: riparian, mesquite forest, abandoned field, grassland, and desert scrub. The sites had been either intensely grazed by cattle under year-long continuous grazing or not grazed by cattle for 22 or more years. The study found that the abundance of native bees was greater in ungrazed sites for all habitat types except for riparian areas. The diversity of native bees was similar in grazed and ungrazed areas, although interestingly, rare species were more common in grazed sites. The lower abundance of native bees observed in ungrazed riparian areas was due to the presence of large gallery cottonwood forests which had established in the years since grazing was removed from the site, and reduced the diversity of plants in the riparian area. Overall this study reported that bee abundance but not diversity was reduced by grazing—at least with only one year of grazing. The reduction in abundance of native bees was presumably due to reduced carrying capacity for native bees in areas with reduced floral resources from grazing.

In another study in the grasslands of the Sonoita plain in southern Arizona, DeBano (2006a) conducted a two-season study that examined the effects of cattle grazing on invertebrate communities. Grazing treatments included both high-density, short-duration grazing with a high stocking rate (also known as "holistic grazing management") and season-long grazing with a stocking rate of one AUM per 39 hectares. The results show that species richness and diversity of Hymenoptera (including native bees) were significantly reduced in both grazing treatments compared to ungrazed sites. In addition, this study reported a significant reduction in the abundance of all types of invertebrates, and decreases in richness or diversity of beetles (Coleoptera) and flies (Diptera). In addition, the species composition of invertebrate communities varied amongst grazing treatments, and one group, true bugs (Hemiptera), had higher diversity in grazed sites. DeBano (2006a) suggested that since habitats in the Desert Southwest evolved without concentrated herds of large native ungulates such as bison, they did not develop adaptations to grazing pressure and so are more susceptible to livestock grazing.

Based on these two studies, it is clear that grazing can have significant negative effects on native pollinators and other invertebrates in the Desert Southwest. However these are only two studies and they were only conducted for one year. More multi-year studies are needed to investigate the effects of commonly implemented grazing management regimes on both pollinators and the native plant community.

Rangeland in the Desert Southwest.

Native, Feral, and Wild Ungulates

In areas with large populations of elk, deer, or other native ungulates or of feral or wild ungulates such as horses, it may be necessary to adjust the timing, intensity, and duration of domestic livestock grazing because there is overlap in forage preferences and potentially competition for forage (DeBano et al. 2016). Avoiding overlap between livestock, feral and wild horses and burros, and native ungulates may help to maintain important floral resources for pollinators.

Monitoring Rangeland Health for Pollinators

A major component of adaptive management includes careful monitoring to determine when changes to grazing intensity, duration, and timing should be adjusted. This should be implemented at semiannual intervals, ideally more frequently. Regular quantitative assessments of rangeland health should include metrics relevant to pollinators. For example:

- Incorporate quantitative assessments of floral resources and/or pollinator abundance into existing range utilization monitoring. (See the "Monitoring" section on page **88** for more information.)
- Include management objectives to achieve healthy rangelands for pollinators, such as a minimum of 3 (ideally more) species flowering at a single time throughout the frost-to-frost growing season.

The Bureau of Land Management is currently piloting a Pollinator Specific supplement to the Assessment Inventory and Monitoring (AIM) protocol. Land management agencies could adopt similar measures to assess pollinators and pollinator habitat in rangeland health monitoring, meadow monitoring, or other relevant forest or land health monitoring protocols in rangelands.

Landscape-Scale Considerations

Incorporate resilience and resistance concepts into grazing management plans. Resilient and resistant rangelands—particularly those in the sagebrush biome—are less likely to be converted to annual invasive grasslands after disturbance events. This approach is being used for greater sage-grouse conservation and is widely applicable to pollinator conservation (Chambers et al. 2017). See Box **5** for more information on the overlap between restoration for pollinators and greater sage-grouse.

- Stock livestock at a duration, timing, and intensity that will maintain existing conditions in areas identified as high priority, resilient, and/or resistant to habitat stressors such as fire, invasive species, and drought. This is especially important in shrublands in the West that are under threat of being invaded by cheatgrass (*Bromus tectorum)*. Some range managers have even had success using targeted grazing to reduce the cover of nonnative grasses (Davies et al. 2016; Launchbaugh and Walker 2006; Olson et al. 2006).

Wild horses—like this herd on rangeland in Nevada—can degrade mesic habitats important for pollinators and other wildlife.

- US Forest Service's resilience and resistance fact sheet (Chambers et al. 2015) provides an overview of ecosystem resilience and resistance concepts and how land managers can use those concepts to assess risks to ecosystems, prioritize management activities, and select appropriate treatment or management.

Mowing and Haying

On western rangelands, mowing is used to maintain roadside vegetation, reduce fuel loads and prevent risk of fire, control invasive weeds, and eliminate encroaching woody plants; haying is used to harvest forage for livestock. In general, when done carefully, mowing can be an effective management tool for increasing or maintaining plant diversity and controlling invasive weeds and encroaching woody plants. Early spring mowing is key to removing cool-season weedy annual grasses, and fall mowing can remove thatch and aid wildflower seed dispersal. Haying can differ from mowing by the amount of resources cut and the height and frequency at which it is cut.

If done inappropriately—such as too frequently or at the wrong times of year—mowing can have detrimental effects on pollinators. Mowing and haying during the growing season affects pollinators by altering vegetation structure, reducing habitat diversity, and removing floral resources (Morris 2000; Johst et al. 2006; Noordijk et al. 2009; Kayser 2014). Both haying and mowing can result in direct mortality of eggs, larvae, and adults as well as destruction of important butterfly shelters and underground bumble bee nests (Thomas 1984; Wynhoff 1998; Di Giulio et al. 2001; Humbert et al. 2010; Hatfield et al. 2012; Kayser 2014). Also, if implemented repeatedly at the same time each year, both haying and mowing can reduce the abundance of flowering plants over time. Further, since intensively managed hayfields can provide large swaths of vegetated habitat that are subsequently cut, spring and summer haying may act as an ecological trap for some grassland birds (Bollinger et al. 1990; Perlut 2007); it is possible a similar effect occurs with bees and other pollinators that are attracted to early spring blooms in hayfields.

Mowing and haying can influence which species of floral resources are available for pollinators (Johansen et al. 2017). Frequent mowing or haying can reduce native plant species diversity and abundance and may also favor the development of grasses over forbs (Parr and Way 1988; Williams et al. 2007; Mader et al. 2011). Multiple studies have shown that mowing twice per season (one early and one late) can increase plant species diversity in grassland habitats (Parr and Way 1988; Forman 2003; Noordijk et al. 2009). However, early season mowing can lead to mortality of immature stages of butterflies as well as host plants. Other studies suggest that a single mow during the growing season (Valtonen et al. 2007) or in the fall (Entsminger et al. 2017) is more beneficial compared to two or more mowings in a year.

Given the huge diversity of species, life histories, habitat types, and varying invasive species challenges that land managers in the West must consider, it can be difficult to identify a single best time for all situations to achieve management goals. Generally, late summer or fall are the best times to mow or hay to minimize negative impacts to pollinators, but there are exceptions and limitations to restricting mowing to late in the season. For example, late-season mowing may result in the spread, not suppression of some invasive plants. Instead, land managers may want to focus on achieving a diverse mosaic of habitat types across the landscape in order to maintain pollinator health and biodiversity. Leaving unmowed strips as refugia and increasing heterogeneity of mowing (e.g., not mowing the same location at the same time every year) can help increase abundance and diversity of native bees and butterflies on rangelands (Bruppacher et al. 2016; Unternährer 2014; Buri et al. 2014; Kühne et al. 2015; Meyer et al. 2017).

Leaving some areas free from mowing or haying helps provide a refuge for pollinators.

Mowing and Haying Best Management Practices

Timing and Frequency

- Limit mowing to no more than twice per year. Ideally, sites would be mowed only once a year or every few years on rotation.
- Delay mowing until later in the growing season to allow flowering plants to bloom and give pollinators time to complete their full life cycles. It will also help ensure a steady supply of nectar and pollen when pollinators need it most.
    - In general, fall mowing after the first frost is ideal to avoid mowing floral resources and host plants for pollinators.
    - In hayfields, which provide pollinator habitat, delay harvesting until after most plants have bloomed, if at all possible.
    - Note that some pollinator species overwinter on the ground in vulnerable immobile life stages; assess whether mowing in autumn or winter will impact sensitive species.
- Mow during the middle of the day. Pollinator adults are typically most active during the warmer parts of the day, which means they are better suited to escape a mower.
- Avoid mowing during vulnerable life stages. This is particularly important when considering any sensitive species, such as the western bumble bee (*Bombus occidentalis*) or the monarch butterfly (*Danaus plexippus plexippus*).
    - Bumble bees and other native bees: Avoid mowing during the spring and summer when bumble bee nests are active, as mowing can destroy nests. See Figure **2** for bee phenology and Appendix **C** for where to find species-specific bumble bee phenology by state and USFS region.
    - Monarchs: Avoid mowing monarch breeding habitat (milkweed) during the monarch breeding season. See Figure **3** for region-specific mowing timing guidance for monarchs.
    - Other butterflies: Depending on the species and location, immature butterfly stages may be present at your site year-round. Determine which species are of high management priority and time mowing to avoid these vulnerable stages.

- If spring or summer mowing cannot be avoided, mow at times that will promote wildflower growth and may thus still provide some benefit to pollinators. For example, spring mowing in the Pacific Northwest can reduce invasive cool season grasses, and promote flowering plants. Consult regional botanists, ecologists, or vegetation management specialists to determine optimal timing to promote wildflower growth in your region.

If Invasive Nonnative or Noxious Weeds are Present

- Become familiar with the life history traits of your target weeds. Some species are stimulated by mowing, so alternative control methods may be preferable when they are present.
- Time mowing for periods before weeds flower. Avoid mowing when weeds have seed heads to help to reduce the spread of noxious weeds at the site. This limits the number of weed seeds that attach to mowing equipment and potentially get moved to a different site.

General Considerations

- Use spot mowing. Focus on areas with weeds and other target plants rather than mowing entire areas uniformly.
- Avoid routine mowing of an entire habitat patch. No more than a third of habitat should be mown in one year.
- Use a flushing bar. Flushing bars encourage wildlife to move out of the way of mower blades and may be helpful for adult pollinators as well.
- Mow at reduced speeds. Reduce mowing speeds to less than 8 mph to allow adult pollinators time to escape mower blades.
- Mow in an outward direction. Rather than mowing an entire hay field from the perimeter inward, harvest from one end of the field to the other so adult pollinators have an escape route.
- Adjust mowing height. Do not mow vegetation all the way to the ground. Ten inches or more is desirable to reduce plant stress and provide shelter to pollinators and other wildlife.
    - Bumble bees: Mow at the highest cutting height possible to prevent disturbance of established nests or overwintering queens. A minimum of 12–16 inches is ideal.
    - Monarchs: If mowing areas of milkweed plants in the spring, mow at a minimum height of 12–16 inches.
- Create a mosaic of patches with structurally different vegetation.
    - Leave one or more patches—as large as possible—of habitat unmowed for the entire year. These patches can provide important refugia for pollinators.
    - Where possible, vary mowing times every few years to increase plant diversity.
    - If hayfields provide pollinator habitat, practice rotational haying so some areas of intact habitat are left for pollinators each year. Divide hayfields into plots that can be harvested in a 3- to 4-year rotation.
    - Leave patches of shrubs, standing dead snags, and dead down wood for nesting material.
- Avoid mowing degraded Wyoming big sagebrush rangelands as it can increase invasive annual plants, and does not increase native vegetation (Davies et al. 2012).

Some milkweed species flourish along roadsides, benefiting from periodic disturbances. However, mowing and other types of management during the breeding season can cause immature monarch mortality (top). Lupine in rangeland along road in Nevada (bottom).

Roadsides and other Rights-of-Way

Roadsides and other rights-of-way frequently offer good opportunities for pollinator habitat because they offer linear, continuous habitat across the landscape. For example, in the Great Basin, roadsides provide important nectar plants such as rabbitbrush and sunflowers blooming along roadsides in the fall (Emma Pelton and Stephanie McKnight, personal observations). However, roadsides and other rights-of-ways are also mainly managed for nonwildlife reasons, such as driver safety and equipment access. Mowing or other management which reduces vegetation can have very detrimental effects during the active season of pollinators, and over time, lead to a reduction in plant diversity. To incorporate pollinators into mowing (or other) management plans for roadsides and other rights-of-ways, consider the following:

- Along roadsides, maintain a regularly mown clear zone as needed for sight distance and safety, but limit mowing of vegetation beyond this zone when possible. Keep in mind that some roadside plant communities will need regular disturbance or management to promote high vegetation quality and reduce weeds.
- Conduct mowing or other vegetation management practices within the context of an integrated roadside vegetation management (IRVM) plan that takes into account the needs of pollinators such as milkweed for monarchs.
- Consult the Federal Highway Administration handbook *Roadside Best Management Practices that Benefit Pollinators* (Hopwood et al. 2015) and other guidelines for more detailed information on best management practices for monarchs and other pollinators along rights-of-way. Available at www.xerces.org.
- Consult the Ecoregional Revegetation Assistant Tool (www.nativerevegetation.org), an online map-based tool to help practitioners to select native plants suitable for revegetation of a site by using filters for needed plant attributes, including value to pollinators. This is part of a collaboration between the Federal Highway Administration, US Forest Service, WSP, and Xerces Society.

Prescribed Fire and Wildfire

Many rangeland ecosystems in the West evolved with wildfire, with fires occurring at varying intensities and at intervals ranging from four to 450 years. Today, prescribed fire is an important tool for maintaining open landscapes dominated by flowering plants. This means fire can be used to greatly improve the value of habitat for pollinators, often increasing plant and pollinator species abundance and diversity (Smallidge and Leopold 1997; Huntzinger 2003; Campbell et al. 2007; Ponisio et al. 2016). Both prescribed fires and wildfires may benefit pollinators by causing a pulse in floral resources, or increasing the abundance of flowering herbaceous vegetation (Smith DiCarlo et al. In Review). This can lead to an uptick in pollinator abundance a few weeks or months after a fire (Van Nuland et al. 2013; Moranz et al. 2014). Work in an arid Pacific Northwest bunchgrass prairie found increases in native bee diversity and species richness, and changes in species composition, one year after wildfire (Smith DiCarlo et al. In Review). In fire-adapted forests and shrublands, the combination of conifer removal and prescribed fire can increase herbaceous flowering vegetation (Roundy et al. 2014; Bates et al. 2016; Bybee et al. 2016) and/or the diversity and abundance of butterflies (Huntzinger 2003; Kleintjes et al. 2004; Waltz and Wallace Covington 2004; Campbell et al. 2007; Taylor and Catling 2012; McIver and Macke 2014; Roundy et al. 2014a; Bates et al. 2016; Bybee et al. 2016).

However, fire in the wrong place, at the wrong scale, or at the wrong time can have the opposite effect, causing damage to native plant and pollinator communities from which it may take decades to fully recover (Smallidge and Leopold 1997; Harper et al. 2000; Ne'eman et al. 2000; Moretti et al. 2006; Pryke and Samways 2012; Scandurra et al. 2014).

The response of pollinators to fire varies by taxa and is often specific to certain life history traits (Smallidge and Leopold 1997; Cane and Neff 2011) and may depend on past land use (Moretti et al. 2009). While ground-nesting native bees typically nest deeply enough in the ground that they will not be directly killed by prescribed fires (Cane and Neff 2011), bee species that nest in stems and other woody materials, as well as butterflies that often reside as caterpillars and pupae in vegetation or very near the soil surface, are vulnerable. Fire may also affect pollinator fecundity (reproduction rates) by altering the availability of host plants or floral resources needed for egg-laying or nectar (Baum and Sharber 2012; Warchola et al. 2017). The response of pollinators is also dependent on the habitat type, fire intensity, and interval. For example, shrub-dominated ecosystems have greater fuel loads and tend to burn hotter than those in grasslands (Cane and Neff 2011), and the burning of slash piles during fuels reduction projects can cause intense localized heating of the soil, both increasing potential mortality of ground-nesting bees (Ne'eman et al. 2000).

Burning an entire area of habitat risks extirpating the local invertebrate community. It is therefore important to not burn an entire habitat area at once (Black et al. 2011), leave unburned skips, and determine if sensitive species are present and their sensitivity to fire. Scarcity of forage post-burn and damage to nesting materials can also stress native

Prescribed fire can be a useful tool for habitat restoration, but must be used carefully to avoid extirpating local pollinator populations.

bee populations. By leaving refugia and burning one-third or less of an area at once, you usually can reap the benefits of prescribed fire without causing irreparable damage to the local pollinator community. Ideally, fire management should aim for rotational burning where small sections are burnt in a multi-year or multi-decade cycle—mimicking historic fire regimes.

Overall, prescribed fire is an important management tool for maintaining flower-rich, open, early seral habitat for wildlife, including pollinators. Management with fire should aim to increase floral diversity, particularly in areas within dispersal or foraging range of important pollinator habitat. When implementing prescribed fire the goal should be to maintain a diversity of successional stages with a high diversity of herbaceous and flowering plants for pollinators.

Prescribed Fire Best Management Practices

Timing and Frequency

- Burn from October through February (or regionally appropriate timing when pollinators or plants are dormant). See recommended timing for the monarch butterfly and bee phenology in Figures **2** and **3**.
- Burn a site once every 3–10 years, or longer depending on the natural fire interval of the site.
    - Consider site-specific natural fire intervals or rotations for prescribed burns. To determine historical fire regimes consult the LANDFIRE database (https://www.landfire.gov/frg.php).
        - Bee populations can be significantly lower in years following a burn.
        - It can take two decades for insect communities to recover from a burn.
        - In forested habitats implement a heterogeneous fire interval.
- If burning must occur between February and October, consider the following:
    - Adult ground-nesting bees are generally in their nests in the early morning or late evening and less susceptible to fire during these times, as most ground-nesting bees nest deep enough to survive most fires.

Burning in the late fall or winter has the fewest negative impacts on pollinators.

    - Adult butterflies and above-ground nesting bees are less susceptible to fire during the warmer part of the day when they are active and can possibly escape.
- Habitat-specific recommendations:
    - In temperate grasslands, burn in the morning when humidity is higher and temperature is lower, which may increase the heterogeneity of a burn. This helps ensure some unburned refugia and may reduce soil heating which can cause mortality to invertebrates (Hill et al. 2017).
    - In areas with vernal pools, carefully time to avoid the key weeks when specialist bee species are active and threatened flower species are blooming. This varies from year to year and location, but is generally in spring or early summer. Other wetlands and riparian areas have longer bloom periods and corresponding pollinator activity, but burns in these areas should also be timed to avoid these periods (Black et al 2007).

Scale and Intensity

- Manage fire to increase habitat heterogeneity at multiple scales within and between sites.
- Avoid high-intensity fires by limiting fire in areas with high fuels, or burning when humidity is highest during the day (e.g., early morning).
- No more than one-third of the land area should be burned each year.
- Avoid burning small isolated habitat fragments.
- Burn small sections at a time. A program of rotational burning where small sections are burned every few years will ensure better colonization potential for pollinators by always leaving some areas of habitat unburned.
- As a fire moves through an area it may leave small unburned areas (“skips”). These should be left intact as potential micro-refuges.
    - If possible, mow or burn to create fire breaks that will result in patches of unburned or lightly burned areas to serve as refugia for animals within the burn area.
- Leave refugia within or adjacent to burned areas to promote recolonization. Pollinator dispersal capacity differs by taxa, and is an important factor to consider when incorporating refugia into a prescribed fire management plan.
    - Solitary bees: 100 meters–to 1 kilometer
    - Bumblebees: several hundred meters to several kilometers
    - Butterflies: varies from several hundred meters (some Lycaenidae) to thousands of kilometers (monarch, painted lady).

Minimize the use of Heavy Equipment or Excessive Ground Disturbance

Avoid actions that could degrade habitat and kill individual pollinators as a result of heavy equipment use or people trampling meadows or other sensitive habitat types.

Wood and Slash Pile Burning

If pile burning in conjunction with forest fuels reduction, or conifer removal where piles of slash or other woody material will be burned, they should be stacked in areas away from known sensitive pollinator sites, and burned outside of periods of peak pollinator activity (October–February).

- Pile burning results in intense localized heating which may kill some ground-nesting bees if they are present. To reduce the duration of soil heating, Busse et al. (2013) suggest “mopping up” pile burns with water 8 hours after ignition.
- Avoid placing piles in close proximity to habitat with high plant diversity, such as meadows, springs, and riparian areas. These areas may harbor a higher density of ground- and above-ground-nesting bees that could be harmed by pile burning.

Sensitive Species

- Develop specific fire management plans for sensitive pollinators, using the best-available information about the effects of fire on adult and larval stages.
- Consider the benefits and risks of managing sensitive pollinator habitat with fire. For example, the mardon skipper (*Polites mardon*), a rare species with very restricted range and limited dispersal ability under the ESA and species with limited dispersal ability, takes years to recolonize after a prescribed fire (Black et al. 2013). See Box **4** for a case study on the mardon skipper butterfly. In some cases, using mowing or grazing may be more appropriate management tools.
- Leave at least one-third of the suitable, occupied habitat untouched. This is especially important for sensitive pollinators to prevent local extirpation. Also, make sure that burning does not fragment suitable habitat at a scale that will isolate remaining populations. For example, some butterflies can only disperse a few hundred meters; burning too large an area may leave butterflies stranded in islands of suitable habitat surrounded by an uncrossable expanse of burned area.

Monitoring Effectiveness of Prescribed Fire

- Pre- and post-project monitoring are recommended to determine the effects of prescribed fire on pollinator communities, which can vary significantly across the landscape.
- For example, two drainages in the same mountain range, can support significantly different butterfly communities (Fleishman 2000) that may respond differently to fire.
- Ensure prescribed fire is achieving desired management goals such as increasing native habitat quality for pollinators and other wildlife.

Wildfire Recommendations

- Avoid application of aerial chemical fire retardants on areas with known populations of sensitive species of pollinators. If at all possible, also avoid creating fire breaks in sensitive pollinator habitat.
- Post-fire seeding should aim to include plant species known to provide floral resources for pollinators to ensure recovery of the surviving bee and pollinator community (Cane and Love 2016). See “Restoration” on page **44** for more details on post-fire seeding.
- If post-fire salvage logging is done, leave some standing and brushy woody material for above-ground, tunnel-nesting bees and minimize ground disturbances which could damage below-ground nesting bee nests.

Monitoring programs are important to understand the impacts of management on pollinator species like the rare mardon skipper.

CASE STUDY

Box 4: Understanding How Controlled Burning Affects a Rare Pacific Northwest Butterfly

Background

The mardon skipper (*Polites mardon*) is a rare butterfly in the Pacific Northwest known from four disjunct geographic areas: (1) southern Puget Sound, (2) the east side of the Cascade Range in Washington, (3) the Cascade Range in southern Oregon, and (4) the Coast Range in northern California (Del Norte) and southern Oregon. Across its range, the butterfly's population has declined due to human development, livestock grazing, fire suppression, and invasion of meadow habitat by native and nonnative vegetation.

Forest encroachment is one of the most serious threats to mardon skipper because it not only reduces the amount of open habitat, but closes off corridors between meadows, thus reducing butterfly dispersal (Roland and Matter 2007). During adult flight, mardon skippers avoid heavily forested habitats, avoid forest edges and trees during oviposition, and are assumed to have limited dispersal abilities (Runquist 2004; Beyer and Black 2007; Beyer and Schultz 2010). Large dense shrubs likely have a similar adverse impact as encroaching trees to the habitat and behaviors of this butterfly.

Historically, fire has played an important role in maintaining many native ecosystems (including mardon skipper habitat), but with wildfire suppression, prescribed fire and controlled burns are an increasingly common management tool. Using fire to manage habitat is based on the assumption that prairie species are adapted to wildfires, and thus can cope with regular burns (Harper et al. 2000; Swengel 2001). With many insects, however, this is dependent, on adequate unburned areas that can provide a refuge and sources of colonizers for the burned habitat. In habitat fragments where populations are more isolated, like many sites of the mardon skipper, prescribed burning could have much more deleterious effects and possibly lead to population extirpation. For example, Harper et al. (2000) found that overall arthropod species richness decreased in burned prairie sites, as well as the abundance of all but one of the species measured. Their results suggest that burning a small habitat fragment in its entirety could risk extirpating some species because of limited opportunities for recolonization from adjacent habitat. Rare butterflies can also be negatively impacted by prescribed burning. Swengel (2001) found that fire had consistent negative effects on prairie specialist butterfly species, and that these effects persisted for 3–5 years after burning.

Mardon skipper nectaring on *Calochortus* lily at Peterson Prairie, Washington.

Burn Study

In June 2008, a population of mardon skipper was discovered by Xerces staff in northern California, at Coon Mountain in the Six Rivers National Forest. The population was estimated to be the largest in California. The Coon Mountain area had not had a natural fire in decades, and small conifers and shrubs were encroaching into open meadow areas and likely having a negative impact on meadow dependent species such as the mardon skipper.

Staff from the Xerces Society, US Forest Service, and US Fish and Wildlife Service designed a study to help determine the effects of a controlled burn on the mardon skipper population. In late fall 2008, the US Forest Service conducted a burn that impacted approximately 30–40% of the core area occupied by the mardon skipper. The site was divided into four monitoring zones, with each zone subdivided into burned and unburned areas. Each monitoring zone was surveyed using two methods, transect counts and zone counts.

For the transect counts, a 150 ft. transect was set up in each subzone, resulting in a total of eight transects across the habitat area, four each in burned and unburned areas. All transects were placed in the best available habitat within the subzone that accommodated the desired transect size. Before each surveying transect, a distance of 15 ft. was measured out to each side to give a transect width of 30 feet. Xerces staff walked each transect slowly and counted all butterflies within the area of the transect. Butterflies were not counted if they flew in from behind the observer, so as to avoid the possibility of counting the same individual twice.

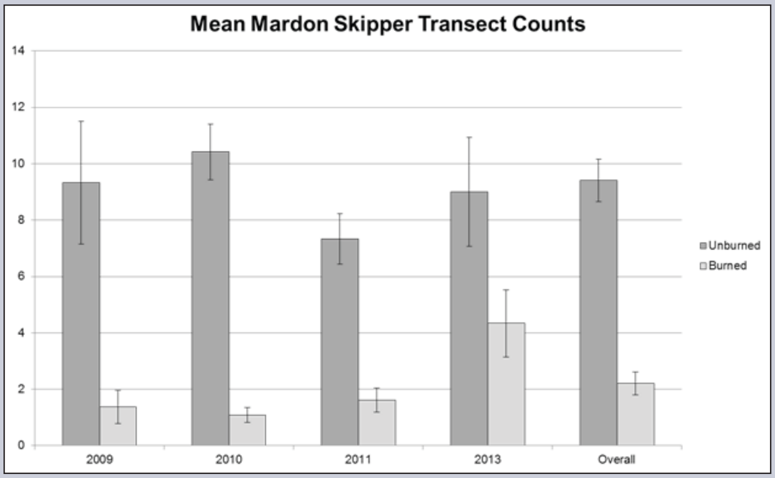
In addition to transect monitoring, we completed counts over each zone using a modified Pollard Walk (Pollard 1977). Xerces staff walked slowly through each zone, taking about 5 minutes to walk 100 meters, while looking back and forth on either side for approximately 20 to 30 feet. Surveyors walked a path such that all areas within the zone with apparently suitable habitat was covered by this visual field. We counted every butterfly that was encountered, and did not count butterflies that flew in from behind.

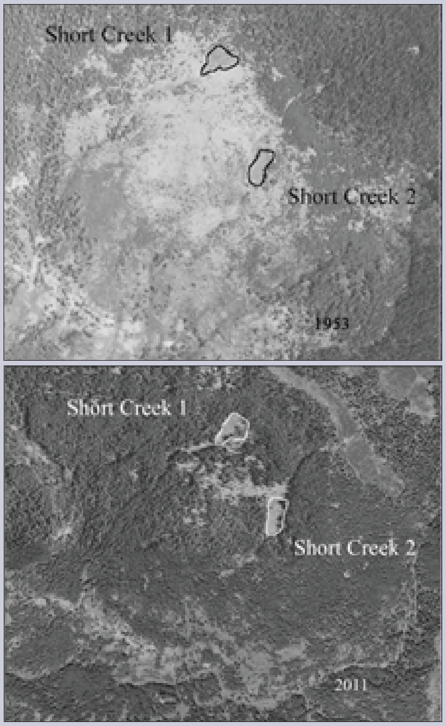
Xerces staff counted mardon skippers in each transect and in each zone twice during the mardon flight season in 2009 (May 27th and June 7th), three times during the flight season in 2010 (June 21st, June 27th, and July 2nd), three times during the flight season in 2011 (June 19th, June 23rd, and July 2nd), and three times during the flight season in 2013 (May 25th, June 2nd, and June 6th).

Results

Counts across all survey dates and years showed mardon numbers that ranged from 1.7 to 27 times higher in unburned zones compared to burned zones on the same dates. When we pooled transect data within years, both burning and time showed a significant effect on mardon skipper abundance. The fact that there was no interaction effect between time and burning suggests that the effect of burning on mardon skippers is real and not confounded by annual variation in butterfly populations. One thing worth noting is that mardon skipper populations in burned areas did appear to be making a comeback after five years. This accentuates the need to leave substantial habitat when using fire as a management tool for mardon skippers. Leaving adequate habitat to support a large enough butterfly population to sustain a prolonged (although ideally short-term) decrease in habitat quality is essential. For the full report, refer to Beyer and Black (2007), Beyer and Schultz (2010), and Runquist (2004).

Mardon skipper transect counts in burned and unburned areas 2009-2012.





Example of forest encroachment of mardon skipper habitat. Aerial photographs taken in 1953 (top) and 2011 (bottom) show how their habitat has changed in the Short Creek Complex of the Rogue River–Siskiyou National Forest in southern Oregon. Outlined areas are current mardon habitat.

Hunt's bumble bee (*Bombus huntii*) nectaring on a native thistle in Oregon.

Restoration

"Restoring rangelands for bees will require large-scale seeding efforts coupled with judicious grazing and fire management to improve plant diversity." (Cane 2011)

Western rangelands have a long history of habitat degradation due to improper livestock grazing and wildfire suppression, as well as invasive species, intense wildfires, drought, and climate change. As a result, rangeland restoration and rehabilitation is now at the forefront of land management. In addition, significant focus has been placed on rangeland restoration for species of conservation concern such as the greater sage-grouse (see Box **5**).

Restoration is a rapidly evolving field that is constantly being refined by science, and while restoration has been a focus of land management for decades, very little consideration has been given to incorporating the needs of pollinators into restoration in western rangelands (Winfree 2010; Menz et al. 2011). The majority of pollinator restoration research in the West has focused on agricultural landscapes (Tonietto and Larkin 2018); restoration of rangelands for pollinators is an essential next step.

Pollinator communities require a diversity of floral resources from frost to frost, with a variety of flower shapes, colors, and phenologies as well as nest resources such as bare ground, cavities, and pithy stemmed plants and shrubs. Restoration planning for pollinators should ideally include seed mixes, plants and nest resources such as areas of bare ground or dead wood with cavites that address all of these needs. To ensure the persistence of sensitive pollinators, habitat restoration may be necessary to maintain, improve, or increase habitat availability such as host or nectar plants, and carefully monitor restoration outcomes for the target species.

Very few studies have examined the optimal patch size of floral and nest resources needed to sustain a diverse and abundant pollinator community. However, in general, research supports that plant-focused habitat restoration is successful in improving habitat for pollinators. This holds true for grasslands, forests, and shrub-dominated habitat types. A recent meta-analysis by Tonietto and Larkin

(2018) found that seven restoration treatments focused on improving plant communities—general habitat restoration, ecological compensation meadows, mowing, prescribed fire, grazing, seeding, and invasive plant removal—in eleven different habitat types in natural areas (the study excluded urban and agricultural landscapes), had a net positive benefit to nontarget native bees. In addition to this meta-analysis, research has found that pinyon–juniper removal and forest restoration (fuels reduction, canopy reduction) treatments all improve habitat for pollinators by increasing the availability of nectar and host plants, as well as microclimates suitable for butterfly roosting or overwintering (McIver and Macke 2014; Kleintjes et al. 2004; McIver et al. 2014; Bates et al. 2016). However, there are some limitations of using plants as the yardstick of a successful restoration for invertebrates (e.g., Longcore 2003). For example, recent work focused on arid grassland restoration in the inland Pacific Northwest suggests that native bee communities may not necessarily benefit from restoration practices focused primarily on vegetation, and that understanding more about both floral requirements of the local bee community and how restoration impacts nesting habitat quality will be important for restoring native pollinator communities (Smith DiCarlo 2018).

Generally, habitat restoration should aim for heterogeneity in flowering plant phenology, color, and shape, as well as heterogeneity in vegetation structure across the landscape. Restoration projects should also provide connectivity in the larger landscape by establishing corridors or high-density stepping-stone patches that will ensure adequate colonization by pollinators, and facilitate dispersal. Ideally, restoration sites will be within one kilometer of intact habitat to accommodate colonization by the majority of more mobile pollinators. Restoration plantings should aim for high plant diversity because it has been correlated with both butterfly and bee diversity and abundance (Minckley et al. 1994; Larsson and Franzén 2007; Xie et al. 2008; Batáry et al. 2010; Grundel et al. 2010; Roulston and Goodell 2011; Palladini 2013; Rubene et al. 2015; Smith et al. 2016; Vrdoljak et al. 2016).

Restoration Best Management Practices

Planning

Site Selection Considerations

- Select sites for pollinator habitat restoration that are protected from pesticide drift, considering past pesticide use. Local, state, and extension soil laboratories can test soil for pesticides, soil fertility, and microorganisms. See "Pesticides" section on page **70**.
- Consider what the existing vegetation is, and how that may shape the potential vegetation at the site (e.g. existing invasive plants).
  - Prioritize sites without noxious or invasive plants that may impede restoration efforts. Consider whether the seed bank may contain problematic plants.
- Consider climatic conditions, elevation, and aspect and how those factors may affect restoration efforts.
  - Select planting locations within climactic microsites that will retain moisture longer into the summer—such as north-facing slopes or gullies that will retain snow or water. This is especially important for post-fire restoration in arid shrub-dominated rangelands.
- Soil type is an important factor to consider when selecting plant species for restoration. Consider the following:
  - Some native plants grow better in specific soil types such as sand, silt, clay, or loam. Select plant species that will perform well in the soil type targeted for restoration. Factors

CASE STUDY

Box 5: Greater Sage-Grouse and Pollinators: Overlaps in Conservation

Conserving or restoring habitat for bees and butterflies by planting nectar and host plants and providing other resources should not be viewed as an insular endeavor—it should be viewed as part of a larger landscape-level effort to conserve or restore habitat for a wide variety of pollinators, insects, birds, and other wildlife. Restoring habitat for pollinators is really about restoring healthy diverse native plant communities for all wildlife species. For example, milkweeds are know as the larval host plant of the monarch butterfly, but many other species benefit from them too (Borders and Lee-Mäder 2014). In arid regions of the West, milkweed species are sometimes the only plants blooming in the hot summer months, and a plethora of butterflies, moths, and native bees including the western bumble bee (Bombus occidentalis), a species of conservation concern in the West, forage for nectar and pollen from their flowers. In addition, an assemblage of other insects including milkweed specialists feed on the plants, and songbirds, including the vermillion flycatcher (*Pyrocephalus rubinus*) and black-capped chickadee (*Poecile atricapillus*), have been observed using the fibers from the seed pods and plants to construct their nests (Borders and Lee-Mäder 2014). Overlap in conservation targets for multiple species allows resource-limited land managers to simultaneously achieve multiple conservation objectives.

