

PETITION TO LIST

Morrison bumble bee

Bombus morrisoni (Cresson), 1878

AS AN ENDANGERED SPECIES UNDER THE U.S. ENDANGERED SPECIES ACT

Submitted by The Xerces Society for Invertebrate Conservation



Photo: Morrison bumble bee, Quinn River Crossing, NV, 2021, by Leif Richardson / The Xerces Society.

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16 November 2023

NOTICE OF PETITION

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PETITIONER

The Xerces Society for Invertebrate Conservation is a nonprofit organization that protects wildlife through the conservation of invertebrates and their habitat. For fifty years, the Society has been at the forefront of invertebrate protection worldwide, harnessing the knowledge of scientists and the enthusiasm of citizens to implement conservation programs. Xerces works to raise awareness about the plight of invertebrates and to gain protection for the most vulnerable species before they decline to a level at which recovery is impossible.

The Honorable Deb Haaland
Secretary, U.S. Department of Interior
1849 C Street NW, Washington, DC 20240

Dear Secretary Haaland,

Pursuant to Section 4(b) of the Endangered Species Act (“ESA”), 16 U.S.C. § 1533(b); Section 553(e) of the Administrative Procedure Act, 5 U.S.C. § 553(e); and 50 C.F.R. § 424.14(a), the Xerces Society for Invertebrate Conservation hereby petitions the Secretary of the Interior, through the United States Fish and Wildlife Service (“FWS,” “Service”), to protect Morrison bumble bee (*Bombus morrisoni*) under the ESA.

Morrison bumble bee historically occurred across much of the Mountain West. However, the bee has declined in recent years (2011-2021) by 74% in relative abundance, and persists in 66% less area, relative to the historic time period. This decline is associated with habitat loss due to climate change and overgrazing, as well as overuse of pesticides, disease, and competition with honeybees, among other factors.

FWS has jurisdiction over this petition. This petition sets in motion a specific process, placing definite response requirements on the Service. Specifically, the Service must issue an initial finding as to whether the petition “presents substantial scientific or commercial information indicating that the petitioned action may be warranted.” 16 U.S.C. § 1533(b)(3)(A). FWS must make this initial finding “[t]o the maximum extent practicable, within 90 days after receiving the petition.”

The Xerces Society for Invertebrate Conservation (“Xerces”) is an international nonprofit organization that protects the natural world through the conservation of invertebrates and their habitats. Xerces works to raise awareness about the plight of invertebrates and to gain protection for the most vulnerable species before they decline to a level at which recovery is impossible.

Xerces submits this petition on behalf of our staff and our members who hold an interest in protecting Morrison bumble bee and its habitat.

Submitted this 16th day of November, 2023



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I. EXECUTIVE SUMMARY

Morrison bumble bee (*Bombus morrisoni*) faces the risk of extinction. This species was historically uncommon across much of the Western U.S, but has become increasingly rare across much of its range, despite significant efforts to find it. Analyses herein demonstrate that Morrison bumble bee has declined in recent years (2011-2021) by 74% in relative abundance, and persists in 66% less area, relative to the historic time period (1815-2010). Morrison bumble bee is in danger of extinction, and needs protection under the U.S. Endangered Species Act.

The broad geographic distribution of Morrison bumble bee has made it vulnerable to multiple threats across its range. These threats include habitat loss and degradation from livestock grazing, exotic species invasion, altered fire regime, agricultural conversion, and urbanization. Additionally, Morrison bumble bee faces risk from pathogens, competition with honey bees, and the use of insecticides and other pesticides, which are ubiquitous in agricultural lands, rangelands, and urban areas. Impacts of climate change, especially drought and increased temperatures, stress bumble bees and alter the plants communities they rely on. Reduced genetic diversity, resulting from isolation of populations caused by climate change or habitat fragmentation and exacerbated by bumble bees' unique sex-determination system, also threatens this species with extinction. Many populations of Morrison bumble bee experience multiple threats, and the synergistic effect of these combined threats present increased risk for the species. Existing regulations are inadequate to protect this species.

Pollinators are critical components of our environment and are essential to global ecosystem stability. Insects, primarily bees, provide the indispensable service of pollination to more than 85% of flowering plants (Ollerton *et al.* 2011). In Europe, declines in pollinators have been associated with a parallel decline in insect pollinated plants (Biesmeijer *et al.* 2006). Bees are critical to ensuring our food security, contributing to 35% of global food production (Klein *et al.* 2007). The estimated annual production value of wild pollinators to just seven US crops is \$1.5 billion, with some of these crops already classified as pollinator-limited (Reilly *et al.* 2020).

Bumble bees are among the most iconic and well understood group of native pollinators in North America. They are generalist pollinators that play a valuable role in the reproduction of a wide variety of plants, including crops such as tomato, squash, melon, blueberry, pepper, alfalfa, clover, and numerous wildflowers.

This petition presents information that Morrison bumble bee meets multiple criteria of an Endangered Species under the U.S. Endangered Species Act.

II. CANDIDATE BACKGROUND, STATUS, AND LISTING HISTORY

Morrison bumble (*Bombus morrisoni*) bee has no legal protection under the U.S. Endangered Species Act, nor any protection from state endangered species laws. It has never been petitioned for listing under the Endangered Species Act. NatureServe ranks Morrison bumble bee as G3 [Vulnerable-at moderate risk of extinction or elimination due to fairly restricted range, relatively few populations or occurrences, recent and widespread declines, threats and other factors (NatureServe 2022a)]. It is also listed on the International Union for the Conservation of Nature Red List as Vulnerable (Hatfield *et al.* 2014). Morrison bumble bee is included as a Species of

Greatest Conservation Need in the State Wildlife Action Plans for California, Colorado, Idaho, and Washington. It is listed as Sensitive by the U.S. Forest Service for Region 6, and Sensitive by the Bureau of Land Management in Oregon and Washington, although these designations confer no legal protection to the species.