In the past decade, the sagebrush biome, the majority of which is rangeland, of the West has become a central focus of landscape-level conservation and restoration efforts for the declining greater sage-grouse (*Centrocercus urophasianus*). Greater sage-grouse chicks are highly reliant on forbs and insects, consuming species from 34 genera of forbs and 41 families of invertebrates (Drut et al. 1994; Gregg and Crawford 2009), and high-quality greater sage-grouse habitat contains native forbs from at least 10 genera. Conservation plans that have already been developed with goals to maintain high-quality sage-grouse habitat will also improve habitat for monarchs and other pollinators by increasing the cover and diversity of forbs that provide nectar resources for adult butterflies and other pollinators, and larval hosts for some butterflies (Gilgert and Vaughan 2011; Dumroese et al. 2016). Dumroese et al. (2016) determined the forbs that are most likely preferred and consumed by greater sage-grouse and are also recommended for pollinators including monarchs. By leveraging available resources, land managers can achieve conservation targets for both greater sage-grouse and other birds, native insects, and pollinators—including the monarch butterfly.

The plants in the following table (Table 3) proffer high-quality forage for both pollinators and the greater sage-grouse (Drut et al. 1994; Gregg and Crawford 2009; Cane and Love 2016; Dumroese et al. 2016; Stettler et al. 2017). Rabbitbrush communities (*Ericameria* spp. and *Chrysothamnus* spp.) in particular are important as

Restoring habitat for pollinators benefits other wildlife, such as the greater sage-grouse.

TABLE 3: Plant List: Forage for Native Bees, Monarch, and Greater Sage-Grouse (adapted from Dumroese et al. 2016).

Family	Genus/Species	Common name	Native Bees	Greater sage-grouse	Monarch
Apiaceae	*Lomatium* spp.	desert parsley	x	x	
Asteraceae	*Balsamorhiza* spp.	balsamroot	x	x	
Asteraceae	*Chrysothamnus* spp.	rabbitbrush	x	x	x
Asteraceae	*Crepis* spp.	hawksbeard	x	x	
Asteraceae	*Ericameria* spp.	rabbitbrush	x	x	x
Asteraceae	*Erigeron* spp.	fleabane	x	x	
Asteraceae	*Symphiotrichum* spp.	asters	x	x	x
Boraginaceae	*Mertensia* spp.	bluebells	x	x	x
Fabaceae	*Astragalus* spp.	milkvetch	x	x	
Fabaceae	*Dalea* spp.	prairie clover	x	x	
Fabaceae	*Hedysarum* spp.	sweet vetch	x	x	
Fabaceae	*Lotus utahensis*	Utah lotus	x	x	
Fabaceae	*Trifolium* spp.	clover	x	x	x
Fabaceae	*Vicia* spp.	vetch	x	x	
Liliacea	*Calochortus* spp.	mariposa lily	x	x	
Polemoniaceae	*Microsteris gracilis*	slender phlox	x	x	
Polygonaceae	*Eriogonum* spp.	buckwheat	x	x	x
Rosaceae	*Geum* spp.	avens	x	x	

*Plants highlighted in blue provide forage for native bees, the monarch butterfly, and greater sage-grouse. For specific species within these genera, refer to Dumroese et al. 2016.

they host more Lepidoptera larvae as potential sage-grouse forage, compared to sagebrush (*Artemesia* spp.) communities. They also provide crucial late-season nectar for native bees and migrating butterflies (Ersch 2009; Griswold and Messinger 2009).

The following efforts outlined in "USDA Forest Service Sage Grouse Conservation Strategy 2015-2020" also address and complement pollinator conservation (Finch et al. 2015).

- Application of Landscape Analyses to Planning.
- Genomics for Conserving Plant and Animal Populations.
- Effects, Prevention, and Control of Fire and Invasive Species.
- Restoration Science and Applications.
- Seed and Plant Materials Development Sciences.
- Application of Resilience and Resistance Concepts.
- Climate Change Adaptation Science and Models.
- Effectiveness Monitoring and Adaptive Management (Finch et al. 2015).

to consider include soil salinity, pH, organic content, bulk density, soil microorganisms (rhizobium), and compaction.

  - Plants may have a higher chance of establishing in microclimatic niches with moisture retention, such as those that hold snow later in the season (north facing drainages or slopes).
  - Soil information can be determined using local soil surveys and the National Resources Conservation Service (NRCS) Web Soil Survey (http://websoilsurvey.nrcs.usda.gov/app/).

- Conduct a site inventory to determine what floral, host, and nest resources are available for pollinators at a site. Choose plant species that will fill in bloom gaps in existing native vegetation. For example, if a site lacks late-season bloom, consider including late-blooming asters in the seed mix or planting plan.
  - If a site inventory reveals that an area has low diversity and abundance of floral, host, and nesting resources for pollinators, consider experimenting to determine if interseeding (see the "Seeding" section on page **56**) or restoration plantings of the degraded rangeland could improve habitat for pollinators while also improving forage for livestock. See Box 7 to learn more about how land management agencies can use research to better understand how to restore rangelands.

Invasive Species

Ensure restoration plans include measures to prevent the introduction and spread of invasive species by ensuring equipment and erosion control material (straw, wattles/logs) is clean and free of mud or invasive plant material. Utilize weed washing stations, Early Detection Rapid Response, and adaptive management as ways to manage invasive plants.

Sensitive Pollinators

If a sensitive pollinator is present in a restoration site, or the restoration is being implemented to enhance or expand habitat for a sensitive pollinator, then careful monitoring and planning is needed. Include a plan to monitor the effects of any implemented restoration treatment on all life stages and behavioral responses of sensitive pollinators. Long-term demographic monitoring is recommended. Plans should also assess the effects of restoration on larval host plants and nectar and nest resources. In addition, land managers should consider ways to improve connectivity, such as ensuring that there is adequate colonized habitat of a target sensitive pollinator within dispersal distance of the restoration site. See the "Monitoring" section on page **88** for further recommendations on establishing pollinator monitoring.

Milkweeds provide resources for a variety of insects such as this bumble bee.

Sourcing Seed and Plant Material

Where available and economical, local ecotypes of native plants and seed should be used, following provisional or empirical seed zone guidelines developed by your region, in accordance with the National Seed Strategy (Bower et al. 2014). Consult the US Forest Service Western Wildland Environmental Threat Assessment Center: TRM Seed Zone Applications (www.fs.fed.us/wwetac/threat-map/TRMSeedZoneMapper.php) and the National Seed Strategy (see explanation on page **49).**

It is also ideal to select plant sources and collect plant materials from multiple locations or sources to achieve high genotypic diversity. Using seed or plant sources with a variety of genotypes will ensure floral resources remain available for longer periods of time, especially under drought (Genung et al. 2010). Research suggests that higher genotypic plant diversity supports a greater diversity of pollinators (Genung et al. 2010; Smith et al. 2015; McCormick 2017).

The availability of pollinator-friendly native seed and other native plant materials is limited in western states (Nahban et al. 2015). Consequently, there is a need to increase commercial seed production of restoration-appropriate seeds in each ecoregion. This is being addressed in part by programs such as Seeds of Success, the national native seed collection program led by the Bureau of Land Management (BLM) in partnership with other federal agencies and nonprofit organizations (accessible through www.blm.gov). Seeds of Success aims to "get the right seed in the right place at the right time" and to "stabilize, rehabilitate and restore lands in the United States." This is also a goal of the National Seed Strategy, a framework that connects the private marketplace with federal, state, tribal, and nonprofit organizations to develop native seed sources for restoration and rehabilitation (accessible through www.fs.fed.us). Oldfield and Olwell (2015) provide an overview of the National Seed Strategy and best practices and strategies for land management agencies to move forward in developing local commercial markets of native seeds for restoration and rehabilitation. According to Oldfield and Olwell (2015), of the roughly 18,000 species of native plants in the United States, there are only just under 2,000 available on the commercial market. The process of getting a native plant species into commercial production is slow, and may take 10–20 years before a species is available at a scale adequate for large landscape-level restoration or rehabilitation efforts (Olwell and Riibe 2016). The National Seed Strategy also addresses several relevant national initiatives including the *National Strategy to Promote the Health of Honey Bees and Other Pollinators*, the Interior Department Secretarial Orders 3330 (mitigation) and 3336 (rangeland fire), and Executive Order 13112 on invasive species. It is important that land management agencies and other groups work within the framework of the National Seed Strategy to identify and develop commercial sources for pollinator-friendly plant species that are suitable for both restoration and rehabilitation and benefit pollinators. One of the most important things to do when beginning a large-scale restoration effort is to identify the native species needed and begin working with native seed producers well in advance of when they will be required. See *Pollinators and Roadsides: Best Management Practices for Managers and Decision Makers* (Hopwood et al 2015) for a case study of how Arizona's Department of Transportation successfully works with native seed producers about upcoming needs and offers a premium above-market value for the species they needed most.

The commercial availability of native, pollinator-friendly plant materials is lacking in many parts of the West, but that is starting to change—thanks to better coordination and growing interest in many species—including showy milkweed.

- Collaborate with federal, state, and nonprofit partners of the National Seed Strategy to increase commercially available, locally sourced native seed from all provisional and empirical seed zones in your region (Olwell and Riibe 2016).

- Communicate with commercial native seed producers to grow and provide seed for species of known value to bees, that fit restoration goals, and in advance, so they have adequate time for grow out and seed production.

Plant Species Selection

Diverse plantings that resemble natural native plant communities are the most likely to resist pest, disease, and weed epidemics and thus confer the most pollinator benefits over time (Tilman et al. 2006, Oakley and Knox 2013). Restoration plantings should provide pollinators with foraging, breeding, and/or overwintering resources. Select a combination of locally native species that are attractive to pollinators, with seasonal diversity in bloom times, flower morphology and color, pithy and woody stems for cavity-nesting species, and native bunch grasses. Species selection will also be influenced by anticipated establishment success, appropriateness for the site, cost, and other limitations. Plant lists should broadly include framework plants (species which provide an abundance of nectar for a variety of pollinators), bridging plants (species which provide nectar during times of scarce floral resources such as early spring or fall), and workhorse plants (species that are widespread, readily establish, compete well, and provide floral resources to pollinators) (Menz et al. 2011).

Flower Color and Shape

Aim for seed mixes or plants with a diversity of flower colors and shapes and sizes, as well as varying plant heights and growth habits to encourage the greatest diversity of pollinators (Potts et al. 2003; Ghazoul 2006). Select plants with a diversity of floral morphology (e.g., simple, medium disk florets, long disk florets, bilateral symmetry, medium tubular), and bloom times to support a diverse pollinator community (Roof et al. 2018).

- Bees typically visit flowers that are purple, violet, yellow, white, and blue (Proctor et al. 1996), and unlike other pollinators, can pollinate zygomorphic flowers (those with bilateral symmetry such as pea flowers).
- Butterflies visit a similarly wide range of colors, but also red (Proctor et al. 1996), and tend to visit flowers with narrow tubes or spurs or those with wide landing pads.
- Flies are generally attracted to white and yellow flowers (Stubbs and Chandler 1978) that are shallow or funnel-shaped or complex and trap-like.
- Bats visit dull white, green, or purple flowers that are bowl shaped and night-blooming.
- Hummingbirds generally visit scarlet, orange, red or white funnel-shaped flowers (refer to Table **1** for more information).

Bees often visit purple, violet, white, blue, or yellow flowers.

Monarchs require the leaves of milkweed—like the swamp milkweed pictured here—for food as caterpillars, but they can also use the flowers' nectar as fuel as adult butterflies.

Temporal Diversity

Seed mixes and plantings should strive for temporal diversity of flowering species. Try to provide floral resources from frost to frost in your region, to support pollinators through the year. For example, bumble bee queens and mining bees emerge early and need late-winter or early spring floral resources. Monarch butterflies need a diversity and abundance of nectar resources during spring migration, in the breeding season, and again for the fall migration, which all varies by region (Figure **2**).

- Aim for a minimum of three flowering plants blooming during each season (spring, summer, and fall).
- Early- and late-blooming plant species are especially important for pollinators. They provide resources for early season pollinators emerging from hibernation (bumble bees), and to pollinators building up their energy reserves before entering winter dormancy (e.g. monarch butterfly) (Pywell et al. 2005).
- Early season plants that provide floral resources for pollinators include willow (*Salix spp.*); native chokecherry/sandcherry/plum (*Prunus* spp.); currant or gooseberry (*Ribes* spp.); serviceberry (*Amelanchier s*pp.); rose (*Rosa* spp.); bitterbrush (*Purshia* spp.); balsamroot (*Balsamorhiza* spp.); desert parsley (*Lomatium* spp.); globemallow (*Sphaeralcea* spp.); penstemon or beardstongue (*Penstemon* spp.); and lupine (*Lupinus* spp.). In one study, 14 species of long-tongued bees (in the genera *Apis, Bombus, Ceratina, Nomada,* and *Osmia*), 43 species of short-tongued bees (in the genera *Andrena* and *Colletes*, and family Halictidae), and 31 species of flies (primarily in the family Syrphidae) were observed foraging on a single willow species (Robertson 1929).
- Summer-flowering species include some milkweed species (*Asclepias* spp.), which often flower in the summer in arid regions, when floral resources are generally scarce. Isolated aspen stands in Pacific Northwest grasslands also support higher flowering plant species richness and distinct floral communities in the late summer (Gonzalez et al. 2013), at which time some research has found a greater abundance of bumble bees compared to the surrounding grassland habitat (Gonzalez et al. 2013).
- Late-season species include rabbitbrush (*Ericameria* spp., *Chrysothamnus* spp.); goldenrod (*Solidago* spp.); sunflowers (*Helianthus* spp.); blanketflower (*Gaillardia* spp.); asters (*Symphiotrichum* spp.); and vinegar weed (*Trichostemma* spp.)—or any late-blooming, regionally appropriate Asteraceae. Rabbitbrush (*Ericameria* spp. and *Chrysothamnus* spp.) provides critical floral resources for pollinators and the monarch butterfly in the late-summer and fall. This is especially true during times of drought when other plants may not flower (Griswold and Messinger 2009). Griswold and Messinger (2009) reported that 60% of fall flying bees use rabbitbrush in the fall.

Central bumble bee (*Bombus centralis*) on penstemon—an early-season blooming plant.

Common Rangeland Plant Genera Important Food Resources for Native Bees

There are some common plant genera identified as supporting a diversity and abundance of native generalist and specialist bees in western rangelands: phacelia (*Phacelia*), globe mallows (*Sphaeralcea*), mallows (*Sidalcea*), legumes (*Astragalus, Dalea, Vicia, Lupinus, Trifolium*), penstemons (*Penstemon*), asters (e.g., *Helianthus*, *Gaillardia*), beeplant (*Cleome*), rabbitbrush (*Ericameria*, *Chrysothamnus*), creosote bush (*Larrea tridentata*), mesquite (*Prosopis*). In the Mojave desert, creosote bush (*Larrea tridentata*)—a species that flowers in both the spring and summer depending on rain—is visited by greater than 120 species of native bees.

Butterflies Host and Nectar Plants

Where regionally appropriate, managers should consider including host plants and nectar sources for declining butterflies such as the monarch butterfly. See Box **6** for more information on where to restore habitat for monarch butterflies.

Pollinator Nest Resources

Incorporate pollinator nest resources into restoration plans. Aim for heterogeneity in habitat structure to provide nest resources for a variety of pollinators.

Ground-Nesting Bees

Leave some areas of undisturbed bare ground that occurs naturally in a given habitat type (e.g. naturally occurring bare ground between bunchgrasses and shrubs in sagebrush steppe) between vegetation or in large patches for ground-nesting bees to excavate and access their nests (Potts et al. 2005). South-facing slopes are frequently the preferred locations for nesting bees, but any angle of slope from horizontal to vertical might be occupied. Nesting areas in bare soil range in size from less than an inch to hundreds of square meters (Wilson and Messinger Carril 2015). Given that the patch size and characteristics of bare ground which benefit most native bees is not well-established, aim to maintain bare ground at distances typically found between the native vegetation at a site.

Above-Ground Tunnel-Nesting Bees

Where appropriate, plant flowering plants with pithy stems, and preserve standing dead snags/trees, leave down wood and twigs, and prevent destruction of microtopography created by bunchgrasses in grassland and shrubland habitats. Where natural bee nesting material is lacking, it may be appropriate to create brush pile structures to provide immediate nesting habitat for some bees (Steffan-Dewenter and Schiele 2008). Examples of plants that provide resources for stem-nesting bees include buckwheat (*Eriogonum* spp.); ironweeds (*Vernonia* spp.); sunflowers (*Helianthus* spp.); agave or century plant (*Agave* spp.); yucca (*Yucca* spp. especially *Yucca whipplei*); sotol (*Dasylirion wheeleri*); and beargrass (*Xerophyllum tenax*), along with shrubs such as wild rose (*Rosa* spp.); elderberry (*Sambucus* spp.); sumac (*Rhus* spp.); sagebrush (*Artemesia* spp.); false willow (*Bacharis* spp.); and native blackberries (*Rubus* spp.).

Native Bunch Grasses

Native, perennial bunch grasses and sedges are larval host plants for some butterflies, potential nesting sites for colonies of bumble bees, and possible overwintering sites for bumble bees (Kearns and Thompson 2001; Collins et al. 2003; Purtauf et al. 2005). Native grasses are important components to seed mixes, but must also be carefully balanced to ensure the grasses do not easily outcompete forbs. Below are basic recommendations for including grasses in habitat restoration seed mixes:

- Most seed mixes should be 45–65% grasses, although this percentage may need to be higher or lower for some sites. Consult regional botanists and restoration ecologists to determine the optimal proportion of grass for a seed mix.
- Prioritize small-statured, highly clumping grasses.
- Include native rhizomatous grasses at a much lower rate (~5%), but do include them.
- Consider foregoing nitrogen fertilizer treatments which overly benefit grasses to the detriment of forbs.

Perennial Flowering Plants

Perennials are more likely than annuals to bloom during times of drought, and can provide critical resources for native bees when annuals are not available or scarce. Rabbitbrush (*Ericameria* spp.) in arid rangelands are an example (Griswold and Messinger 2009). Once established, perennial forbs can confer resilience and resistance to disturbance.

- Incorporate native thistles into restoration projects. Native thistles are visited frequently by butterflies and native bees, and some provide more sugar in their nectar than other native plants (Eckberg et al. 2017). Consult the Xerces Society's *Native Thistles: A Conservation Practitioners Guide* (Eckberg et al. 2017).

Western pygmy blue on rabbitbrush—a late season nectar resource in arid landscapes.

Resources to Help you Choose Species Appropriate for Your Project

- Appendix **C** includes resources that list habitat and plant associations for native bee genera organized by ecoregion and US Forest Service region.
- Ecoregional Revegetation Assistant Tool

The Ecoregional Revegetation Assistant Tool is an online map-based tool to aid practitioners with selecting native plants for restoration and pollinator habitat enhancement. The map can be searched by US Environmental Protection Agency (EPA) Level III Ecoregions, as well as by state. The plant species found within an ecoregion can be filtered by attributes such as soil type, moisture needs, palatability, salt tolerance, and value to pollinators, and a list of workhorse plant species can also be generated. This tool was a collaboration between the Federal Highway Administration, US Forest Service, WSP, and the Xerces Society. The tool is available through the Federal Highway Administration's website, www.nativerevegetation.org/era/.

- **Xerces Society's Pollinator Conservation Resource Center** includes pollinator and monarch nectar plant guides among other resources (www.xerces.org/pollinator-resource-center).

Box 6: Where Should Monarch Habitat Be Restored?

Consistent with the wide-ranging nature of the monarch butterfly, suitable breeding and migratory habitat is widespread across the West. Each spring, monarch butterflies overwintering in forested groves along the California coast fan out across the state to begin finding newly emergent milkweed on which to lay eggs. Additional generations continue the migration north, east, and south into the interior parts of the West and even up into southern British Columbia, tracking milkweed emergence. In the fall, usually between August and October, milkweed dies back, monarch breeding slows or stops, and the final generation makes their way back to the overwintering grounds, clustering together to survive the winter from November to February. (Historically, up to 10 million monarchs made the journey to California, but these days only a few hundred thousand butterflies make their way to the overwintering sites.) Tagging efforts have also shown that some monarchs from the Southwest migrate to central Mexico, mixing with the eastern monarch population (Morris et al. 2015). Consult Figure **3** for ecoregion-specific timing of monarch breeding.

Both monarch adults and caterpillars use milkweed as food.

Habitat suitability modeling shows there are notable concentrations of potentially highly suitable monarch habitat in the Central Valley of California as well as in southern Idaho and eastern Washington; smaller areas are evident in other regions, including northern Nevada, southern Arizona, parts of Utah, and most low-elevation lands in Oregon, excluding the coast (Dilts et al. 2018). Widespread planting of milkweed is often the response to help monarchs. However, this is not a recommended strategy across the western US. Instead, the Xerces Society recommends a more holistic and targeted approach to monarch conservation. The three components of this are, in order of importance:

1. Identify, protect, and manage existing habitat to maintain its value for monarchs.
2. Enhance existing habitat (if needed and appropriate) to improve its value for monarchs.
3. Restore habitat in areas where it occurred historically, but has been lost.

High-quality monarch breeding and migratory habitat offers native milkweeds to provide food for caterpillars (and nectar for adults) and other flowers—preferably native—to provide nectar for adults. Habitat should be safe from pesticides and keep butterflies free from high levels of pathogens. Additional factors such as roosting habitat and shade may also be important features of high-quality habitat.

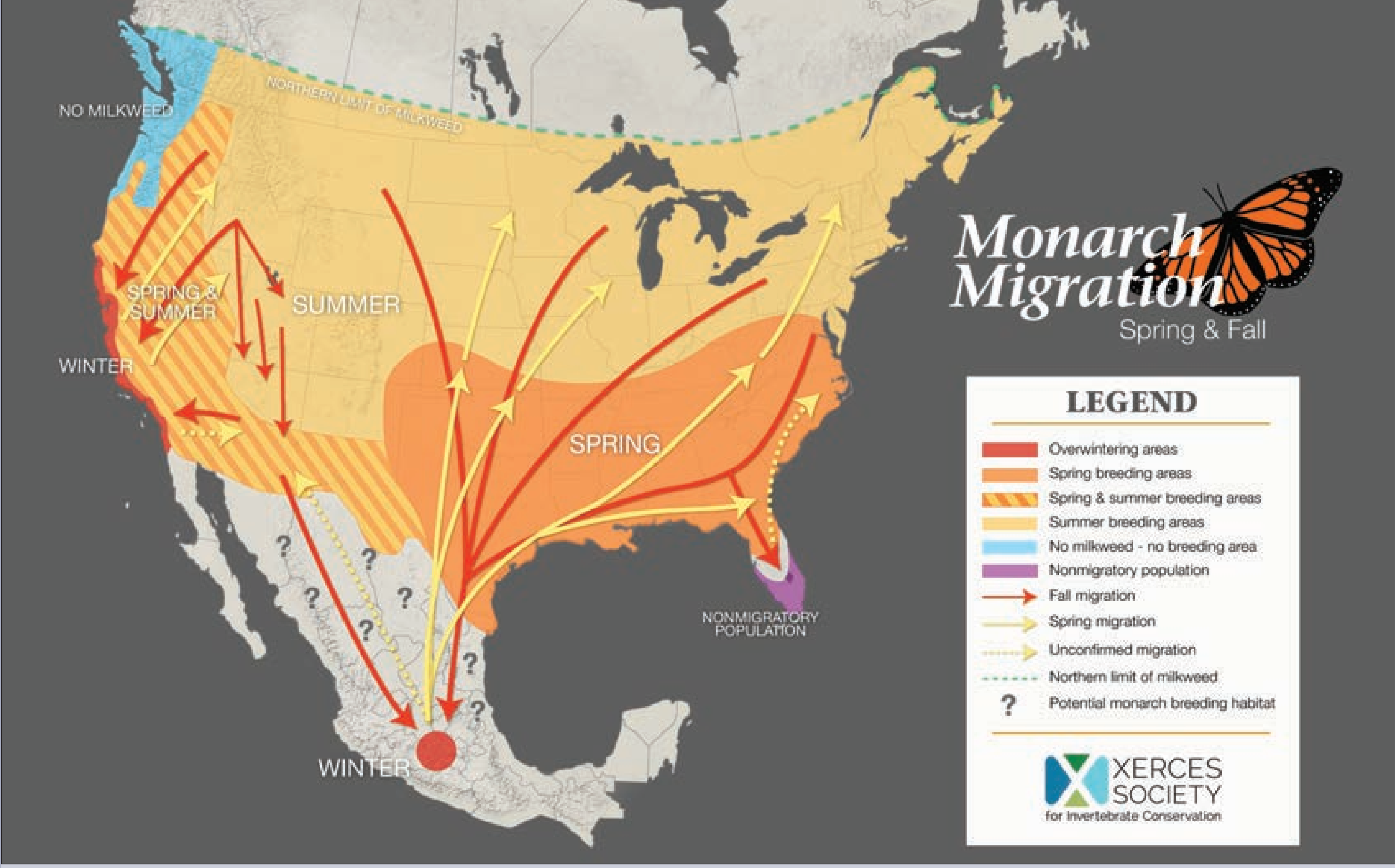


Figure 4: Monarch Migration and Distribution in North America.

With the wide range of milkweed species found in the West, there are species which are appropriate for a variety of habitat types. For example, some species grow in wetter areas (e.g., swamp milkweed and showy milkweed), while others tolerate a wider range of soil moistures (e.g., narrow leaf milkweed), and some prefer very dry, bare soil (e.g., pallid milkweed and desert milkweed). The Xerces Society recommends planting native milkweed where it historically occurred and in appropriate habitat types; we do not recommend planting milkweed close to overwintering sites (within 5–10 miles of the coast) in central and northern coastal California, where it did not occur historically. Planting native flowering plants which provide nectar resources for monarchs and other pollinators are appropriate for nearly all restoration projects.

For more detailed guidance on where and how to manage and restore habitat for monarchs, please refer to the Xerces Society's *Managing for Monarchs in the West: Best Management Practices for Conserving the Monarch Butterfly and its Habitat* (available at www.xerces.org/monarchs). Additional resources available on the website include:

- Monarch Nectar Plant Guides
- Milkweed Seed Finder

The Western Monarch Milkweed Mapper website (www.monarchmilkweedmapper.org) provides information including an interactive map of milkweed and monarch occurrences and western milkweed species profiles. It also includes information about the habitat suitability modeling work which is a joint project of US Fish and Wildlife, Xerces Society, and the University of Nevada–Reno.

Implementation

Restoration of rangelands in the West is often implemented as emergency stabilization and rehabilitation efforts post-wildfire, or as a way to reduce fuels and fire risk. Since these are major foci for restoration in western rangelands, the following sections concentrate on ways to incorporate pollinator-friendly practices into these restoration efforts.

Planting Time

Develop flexible and adaptive restoration plans to allow wide-scale implementation to occur when weather conditions are the most conducive to seed germination and seedling establishment. In arid rangelands, being able to take advantage of precipitation events, or other climatic factors that improves seeding or planting success, will increase the chances that native plants will establish to provide long-term habitat for pollinators.

Native plant seeds have a variety of species-specific germination requirements—scarification, cold stratification, or a specific amount of rainfall, for example. Due to this, there is no one-size-fits-all recommendation on seeding time or strategy. Native seeds with very specific germination requirements may need to be treated prior to direct seeding, or seeded separately. In general, seeding is often implemented in the fall or winter to allow seeds to undergo natural moisture and temperature cycles that stimulate germination (stratification). Rangeland seeding can even be completed over snow. Consult regional botanists or plant material specialists to determine optimal seeding times based on the species, your region, and climate conditions. When planting plugs or container materials, generally aim to plant in the fall or winter when plants are dormant.

Seeding

Seeding Rates

The seeding rate (number of seeds per square foot) will vary by region, the circumstances in which the seed is being applied (post-fire aerial vs. drilling, broadcast seeding after invasive species removal, planting high density patches across the landscape, etc.), and by how much locally sourced native seed is available. General recommended seeding rates are between 25–50 seeds per square foot.

Seeding Methods

Selecting the most effective seeding method is crucial for successfully establishing annual or perennial flowering plants to improve habitat for pollinators and other wildlife. To determine the seeding

Sagebrush restoration planting in Nevada.

Great purple hairstreak nectaring on rabbitbrush in eastern Nevada.

method(s) most appropriate for the site, consider existing conditions such as the remnant seedbank, invasive species presence and abundance, pre-fire vegetation, fire and land use history, weather cycles, and drought (Shaw et al. 2011). Because the seeds of forbs are often smaller and have more specific germination requirements than those of grasses, seeding depth is an important consideration. Small seeds of some perennial forbs, for example, require very shallow planting depths that can be achieved with the appropriate seed drill (minimum-till) or method (aerial or ground broadcast). No-till drills and some modified rangeland drills are generally more effective at establishing flowering plants than broadcast seeding.

The appropriate seeding method also varies by ecoregion, past land use and existing conditions. For example, aerial seeding at low-elevation sites in the sagebrush biome is generally ineffective at establishing native plants (Knutson et al. 2014; Pyke et al. 2017); it is more successful in high elevation and/or higher precipitation sites. In low elevation sagebrush habitats, using a seed drill or planting bare-root perennial plants may be the most cost-effective way to establish native plants (see "Habitat Configuration" on page **60**). Consult Knutson et al. (2014) and Ott et al. (2016, 2017), and references therein, for more detailed information regarding optimal rangeland seeding methods.

The following seeding methods may be appropriate for the varying site conditions and seed mixes used to restore rangelands:

- Rangeland drill seeders (generally effective for flat and accessible terrain)
  - Minimum-till drill (preferable for smaller seeds of most flowering plants; shallow planting depths that maintains existing native vegetation)
  - Conventional drills (preferable for larger seeds, deeper planting depths)
- Seedbed manipulations (chains, harrows, cables, drags etc. are important to use to break up highly water repellent soils that occur post-wildfire in pinyon-juniper habitat [Zvirzdin et al. 2017; Fernelius et al. 2017])
- Aerial broadcast (generally minimally effective, but can be useful in difficult or remote terrain, and is more effective at higher elevations or in areas of higher precipitation)
  - Consider using seed pillows to increase aerial seeding efficacy (Madsen et al. 2016).
- Ground broadcast (appropriate for small scale restoration)

Interseeding

In some areas, interseeding is the best way increase the diversity and abundance of floral resources. This may be appropriate for areas that have been subject to overutilization by livestock grazing, long-term mowing, or other vegetation-altering management—as well as natural disturbances such as wildfire that over time have reduced the availability of or exhausted the seed bank of native forbs. It can also help to fill in bloom gaps, such as too few fall-blooming or mid-summer plants crucial for a diversity of pollinators, or add species that can provide nest sites for tunnel-nesting bees.

Interseeding can be low maintenance and successful under certain circumstances, but often it requires thoughtful management. Successful interseeding relies on disturbance (e.g., seeding using a

CASE STUDY

Box 7: Pollinators, Restoration, and Ungulate Research at the US Forest Service Starkey Experimental Forest and Range

Contributed by:

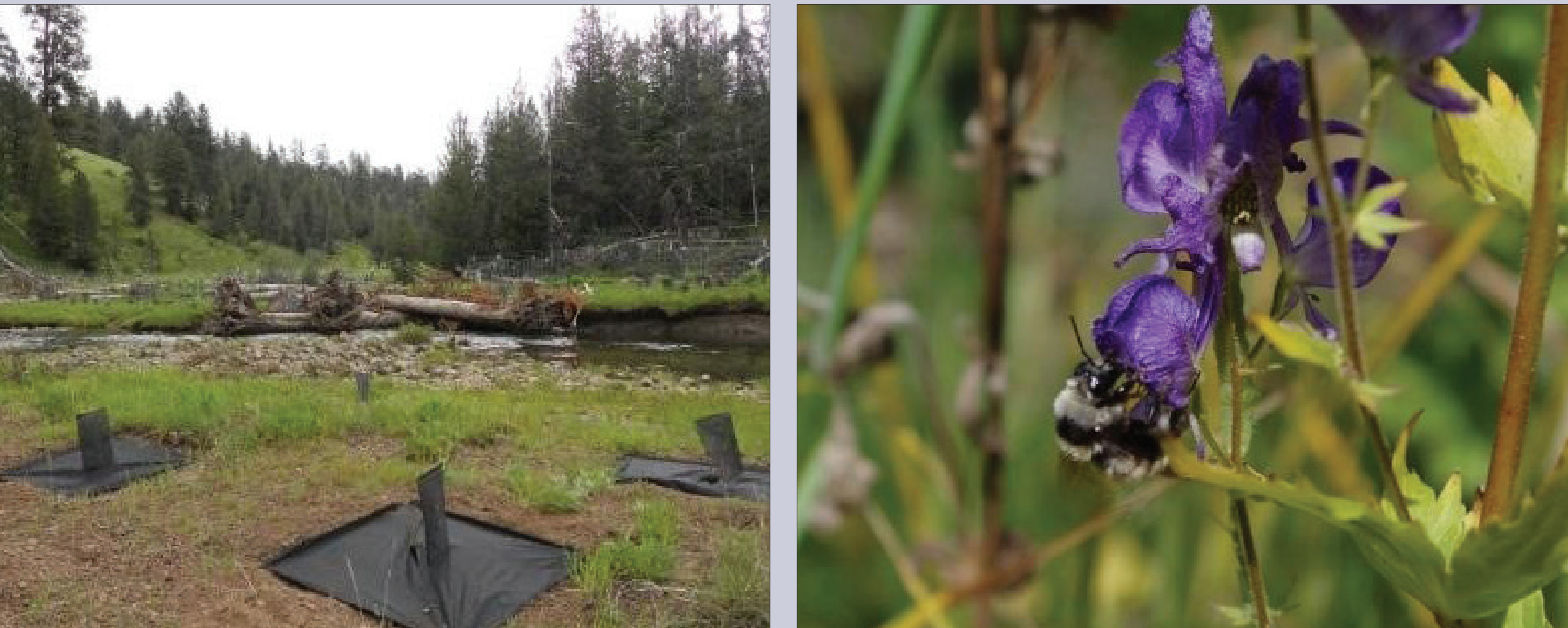
Sandra DeBano, Invertebrate Ecologist
OSU Department of Fisheries and Wildlife
sandy.debano@oregonstate.edu; 541-567-6337

Mary Rowland, Research Wildlife Biologist
US Forest Service Pacific Northwest Research Station
La Grande Forestry & Range Sciences Laboratory
mrowland@fs.fed.us; 541-962-6582

The Starkey Experimental Forest and Range in the Blue Mountains of eastern Oregon is the site of a multi-institutional, interdisciplinary research effort to understand the impacts of riparian restoration and ungulate grazing management on Meadow Creek, a significant salmonid-bearing stream in the Blue Mountains of eastern Oregon. A large restoration project involving in-stream improvements and planting >50,000 native trees and shrubs provides an ideal setting for studies of ecological restoration. An innovative grazing exclosure system allows investigators to partition out effects of livestock (cattle) and native ungulate (deer and elk) grazing. Collaborators from various federal and state agencies and several universities work together to monitor responses of multiple groups, including fish, plants, small mammals, and wild bees.

Land managers strive to balance multiple ecosystem services, including hunting, timber production, and livestock grazing. Recent interest in pollinators adds to the complexity of management decision-making. As a way to address this, a research program at the Starkey Meadow Creek site was developed specifically to address basic gaps in knowledge about one group of pollinators—wild bees. Research focuses on 1) describing wild bee community dynamics in an environment free of pesticides; 2) quantifying the presence and abundance of species of concern, such as the western bumble bee; and 3) investigating how management actions (e.g., restoration, ungulate grazing, invasive plant removal) impact wild bee abundance and diversity. The results from this research can and are being used to inform restoration and land management regionally and nationally to improve pollinator habitat.

Restoration plantings (left) and a bumble bee documented (right) at the Starkey Experimental Forest and Range.



This research provides key information on several topics for which information is scarce or lacking, including:

- Increased understanding of riparian bee communities and factors that influence them
- Responses of native bees to riparian restoration
- Interactions of native ungulates, livestock, and native bees, partitioning out relationships across each
- Abundance and seasonal distribution of the imperiled western bumble bee in riparian systems
- Distribution and abundance of other native bumble bee species, many of which are little studied
- Baseline data on bee communities in a system with no agricultural pesticides or other contaminants for comparison to altered natural systems
- Native bees and their resources in a riparian ecosystem common throughout the Pacific Northwest, one that has undergone riparian restoration for the purpose of improving habitat for salmonids

Key Findings to Date

- Bee communities of Meadow Creek are spatially and temporally dynamic and taxonomically diverse: more than 180 species of native bees have been identified to date.
- Large shifts in abundance and species composition across seasons indicate that "snapshots" of bee communities at one time point will not adequately represent the full bee community.
- Western bumble bee, an imperiled species, was confirmed in each of the four years of sampling, but is rare. Of the 16 bumble bee species identified to date, the western bumble bee makes up only 1% of all individuals sampled.
- An extensive literature review indicated that, for flowers found along Meadow Creek, there is high potential overlap in diets of native bees and domestic and wild ungulates, especially bees and elk (DeBano et al. 2016).
- Over 90 flowering species of forbs and shrubs were evaluated for their attractiveness to bees. While many plants were not commonly visited by bees, dozens of others have been identified as important resources for wild bees (Roof et al. 2018).
- Flower morphology, rather than color, is key in explaining which species of bees visit particular species of flowering plant (Roof et al. 2018).
- Grazing by wild ungulates, primarily elk, has had large impacts on some key flower species important to bees, such as *Potentilla gracilis*, but grazing effects on other species are variable.

Field surveys at the Starkey Experimental Forest and Range (left) document species like the declining western bumble bee (*Bombus occidentalis*) (right).

Current Status of Research

- Four years of sampling native bees and flowers have been completed.
- Cattle were introduced into Meadow Creek pastures in 2017, allowing for evaluation of all grazing treatments on native bees and floral resources. Grazing will continue several years.
- Several new projects began in 2018, including one that examines the importance of early blooming shrubs as forage for native bees, and another that applies DNA metabarcoding techniques to pollen analyses to better understand which plants bees are using for food.
- Workshops aimed at local stakeholders and land managers are currently being planned.

seed drill and drag harrows or seeding into herbicide bands), which gives seeds a better chance of bare soil contact and germinating; disturbance afterwards helps suppress dominant vegetation and helps seedlings establish. The amount of suppression required depends on the existing vegetation. It is often difficult to interseed into invasive weeds and introduced cool season grasses because they are generally difficult to suppress. Stochastic factors can influence the outcome (as with every restoration), especially soil moisture and precipitation in arid climates.

Habitat Configuration

Size

Areas of habitat of any size can bring benefits to pollinators. However, the larger the restoration area, the greater the potential benefit to pollinators and pollination services. In rangeland areas where large-scale seeding or planting is not practical (limited access or remote) or resources are limited, then establish a patchwork of high-diversity plantings (nucleation planting) ranging in size from 10 to 100+ square meters, surrounded by a matrix of less intensive restoration such as broadcast or drill seeding, or lower density plantings. Similar patches are recommended for restoration of sagebrush habitat for greater sage-grouse.

Proximity to Remnant Habitat, or Distance Between Restoration Plantings

The flight or foraging range of different species of bees is an important consideration for restoration design, especially for selecting restoration sites and for determining the distance between planting/seeding sites.

- Solitary bees: 100 meters to 1 kilometer
- Bumblebees: several hundred meters to several kilometers
- Butterflies: varies from several hundred meters (some Lycaenidae) to thousands of kilometers (monarch, painted lady).

Nucleation (Patch/Clump) and Corridor Planting

In rangelands affected by large landscape-level disturbances such as wildfire, land managers may have limited resources for landscape level restoration or rehabilitation. Where restoration is not feasible at the landscape level, high-density nucleation planting (also known as patch or clump plantings) or corridors with less intensive planting efforts in the surrounding matrix of habitat may be the most cost-effective way to reestablish native plants on the landscape over both the short- and long-term (Hulvey et al. 2017). Hulvey et al. (2017) suggest that "*concentrating restoration efforts into high-input, strategically located, smaller areas may provide more satisfying outcomes*" for land managers.

Many solitary bees—like this ground-nesting bee—only travel a few hundred meters in their lifetimes, so restoring pollinator habitat close to existing intact habitat benefits pollinators the most.

Restoration post-fire is a great opportunity to establish habitat for pollinators.

- Aim to connect remaining intact/unburnt (skips) of habitat with high-density corridors or nucleation plantings to provide connectivity and serve as "stepping stones" for pollinators and other wildlife (Stanturf et al. 2014; Hulvey et al. 2017).
- Restore the matrix surrounding the high density patches with less intensive methods such as broadcast or drill seeding with native grass and forb species that can both prevent invasion from and compete with invasive species such as cheatgrass.

Post-Fire Restoration and Rehabilitation

Besides soil stabilization, ensure that adequate floral resources are provided the year after a wildfire by seeding quick growing, native annual or perennial flowers.

- Minimize the use of yarrow (*Achillea millefolium*) and flax (*Linum lewisii*) in seed mixes. These widely used post-fire restoration and rehabilitation species may be important components of a seed mix to initially establish native vegetation and suppress nonnative plant invasion, but they attract few pollinators, and cannot support a diverse pollinator community (Cane and Love 2016).
- Consider the appropriate seeding method for the site. **Aerial seeding at low elevation sites in the arid West is generally ineffective at establishing native plants** (Knutson et al. 2014; Pyke et al. 2017); using a seed drill or planting bare-root perennial plants may be the more cost-effective. Seeding is likely to be the most successful and cost-effective use of resources in high elevation sites.
- Reduce or eliminate the use of nonnative grasses in post-fire rehabilitation seed mixes, and instead use native grasses and forbs.

Conifer Fuels Reduction

Fuels reduction projects in combination with prescribed fire in combination (e.g., pile burning after conifer forest thinning or pinyon–juniper removal) may be beneficial to some pollinators by creating a more open canopy forest and increasing patches of higher diversity and cover of herbaceous plants and flowering shrubs (Kleintjes et al. 2004). See Box **9 f**or a case study on how pinyon-juniper removal improves habitat for butterflies.

- Where appropriate maintain open canopy forests with low shrub cover and high diversity of flowering forbs and shrubs. This can be achieved by thinning practices and prescribed fire (Hanula et al. 2016).
- In reforestation, leave some patches of open chaparral, or a heterogeneous patchwork of forest openings with herbaceous or woody flowering plants (Loffland et al. 2017).
- If fuels reduction or thinning will involve the use of pile burning, or fuel understory burns, they

Box 8: Pollinator-Friendly Plant Material Development for Rangelands

Approximately 150 tons of native flowering plant seed is sought out annually to restore federally managed western landscapes, primarily as a way to rehabilitate areas burned by wildfires (Cane 2008). However, this is just a tiny proportion of the seed used in restoration. An analysis of seed mixes used for post-fire rehabilitation of rangelands in 2001, discovered that only 0.5% of the seed was a native flowering plant (Cane 2008)—and the flowering species included in the mixes (e.g., flax and yarrow) do not support a diverse and abundant pollinator community (Cane and Love 2016).

To alleviate this situation, the Great Basin Native Plant Selection and Increase Project, a collaborative partnership of researchers and agencies associated with the Native Seed Strategy and Seeds of Success, identified 16 flowering plants (Table **4** right) native to the Great Basin (including the Intermountain West and Columbia Plateau) and the Colorado Plateau as pollinator plants appropriate for commercial seed increase and rangeland rehabilitation (Cane 2008, 2011). Similar plant-materials research and development is urgently needed across other western rangeland ecoregions, to develop commercially available pollinator-friendly plant materials for large landscape-level restoration or post-fire rehabilitation. This is especially important because the process of getting a native plant species into commercial production can be very slow, taking 10–20 years before a species is produced at a scale adequate for landscape-level restoration or rehabilitation efforts (Olwell and Riibe 2016).

Purple prairie-clover (*Dalea purpurea*) is a pollinator-friendly plant utilized by many species of bees in interior parts of the West.

TABLE 4: Pollinator-Friendly Plant materials (adapted from Cane 2008 and 2011).

Family	Species
Apiaceae	*Lomatium dissectum*
Apiaceae	*Lomatium triternatum*
Asteraceae	*Balsamorhiza sagittata*
Asteraceae	*Crepis acuminata*
Cleomaceae	*Cleome lutea*
Cleomaceae	*Cleome serrulata*
Fabaceae	*Amorpha canescens*
Fabaceae	*Astragalus filipes*
Fabaceae	*Dalea ornata*
Fabaceae	*Dalea purpurea**
Fabaceae	*Dalea searlsiae*
Fabaceae	*Hedysarum boreale*
Fabaceae	*Lupinus argenteus*
Fabaceae	*Lupinus sericeus*
Malvaceae	*Sphaeralcea grossularifolia*
Malvaceae	*Sphaeralcea munroana*
Scrophulariaceae	*Penstemon speciosus*
Polygonaceae	*Eriogonum umbellatum*

*Purple prairie-clover (*Dalea purpurea*)--used by 50+ spcecies of bees (Cane 2011).

should be implemented outside the active season of most pollinators (October through February).

- Minimize soil disturbance (disking, tilling) during restoration activities to avoid damage to ground-nesting species, and spreading invasive plants.
- Leave some coarse woody debris for pollinator nest sites (standing snags, varying sizes of logs).

Riparian Restoration

- Consider passive restoration of riparian areas such as excluding livestock grazing for 3–5 years, or until native vegetation has recovered to desired conditions (Kauffman et al. 2004; Herbst et al. 2012).
- Active restoration should aim to increase the diversity and abundance of flowering plants, including planting flowering shrubs, and bee nesting sites such as pithy or hollow stemmed plants. See "Plant Species Selection" on page **50**.
- Some important riparian restoration species that occur broadly across western rangelands include native willow (*Salix* spp.), rose (*Rosa* spp.), elderberry (*Sambucus* spp.), currant (*Ribes* spp.), and goldenrod (*Solidago* spp.).
- For sagebrush rangelands consult the following technical note for ways to restore riparian or spring-fed meadows “Hand-Built Structures for Restoring Degraded Meadows in Sagebrush Rangelands Examples and lessons learned from the Upper Gunnison River Basin, Colorado” (https://bit.ly/2KfG105)(Maestas et al. 2018).
- Install fences or cages to protect riparian plantings from both native ungulates, and livestock until plants are established to survive grazing (DeBano et al. 2016 Averett et al. 2017). See Box **7** for more information on active research on the topics of riparian restoration, pollinators, livestock, and native ungulates.

Long-Term Site Maintenance

- Defer livestock grazing for 2–3 years after project implementation, or use the same deferment recommendations for wildfire in your region. Native ungulates such as deer and elk may also need to be excluded from the site; plantings should be caged or fenced until plants are well-enough established to withstand grazing.
- Implement integrated pest management practices to minimize the amount of pesticides used on a site. See the "Pesticides" section on page **70**.
- To protect pollinator nesting and overwintering habitat, do not disturb the soil (disking, tilling) in more than one-third of the site each year.
- If feasible, water or irrigate plantings during the first year to increase survival of plants. This is particularly important in arid regions of California, Nevada, and the Southwest.
    - Take advantage of high-precipitation years to plant, as greater precipitation may lead to higher survivorship of plants and seeding efforts in restoration projects.
    - Potential irrigation systems include deep pipe and porous hose that are low maintenance. These can increase planting survival, especially in arid environments (Bainbridge 2002, 2012).
    - Consider mulching transplants to retain moisture—but do not mulch seedlings.
    - Plant or seed in climatic microsites that will remain moist longer into the summer, such as north-facing slopes or gullies that will retain snow or water.

Agricultural Lands and Leases

Agricultural areas in the West include important habitat for pollinators (see Box **6** for a discussion about integrating monarch habitat). Implementing pollinator-friendly practices, such as restoring habitat for pollinators on lands leased for agriculture on rangelands, is not only good for pollinators, it is also economically valuable. In 2000, the economic value of pollinators was estimated to be $20 billion in the United States alone (Gallai et al. 2009), and pollinators are necessary for the production of over two-thirds of all crops worldwide (Klein et al. 2007).

Detailed recommendations for pollinator habitat restoration in agricultural landscapes is outside the scope of this document, but below you will find many useful resources to help you establish habitat in public lands leased for agriculture, or to provide recommendations to private agriculture producers adjacent to public land that could provide habitat connectivity on the landscape.

- Xerces Society resources available include:
  - Pollinator Conservation Resource Center, which includes regional information about plant lists, habitat conservation guides, and more.
  - Pollinator Habitat Installation Guides by region
  - Pollinator Conservation Seed Mixes by region
- Bee Better Certified is a certification program started by the Xerces Society which helps farmers and food companies integrate habitat for bees and other pollinators into agricultural lands. Find information at www.beebettercertified.org.

There are also financial incentives and technical assistance programs for private (and some public) landowners to help defray the cost of restoring pollinator habitat:

- Farm Bill Programs
  - USDA Natural Resource Conservation Service offers several programs including Conservation Stewardship Program, Environmental Quality Incentives Program, and Working Lands For Wildlife Monarch Conservation Program (www.nrcs.usda.gov).
  - See the Xerces Society's *Using Farm Bill Programs for Pollinator Conservation* for full details of appropriate conservation practices.
  - USDA Farm Service Agency offers the Conservation Reserve Program (www.fsa.usda.gov).
- US Fish and Wildlife Service runs the Partners for Fish and Wildlife Program (www.fws.gov/midwest/partners).

Partnering with farmers to create and maintain habitat for pollinators and other beneficial insects in agricultural areas is an integral part of the Xerces Society's work. Many Xerces Society staff serve as partner biologists for the USDA Natural Resources Conservation Service, where they provide one-on-one technical assistance, workshops, and other support for private landowners and NRCS conservation planners. Check out the Xerces Society website, www.xerces.org, to learn more and connect with Xerces' conservation professionals.

A riot of lupine and poppies bloom in a strip of pollinator habitat installed between the forest and agricultural lands in California.

CASE STUDY

Box 9: Pinyon and Juniper Removal Improves Habitat for Butterflies

Sagebrush steppe rangelands are threatened by encroaching pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) trees as the result of a long history of fire suppression. Pinyon–juniper encroachment can reduce livestock forage production and degrade habitat for wildlife including pollinators (Davies et al. 2014a). In response, land managers are focusing on removing encroaching trees, especially to restore habitat for sagebrush obligate species such as the pygmy rabbit and the greater sage-grouse (Larrucea and Brussard 2008; Bates et al. 2016). Removal of pinyon and juniper trees from arid shrublands frees up groundwater resources (Roundy et al. 2014b) and increases solar radiation for herbaceous plants, resulting in an increased abundance of flowering plants and shrub, which also benefits pollinators (Kleintjes et al. 2004; McIver and Macke 2014; Bates et al. 2016).

One study explored the response of butterfly communities to a variety of restoration treatments in sixteen sagebrush habitats scattered throughout the intermountain west (McIver and Macke 2014). The goal of restoration treatments in this study were to reduce pinyon–juniper encroachment and cheatgrass (*Bromus tectorum*) invasion, and increase the cover and diversity of native plants. Researchers surveyed butterflies in the year prior to and for four years after the three treatments, prescribed fire, mechanical tree removal, and herbicide application. They found that out of all the treatments, prescribed fire led to the greatest increases in the butterfly community. Fire in sagebrush–cheatgrass dominated sites increased butterfly abundance and species richness significantly, with a pronounced increase in skippers and mustard-feeding white butterflies. Fire in areas with mechanical removal of pinyon-juniper tree increased abundance of Melissa blue butterflies (*Plebejus melissa*), largely as a result of an increase in the cover of flowering plants as hosts. The authors of this study note that many Great Basin butterflies are strongly associated with native plants, and any management that results in an increase in the cover and availability of native plants will largely benefit Great Basin butterflies.

Similar studies have reported a positive response of flowering plants (Bybee et al. 2016; Roundy et al. 2014a), and butterfly abundance and diversity (Kleintjes et al. 2004) to mechanical removal of pinyon–juniper in sagebrush and grassland habitats across a wide variety of microclimates. Some of these studies with high pretreatment cover of cheatgrass reported that seeding of native forbs during the restoration process can help to suppress cheatgrass (Bybee et al. 2016; Roundy et al. 2014a). In addition, research shows that to conserve perennial herbaceous vegetation and minimize cheatgrass, pinyon–juniper should be treated when it is less than 40% tree cover (Roundy et al. 2014a). To ensure restoration efforts improve habitat for pollinators, pinyon–juniper removal planning will need to ensure that measures are in place to suppress invasive plants via seeding of native plants or reduce their spread by limiting ground disturbance.

Overall, the removal of pinyon and juniper trees improves pollinator habitat by freeing up already scarce water resources in shrub- or grass-dominated arid plant communities to the benefit of herbaceous flowering plants.

Lupine flowering in a burned pinyon stand in the sagebrush steppe in southeastern Oregon.

Invasive Nonnative Plants


Invasive thistles are often very attractive to pollinators—such as this swallowtail butterfly—when there are not many native blooming plants.

Invasive nonnative plants (referred to as invasive plants hereafter), including those designated as noxious weeds, pose a serious threat to the health and productivity of rangelands in the West. Invasive plants can significantly alter plant community composition, ecosystem processes, soil chemistry, and fire regimes (DiTomaso 2000; Duncan et al. 2004). Invasive plants compete with native plants for resources and can cause significant reductions in the abundance and diversity of pollinators and other herbivorous insects (Samways et al. 1996; Kearns et al. 1998; Spira 2001; Lopezaraiza-Mikel et al. 2007; Hopwood 2008; Zuefle et al. 2008; Moron et al. 2009; Burghardt et al. 2009; Wu et al. 2009; Tallamy and Shropshire 2009; Fork 2010; Hanula and Horn 2011; Fiedler et al. 2012). Generally, native pollinators and other insects prefer to feed on native rather than invasive plants (Hopwood 2008; Burghardt et al. 2009; Wu et al. 2009; Williams et al. 2011; Morandin and Kremen 2013), and native plants support a greater diversity of Lepidoptera compared to nonnative plants (Tallamy and Shropshire 2009). Invasive plants often only provide floral resources for generalist pollinators (Aizen et al. 2008), reduce habitat for specialist pollinators (Traveset and Richardson 2006), and may facilitate establishment of nonnative pollinators (Morales and Aizen 2002). Furthermore, invasive plants may reduce conspecific (same species) pollen deposition on native plants, reducing reproductive output (Litt et al. 2014). Cane (2011) suggest that the greatest threats to pollinators from invasive plants is their ability to displace native vegetation (which may reduce both floral and host plant resources), alter fire regimes, and change soil chemistry through allelopathy (e.g., knapweed [*Centaurea* spp.], tamarisk [*Tamarix* spp.]). Invasive plants on rangelands can also decrease rangeland productivity which may result in economic losses for ranchers (Duncan et al. 2004) and may reduce the abundance of floral, nest, and larval host resources available for pollinators.

Research suggests that invasive plant removal can have positive effects (Baskett et al. 2011; Hanula and Horn 2011; Fiedler et al. 2012; Tonietto and Larkin 2018; Goodell and Parker 2017) on native bees and butterflies. A meta-analysis by Tonietto and Larkin (2018) investigated the overall effects of restoration treatments, including invasive plant removal, on native bees. The analysis found that of all restoration treatments, invasive plant removal had the greatest positive effect on the diversity and abundance of native bees. One study included in the meta-analysis (Hanula and Horn 2011), demonstrates the significant benefits. They found that the removal of Chinese privet (*Ligustrum sinense*) greatly improved habitat for butterflies and bees in riparian forest in southeastern US: Five years after shrub removal, treatment plots had three times as many bees and butterflies compared to control plots.

While invasive plant removal improves habitat for pollinators in the long-term, removal of flowering invasive plants has been suggested as a cause of decline for some pollinator populations by reducing floral resources (Tepedino et al. 2008; Severns and Moldenke 2010; Bezemer et al. 2014; Harmon-Threatt and Chin 2016). Controlling or removing invasive plants is particularly a balance for land managers working in degraded rangeland landscapes where native nectar for pollinators may be scarce. In some landscapes, invasive plants such as Canada and bull thistles may be the only species available as forage in rangelands. Removal of invasive plants under these circumstances may reduce nectar availability for pollinators—but removal of invasive plants is generally more important than the floral resources the plants are providing. To minimize the negative, short-term impacts of removing invasives, a plan should be in place to plant commensurate native floral resources before or immediately after large-scale removal of invasive plants that are known to provide important nectar resources for pollinators.

Invasive plants are often found and spread along roadsides, where some important pollinator plants often grow. For example, common milkweed species and many late-season nectar resources such as sunflowers and rabbitbrush often grow on roadsides. Invasive plants on roadsides are commonly managed with mowing and herbicide applications during times when these plants are actively growing and are being used by pollinators. These management practices have the potential to temporarily remove floral, host, and nest resources and kill immature stages of pollinators, but they are also important to reduce the spread of invasive plants.

Overall, removal of invasive plants with the goal of maintaining or conserving healthy, native plant communities on rangelands is desirable at an ecosystem level, but care should be taken in the short-term to ensure phased removal and replacement with alternative and commensurate floral resources for pollinators. In the long-term, managing to reduce the abundance of invasive plants can increase the abundance and diversity of both native plants and pollinators.

Invasive Nonnative Plant Best Management Practices

Use an Integrated Vegetation Management (IVM) Plan to

- Prevent establishment and/or spread of invasive and noxious plants.
- Make site and plant specific determinations regarding the need for and level of intervention.
- Consider a combination of management techniques (biological, physical, chemical, and cultural practices).
- Ensure treatments are completed in a manner that minimizes risks to nontarget organisms and the environment.

Use Early Detection Rapid Response (EDRR) for New Invasive Plant Occurrences

- Learn more about this approach and EDRR networks on the website (www.invasive.org/edrr).

Nonnative Russian olive flourishes in riparian areas in many parts of the West—including at Crab Creek in Washington.

A crew of volunteers learns how to effectively remove invasive plants without herbicides.

Create and Implement a Revegetation Plan

Before or immediately following invasive plant removal on a large scale, ensure there will be similar native floral resources available for pollinators.

- Replace invasive plants with native perennial plants with similar phenology and pollinator syndrome (bee, butterfly, bird, bat pollinated) as the species targeted for removal. Native perennials may also help deter recolonization of some invasive plants.
- Consider a phased removal and revegetation plan to avoid removing major floral resources.

Prioritize control of invasive plants in habitats with high native plant diversity and abundance, and resiliency to future invasion.

Native pollinator abundance and diversity is positively correlated with the distance from native plant communities.

Minimize Invasive Plant Spread by Limiting Vectors

- There are many vectors for invasive plant spread on rangelands, including wind, water, recreation (boots, bike tires, OHV tires, horses, mules, etc.), livestock (hooves, hair), livestock feed (hay), roads, and cars. The spread of invasive plants can increase in response to disturbances such as fire, recreation, roads, fuels reduction, forest thinning, logging, restoration, floods, and grazing.

Biological Control Insects

One strategy that is utilized to control invasive species, including plants, is biological control. There are three different types of biological control: 1) *classical biocontrol*, where a nonnative insect species is introduced to control a target species, 2) *augmentative biocontrol*, where a native insect species is released to augment local populations of that insect, and 3) *conservation biocontrol*, where managers take actions to support local native populations of beneficial insects.

Conservation biocontrol carries few risks and may be an effective way to control invasive species. The use of classical and augmentative biocontrol insects, however, can have unintended consequences to native plants and pollinators. Introduction of nonnative herbivores may negatively impact native pollinators or other native insects by competing for resources, introduce nonnative parasitoids, and/or reduce resource availability of native plants. Adequate research and regulatory measures are rarely sufficient to evaluate and control the nontarget effects of classical or augmentative biological control insects, and regulatory measures are rarely in place for movement of insects outside of their native ranges. Due to these concerns, classical and augmentative insect biological controls are generally not advised as methods to manage invasive plants and conserve pollinators on rangelands. The following references provide an overview of the pros and cons of using insects as biological control agents: Henneman and Memmott (2001), Barratt et al. (2010), and Kaser and Heimpel (2015).

Recommendations if classical or augmentative biological control insects are used:

- Employ biological control agents that have not demonstrated host plant expansion, especially switching from the nonnative target plant to a native plant.

- To conserve native thistles, avoid additional release of the Eurasian weevil (*Rhinocyllus conicus*), which is most often used to control musk thistle infestations.
- Consider possible negative interactions of biological control insect species with sensitive pollinators.
- If using augmentative biocontrol insects, source them as locally as possible.

Pesticides (Herbicides, Insecticides, Fungicides)

In the West, management practices include the use of pesticides—which include herbicides, insecticides, and fungicides—to remove unwanted vegetation from roadsides, control invasive weeds, and reduce outbreaks of insects that compromise rangeland productivity for livestock. Herbicides are, by far, the pesticides applied most often on US Forest Service lands (Cota 2004); the majority of insecticides and fungicides are applied in nursery settings, not on rangelands. However, insecticides such as carbaryl are sometimes used on rangelands to control grasshoppers, Mormon crickets, and in some limited cases, bark beetles, and borax-based fungicides are used to prevent the spread of root rot in stumps after logging. Pesticides can have both direct (lethal and sublethal) effects and indirect (harm via the effect on another species) effects on pollinators (Thompson 2003;Decourtye et al. 2004; Desneux et al. 2007; Kopit and Pitts-Singer 2018).

Pesticide use has increasingly been identified as a potential factor in pollinator declines (e.g., (Forister et al. 2016; Thogmartin et al. 2017). Tens of thousands of pesticide products are on the market today, which makes assessing their impacts—direct, indirect, additive, or synergistic—nearly impossible. Ultimately, land managers must maintain a delicate balance between managing invasive and noxious species and protecting areas from insect pest outbreaks while still protecting pollinators from potentially harmful pesticides.

Pesticide exposure can occur in areas where an application has occurred as well as in areas that have not been treated but have become contaminated when chemicals drift, leach, or otherwise move off the intended site (Chauzat et al. 2006; Gill et al. 2012; Krupke et al. 2012; Botias et al 2015; Hladik et al 2016; Long and Krupke 2016). Multiple avenues for exposure exist, including direct exposure, residual contact exposure, via contaminated nectar and pollen, or through contaminated nesting materials. Because potential hazard and exposure routes can vary significantly, land managers should be cautious of applying the results of pesticide evaluations—which are often conducted with managed honey bees or bumble bees—to other taxa. For example, many native bee species are much smaller than honey bees and can be affected by lower doses. The majority of these native bees are solitary, with a single female that provisions her nest. If she is killed by pesticide exposure her nest is lost. In contrast, honey bees have thousands of workers and losses may not necessarily harm the colony. Solitary bees are also more sensitive to pesticides that are residual in the soil, because they nest directly in soil, or expressed in plant tissues, becasue some use plant materials to build their nests (Kopit and Pitts-Singer 2018).

Risk can also vary by life stage, and larval bees may be impacted by consuming food contaminated

Narrowleaf milkweed sprayed by herbicide.

Systemic insecticides—such as neonicotinoids—can be taken up in nectar, putting visiting bees at risk.

with pesticides (Johansen et al. 1990; Abbott et al. 2008). For example, solitary bee larvae may be more at risk to exposure than honey bee larvae they are typically fed raw pollen and/or undiluted nectar, which may contain higher concentrations of pesticide residues than the brood food that honey bees are fed (Kopit and Pitts-Singer 2018). Butterfly larvae (caterpillars) may be at higher risk of ingesting residue left on leaves since they feed directly on plants that may have been treated.

Herbicides and fungicides generally have low acute toxicity to adult bees (Johnson 2015), but indirect and sublethal effects are increasingly documented. While testing herbicides for their sublethal effects is not the norm, a few herbicides have demonstrated sublethal effects (Cousin et al. 2013; Balbuena et al. 2015). More commonly, herbicides are associated with indirect effects on pollinators as they can kill the floral resources that pollinators depend on, effectively reducing the amount of plants they use for foraging and egg laying (Kremen et al. 2002; Tscharntke et al. 2005). Some fungicides can be lethal to bees (Zhu et al. 2015), while others have been shown to cause sublethal effects that, in some cases, could lead to population level declines (Bernauer et al. 2015; Park et al. 2015; Sánchez-Bayo et al. 2016; Traynor et al. 2016; McArt et al. 2017a). Pollinators exposed to multiple pesticides at the same time may also experience synergistic or additive effects (Pilling et al. 1995; Iwasa et al. 2004; Biddinger et al. 2013).

Pesticide Best Management Practices

In order to maintain a balance between responding to pest and invasive plant pressures and the risk of pesticide exposures to wildlife, pesticides should be used within an integrated pest management (IPM) plan. Federal land management agencies such as the US Forest Service already work within an IPM framework focused on long-term pest prevention. IPM plans can include diverse management techniques such as biological, physical, and cultural practices. Pesticides should be used only after monitoring indicates that they are needed to respond to a pest at levels that pose serious economic or public health threats. Furthermore, treatments should be made with the goal of removing only the target organism; pest control materials should be selected and applied in a manner that minimizes risks to nontarget organisms and the environment. Below are specific recommendations to limit risk when pesticides are being used.

Nonnative thistle and cheat grass are tenacious invasive species on many rangelands in the Great Basin.

General Recommendations

- Whenever possible, prevent conditions that would allow invasive species to reestablish, thus necessitating further pesticide use.
    - Monitor livestock use to prevent overgrazing which can lead to establishment of invasive plants.
    - Wash all equipment prior to accessing a new site.
    - Minimize the use of mechanized equipment, which brings in invasive seeds from other areas and tears up the soil, allowing seeds from invasive plants to germinate.
        - Utilize mobile weed washing stations when and where mechanized equipment is used, and wash all equipment prior to accessing a new site.
    - Prevent further introductions of nonnative bees, parasites, and pathogens.
- In planning your pest management strategy, determine the types of pollinators in the project area (including any sensitive species) and their vulnerability to pesticides.
- Screen commonly used pesticides to determine which ones have the lowest toxicity to bumble bees and other pollinator species of interest (Zhu et al. 2015).
    - Three tools developed to help determine toxicity
        - UC IPM Bee Precaution, http://www2.ipm.ucanr.edu/beeprecaution/
        - Pesticide Risk Tool, https://pesticiderisk.org/
        - PNW Extension publication 591, *How to Reduce Bee Poisonings from Pesticides* (Hooven et al. 2013), which is now also a smartphone app, www.catalog.extension.oregonstate.edu

- Do not use pesticides known to be toxic to bees or other pollinators when at-risk species are present.
- Avoid applying multiple pesticide formulations at any given time. Some pesticides can interact to increase toxicity to bees and other pollinators.
- Apply pesticides at the lowest legal effective application rate.
- Time pesticide applications to avoid pollinator exposure.
  - Do not apply pesticides when flowers are in bloom and/or pollinators are present.
    - Consider the bloom times of plants in and adjacent to the treatment area, and avoid pesticide applications at those times (for example, apply pesticides in fall or winter when floral resources are less available and pollinators may be less active).
    - Apply pesticides at night when pollinators are generally not active. Keep in mind that some species, such as bumble bees, can be active later in the evening, and many moths and other beneficial insects are nocturnal.
  - Avoid pesticide applications during cool, damp periods or when dew is present, as this can extend a pesticide's period of toxicity.
  - Consider the residual activity and release time of the pesticide product being used. Some products have a slow-release system that can last hours to days. Avoid using pesticide products with long residual toxicities.
- When possible, include spatial or vegetative buffers around areas used by pollinators for foraging, nesting, or overwintering. If using a vegetative buffer, ensure it uses plants that are not attractive to pollinators (e.g., grasses and conifers).
- Use the least hazardous formulation available.
  - Dust, wettable powders, and microencapsulated formulations are most hazardous to bees because they are similar in size to pollen and can stick to hairs on a bee's body.
  - Granular formulations are generally the least hazardous to bees.
- Take precautions to avoid off-site movement, especially into sensitive sites.
  - Conduct applications on calm days when wind speed is <10 mph (avoid applications during gusty or sustained high winds).

Time pesticide applications to fall outside of the bloom time of pollinator-attractive plants.

Techniques like spot spraying and timing applications to avoid pollinator exposure can help reduce negative impacts of herbicides on foraging pollinators and ensure only targeted plants are treated.

  - Apply pesticides as directly and locally as possible, using targeted application methods.
  - Avoid application during a temperature inversion and when conditions are likely to cause evaporation.
  - Avoid aerial applications and mist blowers whenever possible. Consider using backpack sprayers and applying from the ground instead.
  - On boom sprayers, use the lowest effective pressure and largest droplet size possible. Set nozzles low so they operate just above plant height.
  - If the pesticide use is adjacent to but not in habitat, create a no-application zone
- Consider removing floral blooms in a treatment area prior to pesticide applications, as this can help reduce the number of pollinators in the area once treatment does occur.

Herbicides

While they do not target insects, some herbicides can harm or kill bees directly, especially when bees are exposed during application or while foraging. More likely, and better documented, is the risk that herbicides can reduce or eliminate plant resources needed by pollinators for foraging, nesting, and egg laying (Forrester et al. 2005; Russell et al. 2005; Dover et al. 2010). Kearns et al. (1998) note that in some cases, herbicides may be even more detrimental to wild bees than insecticides since they reduce these critical resources, sometimes across a broad area or when few other floral resources are available. Land managers can reduce herbicide use and subsequent impacts of herbicides on pollinators by using selective herbicides, spot spraying the applications, and timing applications to both avoid pollinator exposure and target the most vulnerable life stage of the weed. Herbicide applicators should be trained to ensure only the target plants are treated. For example, native thistles are important floral resources for a variety of pollinators, and can often be mistaken for target invasive thistles (Eckberg et al. 2017). In areas such as hay fields, limiting herbicide applications in field margins benefits insect populations in the field borders and adjacent habitats.

Herbicide-Specific Recommendations

- Consider the ecological benefits of plant species that have historically been managed as weeds in rangelands and other natural areas, such as milkweeds (DiTommaso et al. 2016).
- When available, use selective herbicides that are targeted to the species that need treating.
- Use targeted application techniques.

    - Selectively control undesirable plants with spot treatments, frill treatment, weed wipe, or other well-targeted techniques to avoid nontarget species.
    - Avoid broadcast applications of herbicides which can remove nontarget floral and host plant resources and be incredibly detrimental to the local pollinator community.
- Keep applications on target and minimize drift.
    - Carefully choose and calibrate your spray nozzles to minimize drift, ensuring only target plants are treated.
    - Reduce impacts on nontarget vegetation that provides food and shelter for pollinators.
    - Where possible, utilize spatial or vegetative buffers around pollinator habitat.
- Train staff and contractors in plant identification. The ability to recognize native plants as well as invasive weeds will reduce unintended damage to nontarget plants.
    - Consult regional botanists to develop training materials and the Xerces Society's *Native Thistles: A Conservation Practitioner's Guide* (Eckberg et al. 2017).

Sphinx moth nectaring on thistle.

- Apply herbicides during the plant life stage when a weed is most vulnerable.
    - Plants should not be sprayed while they are in flower or after they have gone to seed.
    - This practice alone can greatly reduce herbicide exposure for the local pollinator community.
- Apply in the early morning or in the evening when pollinators are less active, but not during mid-day when bees and other pollinators are most active (Hooven et al. 2013), especially if the optimal time to spray the target plant is when it is flowering.

Insecticides

Rangelands can be susceptible to a number of insect and other invertebrate pests that defoliate, bore into, or otherwise harm trees and other natural resources. Many of these pests are native, and some, such as gypsy moth, are invasive. The US Forest Service has outlined specific management guidelines for most of these pests often including rigorous monitoring programs to determine if and when a pest population is present at harmful levels. Control options include cultural, biological, and chemical options.

The three insectides used in the greatest quantity and/or widest spatial scale by the US Forest Service are carbaryl, diflubenzuron, and permethrin (Cota 2004). Carbaryl is typically used to treat outbreaks of grasshoppers or in rare instances, bark beetles; it is highly toxic to honey bees (Bond et al. 2016). Diflubenzuron (contained in Dimilin products) is an insect growth regulator used most commonly for fly and mosquito control (NCBI 2018); arthropods are most susceptible to Dimilin in the pre-molting stage (CU, MSU, OSU, and UCD 1993). Permethrin, sometimes used used for bark beetle management, is also highly toxic to bees and other beneficial insects (Toynton et al. 2009).

Use IPM strategies to prevent pest pressure, establish action thresholds, and consider multiple management techniques. If an insecticide application is determined to be necessary, seek the least-

Pollinator habitat—including monarch habitat—should not be installed in areas treated with neonicotinoids were applied in the past two years.

toxic targeted options to minimize impact. At present, neonicotinoids do not appear to be used very frequently or in high quantities on western rangelands, and continuing this practice will benefit pollinators.

Insecticide-Specific Recommendations

- Evaluate the range of management techniques (e.g., chemical, physical, and mechanical) in order to select the most effective feasible management method for the target pest.
- Follow label recommendations to avoid off-site movement and limit environmental hazards.
- Apply insecticides at the lowest legal effective application rate.
- When available, choose targeted insecticides; avoid use of broad spectrum insecticides, and choose the least-toxic formulation.
    - Avoid dusts, wettable powders, and microencapsulated products (the latter of which may be gathered by foraging bees and brought back to the nest).
- Do not apply insecticides that are toxic to pollinators when plants are flowering in an area.
- Do not use systemic insecticides, such as neonicotinoids (especially the long-lived, highly toxic nitroguanidine neonicotinoids, clothianidin, dinotefuran, imidicloprid, and thiamethoxam).
- Do not use seed treated with systemic insecticides.
- Do not plant pollinator habitat in locations where neonicotinoids were applied within the previous two years (this includes areas planted with treated seeds), as neonicotinoids could still be present in soil or plant tissues.

Grasshopper and Mormon Cricket Management

Grasshoppers and Mormon crickets are considered pests when their populations reach high levels, although limited research suggests that under some circumstances grasshopper outbreaks may not be detrimental to rangeland production (Thompson and Gardner 1996). These insects are generally managed using one of three chemical treatments: carbaryl, diflubenzuron, or malathion. Diflubenzuron acts as an insect growth regulator and thus, is most toxic to insects in their larval or pre-molting stages. Carbaryl and malathion are adulticides and have been demonstrated to be highly toxic to bees and other beneficial insects (Gervais et al. 2009; Bond et al. 2016). Before grasshopper or Mormon cricket management occurs, the need for management should be established. Furthermore, some research suggests that grasshoppers could be managed without insecticides by carefully timing fire and grazing to manage vegetation and reduce habitat suitability for target species (Fielding and Brusven 1995; Branson et al. 2006). Still, grasshoppers have species-specific responses to grazing and fire, and more research is needed to develop species- and region-specific management treatments that use alternatives to pesticides (Vermeire et al. 2004).

Grasshopper and Mormon Cricket Management Recommendations

- Use insecticides only after it is judged that an outbreak will have severe economic-level impacts.
- Implement frequent and intense monitoring to identify populations that can be controlled with small ground-based pesticide application equipment.
    - Monitor sites before and after application of any insecticide to determine the efficacy of the pest management technique as well as if there is an impact on water quality or nontarget species.
- Limit applications to ground-based application of diflubezuron or carbaryl granular formulations targeted to infested sites. Avoid aerial applications. Target applications to infested sites.
- Avoid using malathion and liquid carbaryl.
- Include large buffers around all water sources, including intermittent and ephemeral streams, wetlands, and permanent streams and rivers – which often support a high diversity of pollinators - as well as threatened and endangered species habitat, honey bee hives, and any human-inhabited area. For example, Tepedino (2000) recommends a three-mile buffer around rare plant populations, as many of these are pollinated by solitary bees that are susceptible to grasshopper control chemicals.
- When aerial applications cannot be avoided, take precautions to limit drift.
    - Fly at the lowest height and slowest speed possible.
    - Use large droplets and low pressure.
    - Conduct applications when wind speeds are between 2 and 10 mph. Do not apply on gusty days.

Grasshoppers on rangeland in Nevada.

Fungicides

Although fungicides have commonly been thought to pose little risk to pollinators, recent research suggests otherwise. Increasingly, fungicides are accounting for large percentages of total pesticide residues in bee bread and other hive materials (Traynor et al. 2016; McArt et al. 2017b). Some fungicides (such as ethanethiol or ethyl mercaptan) have sublethal effects on bees, including negative effects on foraging, reproduction, and larval survival (Ladurner et al. 2005; Sprayberry et al. 2013; Zhu et al. 2014; Bernauer et al. 2015). Particularly concerning are studies showing synergistic effects of fungicides with other pesticides, particularly neonicotinoids and pyrethroids (Pilling et al. 1995; Sgolastra et al. 2017). On US Forest Service lands, fungicides—primarily borax (sodium tetraborate decahydrate)—are used for disease and fungus control, and applied over relatively small areas (Cota 2004). For example, sodium tetraborate decahydrate is the active ingredient in some fungicides that are applied to cut stumps to prevent the spread of root rot fungus. While borax-based fungicides have not been linked with harm to pollinators, land managers can still exercise caution by limiting application of fungicides to nonbloom periods and avoiding tank mixes and other dual applications that may lead to negative synergistic effects.

Fungicide-Specific Recommendations

- Do not apply foliar fungicides to pollinator habitat during bloom when pollinators are likely to be present.
- Do not apply triazole fungicides in combination with insecticides, as this can potentially harm bees by compromising their capacity to extract sufficient energy from their diet (Mao et al. 2017).
- Avoid the use of fungicides containing captan or ziram, as these have been shown to affect brood growth and development in honey bees (Mussen et al. 2004).
- Do not use tank mixes that combine demethylation inhibitor fungicides with either pyrethroid or neonicotinoid insecticides. These combinations may synergize, becoming even more toxic to bees than either chemical on its own (UC 2018).
- Avoid the use of ergosterol inhibiting fungicides, which have been linked to the spread and abundance of honeybee pathogens and other diseases, and may be a concern for other bees as well (Sánchez-Bayo et al. 2016).

Managed Pollinators

Managed pollinators are critical for the pollination of many agricultural crops and honey is an important industry. However, as more areas of natural habitat are converted to agricultural and suburban uses, the pressures to use public lands and other natural areas for placing honey bee hives and as a source for collecting native bees (e.g., mason bees) for commercial purposes are increasing. This can be an issue because managed pollinators can compete with native pollinators for resources directly or indirectly by affecting the plant community and transmitting diseases. A recent review of the literature by Mallinger et al. (2017) reported that a majority of studies identified potential negative effects of managed bees on native bees via pathogen transmission and competition. The authors also found "managed bees within their native range had lesser competitive effects, but potentially greater effects on wild bees via pathogen transmission." This is of particular concern for areas with declining pollinator species, especially bumble bees. Thus, there is a need for evidence-based decision making by land managers to decide whether managed bee hives and bee collections are appropriate, and if so, the timing, duration and numbers of hives or collections that should be allowed.

A honey bee colony in a managed hive.

Honey Bees

The question of whether introduced honey bees belong on public lands and natural areas in North America, many of which are in rangeland use, has been debated for decades (Pyke 1999, and references therein). Colony collapse disorder in honey bee hives helped to raise public awareness of the plight of pollinators in the 2000s,

A single honey bee hive may contain tens of thousands of worker bees, relying on a large quantity of nectar and pollen from nearby flowering plants.

and beekeepers—professional and hobbyist alike—are some of the most engaged advocates for pollinator conservation. It is important to recognize, however, that while there are some common goals (e.g., reducing pesticide contamination), the conservation of honey bees is more of an agricultural concern than a conservation concern (Geldmann and González-Varo 2018).

Honey bees may pose direct risks to native bees including competing for floral resources and transmitting diseases (Cane and Tepedino 2017), and indirect risks including changes to plant communities (Mallinger et al. 2017). Research continues to find correlative evidence that honeybees can directly compete with native bees for floral resources. Cane & Tepedino (2017) calculated that during a single summer, "a 40-hive apiary residing on wildlands for 3 months collects the pollen equivalent of four million native bees." Honey bees may also transmit diseases to native bees (e.g., spread of deformed wing virus from honey bees to bumble bees causing wing damage) (Fürst et al. 2014; Manley et al. 2015). On rangelands native bees provide substantial ecological services in the form of pollination to nearby agriculture, thus threats to them, such as threats from honey bee apiaries on rangelands or other natural areas, may also pose economic risks. More research needs to be conducted to determine appropriate stocking rates and the least intrusive locations of honey bee hives for native bee health.

Honey Bee Hive Placement
Where local and federal laws permit the placement of honey bees, and managers are deciding whether to include hives on their land, we suggest that managers consider the following issue. See Hatfield et al. (2018) for more details of potential impacts of honey bees.

- Honey bee hives in natural areas pose a direct threat to native bees by spreading disease, depleting pollen availability for larval provisioning (reduced carrying capacity), and competing with native bees for preferred floral hosts (Cane and Tepedino 2017).
  - Cane and Tepedino (2017) recommend the following:
    - Avoid placing hives adjacent to "highly diverse plant communities where a diverse native bee community can be expected."
    - Permit smaller more widely spaced apiaries to "dilute competition with native bees."
    - Determine the direct effects of an apiary to the native bee community, with monitoring.
- Are populations of endangered or threatened pollinators present on the land?
  - If rare species of bees and butterflies, including threatened or endangered species, are known to exist within the flight area where the hives are to be placed, assessment of potential risks to these populations should be undertaken.
  - If it is possible that rare or declining pollinator species can be found in the area, efforts

Honey bees are efficient pollinators, but that also means they are efficient at competing for floral resources with native bees.

should be made to determine if they are present. it is recommended that land managers consult scientists with expertise in pollinator surveys and species identification. In cases where a particular pollinator species is critically imperiled, every remaining population and individual may be essential to the species' immediate and long-term survival. There is potential that honey bees may transmit diseases to native bees (e.g., spread of deformed wing virus from honey bees to bumble bees causing wing damage) and may compete for floral resources (e.g., decreased fecundity in bumble bees).

- Are there invasive plant populations, or ongoing efforts to eradicate invasive plant species, that would be affected by the inclusion of honey bees?
- Honey bees may not be compatible with invasive plant species management. If honey bees pollinate and increase seed production of the invasive species in question (e.g., yellow star thistle), land managers may want to exclude honey bees during periods of bloom.
- What are the potential impacts to other wildlife?
    - Are there bears in the area that will be attracted to the apiary as a food source? Land managers need to work with beekeepers to determine if placement of an apiary will increase the potential for human–bear conflicts. If this is a risk, then electric fencing and maintenance of that fencing to prevent intrusion from bear should be mandated on public lands to avoid bear damage to apiaries and to prevent habituation of bears to hives.
- Is there sufficient infrastructure to support the drop-off and storing of the proposed operation?
    - Commercial beekeepers may bring anywhere between 4 and 400 (or more) hives, depending upon the size of the operation. Hives are delivered using a range of vehicles from flatbed trucks to semi-tractor trailers. Access roads must be appropriate for the required transport, and should not result in excess erosion, road damage, or other infrastructure challenges.
    - Apiary sites also must be of sufficient size, with level and firm ground to accommodate small forklifts or bobcats used to move pallets of bees. An apiary location will also need sufficient space for trucks to turn around.

If the above considerations have been made and a decision to move forward with apiary placement is under consideration (may occur with agencies that follow a multiple use mandate), we recommend:

- Any apiary (no matter the number of hives), needs to be more than 4 miles from:
    - Known locations of pollinators that are listed under state or federal endangered species acts, or designated as special status, sensitive, or other species of concern; this includes plants with specific and important native pollinator relationships that can lead to decline in plant production.

  - Wilderness and wilderness study areas as well as congressionally designated preserves and monuments.
  - Habitats of special value for biodiversity and/or pollinators (e.g., high elevation meadows, wet meadows, etc.).
- Apiaries should be separated by at least 4 miles.
- For further recommendations about placing honey bee hives in natural areas, refer to Xerces' publication An Overview of the Potential Impacts of Honey Bees to Native Bees, Plant Communities, and Ecosystems in Wild Landscapes: Recommendations for Land Managers (https://www.xerces.org/wp-content/uploads/2018/06/16-067_02_Overview-of-the-Potential-Impacts-of-Honey-Bees_web.pdf)

Collection of Bees for Commercial Propagation

Bumble Bees

As the cost of honey bee rental increases and the benefits of bumble bee pollination are recognized, bumble bees are being shipped throughout the world for pollination of greenhouse and field crops. Currently, the most common species of bumble bee that is available for managed pollination in the US is the common eastern bumble bee (*Bombus impatiens*), which is native to the eastern US. The use of this species in the West—outside of its native range—poses considerable risk to native bumble bees in the Rockies and westward (IUCN SSC BBSG 2016). The common eastern bumble bee may spread pathogens to wild bumble bees, or it may become established in the wild and outcompete native species for nest sites or floral resources (Whittington and Winston 2004; Colla et al. 2006). Managed bumble bees are thought to have contributed to the dramatic decline of at least five bumble bee species (the western (*B. occidentalis*), rusty patched (*B. affinis*), American (*B. pensylvanicus*), yellow-banded (*B. terricola*), and Franklin's (*B. franklini*) bumble bees (Evans et al. 2008; Cameron et al. 2011, 2016). There is also evidence that the common eastern bumble bee has become established in southern British Columbia and northern Washington (The Xerces Society et al. 2016). Because the importation of common eastern bumble bees is prohibited in Oregon and prohibited for open-field use in California, there are increasing efforts to develop additional bumble bee species for commercial distribution, including commercial requests for queen collection from US Forest Service lands (personal communication, Ivana Noell, February 24, 2016).

Managed bumble bees pose threats to native bumble bees in the West, like the western bumble bee (*Bombus occidentalis*), which has experienced declines throughout its range.

Bumble Bee Collecting

Where local and federal laws permit the collection of bumble bees for commercial development or research, and managers are deciding whether to approve a permitting request, managers should consider the following potential impacts of queen bumble bee collection. See Hatfield et al. (2012) for more details.

- Ask the collectors to specify the species they plan to collect. Permits which are seeking general collection or exploratory permits have the potential to impact the largest number of species. Also check if cuckoo bumble bees in the subgenus *Psithyrus*, which are cleptoparasitic, utilize any of the species to be collected because these are some of the most imperiled species of bumble bees.
- Avoid collections of sensitive, listed, or host species used by imperiled cuckoo bumble bee species. Check the conservation status of each target species on the IUCN Red List (IUCN 2017).
- Limit the number of bees collected, especially in spring when queen bumble bees are actively forming colonies. For context, each bumble bee queen represents a potential colony of bumble bees; colony size ranges from 50 to 1,000 individuals depending on species (Williams et al. 2014).

Mason Bees and Leafcutter Bees

Leafcutter and mason bees have also become significant contributors to agriculture as managed pollinators, particularly for orchards (mason bees) and alfalfa (alfalfa leafcutter bees). These are above-ground, tunnel-nesting bees and will nest in large aggregations, making commercial colonies possible. The most widely utilized leafcutter bee, the alfalfa leafcutter bee (*Megachile rotunda*), is an exotic species that has naturalized throughout much of North America. The native blue orchard bee (*Osmia lignaria*) as well as many nonnative *Osmia* spp. are also commercially available. Illegal harvest of native mason bees (*Osmia* spp.) for the agricultural industry on public lands is an issue of growing concern. There have been recent reports of illegal harvest from southern Idaho, Utah, and Nevada (Tepedino and Nielson 2017; Beth Waterbury, personal communication, November 21, 2017). Mass bee harvests could negatively impact native bee populations and reduce pollinator services for early blooming plant species. In addition, the tubes used to harvest bees are often abandoned on public land and may act as disease vectors.

Mason and Leafcutter Bee Collecting

It is advised that land managers do not allow collection of native mason or leafcutter bees for commercial use, because of the limited information and understanding on the current distribution and status of

Illegal mason bee trap on National Forest land in Idaho.

most of these native bees. In addition, some of these species are in trouble: a recent analysis showed that 27%of mason bees in North America are at risk (Young et al. 2015). If it is decided, however, to still allow the collection of these species where local and federal laws permit it, managers should consider the following potential impacts of bee collection:

- Nest blocks used to collect mason and leafcutter bees can transmit diseases and can draw pollinators in from a large area and have the potential to remove large quantities of native bee larvae from an area—potentially affecting the bee population the following year.
- There is potential to remove at risk mason and leafcutter bees (Young et al. 2015).
- If there are questions about how a collecting effort may impact the pollinator population, or pollination services they provide, a scientist with expertise and knowledge of local bee populations should be consulted (e.g., USDA Pollinating Insect-Biology, Management, Systematics Research, in Logan, Utah, which you can contact through www.ars.usda.gov).

Illegal mason bee collecting on National Forest land.

If the above considerations have been made and a decision to move forward with a native mason or leafcutter bee collection permit is still under consideration, we recommend:

- Prior to issuing a permit, survey the collection area to determine if any at-risk species of mason bees or other bees that may utilize nest blocks are present. This may require consultation with bee experts to aid in identification.
- Include specific conditions in the permit to require the collector to sanitize or use new nest blocks to minimize the spread of pathogens (such as chalkbrood) and parasites to local bee populations. Reused nesting materials should be submerged in a bleach–water solution (1:3 by volume) for at least five minutes every year (Mader 2010).
- Limit the number of nesting blocks allowed.

Recreation

Recreation including hiking, trail running, equestrian use, mountain biking, and off-highway vehicles (OHV) can affect pollinator habitat by altering the quantity and structure of vegetation (Cole and Spildie 1998; Ballantyne et al. 2014; Hennings 2017), increasing soil erosion, altering soil composition and microflora (cryptobiotic crusts) through compaction and disturbance (Wilshire 1983), and spreading invasive plants (Trunkle and Fay 1991; Trombulak and Frissell 2000). One study found that in just one trip on a 10 mile course, an OHV dispersed 2,000 spotted knapweed seeds (Trunkle and Fay 1991). Recreation, particularly OHV use, can also cause direct mortality to pollinators (Blair and Launer 1997; Center for Biological Diversity 2004; Wayne et al. 2009), and severely damage pollinator habitat (Black et al. 2013). The effects of recreation can be especially damaging in desert dune ecosystems, where OHV use offer occurs in places with endemic and specialist pollinators (Griswold et al. 2003; Center for Biological Diversity 2004; Pitts et al. 2009; Wilson et al. 2009).

Pollinator habitat in arid dune habitat can be vulnerable to the impacts of recreational activities.

Recreation Recommendations

- Limit OHV access in sensitive pollinator habitat, especially during high pollinator activity, and in arid regions that are more vulnerable to disturbance.
    - In arid regions such as the Mojave desert and sagebrush biome, OHV activity has the greatest effect on habitat in the summer months (a period when plant vulnerability and mortality increases) or the time at which plants become dry and brittle at the end of their growing season (Payne et al. 1983; Taylor 2006).
- Set aside refugia free from OHV use in desert dune ecosystems, or any area with known endemic pollinators.
- Minimize the potential for OHVs to spread invasive species by developing weed washing stations in high-use OHV areas and performing outreach to OHV associations to increase awareness.
- Install signage to inform the recreating public about pollinators and how they can participate in pollinator conservation (e.g., cleaning equipment and shoes for invasive plant seeds, staying on trails, etc.).

Managing Pollinator Habitat Under Climate Change

Climate change has been identified as one of the largest risks to biodiversity worldwide (Maclean and Wilson 2011). Impacts associated with climate change include changes in seasonal temperatures, altered precipitation patterns, rising sea levels, and higher frequency of extreme weather events such as storms, floods, heatwaves, and droughts (IPCC 2014). However, relatively little is known about how climate change will impact pollinator communities and the services they provide (Goulson et al. 2015). Given the huge variety of habitats, elevations, and resources used by pollinators and their widely varying life histories, it is likely that the impacts of climate change will vary dramatically from species to species and location to location. Some species may ultimately benefit from climate change, while others will decline.

Four of the primary concerns surrounding pollinators and climate change are (1) phenological divergence of pollinators and their host plants; (2) range shifts that lead to spatial mismatches between plants and pollinators; (3) extreme weather events such as flooding and droughts; and (4) changes in resource quality and availability. This is not an exhaustive list of the effects of climate change on pollinators. For example, climate change can alter physiological processes of species, such as growth rates, activity times, or size at maturity (Walther et al. 2002). Climate change can also alter species interactions, potentially shifting the outcome of interactions with pathogens or enemies like predators or parasitoids (Tylianakis et al. 2008; Van der Putten et al. 2010). These effects of climate change on pollinators are not mutually exclusive, and pollinators are likely to experience multiple effects of climate change. In addition, climate change acts as an added stressor for species already impacted by habitat loss, disease, high pathogen loads, or any of the other threats facing pollinators today.

Phenological Divergence

Changing environmental cues are expected to alter species' phenologies (Hughes 2000; Kumar et al. 2012; Hayes et al. 2012; CaraDonna et al. 2014). These shifts can result in mismatches between flowering times and pollinator foraging windows (Bertin 2008; Hegland et al. 2009; Chambers et al. 2013; Kudo and Ida 2013; Monahan et al. 2016; Byers 2017). Phenological mismatch can also be an issue for butterfly pollinators in the larval stage, as many butterflies rely on a specific species or groups of plants as larval hosts. One survey of 1,420 species of pollinators known to visit at least 429 kinds of plants predicts that climate-driven changes in flowering times will reduce available floral resources for at least 17% and perhaps as many as 50% of all pollinators, resulting in diminished nutritional diversity within their diets (Memmott et al. 2007).

Range Shifts

Range shifts in species distribution and abundance are also expected in a changing climate (Hickling et al. 2006; Franzén and Öckinger 2012). Range shifts in response to climate change have already been demonstrated in butterflies (Parmesan 1996, 2006; Mitikka et al. 2008; Forister et al. 2010; Kerr 2016) and in bumble bees (Williams and Osborne 2009; Kerr 2016). Relatively little climate research has focused on shifts in other native bee species (Szabo et al. 2012); it is expected, however, that range shifts can lead to spatial mismatch between plants and pollinators, or reduced ranges (Kerr 2016). The ability of pollinators to shift their distributions depends on the availability of habitat, including appropriate host plants, in areas with favorable climates, as well as availability of habitat corridors to move through.

Shifts in plant and pollinator phenologies can cause a mismatch in larval host and floral resource availability.

Extreme Weather

Extreme weather events such as flooding and drought can negatively impact pollinators by displacing or drowning subterranean species such as ground-nesting bees, influencing host and nectar plant survivability and palatability, or causing mass die-offs, such as those seen in winter storms at monarch overwintering sites. Extreme temperature events and heat waves may also negatively affect pollinators and their host plants if temperatures exceed critical thresholds. There is evidence that improving habitat availability can reduce the negative effects of abiotic stressors such as high temperature and drought on pollinators (Oliver et al. 2013; Papanikolaou et al. 2017), most likely by providing microclimate refugia.

Resource Quality and Availability

Plants will also respond to climate change, and shifts in plant community composition, including the relative abundance and diversity of plant communities, will certainly affect pollinator communities and competitive interactions. Reductions in particular host plant species may be especially problematic for specialist pollinators. Floral resources will also vary with abiotic conditions. For example, an increase in the intensity and frequency of drought conditions expected in some regions may lead to fewer flowers and reduced nectar production to support pollinator communities.

Climate Change Related Recommendations

- Prioritize conservation of and manage for resilient and resistant rangeland ecosystems and pollinator communities (McIver et al. 2014; Chambers et al. 2017)
- Restore areas with plant species that are likely to persist under future climate scenarios. For example, plant drought tolerant species in areas where drought will become more frequent
- Improve habitat connectivity by creating habitat stepping stones or corridors across the landscape that will aid pollinator dispersal and migration.
- Restore and enhance habitat to create microclimate refugia for pollinators
- Monitor important host plants for specialist pollinators of concern and consider including those host plants in restoration efforts.
- Promote plant diversity, and try to ensure that habitat restorations will have at least three plant species flowering during the active pollinator season. This will help to buffer pollinators against changes in resource quality and phenology.
- Protect areas that may be important habitat for rare or declining species under future climate change scenarios.
- Select locations for restoration that will be able to support restored habitats under future climate scenarios. For example, consider sea-level rise and the likelihood of more frequent and severe flooding events when choosing restoration sites.

Protecting and managing for resilient and resistant rangeland ecosystems will also benefit pollinators.

3

Monitoring

Monitoring Floral Resources and Pollinator Populations

Floral Resource Monitoring

Several studies have reported a direct relationship between bee abundance and the abundance of floral resources (Minckley et al. 1994; Larsson and Franzén 2007; Xie et al. 2008; Batáry et al. 2010; Grundel et al. 2010; Roulston and Goodell 2011; Palladini 2013; Rubene et al. 2015; Smith et al. 2016; Vrdoljak et al. 2016). The diets of both adult and larval are comprised primarily of floral nectar and pollen, and many studies have provided clear evidence that bee populations are regulated by the availability of these two necessities (Visscher and Danforth 1993; Müller et al. 2006; Roulston and Goodell 2011). Cane (2011) found evidence that solitary bee carrying capacity may be limited by the availability of floral resources in rangelands. In addition, butterfly species richness has also been correlated with plant species richness, such as within natural areas of Arizona (Bock et al. 2007). Butterfly species which rely on specific host plants (e.g., monarch and milkweed; little blue butterfly and lupine) are strictly limited by host plant availability.

In addition to food sources, pollinators need shelter and nesting resources. Cane et al. (2006) hypothesized that the availability of cavity-nest resources may directly limit the abundance of cavity-nesting bees in the desert southwest. McFrederick and LeBuhn (2006) found a direct relationship between the density of rodent burrows and bumble bees. Other bee species require naturally occurring bare or mostly

Pollinators depend on sufficient floral resources throughout the season; monitoring the abundance and diversity of flowering plants is often a good indicator of a site's value for bees and butterflies.

bare ground to successfully form nests. Overall, research suggests that bee populations are directly regulated by the availability of floral and pollen resources, and to a lesser degree, by the availability of nest resources (Roulston and Goodell 2011). These examples suggest that management plans, especially grazing plans, need to incorporate management objectives and monitoring components to conserve and monitor floral and nest resources for pollinators. Land managers should strive to incorporate floral resources into existing land health assessments and then set management objectives to achieve reliable and abundant floral resources for pollinators.


Management plans should incorporate floral resource monitoring into health assessments to adaptively manage sites for pollinators.

Below are two monitoring guides/programs you may consider using:

1. **Xerces Society's *Native Bee Conservation Pollinator Habitat Assessment Form and Guide: Natural Areas and Rangelands*.** This assessment guide is a tool to help incorporate pollinator conservation into management by standardizing assessment and quantifying habitat improvements for pollinators. It includes a ranking system to assess floral and nesting resources for pollinators pre- and post-restoration project or management action. Available at www.xerces.org.
2. **BLM's *Pollinator Supplementary Indicator*.** The Bureau of Land Management (BLM) is currently piloting an optional pollinator supplementary indicator for the Assessment Inventory and Monitoring (AIM) protocol to assess pollinator resource availability. The supplement includes general assessments of the presence of pollinator groups (native bee, bumble bee, honey bee, butterfly/moth, monarch, or hummingbird), plant phenology by species, floral cover of flowering plants, and the presence of bare ground and water. While this supplement is still in the pilot stages, it may serve as a guide for other agencies to develop supplements to their respective land health, vegetation, or rangeland monitoring protocols.

Pollinator Monitoring

Floral and nesting resources are only a proxy for real bees and butterflies. Establishing baseline information on bee and butterfly populations, as well as trends, is essential to our understanding of native pollinators on western rangelands, and how range management affects those populations. In most landscapes this information is lacking, but would ideally be implemented as part of any long-term monitoring strategy. No single monitoring strategy exists for monitoring all pollinators; here we present a range of options for monitoring bees and butterflies. It should be noted that insects, being cold-blooded and often having multiple generations per year, are subject to population swings from one year to the next due to weather and stochastic events. Regular monitoring over multiple years is more valuable than single year snapshots.

Monitoring Bees

Native bees are recognized as being difficult to identify and survey. Species level identification of most groups of bees require specimen collection, curation, and employing a taxonomist, which can be costly and time-intensive. Nevertheless, with recent attention being paid to pollinator populations, efforts have been made to standardize surveys and incorporate methods that reduce the need for collecting bees and relying on a taxonomist collection. See Box **10** for a case study about monitoring bumble bees on the Wallowa-Whitman National Forest.

Identifying bees to species takes a lot of dedication or a good taxonomist; you can much more easily learn to identify bees to group—like this metallic green bee—and derive useful information through monitoring.

The following list of bee monitoring protocols is organized from least to most intensive; there is a corresponding increase in the level of detail that would emerge from survey efforts. (This list is not comprehensive.)

1. **Stream-lined bee monitoring protocol.** The Xerces Society, in collaboration with Rutgers University, Michigan State University, and University of California–Davis, developed a citizen-science monitoring protocol for bees. This method requires minimal training and was developed for use by land management agencies, farmers, and citizen scientists to obtain estimates of native bee abundance and diversity. This method has been vetted using scientific studies that have proven a high correlation between native bee abundance and native bee diversity (Kremen et a. 2011). Available at www.xerces.org/xerces-bee-monitoring-tools.
2. **Regional bee monitoring guides.** Developed by the Xerces Society and the University of California Berkeley, these California and Maritime Northwest-specific guides are detailed citizen science bee monitoring protocols. These methods allows managers to track changes in bee diversity and abundance over time. While this protocol does not require a bee taxonomist, it is most effectively implemented with a two-day training workshop to familiarize surveyors with bee morphospecies. This protocol can also be adapted to include butterflies (using butterfly families as morphospecies). Both methods do require some level of expertise or prior training in bee identification. Available at www.xerces.org/xerces-bee-monitoring-tools.
3. **Bumble bee species sampling** (Strange et al. 2013). Developed by the USDA-ARS Pollinating Insect-Biology, Management, Systematics Research lab in Logan, Utah, this method uses active netting to survey bumble bees populations using a standardized method. It has been implemented in national parks throughout the Pacific Northwest. This method would require the services of a bumble bee expert to identify specimens and/or verify photo vouchers. Because bumble bees are a group about much is know, this method could be used to identify the presence of potentially imperiled bumble bee species in an area. Available at www.nps.gov.
4. **The national protocol for bee sampling** (Droege et al. 2016). This method uses only passive sampling for bees (pan traps, glycol traps, and/or blue vane traps) for either short- or long-term monitoring. It has been adopted on national wildlife refuges and other US Fish and Wildlife properties, thus allowing standardized comparisons across the landscape. This method requires the services of a bee taxonomist to identify specimens. Available at www.pubs.er.usgs.gov/publication/70176107.
5. **The bee inventory plot** (LeBuhn et al. 2003). This method combines passive (pan trap) sampling with active (netting) sampling in 1 hectare plots to provide a detailed assessment of the species present at a site over time. This method requires the services of a bee taxonomist to identify specimens. Available at www.online.sfsu.edu/beeplot/pdfs/Bee%20Plot%202003.pdf.

Monitoring Butterflies

The butterfly fauna in Western rangelands is highly diverse. While comprehensive monitoring plans require trained lepidopterists, many species identifications can be done in the field, and do not require collection and curation. In addition to adult butterfly monitoring, host plant abundance and larval habitat condition have been identified as the most important predictors of butterfly population size and persistence. As such, surveys that monitor host plant condition and habitat for target butterfly species are likely to inform land managers about the potential that an area has for maintaining a population of butterflies. Host plant surveys could be incorporated into existing butterfly survey protocols. For sensitive butterflies, it is important to select a protocol that is statistically rigorous, is comparable with historical data, and generates response variables which are useful to inform management (Kral et al. 2018).

Below is a list of butterfly monitoring program from least to most intensive; there is a corresponding increase in the level of detail that would emerge from survey efforts. (This list is not comprehensive.)

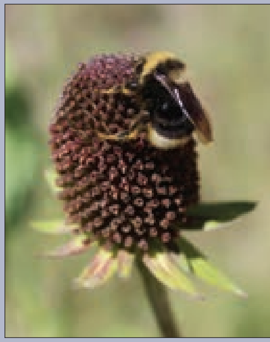
1. **Citizen Science/Morphospecies Monitoring.** The Xerces Society citizen-science monitoring guide includes methodology to monitor butterflies using families as morphospecies. While this does not provide species richness information, it does provide butterfly abundance and an understanding of butterfly diversity at a family level within a site. Most citizen scientists, or biologists, could be trained to identify butterflies to family and how to implement this protocol. Available at www.xerces.org/xerces-bee-monitoring-tools.
2. **Pollard Walks or Transect Counts** (Pollard and Yates 1994). Pollard Walks establish permanent transects through the differing habitats in the survey area that are monitored on a regular basis (e.g., weekly), with guidelines regarding appropriate environmental conditions. During each monitoring event, all butterfly individuals are counted and identified to species (or the lowest taxonomic level possible). This monitoring protocol is the most common standard butterfly survey method and has been adopted all over the world. The data is most useful if the surveys are completed by trained personnel comfortable identifying butterflies to species.

Box 10: Bumble Bee Monitoring on the Wallowa-Whitman National Forest

Wildlife biologist Laura Navarette has been conducting bumble bee surveys on the Wallowa-Whitman National Forest in eastern Oregon since 2014, inspired after a training on bumble bee identification by the Xerces Society. While survey effort varies by year, at each survey location, Laura (and any volunteers she can get) generally completes a 20– to 40–minute search within a 100 m radius area. She uses a butterfly net to catch bumble bees, identifying as many as she can in the field and taking lots of pictures for confirmation back in the office. For harder-to-identify species, she transfers the bees to vials which she chills to make an accurate identification easier. For the really tough ones, she contacts the Xerces Society for confirmation. All bees are released in the area they were collected from and bee mortality is very rare.

The bumble bee monitoring crew at the Wallowa-Whitman National Forest (above); western bumble bee (*B. occidentalis*) found on the forest (below).

In four years of collection, Laura has conducted 34 surveys across the forest and observed 16 different *Bombus* species. Her finds include two sensitive species, the western bumble bee (*B. occidentalis*) at 16 locations and the Suckley cuckoo bee (*B. suckleyi*) at 4 locations. She has also partnered with the Oregon Department of Fish and Wildlife to conduct surveys in a nearby marsh area and found two additional species not recorded on the forest. Laura says "It's pretty exciting to see how much bumblebee diversity we have within Union County!"



3. **Distance Sampling** (e.g., Hatfield et al. 2013). Distance sampling is a transect-based method of monitoring species' density and abundance that accounts for detectability and surveyor bias. While there are several major assumptions that must be met in order to carry out a successful monitoring project, this method is statistically rigorous (Thomas et al. 2010; Isaac et al. 2011) and often used for surveying a single focal species (e.g., mardon skipper [Hatfield et al. 2013]). It does require trained personnel, however, and would ideally be conducted annually at long-term monitoring sites.

Citizen Science Monitoring Programs

There are many citizen-science monitoring programs which focus on tracking occurrences of specific taxa (e.g., bumble bees, monarchs) or standardized one-day pollinator counts (e.g., NABA). Many of these programs offer resources in pollinator identification. (This list is not complete).

Bumble Bees

- Bumble Bee Watch (www.bumblebeewatch.org)

Butterflies

- North American Butterfly Association Butterfly Counts (www.naba.org)
- PollardBase (www.pollardbase.org)
- E-Butterfly (www.e-butterfly.org)

Monarch Butterfly

- Western Monarch Milkweed Mapper (www.monarchmilkweedmapper.org)
- Monarch Larva Monitoring Project (www.monarchlab.org)
- Journey North (www.learner.org/jnorth/monarchs)

Citizen science monitoring for species like the monarch have greatly expanded our understanding of the North American butterfly's migration, population status, parasitism rates, etc.

Literature Cited

Abbott, V. A., J. L. Nadeau, H. A. Higo, and M. L. Winston. 2008. Lethal and sublethal effects of imidacloprid on *Osmia lignaria* and clothianidin on *Megachile rotundata* (Hymenoptera: Megachilidae). *Journal of Economic Entomology* 101:784–796.

Aizen, M. A., C. L. Morales, and J. M. Morales. 2008. Invasive mutualists erode native pollination webs. *PLoS Biology* 6:e31.

Arroyo, M., R. Primack, and J. Armesto. 1982. Community studies in pollination ecology in the high temperate Andes of central Chile. I. Pollination mechanisms and altitudinal variation. *American Journal of Botany* 69:82–97.

Ascher, J. S., and J. Pickering. 2015. Discover life bee species guide and world checklist (Hymenoptera: Apoidea: Anthophila). https://bit.ly/2lukMZT

Averett, J. P., B. A. Endress, M. M. Rowland, B. J. Naylor, and M. J. Wisdom. 2017. Wild ungulate herbivory suppresses deciduous woody plant establishment following salmonid stream restoration. *Forest Ecology and Management* 391:135–144.

Bainbridge, D. A. 2002. Alternative irrigation systems for arid land restoration. *Ecological Restoration* 20:23–30.

Bainbridge, D. A. 2012. *A Guide for Desert and Dryland Restoration: New Hope for Arid Lands*. 416 pp. Washington, D.C.: Island Press.

Balbuena, M. S., L. Tison, M.-L. Hahn, U. Greggers, R. Menzel, and W. M. Farina. 2015. Effects of sublethal doses of glyphosate on honeybee navigation. *Journal of Experimental Biology* 218:2799–2805.

Ballantyne, M., C. Pickering, and O. Gudes. 2014. How formal and informal mountain biking trails result in the reduction, degradation and fragmentation of endangered urban forest remnants. In *Proceedings of the 7th International Conference on Monitoring and Management of Visitors in Recreational and Protected Areas, August 20-23, 2014*, edited by M. Reimann, K. Sepp, E. Pärna, and R. Tuula, 155–157. Tallinn: Tallinn University.

Balmer, O., and A. Erhardt. 2000. Consequences of succession on extensively grazed grasslands for central European butterfly communities: rethinking conservation practices. *Conservation Biology* 14:746–757.

Barratt, B. I. P., F. G. Howarth, T. M. Withers, J. M. Kean, and G. S. Ridley. 2010. Progress in risk assessment for classical biological control. *Biological Control* 52:245–254.

Baskett, C. A., S. M. Emery, and J. A. Rudgers. 2011. Pollinator visits to threatened species are restored following invasive plant removal. *International Journal of Plant Sciences* 172:411–422.

Batáry, P., A. Báldi, M. Sárospataki, F. Kohler, J. Verhulst, E. Knop, F. Herzog, and D. Kleijn. 2010. Effect of conservation management on bees and insect-pollinated grassland plant communities in three European countries. *Agriculture, Ecosystems & Environment* 136:35–39.

Bates, J. D., K. W. Davies, A. Hulet, R. F. Miller, and B. Roundy. 2016. Sage grouse groceries: forb response to pinyon-juniper treatments. *Rangeland Ecology & Management* 70:106–115.

Baum, K. A., and W. V. Sharber. 2012. Fire creates host plant patches for monarch butterflies. *Biology Letters* 8:968–971.

National Research Council. 2007. *Status of Pollinators in North America*. 326 pp. Washington, D.C.: The National Academies Press.

Bernauer, O. M., H. R. Gaines-Day, and S. A. Steffan. 2015. Colonies of bumble bees (*Bombus impatiens*) produce fewer workers, less bee biomass, and have smaller mother queens following fungicide exposure. *Insects* 6:478–488.

Bernhardt, P. 2000. Convergent evolution and adaptive radiation of beetle-pollinated angiosperms. *Plant Systematics and Evolution* 222:293–320.

Bertin, R. I. 2008. Plant phenology and distribution in relation to recent climate change. *Journal of the Torrey Botanical Society* 135:126–146.

Beyer, L., and S. H. Black. 2007. "Site utilization by adults and larvae of mardon skipper butterfly (*Polites mardon*) at four sites in Washington and Oregon. Final Report to the Forest Service and BLM from the Xerces Society." Portland, OR: The Xerces Society for Invertebrate Conservation.

Beyer, L. J., and C. B. Schultz. 2010. Oviposition selection by a rare grass skipper *Polites mardon* in montane habitats: advancing ecological understanding to develop conservation strategies. *Biological Conservation* 143:862–872.

Bezemer, T. M., J. A. Harvey, and J. T. Cronin. 2014. Response of native insect communities to invasive plants. *Annual Review of Entomology* 59:119–141.

Biddinger, D. J., J. L. Robertson, C. Mullin, J. Frazier, S. A. Ashcraft, E. G. Rajotte, N. K. Joshi, and M. Vaughn. 2013. Comparative toxicities and synergism of apple orchard pesticides to *Apis mellifera* (l.) and *Osmia cornifrons* (Radoszkowski). *PloS ONE* 8:e72587.

Bilotta, G. S., R. E. Brazier, and P. M. Haygarth. 2007. The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. *Advances in Agronomy* 94:237–280.

Black, S. H., B. Borders, C. Fallon, E. Lee-Mäder, and M.

Shepherd. 2016. *Gardening for Butterflies: How You Can Attract and Protect Beautiful, Beneficial Insects*. 287 pp. Portland, OR: Timber Press, Inc.

Black, S. H., C. Fallon, R. Hatfield, and C. Mazzacano. 2013. "Controlled Burning and Mardon Skipper: Summary of Mardon Skipper Coon Mountain Burn Site Occupancy Study 2009-2013. Final Report to US Forest Service, Oregon Zoo and US Fish and Wildlife Service." 29 pp. Portland, OR. The Xerces Society for Invertebrate Conservation.

Black, S. H., R. Hatfield, L. Beyer, S. Jepsen, and C. A. Mazzacano. 2013. "Management Plans for Southern Oregon Cascade mardon skipper *(Polites mardon* ssp. *klamathensis)* sites on the Rogue River Siskiyou National Forest, High Cascades Ranger District. The Xerces Society for Invertebrate Conservation: U.S.D.A. Forest Service Region 6 and U.S.D.I. Bureau of Land Management Interagency Special Status and Sensitive Species Program." Portland, OR. The Xerces Society for Invertebrate Conservation.

Black, S. H., N. Hodges, M. Vaughan, and M. Shepherd. 2007. *Pollinators in Natural Areas: A Primer on Habitat Management*. 8 pp. Portland, OR: The Xerces Society For Invertebrate Conservation.

Black, S. H., M. Shepherd, and M. Vaughan. 2011. Rangeland management for pollinators. *Rangelands* 33:9–13.

Black, S. H., and D. M. Vaughan. 2005. "Species Profile: *Pyrgus ruralis lagunae*." In *Red List of Pollinator Insects of North America CD-ROM Version 1 (May 2005)*, edited by M. D. Shepherd, D. M. Vaughan, and S. H. Black. Portland, OR: The Xerces Society for Invertebrate Conservation.

Blair, R. B., and A. E. Launer. 1997. Butterfly diversity and human land use: species assemblages along an urban gradient. *Biological Conservation* 80:113–125.

Bock, C. E., R. A. Bailowitz, D. W. Danforth, Z. F. Jones, and J. H. Bock. 2007. Butterflies and exurban development in southeastern Arizona. *Landscape and Urban Planning* 80:34–44.

Bollinger, E. K., P. B. Bollinger, and T. A. Gavin. 1990. Effects of hay-cropping on eastern populations of the bobolink. *Wildlife Society Bulletin* 18:142–150.

Bond, C., A. Hallman, K. Buhl, and D. Stone. 2016. "Carbaryl general fact sheet." National Pesticide Information Center, Oregon State University Extension Services. http://npic.orst.edu/factsheets/carbarylgen.html (accessed 6/19/18).

Bonte, D. 2005. Anthropogenic induced changes in nesting densities of the dune-specialised digger wasp *Bembix rostrata* (Hymenoptera : Sphecidae). *European Journal of Entomology* 102:809–812.

Borders, B., and E. Lee-Mäder. 2014. *Milkweeds: A Conservation Practitioner's Guide*. 100+vi pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Börschig, C., A.-M. Klein, H. von Wehrden, and J. Krauss. 2013. Traits of butterfly communities change from specialist to generalist characteristics with increasing land-use intensity. *Basic and Applied Ecology* 14:547–554.

Bower, A. D., J. B. St Clair, and V. Erickson. 2014. Generalized provisional seed zones for native plants. *Ecological Applications* 24:913–919.

Branson, D. H., A. Joern, and G. A. Sword. 2006. Sustainable management of insect herbivores in grassland ecosystems: new perspectives in grasshopper control. *Bioscience* 56:743–755.

Brooks, M. L., D. A. Pyke, and K. Galley. 2001. Invasive plants and fire in the deserts of north america. Invasive plants and fire in the deserts of North America. In *Proceedings of the Invasive Plant Workshop: The Role of Fire in the Control and Spread of Invasive Species*, edited by K. E. M. Galley and T. P. Wilson, 1–14. Tallahassee: Tall Timbers Research Station.

Brown, J. H., and M. A. Bowers. 1985. Community organization in hummingbirds: relationships between morphology and ecology. *The Auk* 102:251–269.

Brown, M. J. F., L. V. Dicks, R. J. Paxton, K. C. R. Baldock, A. B. Barron, M.-P. Chauzat, B. M. Freitas et al. 2016. A horizon scan of future threats and opportunities for pollinators and pollination. *PeerJ* 4:e2249.

Bruppacher, L., J. Pellet, R. Arlettaz, and J.-Y. Humbert. 2016. Simple modifications of mowing regime promote butterflies in extensively managed meadows: evidence from field-scale experiments. *Biological Conservation* 196:196–202.

Brussard, P., D. Charlet, and D. Dobkin. 1998. Regional trends of biological resources—Great Basin–Mojave Desert. In *Status and Trends of the Nation's Biological Resources*, edited by M. J. Mac, P. A. Opler, C. E. Puckett Haecker, and P. D. Doran, 505–542. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey.

Burghardt, K. T., D. W. Tallamy, and W. G. Shriver. 2009. Impact of native plants on bird and butterfly biodiversity in suburban landscapes. *Conservation Biology* 23:219–224.

Buri, P., J.-Y. Humbert, and R. Arlettaz. 2014. Promoting pollinating insects in intensive agricultural matrices: field-scale experimental manipulation of hay-meadow mowing regimes and its effects on bees. *PloS ONE* 9:e85635.

Busse, M. D., C. J. Shestak, and K. R. Hubbert. 2013. Soil heating during burning of forest slash piles and wood piles. *International Journal of Wildland Fire* 22:786–796.

Bybee, J., B. A. Roundy, K. R. Young, A. Hulet, D. B. Roundy, L. Crook, Z. Aanderud, D. L. Eggett, and N. L. Cline. 2016. Vegetation response to piñon and juniper tree shredding. *Rangeland Ecology & Management* 69:224–234.

Byers, D. L. 2017. Studying plant-pollinator interactions in a changing climate: a review of approaches. *Applications in Plant Sciences* 5:1700012.

Cagnolo, L., S. I. Molina, and G. R. Valladares. 2002. Diversity and guild structure of insect assemblages under grazing and exclusion regimes in a montane grassland from central Argentina. *Biodiversity and Conservation* 11:407–420.

Calderone, N. W. 2012. Insect pollinated crops, insect pollinators and US agriculture: trend analysis of aggregate data for the period 1992–2009. *PloS ONE* 7:e37235.

Cameron, S. A., H. C. Lim, J. D. Lozier, M. A. Duennes, and R. Thorp. 2016. Test of the invasive pathogen hypothesis of bumble bee decline in North America. *Proceedings of the National Academy of Sciences* 113:4386–4391.

Cameron, S. A., J. D. Lozier, J. P. Strange, J. B. Koch, N. Cordes, L. F. Solter, and T. L. Griswold. 2011. Patterns of widespread decline in North American bumble bees. *Proceedings of the National Academy of Sciences* 108:662–667.

Campbell, J. W., J. L. Hanula, and T. A. Waldrop. 2007. Effects of prescribed fire and fire surrogates on floral visiting insects of the Blue Ridge province in North Carolina. *Biological Conservation* 134:393–404.

Cane, J. H. 1991. Soils of ground-nesting bees (Hymenoptera: Apoidea): texture, moisture, cell depth and climate. *Journal of The Kansas Entomological Society* 64:406–413.

Cane, J. H. 1997. Ground-nesting bees: the neglected pollinator resource for agriculture. *Acta Horticulturae* 437:309–324.

Cane, J. H. 2008. Pollinating bees crucial to farming wildflower seed for U.S. habitat restoration. In *Bee Pollination in Agricultural Ecosystems*, edited by R. James and T. L. Pitts-Singer, 105–123.

Cane, J. H. 2011. Meeting wild bees' needs on rangelands. *Rangelands* 33:27–32.

Cane, J. H. 2015. Landscaping pebbles attract nesting by the native ground-nesting bee *Halictus rubicundus* (Hymenoptera: Halictidae). *Apidologie* 46:728–734.

Cane, J. H., and B. Love. 2016. Floral guilds of bees in sagebrush steppe: comparing bee usage of wildflowers available for postfire restoration. *Natural Areas Journal* 36:377–391.

Cane, J. H., R. L. Minckley, L. J. Kervin, T. H. Roulston, and N. M. Williams. 2006. Complex responses within a desert bee guild (Hymenoptera: Apiformes) to urban habitat fragmentation. *Ecological Applications* 16:632–644.

Cane, J. H., and J. L. Neff. 2011. Predicted fates of ground-nesting bees in soil heated by wildfire: thermal tolerances of life stages and a survey of nesting depths. *Biological Conservation* 144:2631–2636.

Cane, J. H., and V. J. Tepedino. 2017. Gauging the effect of honey bee pollen collection on native bee communities. *Conservation Letters* 10:205–210.

CaraDonna, P. J., A. M. Iler, and D. W. Inouye. 2014. Shifts in flowering phenology reshape a subalpine plant community. *Proceedings of the National Academy of Sciences* 111:4916–4921.

Carvell, C. 2002. Habitat use and conservation of bumblebees (*Bombus* spp.) under different grassland management regimes. *Biological Conservation* 103:33–49.

Casner, K. L., M. L. Forister, J. M. O'Brien, J. Thorne, D. Waetjen, and A. M. Shapiro. 2014. Contribution of urban expansion and a changing climate to decline of a butterfly fauna. *Conservation Biology* 28:773–782.

Center for Biological Diversity. 2004. "Petition to List the Sand Mountain Blue Butterfly (*Euphilotes pallescens arenamontana*) as a Threatened or Endangered Species under the U.S. Endangered Species Act." Tucson: Center for Biological Diversity.

Chambers, J. C., J. D. Maestas, and M Pellant. 2015. *Putting Resilience and Resistance Concepts into Practice*. (Great Basin Factsheets Series, number 1.) 6 pp. Fort Collins: USDA Forest Service, Rocky Mountain Research Station.

Chambers, J. C., J. D. Maestas, D. A. Pyke, C. S. Boyd, M. Pellant, and A. Wuenschel. 2017. Using resilience and resistance concepts to manage persistent threats to sagebrush ecosystems and greater sage-grouse. *Rangeland Ecology & Management* 70:149–164.

Chambers, L. E., R. Altwegg, C. Barbraud, P. Barnard, L. J. Beaumont, R. J. M. Crawford, J. M. Durant et al. 2013. Phenological changes in the southern hemisphere. *PloS ONE* 8:e75514.

Cole, D. N., and D. R. Spildie. 1998. Hiker, horse and llama trampling effects on native vegetation in Montana, USA. *Journal of Environmental Management* 53:61–71.

Colla, S. R., J. S. Ascher, M. Arduser, J. Cane, M. Deyrup, S. Droege, J. Gibbs et al. 2012. Documenting persistence of most eastern North American bee species (Hymenoptera: Apoidea: Anthophila) to 1990–2009. *Journal of the Kansas Entomological Society* 85:14–22.

Colla, S. R., M. C. Otterstatter, R. J. Gegear, and J. D. Thomson. 2006. Plight of the bumble bee: pathogen spillover from commercial to wild populations. *Biological Conservation* 129:461–467.

Colla, S. R., and L. Packer. 2008. Evidence for decline in eastern North American bumblebees (Hymenoptera: Apidae), with special focus on *Bombus affinis* Cresson. *Biodiversity and Conservation* 17:1379.

Cota, J. 2004. "National Report of Pesticide Use on National Forest System Lands." 9 pp. Washington, D.C.: U.S. Forest Service.

Cousin, M., E. Silva-Zacarin, A. Kretzschmar, M. El Maataoui, J.-L. Brunet, and L. P. Belzunces. 2013. Size changes in honey bee larvae oenocytes induced by exposure to paraquat at very low concentrations. *PloS ONE* 8:e65693.

CU, MSU, OSU, and UCD [Cornell University, Michigan State University, Oregon State University and the University of California at Davis]. 1993. "Pesticide Information Profile: Diflubenzuron." https://bit.ly/2MiS8X9 (accessed 1/22/18).

Dana, R. P. 1997. *Characterization of three Dakota skipper sites in Minnesota*. 17 pp. St. Paul: Minnesota Department of Natural Resources, Natural Heritage and Nongame Research Program.

Davies, K. W., J. D. Bates, M. D. Madsen, and A. M. Nafus. 2014a. Restoration of mountain big sagebrush steppe following prescribed burning to control western juniper. *Environmental Management* 53:1015–1022.

Davies, K. W., J. D. Bates, and A. M. Nafus. 2012. Mowing Wyoming big sagebrush communities with degraded herbaceous understories: has a threshold been crossed? *Rangeland Ecology & Management* 65:498–505.

Davies, K. W., M. Vavra, B. Schultz, and N. Rimbey. 2014b. Implications of longer term rest from grazing in the sagebrush steppe. *Journal of Rangeland Applications* 1:14–34.

DeBano, S. J. 2006a. Effects of livestock grazing on aboveground insect communities in semi-arid grasslands of southeastern Arizona. *Biodiversity and Conservation* 15:2547–2564.

DeBano, S. J. 2006b. The effect of livestock grazing on the rainbow grasshopper: population differences and ecological correlates. *Western North American Naturalist* 66:222–229.

DeBano, S. J., S. M. Roof, M. M. Rowland, and L. A. Smith. 2016. Diet overlap of mammalian herbivores and native bees: Implications for managing co-occurring grazers and pollinators. *Natural Areas Journal* 36:458–477.

Decourtye, A., C. Armengaud, M. Renou, J. Devillers, S. Cluzeau, M. Gauthier, and M.-H. Pham-Delègue. 2004. Imidacloprid impairs memory and brain metabolism in the honeybee (*Apis mellifera* L.). *Pesticide Biochemistry and Physiology* 78:83–92.

Desneux, N., A. Decourtye, and J.-M. Delpuech. 2007. The sublethal effects of pesticides on beneficial arthropods. *Annual Review of Entomology* 52:81–106.

Di Giulio, M., P. J. Edwards, and E. Meister. 2001. Enhancing insect diversity in agricultural grasslands: the roles of management and landscape structure. *Journal of Applied Ecology* 38:310–319.

Dilts, T. D., M. Steele, S. Black, E. Craver, J. Engler, S. Jepsen, A. Jones, S. McKnight, E. Pelton, A. Taylor, and M. Forister. 2018. *Western Monarch and Milkweed Habitat Suitability Modeling Project Version 2 - Maxent Model Outputs*. Xerces Society/US Fish and Wildlife Society/University of Nevada Reno. Available at: www.monarchmilkweedmapper.org/.

DiTomaso, J. M. 2000. Invasive weeds in rangelands: species, impacts, and management. *Weed Science* 48:255–265.

DiTommaso, A., K. M. Averill, M. P. Hoffmann, J. R. Fuchsberg, and J. E. Losey. 2016. Integrating insect, resistance, and floral resource management in weed control decision-making. *Weed Science* 64:743–756.

Dolan, A. C., C. M. Delphia, K. M. O'Neill, and M. A. Ivie. 2017. Bumble bees (Hymenoptera: Apidae) of Montana. *Annals of the Entomological Society of America* 110:129–144.

Donahue, D. L. 2000. Review: The Western Range Revisited: Removing Livestock from Public Lands to Conserve Native Biodiversity. *Journal of Range Management* 53:134–135

Dover, J. W., A. Rescia, S. Fungariño, J. Fairburn, P. Carey, P. Lunt, R. L. H. Dennis, and C. J. Dover. 2010. Can hay harvesting detrimentally affect adult butterfly abundance? *Journal of Insect Conservation* 14:413–418.

Droege, S., J. D. Engler, E. A. Sellers, and L. O'Brien. 2016. *National Protocol Framework for the Inventory and Monitoring of Bees*. 79+v pp. Fort Collins, CO: U.S. Fish and Wildlife Service.

Drut, M. S., W. H. Pyle, and J. A. Crawford. 1994. Diets and food selection of sage grouse chicks in Oregon. *Journal of Range Management* 47:90–93.

Dumroese, K., T. Luna, B. A. Richardson, F. F. Kilkenny, J. B. Runyon, K. Dumroese, T. Luna et al. 2015. Conserving and restoring habitat for greater sage-grouse and other sagebrush-obligate wildlife: the crucial link of forbs and sagebrush diversity. *Native Plants* 16:276–299.

Dumroese, R. K., T. Luna, J. R. Pinto, and T. D. Landis. 2016. Forbs: foundation for restoration of monarch butterflies, other pollinators, and greater sage-grouse in the western United States. *Natural Areas Journal* 36:499–511.

Duncan, C. A., J. J. Jachetta, M. L. Brown, V. F. Carrithers, J. K. Clark, J. M. DiTomaso, R. G. Lym, K. C. McDaniel, M. J. Renz, and P. M. Rice. 2004. Assessing the economic, environmental, and societal losses from invasive plants on rangeland and wildlands. *Weed Technology* 18:1411–1416.

Eckberg, J. E., E. Lee-Mäder, J. Hopwood, S. Foltz Jordan, and B. Borders. 2017. *Native Thistles: A Conservation Practitioner's Guide*. 92 pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Elligsen, H., B. Beinlich, and H. Plachter. 1997. Effects of large-scale cattle grazing on populations of *Coenonympha glycerion* and *Lasiommata megera* (Lepidoptera: Satyridae). *Journal of Insect Conservation* 1:13–23.

Elwell, S. L., T. Griswold, and E. Elle. 2016. Habitat type plays a greater role than livestock grazing in structuring shrub steppe plant–pollinator communities. *Journal of Insect Conservation* 20:515–525.

Ersch, E. A. 2009. "Effects of plant community characteristics on insect abundance: implications for sage-grouse brood-rearing habitat" (master's thesis, Oregon State University).

Espeset, A. E., J. G. Harrison, A. M. Shapiro, C. C. Nice, J. H. Thorne, D. P. Waetjen, J. A. Fordyce, and M. L. Forister. 2016. Understanding a migratory species in a changing world: climatic effects and demographic declines in the western monarch revealed by four decades of intensive monitoring. *Oecologia* 181:819–830.

Evans, E., R. Thorp, S. Jepsen, and S. H. Black. 2008. "Status review of three formerly common species of bumble bee in the subgenus *Bombus*." 63 pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Everaars, J., J. Settele, and C. F. Dormann. 2018. Fragmentation of nest and foraging habitat affects time budgets of solitary bees, their fitness and pollination services, depending on traits: results from an individual-based model. *PloS ONE* 13:e0188269.

F. C. Bryant, M. M. Kothmann, and L. B. Merrill. 1979. Diets of sheep, Angora goats, Spanish goats and white-tailed deer under excellent range conditions. *Journal of Range Management* 32:412–417.

Fernelius, K. J., M. D. Madsen, B. G. Hopkins, S. Bansal, V. J. Anderson, D. L. Eggett, and B. A. Roundy. 2017. Post-fire interactions between soil water repellency, soil fertility and plant growth in soil collected from a burned piñon-juniper woodland. *Journal of Arid Environments* 144:98–109.

Fiedler, A. K., D. A. Landis, and M. Arduser. 2012. Rapid shift in pollinator communities following invasive species removal. *Restoration Ecology* 20:593–602.

Fielding, D. J., and M. A. Brusven. 1995. Grasshopper densities on grazed and ungrazed rangeland under drought conditions in southern Idaho. *Great Basin Naturalist* 55:352–358.

Finch, D., D. Boyce, J. Chambers, C. Colt, C. McCarthy, S. Kitchen, B. Richardson et al. 2015. *USDA Forest Service Sage-Grouse Conservation Science Strategy 2015-2020*. 39 pp. Washington, D.C.: U.S. Forest Service.

Finch, D. M., R. L. Pendleton, M. C. Reeves, J. E. Ott, F. F. Kilkenny, J. L. Butler, J. P. Ott et al. 2016. Rangeland Drought: Effects, Restoration, and Adaptation. In *Effects of Drought on Forests and Rangelands in the United States: A Comprehensive Science Synthesis* (Gen. Tech. Report WO-93b), edited by J. M. Vose, J. S. Clark, C. H. Luce, and T. Patel-Weynand, 155–194. Washington, D.C.: U.S. Forest Service.

Fleishman, E. 2000. Monitoring the response of butterfly communities to prescribed fire. *Environmental Management* 26:685–695.

Fleishman, E., D. D. Murphy, and G. T. Austin. 1999. Butterflies of the Toquima Range, Nevada: distribution, natural history, and comparison to the Toiyabe Range. *Western North American Naturalist* 59:50–62.

Forister, M. L., B. Cousens, J. G. Harrison, K. Anderson, J. H. Thorne, D. Waetjen, C. C. Nice et al. 2016. Increasing neonicotinoid use and the declining butterfly fauna of lowland California. *Biology Letters* 12:20160475

Forister, M. L., J. P. Jahner, K. L. Casner, J. S. Wilson, and A. M. Shapiro. 2011. The race is not to the swift: Long-term data reveal pervasive declines in California's low-elevation butterfly fauna. *Ecology* 92:2222–2235.

Forister, M. L., A. C. McCall, N. J. Sanders, J. A. Fordyce, J. H. Thorne, J. O'Brien, D. P. Waetjen, and A. M. Shapiro. 2010. Compounded effects of climate change and habitat alteration shift patterns of butterfly diversity. *Proceedings of the National Academy of Sciences* 107:2088–2092.

Fork, S. K. 2010. Arthropod assemblages on native and nonnative plant species of a coastal reserve in California. *Environmental Entomology* 39:753–762.

Forrester, J. A., D. J. Leopold, and S. D. Hafner. 2005. Maintaining critical habitat in a heavily managed landscape: effects of power line corridor management on Karner blue butterfly (*Lycaeides melissa samuelis*) habitat. *Restoration Ecology* 13:488–498.

Fox, R., T. Brereton, J. Asher, T. A. August, M. S. Botham, N. A. D. Bourn, K. L. Cruickshanks et al 2015. *The State of the UK's Butterflies 2015*. 28 pp. Wareham: Butterfly Conservation and the Centre for Ecology & Hydrology.

Franzén, M., and E. Öckinger. 2012. Climate-driven changes in pollinator assemblages during the last 60 years in an arctic mountain region in northern Scandinavia. *Journal of Insect Conservation* 16:227–238.

Fürst, M. A., D. P. McMahon, J. L. Osborne, R. J. Paxton, and M. J. F. Brown. 2014. Disease associations between honeybees and bumblebees as a threat to wild pollinators. *Nature* 506:364–366.

Gallai, N., J.-M. Salles, J. Settele, and B. E. Vaissière. 2009. Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. *Ecological Economics* 68:810–821.

Gasparatos, A., C. N. H. Doll, M. Esteban, A. Ahmed, and T. A. Olang. 2017. Renewable energy and biodiversity: implications for transitioning to a green economy. *Renewable and Sustainable Energy Reviews* 70:161–184.

Gegear, R. J., and J. G. Burns. 2007. The birds, the bees, and the virtual flowers: can pollinator behavior drive ecological speciation in flowering plants? *American Naturalist* 170:551–566.

Geldmann, J., and J. P. González-Varo. 2018. Conserving honey bees does not help wildlife. *Science* 359:392–393.

Genung, M. A., J.-P. Lessard, C. B. Brown, W. A. Bunn, M. A. Cregger, W. M. N. Reynolds, E. Felker-Quinn et al. 2010. Non-additive effects of genotypic diversity increase floral abundance and abundance of floral visitors. *PloS ONE* 5:e8711.

Gervais, J. A., B. Luukinen, K. Buhl, and D. Stone. 2009. "Malathion general fact sheet." National Pesticide Information Center, Oregon State University Extension Services. http://npic.orst.edu/factsheets/malagen.html (accessed 6/19/18).

Gess, F. W., and S. K. Gess. 1999. The use by wasps, bees and spiders of shells of *Trigonephrus* Pilsb. (Mollusca: Gasteropoda: Dorcasiidae) in desertic winter-rainfall areas in southern Africa. *Journal of Arid Environments* 43:143–153.

Ghazoul, J. 2006. Floral diversity and the facilitation of pollination. *Journal of Ecology* 94:295–304.

Gilgert, W., and M. Vaughan. 2011. The value of pollinators and pollinator habitat to rangelands: connections among pollinators, insects, plant communities, fish, and wildlife. *Rangelands* 33:14–19.

Gill, R. J., K. C. R. Baldock, M. J. F. Brown, J. E. Cresswell, L. V. Dicks, M. T. Fountain, M. P. D. Garratt et al. 2016. Chapter four: Protecting an ecosystem service: Approaches to understanding and mitigating threats to wild insect pollinators. In *Advances in Ecological Research*, edited by G. Woodward and D. A. Bohan, 135–206. Cambridge, MA: Academic Press.

Gonzalez, N., S. J. DeBano, C. Kimoto, and R. V. Taylor. 2013. Native bees associated with isolated aspen stands in Pacific Northwest bunchgrass prairie. *Natural Areas Journal* 33:374–383.

Goodell, K., and I. M. Parker. 2017. Invasion of a dominant floral resource: effects on the floral community and pollination of native plants. *Ecology* 98:57–69.

Goulson, D. 2010. *Bumblebees: Behaviour, Ecology, and Conservation*. 336 pp. Oxford: Oxford University Press.

Goulson, D., E. Nicholls, C. Botías, and E. L. Rotheray. 2015. Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. *Science* 347:1255957.

Gregg, M. A., and J. A. Crawford. 2009. Survival of greater sage-grouse chicks and broods in the northern Great Basin. *Journal of Wildlife Management* 73:904–913.

Griswold, T., S. Higbee, and O. Messinger. 2003. "Pollination Ecology Final Report 2003 Biennium Clark County, Nevada (2004-2005)." Logan: USDA-ARS Bee Biology & Systematics Laboratory, Utah State University.

Griswold, T. L., and O. J. Messinger. 2009. "The Dominator: Rabbitbrush (*Ericameria* spp.) in the late summer-fall pollinator market of the Colorado Plateau." Abstract for the 94th ESA Annual Meeting (August 2-7, 2009). Accessible at https://bit.ly/2lrKTAA (accessed 6/18/18).

Griswold, T., F. D. Parker, and V. J. Tepedino. 1997. "The Bees of the San Rafael Desert: Implications for the Bee Fauna of the Grand Staircase-Escalante National Monument." In *Learning from the Land. Grand Staircase-Escalante National Monument Science Symposium Proceedings: November 4-5, 1997, Southern Utah University*, edited by L. M. Hill, 175 – 186. Salt Lake City: Bureau of Land Management.

Grundel, R., R. P. Jean, K. J. Frohnapple, G. A. Glowacki, P. E. Scott, and N. B. Pavlovic. 2010. Floral and nesting resources, habitat structure, and fire influence bee distribution across an open-forest gradient. *Ecological Applications* 20:1678–1692.

Habel, J. C., A. Segerer, W. Ulrich, O. Torchyk, W. W. Weisser, and T. Schmitt. 2016. Butterfly community shifts over two centuries. *Conservation Biology* 30:754–762.

Hanula, J., and S. Horn. 2011. Removing an exotic shrub from riparian forests increases butterfly abundance and diversity. *Forest Ecology And Management* 262:674–680.

Hanula, J. L., M. D. Ulyshen, and S. Horn. 2016. Conserving pollinators in North American forests: a review. *Natural Areas Journal* 36:427–439.

Harmon-Threatt, A., and K. Chin. 2016. Common methods for tallgrass prairie restoration and their potential effects on bee diversity. *Natural Areas Journal* 36:400–411.

Harper, M. G., C. H. Dietrich, R. L. Larimore, and P. A. Tessene. 2000. Effects of prescribed fire on prairie arthropods: An enclosure study. *Natural Areas Journal* 20:325–335.

Hatfield, R., S. H. Black, and S. Jepsen. 2013. "Range-wide strategy and monitoring protocol for the mardon skipper *(Polites mardon,* W.H. Edwards, 1881*)*." Portland, OR: The Xerces Society or Invertebrate Conservation.

Hatfield, R. G., S. R. Colla, S. Jepsen, L. L. Richardson, R. W. Thorp, and S. Foltz Jordan. 2014. "IUCN Assessments for North American *Bombus* spp. for the North American IUCN Bumble Bee Specialist Group." Portland, OR. The Xerces Society for Invertebrate Conservation, Portland, OR.

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

Hatfield, R. G., S. Jepsen, M. Vaughan, S. Black, and E. Lee-Mäder. 2018. *An Overview of the Potential Impacts of Honey Bees to Native Bees, Plant Communities, and Ecosystems in Wild Landscapes: Recommendations for Land Managers*. 12 pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Hatfield, R., S. Jepsen, E. Mader, S. H. Black, and M. Shepherd. 2012. *Conserving Bumble Bees: Guidelines for Creating and Managing Habitat for America's Declining Pollinators*. 32 pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Havstad, K. M., D. P. C. Peters, R. Skaggs, J. Brown, B. Bestelmeyer, E. Fredrickson, J. Herrick, and J. Wright. 2007. Ecological services to and from rangelands of the United States. *Ecological Economics* 64:261–268.

Hayes, F., J. Williamson, and G. Mills. 2012. Ozone pollution affects flower numbers and timing in a simulated BAP priority calcareous grassland community. *Environmental Pollution* 163:40–47.

Hegland, S. J., A. Nielsen, A. Lázaro, A.-L. Bjerknes, and Ø. Totland. 2009. How does climate warming affect plant-pollinator interactions? *Ecology Letters* 12:184–195.

Henneman, M. L., and J. Memmott. 2001. Infiltration of a Hawaiian community by introduced biological control agents. *Science* 293:1314–1316.

Hennings, L. 2017. "Hiking, mountain biking and equestrian use in natural areas: A recreation ecology literature review." Portland, OR: Metro Parks and Nature. Available at: https://bit.ly/2HARoyl (accessed 4/17/18).

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

Hickling, R., D. B. Roy, J. K. Hill, R. Fox, and C. D. Thomas. 2006. The distributions of a wide range of taxonomic groups are expanding polewards. *Global Change Biology* 12:450–455.

Hill, K. C., J. D. Bakker, and P. W. Dunwiddie. 2017. Prescribed fire in grassland butterfly habitat: targeting weather and fuel conditions to reduce soil temperatures and burn severity. *Fire Ecology* 13:24–41.

Holechek, J. L. 1984. Comparative contribution of grasses, forbs, and shrubs to the nutrition of range ungulates. *Rangelands* 6:261–263.

Holechek, J. L., H. Gomez, F. Molinar, and D. Galt. 1999. Grazing studies: what we've learned. Rangelands 21:12–16.

Hooven, L., R. R. Sagili, and E. Johansen. 2013. *How to Reduce Bee Poisoning from Pesticides* (PNW Extension publication 591). 35 pp. Corvallis: Oregon State University Extension Service.

Hopwood, J., S. Black, and S. Fleury. 2015. *Roadside Best Management Practices that Benefit Pollinators: Handbook for Supporting Pollinators through Roadside Maintenance and Landscape Design* (FHWA-HEP-16-059). 88 pp. Washington, D.C.: U.S. Department of Transportation, Federal Highway Administration.

Hopwood, J. L. 2008. The contribution of roadside grassland restorations to native bee conservation. *Biological Conservation* 141:2632–2640.

Howery, L. D. 2016. *Rangeland Management Before, During and After Drought*. 6 pp. Tucson: College of Agriculture and Life Sciences, University of Arizona.

Hughes, L. 2000. Biological consequences of global warming: is the signal already apparent. *Trends in Ecology & Evolution* 15:56–61.

Hulvey, K. B., E. A. Leger, L. M. Porensky, L. M. Roche, K. E. Veblen, A. Fund, J. Shaw, and E. S. Gornish. 2017. Restoration islands: a tool for efficiently restoring dryland ecosystems? *Restoration Ecology* 25(S2):S124–S134.

Humbert, J.-Y., J. Ghazoul, G. J. Sauter, and T. Walter. 2010. Impact of different meadow mowing techniques on field invertebrates. *Journal of Applied Entomology* 134:592–599.

Huntzinger, M. 2003. Effects of fire management practices on butterfly diversity in the forested western United States. *Biological Conservation* 113:1–12.

Hutchinson, K. J. and K. L. King. 1980. The effects of sheep stocking level on invertebrate abundance, biomass and energy utilization in a temperate, sown grassland. *Journal of Applied Ecology* 17:369.

Inouye, D. W. and G. H. Pyke. 1988. Pollination biology in the snowy mountains of australia: comparisons with montane Colorado, USA. *Australian Journal of Ecology* 13:191–205.

IPCC [Intergovernmental Panel on Climate Change]. 2014. "Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change," edited by R. K. Pachauri and L. A. Meyer. 151 pp. Geneva, Switzerland: Intergovernmental Panel on Climate Change.

Isaac, N. J. B., K. L. Cruickshanks, A. M. Weddle, J. Marcus Rowcliffe, T. M. Brereton, R. L. H. Dennis, D. M. Shuker, and C. D. Thomas. 2011. Distance sampling and the challenge of monitoring butterfly populations. *Methods in Ecology and Evolution* 2:585–594.

IUCN SSC BBSG [International Union for Conservation of Nature Species Survival Commission Bumblebee Specialist Group]. 2016. Commercial bumblebee policy statement. Bumble Bee Specialist Group. Accessible at https://bumblebeespecialistgroup.org/policy/ (accessed 11/21/17).

Iwasa, T., N. Motoyama, J. T. Ambrose, and R. M. Roe. 2004. Mechanism for the differential toxicity of neonicotinoid insecticides in the honey bee, *Apis mellifera*. *Crop Protection* 23:371–378.

Jerrentrup, J. S., N. Wrage-Mönnig, K.-U. Röver, and J. Isselstein. 2014. Grazing intensity affects insect diversity via sward structure and heterogeneity in a long-term experiment. *Journal of Applied Ecology* 51:968–977.

Johansen, C. A., and D. F. Mayer. 1990. *Pollinator Protection: A Bee & Pesticide Handbook*. 212 pp. Kalamazoo: Wicwas Press.

Johnsgard, P. A. 2016. *The Hummingbirds of North America, Second Edition*. 278 pp. Washington, D.C.: Smithsonian Institution.

Johnson, R. M. 2015. Honey bee toxicology. *Annual Review of Entomology* 60:415–434.

Johst, K., M. Drechsler, J. Thomas, and J. Settele. 2006. Influence of mowing on the persistence of two endangered large blue butterfly species. *Journal of Applied Ecology* 43:333–342.

Kaser, J. M., and G. E. Heimpel. 2015. Linking risk and efficacy in biological control host–parasitoid models. *Biological Control* 90:49–60.

Kauffman, J. B., A. S. Thorpe, and E. N. J. Brookshire. 2004. Livestock exclusion and belowground ecosystem responses in riparian meadows of eastern Oregon. *Ecological Applications* 14:1671–1679.

Kayser, M. 2014. How to manage habitats of the endangered lycaenid butterfly *Lycaena helle* (Denis & Schiffermüller, 1775) (Insecta, Lepidoptera). *Bulletin de la Société des Naturalistes Luxembourgeois* 115:241–249.

Kearns, C. A., and D. W. Inouye. 1997. Pollinators, flowering plants, and conservation biology. *Bioscience* 47:297–307.

Kearns, C. A., D. W. Inouye, and N. M. Waser. 1998. Endangered mutualisms: the conservation of plant-pollinator interactions. *Annual Review of Ecology and Systematics* 29:83–112.

Kearns, C. A., and D. M. Oliveras. 2009. Boulder County bees revisited: a resampling of Boulder Colorado bees a century later. *Journal of Insect Conservation* 13:603–613.

Kerr, J. 2016. "Climate change and rapid bumblebee and butterfly losses within and across continents." Paper presented at the 2016 XXV International Congress of Entomology, Orlando, FL, September 2016. https://esa.confex.com/esa/ice2016/meetingapp.cgi/Paper/105048.

Kimoto, C. 2011. "Effect of livestock grazing on native bees in a Pacific Northwest bunchgrass prairie" (master's thesis, Oregon State University).

Kimoto, C., S. J. DeBano, R. W. Thorp, S. Rao, and W. P. Stephen. 2012a. Investigating temporal patterns of a native bee community in a remnant North American bunchgrass prairie using blue vane traps. *Journal of Insect Science* 12:1–23.

Kimoto, C., S. J. DeBano, R. W. Thorp, R. V. Taylor, H. Schmalz, T. DelCurto, T. Johnson, P. L. Kennedy, and S. Rao. 2012b. Short-term responses of native bees to livestock and implications for managing ecosystem services in grasslands. *Ecosphere* 3:88.

Klein, A.-M., B. E. Vaissière, J. H. Cane, I. Steffan-Dewenter, S. A. Cunningham, C. Kremen, and T. Tscharntke. 2007. Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society of London B: Biological Sciences* 274:303–313.

Kleintjes, P. K., B. F. Jacobs, and S. M. Fettig. 2004. Initial response of butterflies to an overstory reduction and slash mulching treatment of a degraded pinon-juniper woodland. *Restoration Ecology* 12:231.

van Klink, R., S. Nolte, F. S. Mandema, D. D. G. Lagendijk, M. F. WallisDeVries, J. P. Bakker, P. Esselink, and C. Smit. 2016. Effects of grazing management on biodiversity across trophic levels–the importance of livestock species and stocking density in salt marshes. *Agriculture, Ecosystems & Environment* 235:329–339.

Knight, T. M., J. L. Dunn, L. A. Smith, J. Davis, and S. Kalisz. 2009. Deer facilitate invasive plant success in a Pennsylvania forest understory. *Natural Areas Journal* 29:110–116.

Knutson, K. C., D. A. Pyke, and T. A. Wirth. 2014. Long-term effects of seeding after wildfire on vegetation in Great Basin shrubland ecosystems. *Journal of Applied Ecology* 51:1414–1424

Kobernus, P. 2011. Managing a mountain: the San Bruno Mountain habitat conservation plan. *Fremontia* 38/39:10–17. Available at: https://bit.ly/2JZWMfH (accessed 12/10/17).

Konvicka, M., J. Novak, J. Benes, Z. Fric, J. Bradley, P. Keil, J. Hrcek, K. Chobot, and P. Marhoul. 2008. The last population of the woodland brown butterfly (*Lopinga achine*) in the Czech Republic: habitat use, demography and site management. *Journal of Insect Conservation* 12:549–560.

Kopit, A. M., and T. L. Pitts-Singer. 2018. Routes of pesticide exposure in solitary, cavity-nesting bees. *Environmental Entomology* 47:499–510.

Kral, K., J. Harmon, R. Limb, and T. Hovick. 2018. Improving our science: the evolution of butterfly sampling and surveying methods over time. *Journal of Insect Conservation* 22:1–14.

Kremen, C., K. S. Ullman, and R. W. Thorp. 2011. Evaluating the quality of citizen-scientist data on pollinator communities. *Conservation Biology* 25:607–617

Kremen, C., N. M. Williams, and R. W. Thorp. 2002. Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Sciences* 99:16812–16816.

Kruess, A., and T. Tscharntke. 2002a. Contrasting responses of plant and insect diversity to variation in grazing intensity. *Biological Conservation* 106:293–302.

Kruess, A. and T. Tscharntke. 2002b. Grazing intensity and the diversity of grasshoppers, butterflies, and trap-nesting bees and wasps. *Conservation Biology* 16:1570–1580.

Kudo, G., and T. Y. Ida. 2013. Early onset of spring increases the phenological mismatch between plants and pollinators. *Ecology* 94:2311–2320.

Kühne, I., R. Arlettaz, J. Pellet, L. Bruppacher, and J.-Y. Humbert. 2015. Leaving an uncut grass refuge promotes butterfly abundance in extensively managed lowland hay meadows in Switzerland. *Conservation Evidence* 12:25–27.

Kulhanek, K., N. Steinhauer, K. Rennich, D. M. Caron, R. R. Sagili, J. S. Pettis, J. D. Ellis et al. 2017. A national survey of managed honey bee 2015–2016 annual colony losses in the USA. *Journal of Apicultural Research* 56:328–340.

Kumar, S. V., D. Lucyshyn, K. E. Jaeger, E. Alós, E. Alvey, N. P. Harberd, and P. A. Wigge. 2012. Transcription factor PIF4 controls the thermosensory activation of flowering. *Nature* 484:242–245.

Kuussaari, M., J. Heliölä, M. Luoto, and J. Pöyry. 2007. Determinants of local species richness of diurnal Lepidoptera in boreal agricultural landscapes. *Agriculture, Ecosystems & Environment* 122:366–376.

Ladurner, E., J. Bosch, W. P. Kemp, and S. Maini. 2005. Assessing delayed and acute toxicity of five formulated fungicides to *Osmia lignaria* Say and *Apis mellifera*. *Apidologie* 36:449–460.

Larrucea, E. S., and P. F. Brussard. 2008. Shift in location of pygmy rabbit (*Brachylagus idahoensis*) habitat in response to changing environments. *Journal of Arid Environments* 72:1636–1643.

Larson, B. M. H., P. G. Kevan, and D. W. Inouye. 2001. Flies and flowers: taxonomic diversity of anthophiles and pollinators. *The Canadian Entomologist* 133:439–465.

Larsson, M. and M. Franzén. 2007. Critical resource levels of pollen for the declining bee *Andrena hattorfiana* (Hymenoptera, Andrenidae). *Biological Conservation* 134:405–414.

LeBuhn, G., T. Griswold, R. Minckley, S. Droege, T. Roulston, J. Cane, F. Parker et al. 2003. "A standardized method for monitoring bee populations—the bee inventory (BI) plot." Available at http://online.sfsu.edu/beeplot/ (accessed 6/18/18).

Litt, A. R., E. E. Cord, T. E. Fulbright, and G. L. Schuster. 2014. Effects of invasive plants on arthropods. *Conservation Biology* 28:1532–1549.

Littlewood, N. A. 2008. Grazing impacts on moth diversity and abundance on a Scottish upland estate. *Insect Conservation and Diversity* 1:151–160.

Lobova, T. A., C. K. Geiselman, and S. A. Mori. 2009. *Seed Dispersal by Bats in the Neotropics (Memoirs of The New York Botanical Garden Volume 101)*. 465 pp. New York: NYBG Press.

Loffland, H. L., J. S. Polasik, M. W. Tingley, E. A. Elsey, C. Loffland, G. Lebuhn, and R. B. Siegel. 2017. Bumble bee use of post-fire chaparral in the central Sierra Nevada. *The Journal of Wildlife Management* 81:1084–1097.

Longcore, T. 2003. Terrestrial arthropods as indicators of ecological restoration success in coastal sage scrub (California, U.S.A.). *Restoration Ecology* 11:397–409.

Lopezaraiza-Mikel, M. E., R. B. Hayes, M. R. Whalley, and J. Memmott. 2007. The impact of an alien plant on a native plant–pollinator network: an experimental approach. *Ecology Letters* 10:539–550.

Losey, J. E., and M. Vaughan. 2006. The economic value of ecological services provided by insects. *Bioscience*

56:311–323.

Lovich, J. E., and D. Bainbridge. 1999. Anthropogenic degradation of the southern California desert ecosystem and prospects for natural recovery and restoration. *Environmental Management* 24:309–326.

Maclean, I. M. D., and R. J. Wilson. 2011. Recent ecological responses to climate change support predictions of high extinction risk. *Proceedings of the National Academy of Sciences* 108:12337–12342.

Mader, E. 2010. Appendix A: Managing parasites and disease in solitary bee operations. In *Managing Alternative Pollinators*, edited by E. Mader, M. Spivak, and E. Evans, 114–125. Ithaca, NY: Natural Resource, Agriculture, and Engineering Service.

Madsen, M. D., K. W. Davies, C. S. Boyd, J. D. Kerby, and T. J. Svejcar. 2016. Emerging seed enhancement technologies for overcoming barriers to restoration. *Restoration Ecology* 24(S2):S77–S84.

Maestas, J. D., S. Conner, B. Zeedyk, B. Neely, R. Rondeau, N. Seward, T. Chapman, L. With, and R. Murph. 2018. Hand-built structures for restoring degraded meadows in sagebrush rangelands: Examples and lessons learned from the Upper Gunnison River Basin, Colorado. Range Technical Note No. 40. USDA-NRCS, Denver, CO.

Mallinger, R. E., H. R. Gaines-Day, and C. Gratton. 2017. Do managed bees have negative effects on wild bees?: A systematic review of the literature. *PloS ONE* 12:e0189268.

Manley, R., M. Boots, and L. Wilfert. 2015. Review: Emerging viral disease risk to pollinating insects: ecological, evolutionary and anthropogenic factors. *Journal of Applied Ecology* 52:331–340.

Mao, W., M. A. Schuler, and M. R. Berenbaum. 2017. Disruption of quercetin metabolism by fungicide affects energy production in honey bees (*Apis mellifera*). *Proceedings of the National Academy of Sciences* 114:2538–2543.

Marks, R. 2006. *Native Pollinators*. (Fish and Wildlife Habitat Management Leaflet. No. 34). Washington, D.C.: U.S. Department of Agriculture Natural Resources Conservation Service.

McArt, S. H., A. A. Fersch, N. J. Milano, L. L. Truitt, and K. Böröczky. 2017a. High pesticide risk to honey bees despite low focal crop pollen collection during pollination of a mass blooming crop. *Scientific Reports* 7:46554.

McArt, S. H., C. Urbanowicz, S. McCoshum, R. E. Irwin, and L. S. Adler. 2017b. Landscape predictors of pathogen prevalence and range contractions in us bumblebees. *Proceedings of the Royal Society of London B: Biological Sciences* 284:20172181.

McCormick, M. L. 2017. "Heterogeneous planting design supports rich and abundant pollinator communities to minimize use of restoration resources in arid lands" (master's thesis, Northern Arizona University).

McDougald, N. K., W. E. Frost, and R. L. Phillips. 2001. "Livestock Management During Drought." (Rangeland Management Series ANR Publication 8034.) 6pp. Oakland: University of California, Agriculture and Natural Resources.

McFrederick, Q. S., and G. LeBuhn. 2006. Are urban parks refuges for bumble bees *Bombus* spp. (Hymenoptera: Apidae)? *Biological Conservation* 129:372–382.

McIver, J., M. Brunson, S. Bunting, J. Chambers, P. Doescher, J. Grace, A. Hulet et al. 2014. A synopsis of short-term response to alternative restoration treatments in sagebrush-steppe: the sagestep project. *Rangeland Ecology & Management* 67:584–598.

McIver, J., and E. Macke. 2014. Short-term butterfly response to sagebrush steppe restoration treatments. *Rangeland Ecology & Management* 67:539–552.

Memmott, J., P. G. Craze, N. M. Waser, and M. V. Price. 2007. Global warming and the disruption of plant-pollinator interactions. *Ecology Letters* 10:710–717.

Menz, M. H. M., R. D. Phillips, R. Winfree, C. Kremen, M. A. Aizen, S. D. Johnson, and K. W. Dixon. 2011. Reconnecting plants and pollinators: challenges in the restoration of pollination mutualisms. *Trends in Plant Science* 16:4–12.

Messinger, O. 2006. "A survey of the bees of Grand Staircase-Escalante National Monument, southern Utah: incidence, abundance, and community dynamics" (master's thesis, Utah State University).

Meyer, S., D. Unternährer, R. Arlettaz, J.-Y. Humbert, and M. H. M. Menz. 2017. Promoting diverse communities of wild bees and hoverflies requires a landscape approach to managing meadows. *Agriculture, Ecosystems & Environment* 239:376–384.

Michener, C. D. 2007. *The Bees of the World*. 992 pp. Baltimore: Johns Hopkins University Press.

Minckley, R. L. 2014. Maintenance of richness despite reduced abundance of desert bees (Hymenoptera: Apiformes) to persistent grazing. *Insect Conservation and Diversity* 7:263–273.

Minckley, R. L., and T. H. Roulston. 2006. Incidental mutualisms and pollen specialization among bees. In *Plant-Pollinator Interactions: from specialization to generalization*, edited by N. M. Waser and J. Ollerton, 69–98. Chicago: University of Chicago Press.

Minckley, R. L., T. H. Roulston, and N. M. Williams. 2013. Resource assurance predicts specialist and generalist bee activity in drought. *Proceedings of the Royal Society of London B: Biological Sciences* 280:20122703.

Minckley, R. L., W. T. Wcislo, D. Yanega, and S. L. Buchmann. 1994. Behavior and phenology of a specialist bee (*Dieunomia*) and sunflower (*Helianthus*) pollen availability. *Ecology* 75:1406–1419.

Mitikka, V., R. K. Heikkinen, M. Luoto, M. B. Araújo, K. Saarinen, J. Pöyry, and S. Fronzek. 2008. Predicting range expansion of the map butterfly in northern Europe using bioclimatic models. *Biodiversity and Conservation* 17:623–641.

Monahan, W. B., A. Rosemartin, K. L. Gerst, N. A. Fisichelli, T. Ault, M. D. Schwartz, J. E. Gross, and J. F. Weltzin.

2016. Climate change is advancing spring onset across the U.S. national park system. *Ecosphere* 7:e01465.

Morales, C. L., and M. A. Aizen. 2002. Does invasion of exotic plants promote invasion of exotic flower visitors? A case study from the temperate forests of the southern Andes. *Biological Invasions* 4:87–100.

Morandin, L. A., and C. Kremen. 2013. Bee preference for native versus exotic plants in restored agricultural hedgerows. *Restoration Ecology* 21:26–32.

Moranz, R. A., S. D. Fuhlendorf, and D. M. Engle. 2014. Making sense of a prairie butterfly paradox: The effects of grazing, time since fire, and sampling period on regal fritillary abundance. *Biological Conservation* 173:32–41.

Moretti, M., F. de Bello, S. P. M. Roberts, and S. G. Potts. 2009. Taxonomical vs. functional responses of bee communities to fire in two contrasting climatic regions. *Journal of Animal Ecology* 78:98–108.

Moretti, M., P. Duelli, and M. K. Obrist. 2006. Biodiversity and resilience of arthropod communities after fire disturbance in temperate forests. *Oecologia* 149:312–327.

Moron, D., M. Lenda, P. Skórka, H. Szentgyörgyi, J. Settele, and M. Woyciechowski. 2009. Wild pollinator communities are negatively affected by invasion of alien goldenrods in grassland landscapes. *Biological Conservation* 142:1322–1332.

Morris, G. M., C. Kline, and S. M. Morris. 2015. Status of *Danaus plexippus* population in Arizona. *Journal of the Lepidopterists' Society* 69:91–107.

Morris, M. G. 2000. The effects of structure and its dynamics on the ecology and conservation of arthropods in British grasslands. *Biological Conservation* 95:129–142.

Morris, M. G. 1967. Differences between the invertebrate faunas of grazed and ungrazed chalk grassland. I. Responses of some phytophagous insects to cessation of grazing. *Journal of Applied Ecology* 4:459–474.

Müller, A., S. Diener, S. Schnyder, K. Stutz, C. Sedivy, and S. Dorn. 2006. Quantitative pollen requirements of solitary bees: implications for bee conservation and the evolution of bee–flower relationships. *Biological Conservation* 130:604–615.

Murphy, D. D., and S. B. Weiss. 1988. Ecological studies and the conservation of the bay checkerspot butterfly, *Euphydryas editha bayensis*. *Biological Conservation* 46:183–200.

Mussen, E. C., J. E. Lopez, and C. Y. S. Peng. 2004. Effects of selected fungicides on growth and development of larval honey bees, *Apis mellifera* L. (Hymenoptera: Apidae). *Environmental Entomology* 33:1151–1154.

Nabhan, G. P., and S. L. Buchmann. 1997. Services provided by pollinators. *Nature's Services: Societal Dependence on Natural Ecosystems*, edited by G. C. Daily, 133–150. Washington, D.C.: Island Press.

Nahban, G., I. Warren, and C. Taylor. 2015. "Milkweed Seed Supply Chains for Monarch Habitat Restoration." Tucson: University of Arizona, The Southwest Center.

Natureserve. 2018. NatureServe Web Service. Arlington, VA. U.S.A. Available http://services.natureserve.org. (Accessed: <April, 2018>)

NCBI [National Center for Biotechnology Information]. 2018. PubChem Compound Database; CID=37123, https://pubchem.ncbi.nlm.nih.gov/compound/37123 (accessed 1/22/18).

Ne'eman, G., A. Dafni, and S. G. Potts. 2000. The effect of fire on flower visitation rate and fruit set in four core-species in east Mediterranean scrubland. *Plant Ecology* 146:97–104.

Nilsson, S. G., M. Franzén, and E. Jönsson. 2008. Long-term land-use changes and extinction of specialised butterflies. *Insect Conservation and Diversity* 1:197–207.

Noordijk, J., K. Delille, A. P. Schaffers, and K. V. Sýkora. 2009. Optimizing grassland management for flower-visiting insects in roadside verges. *Biological Conservation* 142:2097–2103.

Oakley, C. A. and J. S. Knox. 2013. Plant species richness increases resistance to invasion by non-resident plant species during grassland restoration. *Applied Vegetation Science* 16:21–28.

Oldfield, S., and P. Olwell. 2015. The right seed in the right place at the right time. *Bioscience* 65:955–956.

Oliver, T. H., T. Brereton, and D. B. Roy. 2013. Population resilience to an extreme drought is influenced by habitat area and fragmentation in the local landscape. *Ecography* 36:579–586.

Ollerton, J., R. Winfree, and S. Tarrant. 2011. How many flowering plants are pollinated by animals? *Oikos* 120:321–326.

Olson, B. E. 1999. Impacts of noxious weeds on ecologic and economic systems. *Biology And Management Of Noxious Rangeland Weeds. Oregon State University Press, Corvallis, OR* 4–18.

Olwell, P., and L. Riibe. 2016. National seed strategy: restoring pollinator habitat begins with the right seed in the right place at the right time. *Natural Areas Journal* 36:363–365.

Orford, K. A., I. P. Vaughan, and J. Memmott. 2015. The forgotten flies: the importance of non-syrphid Diptera as pollinators. *Proceedings of the Royal Society of London B: Biological Sciences* 282:20142934.

Orr, M. C., T. Griswold, J. P. Pitts, and F. D. Parker. 2016. A new bee species that excavates sandstone nests. *Current Biology* 26:R792–R793.

O'Toole, C., and A. Raw. 1991. *Bees of the World*. 192 pp. London: Blandford Press.

Ott, J. E., R. D. Cox, N. L. Shaw, B. A. Newingham, A. C. Ganguli, M. Pellant, B. A. Roundy, and D. L. Eggett. 2016. Postfire drill-seeding of great basin plants: effects of contrasting drills on seeded and nonseeded species. *Rangeland Ecology & Management* 69:373–385.

Palladini, J. 2013. "The influence of exotic plant invasion on native bee abundance and demography, and the pollination and abundance of native plants" (PhD diss., University of Montana).

Papanikolaou, A. D., I. Kühn, M. Frenzel, and O. Schweiger.

2017. Semi-natural habitats mitigate the effects of temperature rise on wild bees. *Journal of Applied Ecology* 54:527–536.

Park, M. G., E. J. Blitzer, J. Gibbs, J. E. Losey, and B. N. Danforth. 2015. Negative effects of pesticides on wild bee communities can be buffered by landscape context. *Proceedings of the Royal Society of London B: Biological Sciences* 282:20150299.

Parmesan, C. 1996. Climate and species' range. *Nature* 382:765–766.

Parmesan, C. 2006. Ecological and evolutionary responses to recent climate change. *Annual Review of Ecology, Evolution, and Systematics* 37:637–669.

Payne, G. F., J. W. Foster, and W. C. Leininger. 1983. Vehicle impacts on northern Great Plains range vegetation. *Journal of Range Management* 36:327–331.

Pellmyr, O. 2003. Yuccas, yucca moths, and coevolution: A review. *Annals of the Missouri Botanical Garden* 90:35–55.

Pelton, E., S. Jepsen, C. Schultz, C. Fallon, and S. H. Black. 2016. *State of the Monarch Butterfly Overwintering Sites in California*. 40+vi pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Perlut, N. 2007. "Effects of hayfield management on grassland songbirds" (PhD diss., University of Vermont).

Pieper, R. D., and R. F. Beck. 1980. Importance of forbs on southwestern ranges. *Rangelands* 2:35–36.

Pilling, E. D., K. A. C. Bromleychallenor, C. H. Walker, and P. C. Jepson. 1995. Mechanism of synergism between the pyrethroid insecticide λ-cyhalothrin and the imidazole fungicide prochloraz, in the honeybee (*Apis mellifera* L.). *Pesticide Biochemistry And Physiology* 51:1–11.

Pitts, J. P., J. S. Wilson, K. A. Williams, and N. F. Boehme. 2009. Velvet ants (Hymenoptera: Mutillidae) of the Algodones Sand Dunes of California, USA. *Zootaxa* 2131:1–53.

Pollard, E. 1977. A method for assessing changes in the abundance of butterflies. *Biological Conservation* 12:115–134.

Pollard, E., and T. J. Yates. 1994. *Monitoring Butterflies for Ecology and Conservation: The British Butterfly Monitoring Scheme*. 274+xiv pp. Dordrecht: Springer.

Ponisio, L. C., K. Wilkin, L. K. M'Gonigle, K. Kulhanek, L. Cook, R. Thorp, T. Griswold, and C. Kremen. 2016. Pyrodiversity begets plant–pollinator community diversity. *Global Change Biology* 22:1794–1808.

Potts, S. G., J. C. Biesmeijer, C. Kremen, P. Neumann, O. Schweiger, and W. E. Kunin. 2010a. Global pollinator declines: trends, impacts and drivers. *Trends in Ecology & Evolution* 25:345–353.

Potts, S. G., S. P. M. Roberts, R. Dean, G. Marris, M. A. Brown, R. Jones, P. Neumann, and J. Settele. 2010b. Declines of managed honey bees and beekeepers in Europe. *Journal of Apicultural Research* 49:15–22.

Potts, S. G., B. Vulliamy, A. Dafni, G. Ne'eman, and P. Willmer. 2003. Linking bees and flowers: How do floral communities structure pollinator communities? *Ecology* 84:2628–2642.

Potts, S. G., B. A. Woodcock, S. P. M. Roberts, T. Tscheulin, E. S. Pilgrim, V. K. Brown, and J. R. Tallowin. 2009. Enhancing pollinator biodiversity in intensive grasslands. *Journal of Applied Ecology* 46:369–379.

Pöyry, J., S. Lindgren, J. Salminen, and M. Kuussaari. 2004. Restoration of butterfly and moth communities in semi-natural grasslands by cattle grazing. *Ecological Applications* 14:1656–1670.

Pöyry, J., S. Lindgren, J. Salminen, and M. Kuussaari. 2005. Responses of butterfly and moth species to restored cattle grazing in semi-natural grasslands. *Biological Conservation* 122:465–478.

Pöyry, J., M. Luoto, J. Paukkunen, J. Pykälä, K. Raatikainen, and M. Kuussaari. 2006. Different responses of plants and herbivore insects to a gradient of vegetation height: an indicator of the vertebrate grazing intensity and successional age. *Oikos* 115:401–412.

Primack, R. B. 1983. Insect pollination in the New Zealand mountain flora. *New Zealand Journal of Botany* 21:317–333.

Proctor, M., P. Yeo, and A. Lack. 1996. *The Natural History of Pollination*. 479 pp. Portland, OR: Timber Press.

Pryke, J. S., and M. J. Samways. 2012. Importance of using many taxa and having adequate controls for monitoring impacts of fire for arthropod conservation. *Journal of Insect Conservation* 16:177–185.

Pyke, D. A., J. C. Chambers, M. Pellant, R. F. Miller, J. L. Beck, P. S. Doescher, B. A. Roundy et al. 2017. *Restoration Handbook for Sagebrush Steppe Ecosystems with Emphasis on Greater Sage-Grouse Habitat— Part 3. Site Level Restoration Decisions*. 62 pp. Reston, VA: US Geological Survey. Available at: https://www. fs.usda.gov/treesearch/pubs/53743.

Pyke, G. H. 1999. The introduced honeybee *Apis mellifera* and the precautionary principle: Reducing the conflict. *Australian Zoologist* 31:181–186.

Pywell, R. F., E. A. Warman, C. Carvell, T. H. Sparks, L. V. Dicks, D. Bennett, A. Wright, C. N. R. Critchley, and A. Sherwood. 2005. Providing foraging resources for bumblebees in intensively farmed landscapes. *Biological Conservation* 121:479–494.

Ralphs, M. H., and J. A. Pfister. 1992. Cattle diets in tall forb communities on mountain rangelands. *Journal of Range Management* 45:534–537.

Robertson, C. 1929. *Flowers and Insects: Lists of Visitors of Four Hundred and Fifty-Three Flowers*. 221 pp. Lancaster, PA: The Science Press Printing Company.

Roland, J., and S. F. Matter. 2007. Encroaching forests decouple alpine butterfly population dynamics. *Proceedings of the National Academy of Sciences* 104:13702–13704.

Roof, S. M., S. J. Debano, M. M. Rowland, and S. Burrows. 2017. "Influence of flower native status, color, and morphology on visiting preferences of bees in riparian areas of eastern Oregon. Poster presented at 88th Annual Meeting of the Northwest Scientific Association, March 29–April 1, 2017, Southern Oregon University, Ashland, OR. Available at https://bit.ly/2IgyMPW (accessed 6/6/18).

Roof, S. M., S. J. DeBano, M. M. Rowland, and S. Burrows. In press. Associations between blooming plants and their bee visitors in a riparian ecosystem in eastern Oregon. *Northwest Science*.

Roulston, T. H., and K. Goodell. 2011. The role of resources and risks in regulating wild bee populations. *Annual Review of Entomology* 56:293–312.

Roundy, B. A., R. F. Miller, R. J. Tausch, K. Young, A. Hulet, B. Rau, B. Jessop, J. C. Chambers, and D. Eggett. 2014a. Understory cover responses to piñon–juniper treatments across tree dominance gradients in the great basin. *Rangeland Ecology & Management* 67:482–494.

Roundy, B. A., K. Young, N. Cline, A. Hulet, R. F. Miller, R. J. Tausch, J. C. Chambers, and B. Rau. 2014b. Piñon–juniper reduction increases soil water availability of the resource growth pool. *Rangeland Ecology & Management* 67:495–505.

Rubene, D., M. Schroeder, and T. Ranius. 2015. Diversity patterns of wild bees and wasps in managed boreal forests: effects of spatial structure, local habitat and surrounding landscape. *Biological Conservation* 184:201–208.

Runquist, E. 2004. Report presented at Workshop on the Ecology and Status of the Mardon Skipper (*Polites mardon*), an Unusual Pacific Northwest Butterfly, Ashland, Oregon (unpublished).

Russell, K. N., H. Ikerd, and S. Droege. 2005. The potential conservation value of unmowed powerline strips for native bees. *Biological Conservation* 124:133–148.

Saarinen, K., J. Jantunen, and A. Valtonen. 2005. Resumed forest grazing restored a population of *Euphydryas aurinia* (Lepidoptera: Nymphalidae) in SE Finland. *European Journal of Entomology* 102:683–690.

Samways, M. J., P. M. Caldwell, and R. Osborn. 1996. Ground-living invertebrate assemblages in native, planted and invasive vegetation in South Africa. *Agriculture, Ecosystems & Environment* 59:19–32.

Sánchez-Bayo, F., D. Goulson, F. Pennacchio, F. Nazzi, K. Goka, and N. Desneux. 2016. Are bee diseases linked to pesticides? - a brief review. *Environment International* 89-90:7–11.

Scandurra, A., L. Magliozzi, M. Aria, and B. D'Aniello. 2014. Short-term effects of fire on Papilionoidea (Lepidoptera) communities: a pilot study in Mediterranean maquis shrubland. *Italian Journal of Zoology* 81:599–609.

Schmalz, H. J., R. V. Taylor, T. N. Johnson, P. L. Kennedy, S. J. DeBano, B. Newingham, and P. A. McDaniel. 2013. Soil morphologic properties and cattle stocking rates affect dynamic soil properties. *Rangeland Ecology and Management* 66:445–453.

Schmelzer, L., B. Perryman, B. Bruce, B. Schultz, K. McAdoo, G. McCuin, S. Swanson, J. Wilker, and K. Conley. 2014. Case study: reducing cheatgrass (*Bromus tectorum* l.) fuel loads using fall cattle grazing. *The Professional Animal Scientist* 30:270–278.

Schultz, C. B. 1998. Dispersal behavior and its implications for reserve design in a rare oregon butterfly. *Conservation Biology* 12:284–292.

Schultz, C. B., L. M. Brown, E. Pelton, and E. E. Crone. 2017. Citizen science monitoring demonstrates dramatic declines of monarch butterflies in western North America. *Biological Conservation* 214:343–346.

Scohier, A., A. Ouin, A. Farruggia, and B. Dumont. 2012. Is there a benefit of excluding sheep from pastures at flowering peak on flower-visiting insect diversity? *Journal of Insect Conservation* 17:287–294.

Semmens, B. X., D. J. Semmens, W. E. Thogmartin, R. Wiederholt, L. López-Hoffman, J. E. Diffendorfer, J. M. Pleasants, K. S. Oberhauser, and O. R. Taylor. 2016. Quasi-extinction risk and population targets for the eastern, migratory population of monarch butterflies (*Danaus plexippus*). *Scientific Reports* 6:23265.

Severns, P. M., and A. R. Moldenke. 2010. Management tradeoffs between focal species and biodiversity: endemic plant conservation and solitary bee extinction. *Biodiversity and Conservation* 19:3605–3609.

Sgolastra, F., P. Medrzycki, L. Bortolotti, M. T. Renzi, S. Tosi, G. Bogo, D. Teper, C. Porrini, R. Molowny-Horas, and J. Bosch. 2017. Synergistic mortality between a neonicotinoid insecticide and an ergosterol-biosynthesis-inhibiting fungicide in three bee species. *Pest Management Science* 73:1236–1243.

Shaw, N., B. Newingham, A. C. Ganguli, A. L. Hild, R. D. Cox, J. Truax, M. Pellant, D. Pyke, D. Ogle, and L. St. John. 2011. "Equipment and strategies to enhance the post-wildfire establishment and persistence of Great Basin native plants. Final Report: JFSP Project Number 07-1-3-12." U.S. Joint Fire Science Program.

Shiflet, T. N. 1994. *Rangeland Cover Types of the United States*. 152+vii pp. Littleton, CO: Society for Range Management.

Sjödin, N. E. 2007. Pollinator behavioural responses to grazing intensity. *Biodiversity and Conservation* 16:2103–2121.

Smallidge, P. J., and D. J. Leopold. 1997. Vegetation management for the maintenance and conservation of butterfly habitats in temperate human-dominated landscapes. *Landscape and Urban Planning* 38:259–280.

Smith, D. S., M. K. Lau, R. Jacobs, J. A. Monroy, S. M. Shuster, and T. G. Whitham. 2015. Rapid plant evolution in the presence of an introduced species alters community composition. *Oecologia* 179:563–572.

Smith, G. W., D. M. Debinski, N. A. Scavo, C. J. Lange, J. T. Delaney, R. A. Moranz, J. R. Miller, D. M. Engle, and A. L. Toth. 2016. Bee abundance and nutritional status in relation to grassland management practices in an agricultural landscape. *Environmental Entomology* 45:338–347.

Smith DiCarlo, L. A. 2018. "Native Bee and Spider Community Responses to Grassland Restoration and Wildfire" (PhD diss, Oregon State University).

Smith DiCarlo, L. A., S. J. DeBano, and S. Burrows. In Review.

Short-term response of two beneficial invertebrate groups to wildfire in an arid grassland system, USA. *Rangeland Ecology and Management*.

Spira, T. P. 2001. Plant-pollinator interactions: A threatened mutualism with implications for the ecology and management of rare plants. *Natural Areas Journal* 21:78–88.

Sprayberry, J. D. H., K. A. Ritter, and J. A. Riffell. 2013. The effect of olfactory exposure to non-insecticidal agrochemicals on bumblebee foraging behavior. *PloS ONE* 8:e76273.

Stanturf, J. A., B. J. Palik, and R. K. Dumroese. 2014. Contemporary forest restoration: a review emphasizing function. *Forest Ecology and Management* 331:292–323.

Steffan-Dewenter, I., and S. Schiele. 2008. Do resources or natural enemies drive bee population dynamics in fragmented habitats? *Ecology* 89:1375–1387.

Steinhauer, N., K. Rennich, D. M. Caron, J. D. Ellis, P. Koenig, K. Kulhanek, J. Klepps et al. 2017. 2016-2017 loss results. Bee Informed Partnership. Accessible at https://bit.ly/2MhpXYq (accessed 11/21/17).

Stephenson, M. B., D. W. Bailey, R. A. Bruegger, and L. D. Howery. 2017. Factors affecting the efficacy of low-stress herding and supplement placement to target cattle grazing locations. *Rangeland Ecology & Management* 70:202–209.

Stettler, J. M., D. A. Johnson, B. S. Bushman, K. J. Connors, T. A. Jones, J. W. MacAdam, and D. J. Hole. 2017. Utah lotus: North American legume for rangeland revegetation in the southern Great Basin and Colorado Plateau. *Rangeland Ecology & Management* 70:691–699.

Stiles, F. G. 1981. Geographical aspects of bird-flower coevolution, with particular reference to Central America. *Annals of the Missouri Botanical Garden* 68:323–351.

Strange, J., W. Sheppard, and J. Koch. 2013. *Monitoring Bumble Bee Pollinators in the Pacific Northwest*. 6 pp. Logan, UT: USDA–ARS Pollinating Insects Research Unit.

Stubbs, A., and P. Chandler (editors). 1978. *A Dipterist's Handbook*. 257+x pp. Orpington: The Amateur Entomologists' Society.

Sugden, E. A. 1985. Pollinators of *Astragalus monoensis* Barneby (Fabaceae): new host records; potential impact of sheep grazing. *Great Basin Naturalist* 45:299–312.

Swengel, A. B. 2001. A literature review of insect responses to fire, compared to other conservation managements of open habitat. *Biodiversity and Conservation* 10:1141–1169.

Szabo, N. D., S. R. Colla, D. L. Wagner, L. F. Gall, and J. T. Kerr. 2012. Do pathogen spillover, pesticide use, or habitat loss explain recent North American bumblebee declines? *Conservation Letters* 5:232–239.

Tallamy, D. W., and K. J. Shropshire. 2009. Ranking lepidopteran use of native versus introduced plants. *Conservation Biology* 23:941–947.

Taylor, A. N., and P. M. Catling. 2012. Bees and butterflies in burned and unburned alvar woodland: evidence for the importance of postfire succession to insect pollinator diversity in an imperiled ecosystem. *Canadian Field-Naturalist* 125:297–306.

Taylor, R. B. 2006. "The effects of off-road vehicles on ecosystems." 12 pp. Texas Parks and Wildlife. Available at https://bit.ly/2rUsRwy (accessed 6/18/18).

Teague, R., B. Grant, and H.-H. Wang. 2015. Assessing optimal configurations of multi-paddock grazing strategies in tallgrass prairie using a simulation model. *Journal of Environmental Management* 150:262–273.

Teague, W. R., U. P. Kreuter, W. E. Grant, H. Diaz-Solis, and M. M. Kothmann. 2009. Economic implications of maintaining rangeland ecosystem health in a semi-arid savanna. *Ecological Economics* 68:1417–1429.

Temeles, E. J., and W. J. Kress. 2003. Adaptation in a plant-hummingbird association. *Science* 300:630–633.

Tepedino, V. J. 2000. The reproductive biology of rare rangeland plants and their vulnerability to insecticides. In *Grasshopper Integrated Pest Management User Handbook (Technical Bulletin 1809)*, edited by G. L. Cunningham and M. W. Sampson (technical coordinators), pages III.5-1–III.5-10. Washington, D.C.: USDA Animal and Plant Health Inspection Service. Available at: http://www.Sidney.ars.usda.gov/grasshopper/Handbook/III/iii_5.htm (accessed 4/23/18).

Tepedino, V. J., B. A. Bradley, and T. L. Griswold. 2008. Might flowers of invasive plants increase native bee carrying capacity? Intimations from Capitol Reef National Park, Utah. *Natural Areas Journal* 28:44–50.

Tepedino, V. J., and D. Nielson. 2017. Bee-rustling on the range: Trap-nesting for pollinators on public lands. *Natural Areas Journal* 37:265–269.

Tubbesing, C., C. Strohm, S.J. DeBano, N. Gonzalez, C. Kimoto, and R.V. Taylor. 2014. Insect visitors and pollination ecology of Spalding's catchfly (*Silene spaldingii*) in the Zumwalt Prairie of northeastern Oregon. *Natural Areas Journal* 34:200–211.

IUCN [International Union for Conservation for Nature]. 2017. IUCN Red List of Threatened Species. Version 2017-3. <www.iucnredlist.org>

The Xerces Society, Wildlife Preservation Canada, York University, The Montreal Insectarium, The London Natural History Museum, and BeeSpotter. 2016. Data accessed from Bumble Bee Watch, a collaborative website to track and conserve North America's bumble bees. Accessible at http://www.bumblebeewatch.org (accessed 11/8/16).

Thogmartin, W. E., R. Wiederholt, K. Oberhauser, R. G. Drum, J. E. Diffendorfer, S. Altizer, O. R. Taylor et al. 2017. Monarch butterfly population decline in North America: identifying the threatening processes. *Royal Society Open Science* 4:170760.

Thomas, J. A. 1984. Conservation of butterflies in temperate countries: Past efforts and lessons for the future. In *The Biology of Butterflies. Symposia of the Royal Entomological Society of London (Vol. 11)*, 333–353. London: Academic Press.

Thomas, L., S. T. Buckland, E. A. Rexstad, J. L. Laake, S. Strindberg, S. L. Hedley, J. R. Bishop, T. A. Marques, and K. P. Burnham. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. *Journal of Applied Ecology* 47:5–14.

Thompson, D. C., and K. T. Gardner. 1996. Importance of grasshopper defoliation period on southwestern blue grama-dominated rangeland. *Journal of Range Management* 49:494–498.

Thompson, H. M. 2003. Behavioural effects of pesticides in bees–their potential for use in risk assessment. *Ecotoxicology* 12:317–330.

Tilman, D., P. B. Reich, and J. M. H. Knops. 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. *Nature* 441:629–632.

Tonietto, R. K., and D. J. Larkin. 2018. Habitat restoration benefits wild bees: A meta-analysis. *Journal of Applied Ecology* 55:582–590.

Toynton, K., B. Luukinen, K. Buhl, and D. Stone. 2009. "Permethrin general fact sheet." National Pesticide Information Center, Oregon State University Extension Services. http://npic.orst.edu/factsheets/PermGen.html (accessed 6/19/18).

Traveset, A., and D. M. Richardson. 2006. Biological invasions as disruptors of plant reproductive mutualisms. *Trends in Ecology & Evolution* 21:208–216.

Traynor, K. S., J. S. Pettis, D. R. Tarpy, C. A. Mullin, J. L. Frazier, M. Frazier, and D. vanEngelsdorp. 2016. In-hive pesticide exposome: Assessing risks to migratory honey bees from in-hive pesticide contamination in the Eastern United States. *Scientific Reports* 6:33207.

Trombulak, S. C., and C. A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14:18–30.

Trunkle, T., and P. Fay. 1991. Transportation of spotted knapweed seeds by vehicles. *Proceedings of the Montana Weed Control Association. Butte, MT, January 14–16*, 33.

Tscharntke, T., A. M. Klein, and A. Kruess. 2005. Landscape perspectives on agricultural intensification and biodiversity–ecosystem service management. *Ecology Letters* 8:857–874.

Tylianakis, J. M., R. K. Didham, J. Bascompte, and D. A. Wardle. 2008. Global change and species interactions in terrestrial ecosystems. *Ecology Letters* 11:1351–1363.

UC [University of California]. 2018. Bee precaution pesticide ratings. UC Integrated Pest Management. Available at: http://www2.ipm.ucanr.edu/beeprecaution/ (last accessed 22 January 2018).

Unternährer, D. 2014. "Leaving uncut refuges within lowland extensively managed meadows secures wild bee species richness and diversity" (master's thesis, University of Bern).

USFS [USDA Forest Service]. 2012. *Future of America's Forests and Rangelands: Forest Service 2010 Resources Planning Act Assessment* (Gen. Tech. Rep. WO-87). 198+xvi pp. Washington, D.C.: U.S. Department of Agriculture, Forest Service.

USFS [USDA Forest Service]. 2015a.Grazing statistical summary 2015. Accessible at https://bit.ly/2lrnorh (accessed 6/19/18).

USFS [USDA Forest Service]. 2015b. *Greater Sage-grouse Record of Decision: Idaho and Southwest Montana, Nevada, Utah*. 272 pp. Washington, D.C.: US Department of Agriculture, Forest Service.

USFS [USDA Forest Service]. 2018.Pollinator syndromes. Accessible at https://bit.ly/2yBKjtu (accessed 4/10/18).

Vanbergen, A. J., A. Espíndola, and M. A. Aizen. 2018. Risks to pollinators and pollination from invasive alien species. *Nature Ecology & Evolution* 2:16–25.

Vanbergen, A. J., and the Insect Pollinators Initiative. 2013. Threats to an ecosystem service: pressures on pollinators. *Frontiers in Ecology and the Environment* 11:251–259.

Vanbergen, A. J., B. A. Woodcock, A. Gray, F. Grant, A. Telford, P. Lambdon, D. S. Chapman, R. F. Pywell, M. S. Heard, and S. Cavers. 2014. Grazing alters insect visitation networks and plant mating systems. *Functional Ecology* 28:178–189.

Van der Putten, W. H., M. Macel, and M. E. Visser. 2010. Predicting species distribution and abundance responses to climate change: why it is essential to include biotic interactions across trophic levels. *Philosophical Transactions of the Royal Society B, Biological Sciences* 365:2025–2034.

Van Nuland, M. E., E. N. Haag, J. A. M. Bryant, Q. D. Read, R. N. Klein, M. J. Douglas, C. E. Gorman et al. 2013. Fire promotes pollinator visitation: implications for ameliorating declines of pollination services. *PloS ONE* 8:e79853.

Vavra, M., C. G. Parks, and M. J. Wisdom. 2007. Biodiversity, exotic plant species, and herbivory: the good, the bad, and the ungulate. *Forest Ecology and Management* 246:66–72.

Veblen, K. E., B. A. Newingham, J. Bates, E. LaMalfa, and J. Gicklhorn. 2015. "Great Basin Fact Sheet No. 7: Post-Fire Grazing Management in the Great Basin". Available at: https://bit.ly/2Kath6Y (accessed 4/17/18).

Vermeire, L. T., R. B. Mitchell, S. D. Fuhlendorf, and D. B. Wester. 2004. Selective control of rangeland grasshoppers with prescribed fire. *Journal of Range Management* 57:29–33.

Visscher, P. K., and B. N. Danforth. 1993. Biology of *Calliopsis pugionis* (Hymenoptera: Andrenidae): nesting, foraging, and investment sex ratio. *Annals of the Entomological Society of America* 86:822–832.

Vose, J. M., J. S. Clark, C. H. Luce, and T. Patel-Weynand (editors). 2016. *Effects of Drought on Forests and Rangelands in the United States: A Comprehensive Science Synthesis*. (Gen. Tech. Report WO-93b). 289 pp. Washington, D.C.: US Department of Agriculture, Forest Service, Washington Office.

Vrdoljak, S. M., M. J. Samways, and J. P. Simaika. 2016. Pollinator conservation at the local scale: flower density, diversity and community structure increase flower visiting insect activity to mixed floral stands. *Journal of*

Insect Conservation 20:711–721.

Vulliamy, B., S. G. Potts, and P. G. Willmer. 2006. The effects of cattle grazing on plant-pollinator communities in a fragmented Mediterranean landscape. *Oikos* 114:529–543.

WallisDeVries, M. F., and I. Raemakers. 2001. Does extensive grazing benefit butterflies in coastal dunes? *Restoration Ecology* 9:179–188.

Walther, G.-R., E. Post, P. Convey, A. Menzel, C. Parmesan, T. J. C. Beebee, J.-M. Fromentin, O. Hoegh-Guldberg, and F. Bairlein. 2002. Ecological responses to recent climate change. *Nature* 416:389–395.

Waltz, A. E. M., and W. W. Covington. 2004. Ecological restoration treatments increase butterfly richness and abundance: mechanisms of response. *Restoration Ecology* 12:85–96.

Warchola, N., E. E. Crone, and C. B. Schultz. 2017. Balancing ecological costs and benefits of fire for population viability of disturbance-dependent butterflies. *Journal of Applied Ecology* 55:800–809.

Wayne, L., S. B. Weiss, and C. Niederer. 2009. "Recovery action plan for the mission blue butterfly (*Icaricia icarioides missionensis*) at Twin Peaks Natural Area." 44 pp. San Francisco: San Francisco Parks and Recreation Department and the Creekside Center for Earth Observation.

Weiss, S. B. 1999. Cars, cows, and checkerspot butterflies: nitrogen deposition and management of nutrient-poor grasslands for a threatened species. *Conservation Biology* 13:1476–1486.

Whelan, C. J., D. G. Wenny, and R. J. Marquis. 2008. Ecosystem services provided by birds. *Annals of the New York Academy of Sciences* 1134:25–60.

Whittington, R., and M. L. Winston. 2004. Comparison and examination of *Bombus occidentalis* and *Bombus impatiens* (Hymenoptera: Apidae) in tomato greenhouses. *Journal of Economic Entomology* 97:1384–1389.

Williams, N. M., D. Cariveau, R. Winfree, and C. Kremen. 2011. Bees in disturbed habitats use, but do not prefer, alien plants. *Basic and Applied Ecology* 12:332–341.

Williams, P. H., and J. L. Osborne. 2009. Bumblebee vulnerability and conservation world-wide. *Apidologie* 40:367–387.

Williams, P. H., R. W. Thorp, L. L. Richardson, and S. R. Colla. 2014. *Bumble Bees of North America: An Identification Guide*. 208 pp. Princeton: Princeton University Press.

Wilshire, H. G. 1983. The impact of vehicles on desert soil stabilizers. In *Environmental Effects of Off-Road Vehicles* (Springer Series on Environmental Management), edited by R. H. Webb and H. G. Wilshire, 31–50. New York: Springer.

Wilson, J. S., and O. J. Messinger Carril. 2015. *The Bees in Your Backyard: A Guide to North America's Bees*. 288 pp. Princeton: Princeton University Press.

Wilson, J. S., O. J. Messinger, and T. Griswold. 2009. Variation between bee communities on a sand dune complex in the Great Basin desert, North America: implications for sand dune conservation. *Journal of Arid Environments* 73:666–671.

Wilson, J. S., L. E. Wilson, L. D. Loftis, and T. Griswold. 2010. The montane bee fauna of north central Washington, USA, with floral associations. *Western North American Naturalist* 70:198–207.

Winfree, R. 2010. The conservation and restoration of wild bees. *Annals of the New York Academy of Sciences* 1195:169–197.

WMTC [Xerces Society Western Monarch Thanksgiving Count]. 2018. Western Monarch Thanksgiving Count Data, 1997–2017. Available at: www.westernmonarchcount.org.

Wu, Y.-T., C.-H. Wang, X.-D. Zhang, B. Zhao, L.-F. Jiang, J.-K. Chen, and B. Li. 2009. Effects of saltmarsh invasion by *Spartina alterniflora* on arthropod community structure and diets. *Biological Invasions* 11:635–649.

Wynhoff, I. 1998. Lessons from the reintroduction of *Maculinea teleius* and *M. nausithous* in the Netherlands. *Journal of Insect Conservation* 2:47–57.

Xie, Z., P. H. Williams, and Y. Tang. 2008. The effect of grazing on bumblebees in the high rangelands of the eastern Tibetan Plateau of Sichuan. *Journal of Insect Conservation* 12:695–703.

Yoshihara, Y., B. Chimeddorj, B. Buuveibaatar, B. Lhagvasuren, and S. Takatsuki. 2008. Effects of livestock grazing on pollination on a steppe in eastern Mongolia. *Biological Conservation* 141:2376–2386

Young, B. E., D. F. Schweitzer, N. A. Sears, and M. F. Ormes. 2015. *Conservation and Management of North American Mason Bees*. 20 pp. Arlington, VA: NatureServe.

Zhu, H., D. Wang, L. Wang, Y. Bai, J. Fang, and J. Liu. 2012. The effects of large herbivore grazing on meadow steppe plant and insect diversity. *Journal of Applied Ecology* 49:1075–1083.

Zhu, W., D. R. Schmehl, C. A. Mullin, and J. L. Frazier. 2014. Four common pesticides, their mixtures and a formulation solvent in the hive environment have high oral toxicity to honey bee larvae. *PloS ONE* 9:e77547.

Zhu, Y. C., J. Adamczyk, T. Rinderer, J. Yao, R. Danka, R. Luttrell, and J. Gore. 2015. Spray toxicity and risk potential of 42 commonly used formulations of row crop pesticides to adult honey bees (Hymenoptera: Apidae). *Journal of Economic Entomology* 108:2640–2647.

Zuefle, M. E., W. P. Brown, and D. W. Tallamy. 2008. Effects of non-native plants on the native insect community of Delaware. *Biological Invasions* 10:1159–1169.

Zvirzdin, D. L., B. A. Roundy, N. S. Barney, S. L. Petersen, V. J. Anderson, and M. D. Madsen. 2017. Postfire soil water repellency in piñon–juniper woodlands: extent, severity, and thickness relative to ecological site characteristics and climate. *Ecology and Evolution* 7:4630–4639.

Appendix A. Bumble Bee Phenology and Distribution

Distribution by State and US Forest Service Region, and IUCN Status

Common Name	Species	General Status	IUCN Status	AZ	CA	CO	ID	MT	NM	NV	OR	WA	WY	UT
White-shouldered bumble bee	*Bombus appositus*	*Common*	**Least Concern; LC**		x	x	x	x	x	x	x	x	x	x
High country bumble bee	*Bombus balteatus*	*Rare*	**Data Deficient; DD**		x	x	x	x		x		x	x	
Two-form bumble bee	*Bombus bifarius*	*Very Common*	**Least Concern; LC**	x	x	x	x	x	x	x	x	x	x	x
Obscure bumble bee	*Bombus caliginosus*	*Limited range, likely in decline*	**Vulnerable; VU**		x						x	x		
Central bumble bee	*Bombus centralis*	*Common*	**Least Concern; LC**	x	x		x	x	x	x	x	x	x	
Crotch bumble bee	*Bombus crotchii*	*Limited range, likely in decline*	**Endangered; EN**		x					x				
Yellow bumble bee	*Bombus fervidus*	*Can be locally common, but possibly in decline.*	**Vulnerable; VU**	x	x	x	x	x	x	x	x	x	x	x
Fernald cuckoo bumble bee	*Bombus flavidus =fernaldae*	*Uncommon*	**Least Concern; LC**		x	x	x	x	x	x	x	x	x	x
Yellow head bumble bee	*Bombus flavifrons*	*Common*	**Least Concern; LC**	x	x	x	x	x	x	x	x	x	x	x
Franklin's bumble bee	*Bombus franklini*	*Limited Range, Very Rare, Possibly Extinct*	**Critically Endangered; CR**		x						x			
Southern plains bumble bee	*Bombus fraternus*	*Rare*	**Endangered; EN**			x			x					
Bombus frigidus	*Bombus frigidus*	*Rare*	**Least Concern; LC**			x		x					x	
Brown-belted bumble bee	*Bombus griseocollis*	*Common*	**Least Concern; LC**	x	x	x	x	x	x	x	x	x	x	x
Hunt's bumble bee	*Bombus huntii*	*Common*	**Least Concern; LC**	x	x	x	x	x	x	x	x	x	x	x
Indiscriminant cuckoo bumble bee	*Bombus insularis*	*Common*	**Least Concern; LC**	x	x	x	x	x	x	x	x	x	x	x
Black-tailed bumble bee	*Bombus melanopygus*	*Common*	**Least Concern; LC**		x	x	x	x			x	x	x	x
Fuzzy-horned bumble bee	*Bombus mixtus*	*Common*	**Least Concern; LC**	x	x	x	x	x		x	x	x	x	
Morrison bumble bee	*Bombus morrisoni*	*Common*	**Near Threatened; NT**	x	x	x	x	x	x	x	x	x	x	
Nevada bumble bee	*Bombus nevadensis*	*Common*	**Least Concern; LC**	x	x				x	x				
Western bumble bee	*Bombus occidentalis*	*In decline, particularly in the western portion of its range*	**Vulnerable; VU**	x	x	x	x	x	x	x	x	x	x	
American bumble bee	*Bombus pensylvanicus*	*In decline*	**Vulnerable; VU**	x	x	x			x	x	x			
Red-belted bumble bee	*Bombus rufocinctus*	*Common*	**Least Concern; LC**	x	x	x	x	x	x	x	x	x	x	x

R1	R2	R3	R4	R5	R6	J	F	M	A	M	J	J	A	S	O	N	D	Floral Associations (Genera)
x	x	x	x	x	x				x	x	x	x	x	x	x	x		*Cirsium, Delphinium, Geranium, Helianthella, Linaria, Mertensia, Penstemon, Sidalcea, Trifolium, Vicia*
x	x		x	x	x				x	x	x	x	x	x	x			*Aster, Castilleja, Chamerion, Geranium, Mertensia, Mimulus, Penstemon*
x	x	x	x	x	x			x	x	x	x	x	x	x	x			*Arctostaphylos, Aster, Ceonothus, Chrysothamnus, Cirsium, Frasera, Lupinus, Melilotus, Senecio, Symphoricarpos*
				x	x			x	x	x	x	x	x	x	x			*Arctostaphylos, Baccharis, Cirsium, Grindelia, Lotus, Lupinus, Marah, Phacelia, Rhododendron*
x	x	x	x	x	x				x	x	x	x	x	x	x			*Allium, Cirsium, Helenium, Helianthus, Mimulus, Monarda Penstemon, Phacelia, Rudbeckia, Solidago, Symphoricarpos*
			x	x			x	x	x	x	x	x	x	x				*Antirrhinum, Clarkia, Cleome, Delphinium, Dendromecon, Eriogonum Eschscholzia, Phacelia, Trichostema*
x	x	x	x	x	x		x	x	x	x	x	x	x	x	x	x		*Aster, Castilleja, Chamerion, Geranium Hydrophyllum, Kalmia, Lobelia, Mertensia, Mimulus, Penstemon*
x	x		x	x	x			x	x	x	x	x	x	x	x			*Asclepias, Aster, Chamaebatia, Cirsium Potentilla, Senecio, Solidago, Trifolium, Vaccinium, Veratrum*
x	x	x	x	x	x				x	x	x	x	x	x	x			*Chrysothamnus, Epilobium, Eriogonum, Frasera, Geranium, Lathyrus Lupinus, Monardella, Penstemon, Trifolium, Vicia*
				x	x				x	x	x	x	x	x	x			*Ceanothus, Centaurea, Eriogonum, Hyssopus, Lupinus, Trifolium Veratrum,*
x	x						x	x	x	x	x	x	x	x	x	x		*Aslcepias, Brickellia, Cephalanthus, Cirsium, Zizia*
x	x								x	x	x	x	x	x				*Achillea, Chamerion, Draba, Geranium, Helianthella, Hyssopus, Lupinus, Polemonium, Symphoricarpos, Trifolium*
x	x	x	x	x	x		x	x	x	x	x	x	x	x	x	x		*Asclepias, Cirsium, Clinopodium, Dipsacus, Helianthus, Hypericum, Medicago, Phacelia, Solidago, Trifolium*
x	x	x	x	x	x			x	x	x	x	x	x	x	x	x		*Astragalus, Cirsium, Erigeron, Lupinus, Melilotus, Mertensia, Penstemon Phacelia, Rudbeckia, Thermopsis*
x	x	x	x	x	x			x	x	x	x	x	x	x	x			*Asclepias, Aster, Erigeron, Helenium, Senecio, Solidago, Spiranthes, Trifolium Vaccinium, Wyethia*
x	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x		*Ceanothus, Chamerion, Lupinus, Penstemon Rhododendron, Rubus, Senecio, Symphoricarpos, Trifolium, Vaccinium, Wyethia*
x	x	x	x	x	x			x	x	x	x	x	x	x	x			*Apocynum, Epilobium Eriogonum, Lupinus, Mondardella, Rhododendron, Rubus, Senecio, Symphoricarpos, Trifolium*
x	x	x	x	x	x			x	x	x	x	x	x	x	x	x	x	*Chrysothamnus, Cirsium, Cleome, Frasera, Helenium, Helianthus, Lupinus, Melilotus Monarda, Stanleya*
		x	x	x					x	x	x	x	x	x	x			*Astragalus, Balsamorhiza, Ceanothus, Cirsium, Echinacea, Helianthus, Melilotus, Monarda, Ribes, Thermopsis, Vicia*
x	x	x	x	x	x		x	x	x	x	x	x	x	x	x	x		*Centaurea*, Chrysothamnus, Cirsium, Eriogonum Erythronium, Gilia, Melilotus, Pedicularis, Symphoricarpos, Trifolium, Vicia*
	x	x	x	x	x		x	x	x	x	x	x	x	x	x	x	x	*Astragalus, Chrysothamnus, Erigeron, Gossypium, Helianthus, Hydrophyllum, Kallstroemia Linaria, Rubus, Viguiera,*
x	x	x	x	x	x				x	x	x	x	x	x	x			*Apocynum, Arctium, Aster, Cirsium, Eutrochium, Hypericum, Monarda, Solidago, Tanacetum, Trifolium*

Common Name	Species	General Status	IUCN Status	AZ	CA	CO	ID	MT	NM	NV	OR	WA	WY	UT
Sitka bumble bee	*Bombus sitkensis*	*Common*	**Least Concern; LC**		x		x	x			x	x		
Suckley cuckoo bumble bee	*Bombus suckleyi*	*Rare*	**Critically Endangered; CR**		x	x	x	x			x	x	x	x
Forest bumble bee	*Bombus sylvicola*	*Uncommon*	**Least Concern; LC**		x	x	x	x		x	x	x	x	x
Tri-colored bumble bee	*Bombus ternarius*	*Common*	**Least Concern; LC**					x						
Yellow-banded bumble bee	*Bombus terricola*	*Rare throughout much of its range, in decline*	**Vulnerable; VU**					x						
Half-black bumble bee	*Bombus vagans*	*Common*	**Least Concern; LC**				x	x			x	x	x	
van Dyke bumble bee	*Bombus vandykei*	*Uncommon*	**Least Concern; LC**		x					x	x	x		
Variable cuckoo bumble bee	*Bombus variabilis*	*Very Rare*	**Critically Endangered; CR**			x			x					
Yellow-faced bumble bee	*Bombus vosnesenskii*	*Very common*	**Least Concern; LC**		x					x	x	x		

R1	R2	R3	R4	R5	R6	J	F	M	A	M	J	J	A	S	O	N	D	Floral Associations (Genera)
x			x	x	x			x	x	x	x	x	x	x	x			*Ceanothus, Epilobium, Lupinus, Rhododendron, Rosa, Rubus, Sidalcea, Solidago, Vaccinium, Vicia*
x	x		x	x	x				x	x	x	x	x	x	x			*Aster, Astragalus, Centaurea*, Chrysothamnus, Cirsium, Grindelia, Helichrysum Mertensia, Symphoricarpos, Trifolium*
x	x		x	x	x				x	x	x	x	x	x	x			*Arenaria, Chamaebatiaria, Chamerion, Dodacatheon, Helenium, Lupinus, Melilotus, Monardella, Raillardella, Senecio, Wyethia*
x								x	x	x	x	x	x	x	x	x		*Achillea, Chrysothamnus, Claytonia, Eriogonum, Eutrochium, Monarda, Prunus, Tanacetum, Trifolium, Vaccinium*
x								x	x	x	x	x	x	x	x			*Agastache, Asclepias, Aster, Chamerion, Dalea, Lonicera, Rosa, Rubus, Salix, Solidago, Vaccinium*
x	x		x		x			x	x	x	x	x	x	x	x	x		*Asclepias, Aster, Cirsium, Clinopodium, Eupatorium, Hydrophyllum, Penstemon, Spireae*
			x	x	x				x	x	x	x	x	x				*Aster, Clarkia, Collinsia, Hyssopus, Lupinus, Monardella, Penstemon, Phacelia, Stachys, Streptanthus*
	x	x																*Asclepias, Cephalanthus, Cirsium, Dalea, Phlox*
			x	x	x	x	x	x	x	x	x	x	x	x	x	x		*Castilleja, Cirsium, Clarkia, Cleome, Dicentra, Ericameria, Eriogonum, Hyssopus, Lupinus, Mimulus, Phacelia*

Appendix B. At-risk Pollinators in the West

Taxon	Species Name	Common Name	Global Rank	National Rank	State Status (*from NatureServe [2017]*)
Bee	*Andrena aculeata*	A miner bee	GNR	NNR	None
Bee	*Anthophora* sp. nov. 1	A bee	None	None	None
Bee	*Ashmeadiella sculleni*	A leafcutting bee	GNR	NNR	S1? (OR)
Bee	*Bombus caliginosus*	Obscure bumble bee	G3G4	NNR	S1S2 (CA), SNR (OR, WA)
Bee	*Bombus crotchii*	Crotch bumble bee	G3G4	NNR	S1S2 (CA)
Bee	*Bombus fervidus*	Yellow bumble bee	G4?	N4?	SNR (AZ, CA, CO, NV, UT, WA, WY)
Bee	*Bombus franklini*	Franklin's bumblebee	G1	N1	S1 (CA, OR)
Bee	*Bombus huntii*	Hunt's bumble bee	G5	N5	SNR (AZ, CA, CO, UT, WA, WY)
Bee	*Bombus morrisoni*	Morrison's bumble bee	G4G5	N4N5	S1S2 (CA), SNR (AZ, CO, MT, UT, WA)
Bee	*Bombus occidentalis*	Western bumblebee	G4	N2N3	S1 (CA, WA), S1S2 (OR), S2? (AZ), SNR (CO, ID, MT, NV, NM, UT, WY)
Bee	*Bombus suckleyi*	Suckley cuckoo bumble bee	G1G3	NU	S1 (CA), SNR (MT, WA)
Bee	*Calliopsis barri*	A miner bee	GNR	NNR	S1 (OR)
Bee	*Halictus harmonius*	Haromonius halictid bee	G1	NNR	S1 (CA)
Bee	*Hesperapis kayella*	A miner bee	None	None	None
Bee	*Hesperapis* sp. nov. 2	A bee	None	None	None
Bee	*Hoplitis orthognathus*	A mason bee	None	None	None
Bee	*Hoplitis producta subgracilis*	A mason bee	None	None	None
Bee	*Hylaeus lunicraterius*	A yellow-masked bee	None	None	None
Bee	*Lasioglossum channelense*	Channel Island sweat bee	G1	NNR	S1 (CA)
Bee	*Paranomada californica*	California cuckoo bee	G1	NNR	S1 (CA)
Bee	*Perdita barri*	A miner bee	None	None	None
Bee	*Perdita haigi*	A miner bee	GNR	NNR	None
Bee	*Perdita salicis euxantha*	A miner bee	G5TNR	NNR	SU (OR)
Bee	*Perdita scitula antiochensis*	Antioch andrenid bee	G1T1	NNR	S1 (CA)
Bee	*Perdita wyomingensis*	A miner bee	None	None	None

					Western Distribution (based primarily on NatureServe)										
Federal Status	**IUCN Status**	**USFS Status**	**BLM Status**	**Other Statuses**	**AZ**	**CA**	**CO**	**ID**	**MT**	**NM**	**NV**	**OR**	**UT**	**WA**	**WY**
None	Not assessed			SGCN (ID)				x							
None	Not assessed		SEN (NV)								x				
None	Not assessed			SGCN (ID)								x			
None	VU			SGCN (CA); VU (IUCN)		x						x		x	
None	EN			SGCN (CA); EN (IUCN)		x									
None	VU			SGCN (ID); VU (IUCN)	x	x	x				x		x	x	x
Under review	CR	SEN (R6)	SEN (OR-WA)	SGCN (CA, OR); CR (IUCN)		x						x			
None	LC			SGCN (ID); LC (IUCN)	x	x	x						x	x	x
None	VU			SGCN (CA, ID, WA); VU (IUCN)	x	x	x		x				x	x	
None	VU	SEN (R2, R5, R6)	SEN (OR-WA)	SGCN (CA, ID, OR, WA); VU (IUCN)	x	x	x	x	x	x	x	x	x	x	x
None	CR			SGCN (CA, ID, WA); CR (IUCN)		x			x					x	
None	Not assessed			SGCN (ID)				x				x			
None	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (ID)				x			x				
None	Not assessed		SEN (NV)								x				
None	Not assessed			SGCN (ID)				x				x		x	
None	Not assessed			SGCN (ID)				x							
None	Not assessed			SGCN (ID)				x							
None	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (ID)				x							
None	Not assessed		SEN (NV)								x				
None	Not assessed			SGCN (ID)								x			
None	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (ID)				x							

Taxon	Species Name	Common Name	Global Rank	National Rank	State Status *(from NatureServe [2017])*
Bee	*Protodufourea wasbaueri*	**Wasbauer's protodufourea bee**	**G1**	**NNR**	**S1 (CA)**
Bee	*Protodufourea zavortinki*	**Zavortink's protodufourea bee**	**G1**	**NNR**	**S1 (CA)**
Bee	*Rhopalolemma robertsi*	**Roberts' rhopalolemma bee**	**G1**	**NNR**	**S1 (CA)**
Bee	*Sphecodogastra antiochensis*	**Antioch Dunes halictid bee**	**G1**	**NNR**	**S1 (CA)**
Bee	*Trachusa gummifera*	**San Francisco Bay Area leaf-cutter bee**	**G1**	**NNR**	**S1 (CA)**
Butterfly	*Adopaeoides prittwitzi*	**Sunrise skipper**	**G2G4**	**N1N2**	**S2 (AZ), SNR (NM)**
Butterfly	*Agathymus evansi*	**Huachuca giant skipper**	**G2G3**	**N2**	**S3 (AZ)**
Butterfly	*Apodemia mormo langei*	**Lang's metalmark butterfly**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Atrytone arogos iowa*	**Arogos skipper**	**G3T3**	**N3**	**SNR (CO, MT, WY)**
Butterfly	*Atrytonopsis cestus*	**Cestus skipper**	**G3G4**	**N1N3**	**S2 (AZ)**
Butterfly	*Boloria astarte*	**Astarte fritillary**	**G5**	**N4N5**	**S2S3 (MT, WA)**
Butterfly	*Boloria bellona*	**Meadow fritillary**	**G5**	**N5**	**S1 (OR), S2? (WA), S4 (CO), S5 (MT), SNR (ID, WY)**
Butterfly	*Boloria bellona toddi*	**Meadow fritillary**	**G5T5**	**N4N5**	**S1 (OR)**
Butterfly	*Boloria freija*	**Freija fritillary**	**G5**	**N5**	**S2S3 (WA), S3S5 (MT), S5 (CO), SNR (MT, UT, WY)**
Butterfly	*Boloria kriemhild*	**Kriemhild fritillary**	**G3G4**	**N3N4**	**S2 (ID), S3S4 (MT), SNR (UT, WY)**
Butterfly	*Boloria selene*	**Silver-bordered fritillary**	**G5**	**N5**	**S2 (OR), S3 (WA), S5 (MT) SNR (CO, ID, NM, UT, WY)**
Butterfly	*Boloria selene atrocostalis*	**Dark-bordered fritillary**	**G5T5**	**N4**	**S2 (OR)**
Butterfly	*Callophrys (Incisalia) polios maritima*	**Hoary elfin**	**G5T2T3**	**N2N3**	**S1 (OR), S2S3 (WA), SNR (CA)**
Butterfly	*Callophrys gryneus chalcosiva*	**Barry's hairstreak**	**None**	**None**	**None**
Butterfly	*Callophrys gryneus Columbia Basin segregate*	**Juniper hairstreak**	**None**	**None**	**None**
Butterfly	*Callophrys gryneus rosneri*	**Rosner's hairstreak**	**G5T4**	**N2N4**	**S2S3 (WA), SNR OR)**
Butterfly	*Callophrys johnsoni*	**Johnson's hairstreak**	**G3G4**	**N3N4**	**S2 (OR), S2S3 (WA), SNR (CA)**
Butterfly	*Callophrys mossii bayensis*	**San Bruno elfin butterfly**	**G4T1**	**N1**	**S1 (CA)**
Butterfly	*Callophrys mossii marinensis*	**Marin elfin butterfly**	**G4T1**	**N1**	**S1 (CA)**
Butterfly	*Callophrys polios Puget Trough segregate*	**Hoary elfin**	**None**	**None**	**None**
Butterfly	*Callophrys thornei*	**Thorne's hairstreak butterfly**	**G1**	**N1**	**S1 (CA)**

					Western Distribution (based primarily on NatureServe)										
Federal Status	**IUCN Status**	**USFS Status**	**BLM Status**	**Other Statuses**	**AZ**	**CA**	**CO**	**ID**	**MT**	**NM**	**NV**	**OR**	**UT**	**WA**	**WY**
None	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (CA)		x									
None	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (CA)		x									
None	Not assessed	SEN (R3)			x					x					
None	Not assessed	SEN (R3)			x										
Endangered	Not assessed			SGCN (CA)		x									
None	Not assessed	SEN (R1)					x		x						x
None	Not assessed	SEN (R3)			x										
None	Not assessed	SEN (R6)	SEN (OR-WA)						x					x	
None	Not assessed	SEN/STR (R6)	SEN/STR (OR-WA)				x	x	x			x		x	x
None	Not assessed			SGCN (WA)								x		x	
None	Not assessed	SEN (R6)					x		x	x			x	x	x
None	Not assessed			SGCN (ID)				x	x				x		x
None	Not assessed	SEN (R6)	SEN (OR-WA)				x	x	x	x		x	x	x	x
None	Not assessed			SGCN (WA)								x		x	
None	Not assessed	STR (R6)	STR (OR-WA)	SGCN (OR)		x						x		x	
None	Not assessed	SEN (R6)	SEN (OR-WA)					x				x		x	
None	Not assessed			SGCN (WA)										x	
None	Not assessed	SEN (R6)										x		x	
None	Not assessed	SEN (R6)	SEN (OR-WA)	SGCN (ID, WA)		x						x		x	
Endangered	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (WA)										x	
None	Not assessed		SEN (CA)	SGCN (CA)		x									

Taxon	Species Name	Common Name	Global Rank	National Rank	State Status (*from NatureServe [2017]*)
Butterfly	*Carterocephalus palaemon magnus*	**Sonoma arctic skipper**	**G5T5**	**N1**	**S1 (CA)**
Butterfly	*Chlosyne acastus robusta*	**Spring Mountain checkerspot**	**G4G5T1**	**N1**	**S1 (NV)**
Butterfly	*Coenonympha tullia yontockett*	**Yontocket satyr**	**G5T1T2**	**N1N2**	**S1 (CA), SNR (OR)**
Butterfly	*Colias nastes*	**Labrador sulphur**	**G5**	**N5**	**S2S3 (MT, WA)**
Butterfly	*Colias occidentalis pseudochristina*	**Intermountain sulphur**	**G4T2T4**	**NU**	**S1 (WA), S2 (OR), SNR (ID, UT)**
Butterfly	*Cupido comyntas*	**Eastern tailed blue**	**G5**	**N5**	**S2S3 (WA), S5 (CO), SNR (AZ, CA, ID, MT, NV, NM, OR, WY)**
Butterfly	*Danaus plexippus plexippus*	**Monarch**	**G4T3**	**N2N3**	**SNR (WY), S4 (WA), S4B (MT), S5 (CO), SNA (NV, OR, UT, WY), SNR (AZ, CA, ID, NM)**
Butterfly	*Erynnis propertius*	**Propertius duskywing**	**G5**	**N5**	**S3 (WA), SNR (CA, NV, OR)**
Butterfly	*Euchloe ausonides insulanus*	**Island large marble**	**G5T1**	**N1**	**S1 (WA)**
Butterfly	*Euchloe hyantis andrewsi*	**Andrew's marble butterfly**	**G3G4T1**	**N1**	**S1 (CA)**
Butterfly	*Euphilotes ancilla purpura*	**Dark blue**	**G5T2**	**N2**	**S1S2 (NV)**
Butterfly	*Euphilotes battoides allyni*	**El Segundo blue butterfly**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Euphilotes baueri (battoides) vernalis*	**Vernal blue butterfly**	**G5T1**	**N1**	**SNR (CA)**
Butterfly	*Euphilotes enoptes cryptorufes*	**Pratt's blue butterfly**	**G5T1T2**	**N1**	**SNR (CA)**
Butterfly	*Euphilotes enoptes* **nr.** *dammersi*	**Dammer's blue butterfly**	**None**	**None**	**None**
Butterfly	*Euphilotes enoptes smithi*	**Smith's blue butterfly**	**G5T2**	**N1N2**	**S1S2 (CA)**
Butterfly	*Euphydryas anicia cloudcrofti*	**Sacramento Mountains checkerspot**	**G5T1**	**N1**	**SNR (NM)**
Butterfly	*Euphydryas anicia morandi*	**Morand's checkerspot**	**G5T2**	**N2**	**S2 (NV)**
Butterfly	*Euphydryas colon colon*	**Island checkerspot**	**G5T2T3**	**N2N3**	**S2S3 (WA)**
Butterfly	*Euphydryas editha bayensis*	**Bay checkerspot butterfly**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Euphydryas editha bingi*	**Bing's checkerspot butterfly**	**G5T1**	**N1**	**SNR (CA)**
Butterfly	*Euphydryas editha ehrlichi*	**Ehrlich's checkerspot butterfly**	**G5T1**	**N1**	**SNR (CA)**
Butterfly	*Euphydryas editha karinae*	**Karin's checkerspot butterfly**	**G5T1**	**N1**	**SNR (CA)**
Butterfly	*Euphydryas editha monoensis*	**Mono Lake checkerspot butterfly**	**G5T2T3**	**N2N3**	**S1 (NV), S1S2 (CA)**
Butterfly	*Euphydryas editha quino*	**Quino checkerspot butterfly**	**G5T1S2**	**N1**	**S1S2 (CA)**

					Western Distribution (based primarily on NatureServe)										
Federal Status	**IUCN Status**	**USFS Status**	**BLM Status**	**Other Statuses**	**AZ**	**CA**	**CO**	**ID**	**MT**	**NM**	**NV**	**OR**	**UT**	**WA**	**WY**
None	Not assessed			SGCN (CA)		x									
None	Not assessed	SEN (R4)									x				
None	Not assessed			SGCN (CA)		x						x			
None	Not assessed	SEN (R6)							x					x	
None	Not assessed	SEN/STR (R6)	SEN/STR (OR-WA)					x				x	x	x	
None	Not assessed	SEN (R6)	SEN (OR-WA)		x	x	x	x	x	x	x	x		x	x
Under review	Not assessed	SEN (R2, R5)	SEN (AZ)	SGCN (CA, ID, OR, WA)	x	x	x	x	x	x	x	x	x	x	x
None	Not assessed		SEN (OR-WA)	SGCN (WA)		x					x	x		x	
Proposed endangered	Not assessed		SEN (OR-WA)	SGCN (WA)										x	
None	Not assessed			SGCN (CA)		x									
None	Not assessed	SEN (R4)									x				
Endangered	Not assessed	SEN (R4)		SGCN (CA)		x									
None	Not assessed	SEN (R5)				x									
None	Not assessed	SEN (R5)				x									
None	Not assessed	SEN (R5)				x									
Endangered	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R3)								x					
None	Not assessed	SEN (R4)									x				
None	Not assessed		STR (OR-WA)											x	
Threatened	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R5)				x									
None	Not assessed	SEN (R5)				x									
None	Not assessed	SEN (R5)				x									
None	Not assessed	SEN (R5)				x					x				
Endangered	Not assessed	SEN (R5)		SGCN (CA)		x									

Taxon	Species Name	Common Name	Global Rank	National Rank	State Status (*from NatureServe [2017]*)
Butterfly	*Euphydryas editha taylori*	**Taylor's checkerspot**	**G5T1**	**N1**	**S1 (OR, WA)**
Butterfly	*Euphydryas gillettii*	**Gillette's checkerspot**	**G3**	**N2N3**	**S1 (OR), S2 (MT), S3 (ID), SNR (UT, WY)**
Butterfly	*Glaucopsyche lygdamus palosverdesensis*	**Palos Verde blue butterfly**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Glaucopsyche piasus* **nr.** *sagittegera*	**Arrowhead blue butterfly**	**None**	**None**	**None**
Butterfly	*Habrodais grunus*	**Golden hairstreak**	**G4G5**	**N4N5**	**S1 (WA), SNR (AZ, CA, NV, OR)**
Butterfly	*Habrodais grunus herri*	**Herr's hairstreak**	**G4G5T2T3**	**N2N3**	**S1 (WA), SNR (OR)**
Butterfly	*Hesperia colorado oregonia*	**Oregon branded skipper**	**G5T2**	**NNR**	**S2 (WA)**
Butterfly	*Hesperia miriamae longaevicola*	**White Mountains skipper**	**G2G3T1T2**	**N1**	**S1 (CA, NV)**
Butterfly	*Hesperia ottoe*	**Ottoe skipper**	**G3G4**	**N3N4**	**S2 (CO), S2S3 (MT), S3 (WY)**
Butterfly	*Icaricia shasta charlestonensis*	**Mt. Charleston blue butterfly**	**G5T1**	**N1**	**S2 (NV)**
Butterfly	*Incisalia mossii hidakupa*	**San Gabriel Mountains elfin**	**G4T1T2**	**N1N2**	**S1S2 (CA)**
Butterfly	*Lycaena cupreus*	**Lustrous copper**	**G5**	**N5**	**S2 (WA), S5 (CO, MT), SNR (CA, ID, NV, NM, OR, UT, WY)**
Butterfly	*Lycaena ferrisi*	**Ferris' copper**	**G1G2**	**N1N2**	**S2 (AZ)**
Butterfly	*Lycaena hermes*	**Hermes copper butterfly**	**G1**	**N1N2**	**S1 (CA)**
Butterfly	*Lycaena mariposa charlottensis*	**Makah copper**	**G5T5**	**N2**	**S2 (WA)**
Butterfly	*Lycaena phlaeas arctodon*	**Beartooth copper**	**G5T3T5**	**NU**	**S3? (OR), SNR (ID, MT, WY)**
Butterfly	*Lycaena rubidus incana*	**White Mountains copper**	**G5T1**	**N1**	**S1 (CA, NV)**
Butterfly	*Ochlodes yuma*	**Yuma skipper**	**G5**	**N5**	**S1 (WA), S1? (OR), S2S3 (CO), SNR (AZ, CA, NV, NM, UT, WY)**
Butterfly	*Oeneis chryxus valerata*	**Olympic arctic**	**G5T3**	**N3**	**S2 (WA)**
Butterfly	*Oeneis melissa*	**Melissa arctic**	**G5**	**N5**	**S2S3 (MT, WA), S5 (CO), SNR (NM, UT, WY)**
Butterfly	*Oeneis nevadensis*	**Great arctic**	**G5**	**N5**	**S5 (WA), SNR (CA, OR)**
Butterfly	*Philotiella leona*	**Leona's little blue butterfly**	**G1**	**N1**	**S1 (OR)**
Butterfly	*Philotiella speciosa bohartorum*	**Bohart's blue butterfly**	**G3G4T1**	**N1**	**S1 (CA)**
Butterfly	*Phyciodes batessi*	**Tawny crescent**	**G4G5**	**N3N4**	**S2S3 (MT), SNR (NM, UT, WY), SH (AZ)**
Butterfly	*Plebejus acmon sp.*	**Straits acmon blue**	**G5T?**	**N5T?**	**SNR (WA)**

					Western Distribution (based primarily on NatureServe)										
Federal Status	IUCN Status	USFS Status	BLM Status	Other Statuses	AZ	CA	CO	ID	MT	NM	NV	OR	UT	WA	WY
Endangered	Not assessed	FE (R6)	FE (OR-WA)	SGCN (OR, WA)								x		x	
None	Not assessed			SGCN (ID)				x	x			x	x		x
Endangered	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R5)				x									
None	Not assessed	SEN (R6)			x	x					x	x		x	
None	Not assessed			SGCN (WA)								x		x	
None	Not assessed	STR (R6)	STR (OR-WA)	SGCN (WA)										x	
None	Not assessed	SEN (R5)		SGCN (CA)		x					x				
None	Not assessed	SEN (R1, R2)					x		x						x
Endangered	Not assessed	FE (R4)	SEN (NV)								x				
None	Not assessed	SEN (R5)				x									
None	Not assessed	SEN (R6)				x	x	x	x	x	x	x	x	x	x
Under review	Not assessed	SEN (R3)			x										
Candidate	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R6)		SGCN (WA)										x	
None	Not assessed			SGCN (ID)											
None	Not assessed	SEN (R5)		SGCN (CA)		x					x				
None	Not assessed	SEN/STR (R6)	SEN/STR (OR-WA)	SGCN (WA)	x	x	x			x	x	x	x	x	x
None	Not assessed	SEN (R6)												x	
None	Not assessed	SEN (R6)					x		x	x			x	x	x
None	Not assessed	SEN (R6)		SGCN (WA)		x						x		x	
None	Not assessed	SEN (R6)		SGCN (OR)								x			
None	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R1)			x				x	x			x		x
None	Not assessed			SGCN (WA)										x	

Taxon	**Species Name**	**Common Name**	**Global Rank**	**National Rank**	**State Status (*from NatureServe [2017]*)**
Butterfly	*Plebejus icarioides blackmorei*	**Puget blue**	**G5T3**	**N1N3**	**S2 (WA)**
Butterfly	*Plebejus icarioides fenderi*	**Fender's blue butterfly**	**G5T1**	**N1**	**S1 (OR)**
Butterfly	*Plebejus icarioides missionensis*	**Mission blue butterfly**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Plebejus idas lotis*	**Lotis blue butterfly**	**G5TH**	**NH**	**SH (CA)**
Butterfly	*Plebejus lupini spangelatus*	**Lupine blue butterfly**	**G5T1**	**N1**	**S2 (WA)**
Butterfly	*Plebejus podarce klamathensis*	**Gray-blue butterfly**	**G3G4T3**	**N3**	**S2 (OR), SNR (CA)**
Butterfly	*Plebejus saepiolus aureolus*	**San Gabriel Mountains blue butterfly**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Plebejus saepiolus littoralis*	**Coastal greenish blue butterfly**	**G5T1T3**	**N1N3**	**S1 (OR), SNR (CA)**
Butterfly	*Plebulina emigdionis*	**San Emigdio blue butterfly**	**G1G2**	**N2N3**	**S1S2 (CA)**
Butterfly	*Polites mardon*	**Mardon skipper**	**G2G3**	**N2N3**	**S1 (CA, WA), S2 (OR)**
Butterfly	*Polites peckius*	**Peck's skipper**	**G5**	**N5**	**S2S3 (WA), S3 (OR), S4 (CO), S5 (MT), SNR (AZ, ID, WY)**
Butterfly	*Polites sonora siris*	**Dog star skipper**	**G4T3**	**N3**	**S2S3 (WA), S3? (OR), SNR (CA)**
Butterfly	*Polites themistocles*	**Tawny-edged skipper**	**G5**	**N5**	**S2S3 (WA), S5 (CO, MT), SNR (AZ, CA, ID, NM, OR, UT, WY)**
Butterfly	*Pseudocopaeodes eunus obscurus*	**Carson wandering skipper**	**G3G4T1**	**N1**	**S1 (CA, NV)**
Butterfly	*Pyrgus ruralis lagunae*	**Laguna Mountains skipper**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Speyeria adiaste adiaste*	**Unsilvered fritillary**	**G1G2T1**	**N1**	**S1 (CA)**
Butterfly	*Speyeria callippe callippe*	**Callippe silverspot butterfly**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Speyeria coronis coronis*	**Coronis fritillary**	**G5T3T4**	**N3N4**	**S1 (OR), SNR (CA)**
Butterfly	*Speyeria cybele*	**Great spangled fritillary**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Speyeria cybele pugetensis*	**Puget Sound fritillary**	**G5TU**	**NNR**	**S3? (WA)**
Butterfly	*Speyeria egleis*	**Great basin fritillary**	**G5**	**N5**	**S2 (CO), S2? (WA), S5 (MT), SNR (CA, ID, NV, OR, UT, WY)**
Butterfly	*Speyeria egleis tehachapina*	**Tehachapi fritillary butterfly**	**G5T2**	**N2N3**	**S2 (CA)**
Butterfly	*Speyeria idalia*	**Regal fritillary**	**G3**	**N3**	**S1 (CO), S3 (WY)**
Butterfly	*Speyeria nokomis apacheana*	**Apache silverspot butterfly**	**G3T2**	**N2**	**S2 (NV), SNR (CA)**
Butterfly	*Speyeria nokomis carsonensis*	**Carson Valley silverspot**	**G3T1**	**N1**	**S1 (CA, NV)**

					Western Distribution (based primarily on NatureServe)										
Federal Status	**IUCN Status**	**USFS Status**	**BLM Status**	**Other Statuses**	**AZ**	**CA**	**CO**	**ID**	**MT**	**NM**	**NV**	**OR**	**UT**	**WA**	**WY**
None	Not assessed	SEN (R6)		SGCN (WA)										x	
Endangered	Not assessed		FE (OR-WA)	SGCN (OR)								x			
Endangered	Not assessed			SGCN (CA)		x									
Endangered	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R6)												x	
None	Not assessed	SEN (R6)	SEN (OR-WA)			x						x			
None	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R6)	SEN (OR-WA)	SGCN (OR)		x						x			
None	Not assessed	SEN (R5)				x									
None	Not assessed	SEN (R5, R6)	SEN (OR-WA)	SGCN (CA, OR, WA)		x						x		x	
None	Not assessed	SEN (R6)			x		x	x	x			x		x	x
None	Not assessed	STR (R6)	STR (OR-WA)	SGCN (WA)		x						x		x	
None	Not assessed	SEN (R6)	SEN (OR-WA)		x	x	x	x	x	x		x	x	x	x
Endangered	Not assessed		SEN (NV)	SGCN (CA)		x					x				
Endangered	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R5)		SGCN (CA)		x									
Endangered	Not assessed	SEN (R5)		SGCN (CA)		x									
None	Not assessed	SEN (R6)	SEN (OR-WA)			x						x			
None	Not assessed	SEN (R5)		SGCN (OR)											
None	Not assessed			SGCN (WA)										x	
None	Not assessed	SEN (R6)	SEN (OR-WA)			x	x	x	x		x	x	x	x	x
None	Not assessed	SEN (R5)				x									
Under review	Not assessed	SEN (R1, R2)					x								x
None	Not assessed	SEN (R5)				x					x				
None	Not assessed			SGCN (CA)		x					x				

Taxon	**Species Name**	**Common Name**	**Global Rank**	**National Rank**	**State Status (*from NatureServe [2017]*)**
Butterfly	*Speyeria nokomis nokomis*	**Nokomis fritillary**	**G3T1**	**N1**	**S1 (CO, NM), S2S3 (Navajo Nation), SNR (AZ, UT)**
Butterfly	*Speyeria zerene behrensii*	**Behren's silverspot butterfly**	**G5T1**	**N1**	**S1 (CA)**
Butterfly	*Speyeria zerene bremnerii*	**Valley silverspot**	**G5T3T4**	**NU**	**S2S3 (WA), SH (OR)**
Butterfly	*Speyeria zerene hippolyta**	**Oregon silverspot butterfly**	**G5T1**	**N1**	**S1 (CA, OR), SX (WA)**
Butterfly	*Speyeria zerene myrtleae**	**Myrtle's silverspot butterfly**	**G5TX**	**NX**	**SX (CA)**
Butterfly	*Speyeria zerene sonomensis*	**Sonoma zerene fritillary**	**G5T1**	**N1**	**S1 (CA)**
Moth	*Areniscythris brachypteris*	**Oso Flaco flightless moth**	**G1**	**NNR**	**S1 (CA)**
Moth	*Copablepharon columbia*	**A moth**	**None**	**None**	**None**
Moth	*Copablepharon fuscum*	**Sand-verbena moth**	**G1G2**	**N1N2**	**S1? (WA)**
Moth	*Copablepharon mutans*	**A moth**	**None**	**None**	**None**
Moth	*Copablepharon viridisparsa hopfingeri*	**A moth**	**None**	**None**	**None**
Moth	*Eucosma hennei*	**Henne's eucosman moth**	**G1**	**NNR**	**S1 (CA)**
Moth	*Euhyparpax rosea*	**A notodontid moth**	**G1G2**	**N1N2**	**SNR (AZ, CO, NM)**
Moth	*Euproserpinus euterpe*	**Kern primrose sphinx moth**	**G1G2**	**N1**	**S1 (CA)**
Moth	*Euproserpinus wiesti*	**Wiest's primrose sphinx**	**G3G4**	**N3N4**	**S2 (CO), SNR (AZ, CA, NV, NM, UT)**
Moth	*Grammia eureka*	**A moth**	**None**	**None**	**None**
Moth	*Lophocampa roseata*	**A tiger moth**	**G1G2**	**N1**	**SNR (WA)**

***Presumed to be extirpated from states highlighted in red.**

					Western Distribution (based primarily on NatureServe)										
Federal Status	**IUCN Status**	**USFS Status**	**BLM Status**	**Other Statuses**	**AZ**	**CA**	**CO**	**ID**	**MT**	**NM**	**NV**	**OR**	**UT**	**WA**	**WY**
None	Not assessed	SEN (R2, R3)			x		x			x			x		
Endangered	Not assessed			SGCN (CA)		x									
None	Not assessed	SEN/STR (R6)		SGCN (WA)								x		x	
Threatened	Not assessed	FT (R6)	FT (OR-WA)	SGCN (CA, OR, WA)		x						x		x	
Endangered	Not assessed			SGCN (CA)		x									
Endangered	Not assessed			SGCN (CA)		x									
None	Not assessed			SGCN (CA)							x				
None	Not assessed			SGCN (WA)										x	
Under review	Not assessed		STR (OR-WA)	SGCN (WA)										x	
None	Not assessed			SGCN (WA)										x	
None	Not assessed			SGCN (WA)										x	
None	Not assessed			SGCN (CA)		x									
None	Not assessed	SEN (R3)			x		x			x					
Threatened	Not assessed			SGCN (CA)		x									
None	CR			SGCN (ID); CR (IUCN)	x	x	x			x	x		x		
None	Not assessed			SGCN (ID)				x					x		
None	Not assessed	STR (R6)	STR (OR-WA)											x	

Appendix C. Additional Resources

Additional resources available online at:
https://xerces.org/best-management-practices-for-pollinators-on-western-rangelands

1. Literature Review
2. Native bee genera lists by ecoregion and US Forest Service region with floral and nest plant associations
3. At-risk butterflies and moths and their host plants in the West

Bee Biology and Identification Resources (this list is not complete)

Dreesen, D. R. and L. Lunas. *Pocket Guide to the Native Bees of New Mexico.* New Mexico State University, Cooperative Extension Service Agricultural Experiment Station, NRCS.

Koch, J., J. Strange, and P. Williams. 2012. Guide to Bumble Bees of the Western United States. USDA Forest Service/ Pollinator Partnership, San Francisco, California

Michener, C. D., R. J. McGinley, and B. N. Danforth. 1994. The Bee Genera of North America and Central America. 209 pp. Washington, D.C.: Smithsonian Institute Press.

Michener, C. D. 2000. The Bees of the World. 913 pp. Baltimore, MD: The Johns Hopkins University Press.

O'Toole, C., and A. Raw. 1999. Bees of the World. 192 pp. London, UK: Blandford Press.

Williams, P. H., R. W. Thorp, L. L. Richardson, and S. R. Colla. 2014. Bumble Bees of North America: An Identification Guide. 208 pp. Princeton, NJ: Princeton University Press.

Wilson, J. S. and O. J. Messinger Carril. 2015. The Bees in Your Backyard: A Guide to North America's Bees. Princeton University Press.

Wright, A., C. L. Boyd, D. M. Bowers, and V. L. Scott. 2017. The Bumble Bees of Colorado: A Pictorial Identification and Information Guide. University of Colorado; Minnesota Department of Natural Resources, University of Colorado Museum of Natural History.

Bee Pocket Guides

California Citizen-Scientist Bee Monitoring Pocket Guide (The Xerces Society: field ID guide). A pocket-sized printable field version of the taxonomic native bee groups described the California Citizen-Scientist Bee Monitoring Guide. Available at https://xerces.org/pollinator-resource-center.

Western Bumble Bee Pocket Identification Guide (The Xerces Society: field ID guide). A full color print-and-fold guide to the Western bumble bee (Bombus occidentalis), a formerly common species believed to be in decline. Includes images of similar looking species. Available at https://xerces.org/pollinator-resource-center.

Photo Credits

Photographs

We are grateful to the photographers for allowing us to use their wonderful photographs. All photographs are copyrighted, and none may be reproduced without permission from the photographer.

Bob Brown, US Fish and Wildlife Service: hummingbird on Rocky Mountain bee plant (p. 12).

Bureau of Land Management [flickr.com/photos/mypubliclands]: firefighting (p. 61).

Bureau of Land Management, New Mexico [flickr.com/photos/mypubliclands]: grazing on Agua Fria National Monument (p. 31).

Austin Catlin, US Fish and Wildlife Service [flickr.com/photos/usfwspacific/]: Prescribed fire for sagebrush habitat restoration (p. 37).

Jonathan Coffin [flickr.com/photos/stonebird]: monarch on goldenrod (p. 76).

Sandra DeBano, Associate Professor, Department of Fisheries and Wildlife, Oregon State University: bee on monkshood (p. 58); bee on monkeyflower (p. 59).

Lisa Kieth: milkweed with a host of pollinators (p. 19).

Tom Koerner, US Fish and Wildlife Service: sagebrush steppe (p. 2); globe mallow (p. 16); greater sage grouse (p. 46); rufous hummingbird on showy milkweed (p. 12).

Joshua Mayer: [flickr/photos/wackybadger]: purple prairie clover (p. 62).

Susy Morris: honey bee hive (p. 78).

Laura Navarette, US Forest Service: bumble bee monitoring crew (p. 91); western bumble bee found on forest (p. 91).

Mary Rowland, US Forest Service: Starkey Experimental Forest and Range (p. 58); restoration plantings (p. 58); researcher Samantha Roof netting bees (p. 59).

USDA Natural Resource Conservation Service, Montana: wildflowers on rangeland habitat (p. v).

US Fish and Wildlife Service [flickr.com/photos/usfwsmtnprairie]: invasive plant removal (p. 69).

US Forest Service: mardon skipper habitat comparison (p. 43); Big Summit prairie (p. 15); illegal mason bee trap (p. 82); illegal leaf cutter bee trap (p. 83);

The Xerces Society/Candace Fallon: monarch cluster in California (p. 14); mardon skipper on lily (p. 42); Russian olive stand (p. 68).

The Xerces Society/Rich Hatfield: white-shouldered bumble bee (p. 4); ground-nesting bee (p. 5); indiscriminate cuckoo bumble bee (p. 7); upper photo-syrphid fly on flower (p. 11); western bumble bee (p. 13); bumble bee (p. 29); mardon skipper (p. 41); bumble bee on penstemon (p. 52); ground-nesting bee (p. 60); western bumble bee nectaring (p. 81).

The Xerces Society/Thelma Heidel-Baker: cattle eating around milkweed plants in a pasture (p. 26).

The Xerces Society/Katie Hietala-Henschell: honey bee on gilia (p. 80).

The Xerces Society/Jennifer Hopwood: roadside mowing (p. 34).

The Xerces Society/Jessa Kay Cruz: pollinaotr habitat on a California farm (p. 65).

The Xerces Society/Stephanie McKnight: rangeland habitat in Nevada (p. vi); long-tounged bee on penstemon (p. vii); sagebrush steppe habitat with juniper on rangeland (p. 2); monarch chrysalis (p. 10); lower left-fly on flower (p. 11); lower right-beetle on flower (p. 11); globe mallow (p. 16); southeast Oregon rangeland (p. 22); cattle on rangeland (p. 23); grazing in riparian area (p. 23); cattle and narrowleaf milkweed (p. 27); fencing (p. 30); wild horses (p. 32); caterpillar on roadside (p. 36); lupine along roadside (p. 36); bumble bee on thistle (p. 44); bumble bee on showy milkweed (p. 48); showy milkweed seed (p. 49); native bee on sunflower (p. 50); monarchs on swamp milkweed (p. 51); western pygmy blue on rabbit brush (p. 53); monarch adult on showy milkweed (p. 54); monarch caterpillar (p. 54); restoration planting (p. 56); great purple hairstreak (p. 57); lupine (p. 66); swallowtail on thistle (p. 67); sprayed narrowleaf milkweed (p. 70); bee on milkweed (p. 71); bumble bee on lupine (p. 73); bumble bee on native thistle (p. 74); sphynx moth on thistle (p. 75); monarch on rabbit brush (p. 76); grasshoppers (p. 77); dune habitat (p. 84); butterfly on flower (p. 85); rangeland (p. 87); native bee on flower (p. 88); native bee on flower (p. 89); metallic green bee (p. 90).

The Xerces Society/Emma Pelton: sagebrush steppe habitat (cover); invasive thistles in a Nevada rangeland (p. 72).

The Xerces Society/Mace Vaughn: mason bee (p. 6).

The Xerces Society/Matthew Shepard: honey bee hives (p. 79).

Jeff Vanuga/USDA Natural Resource Conservation Service: prescribed burn (p. 38).

XERCES
SOCIETY
for Invertebrate Conservation

628 NE Broadway, Suite 200, Portland, OR 97232
Tel (855) 232-6639 Fax (503) 233-6794
www.xerces.org

18-015_01