III. TAXONOMIC STATUS

All bumble bees belong to the genus *Bombus* within the family Apidae. Morrison bumble bee belongs to the monophyletic subgenus *Cullumanobombus*, which includes 22 other described species. *Cullumanobombus* is well supported as a distinct subgenus (Williams *et al.* 2008). Morrison bumblebee and seven other species compose a Nearctic clade (Cameron *et al.* 2007). *Bombus morrisoni* was first described by Cresson (Cresson 1878). Its status as a species has been upheld by Williams (1998), and by Cameron *et al.* (2007).

Order: Hymenoptera

Family: Apidae

Subfamily: Apinae

Tribe: Bombini

Genus: *Bombus*

Subgenus: *Cullumanobombus*

Species: *morrisoni*

(ITIS 2022)



Figure 1. Female *Bombus morrisoni* illustration: Paul Williams (identification and color patterns) and Elaine Evans (bee body design.)

IV. SPECIES DESCRIPTION

A. Queens and Workers

Morrison bumble bee queens and workers are similar in coloration. The queen is 22 - 26 mm in length, the worker is 12 - 22 mm in length. Their hair is very short and even. The hair of the face is usually black, but the vertex (top of the head) is yellow. The hair on the top of the thorax is all yellow, without a black spot, or band between the wings. The hair on the thorax below the wings is predominantly black. The first two tergal (dorsal plate) segments on the abdomen are yellow with at least some yellow (centrally) on tergal segment 3. Tergal segment 4 has predominantly black hairs. *B. morrisoni* has a short

cheek (the length of the space between the eyes and the mandibles is significantly longer than the width of the mandibular hinges) and females have a rounded corner on the distal posterior corner of their midleg basitarsus (Williams *et al.* 2014). See Figure 1 for an illustration of a Morrison bumble bee worker.

Males

The male is 15 - 20 mm in length. The hair on the head is predominantly yellow, with some black hairs mixed in – especially on the face. The hair on the top of the thorax is entirely yellow with black hairs below and behind the wings. The hair on the first to third tergal segments is yellow, and occasionally there are some yellow hairs on the fourth tergal segment. Tergal segments 5 - 7 are entirely black. The eyes of male *B. morrisoni* are greatly enlarged, much larger than any female bumble bee (Williams *et al.* 2014).

V. BIOLOGY, HABITAT REQUIREMENTS AND POLLINATION ECOLOGY

A. Biology and Habitat Requirements

Morrison bumble bee, like other social bumble bees (e.g. not in the subgenus *Psithyrus*), live in colonies consisting of a queen (foundress) and her offspring, the workers, and near the end of the season the reproductive members of the colony, the males and new queens. There is a division of labor among these three types of bees. The foundress is responsible for initiating colonies and laying eggs. Workers are responsible for most food collection, colony defense, and feeding of the young. Males leave the nest once they reach maturity and their sole function is to mate with new queens. New queens remain with the nest until the end of the season when they leave to mate and find a hibernaculum.

Colonies are annual, progressing from colony initiation by solitary queens in spring, to production of workers, and, for successful colonies, finally to production of queens and males. While it varies by latitude and elevation for all castes, the flight period for *B. morrisoni* queens is from March through November, peaking in late May and early June, with another small peak in September, likely indicating the timing of new queen production. The flight period for workers is from early April to October with a peak in July and August; the male flight period is from June through November, with a peak in late August and early September (Williams *et al.* 2014).

The foundress begins searching for suitable nesting sites and collects nectar and pollen from flowers to support the production of her eggs, which are fertilized by sperm she has stored since mating the previous fall. In the early stages of colony development, the queen is responsible for all food collection and care of the young. As the colony grows, workers take over the duties of food collection, colony defense, and care of the young. The foundress then remains within the nest and spends most of her time laying eggs. The average size of *Bombus morrisoni* colonies has not been well documented in the published literature, with the exception of one nest observation containing 424 cells, 25% of which were sealed cells of pupae that failed to emerge (Koch & Cane 2022). Average *Bombus* sp. colony sizes range from 100-400 workers – though there are species with exceptionally large colony sizes (>1,000), and exceptionally small colony sizes (<50) (Goulson 2010). New queens and males are produced during the later stages of colony development, which is generally from late summer to fall (Koch *et al.* 2012). The new queens forage to build up fat reserves and find a mate before entering diapause, a form of hibernation. At the end of the season, the foundress dies.

There is little information about the nesting biology of Morrison bumble bee. The only published nest description describes a 196 g, 9 cm tall nest found located under chopped straw on the

plywood floor of a shed (Koch & Cane 2022). Other than this single nest description, Morrison bumble bee, like other members of *Cullumanobombus*, are assumed to nest underground (e.g. in abandoned rodent burrows), or occasionally on the surface of the ground. Thus, nesting sites may be limited by the abundance of rodents. This species has been observed or collected from open dry scrub, shrubland, grassland, and dry coniferous forests (Williams *et al.* 2014).

Although little is known about the overwintering habits of Morrison bumble bee queens, queens of other species frequently dig a few centimeters into soft, disturbed soil and form an oval shaped chamber in which she will spend the duration of the winter. Compost in gardens or mole hills may provide suitable sites for queens to overwinter (Goulson 2010). Although there may be species specific preferences for Morrison bumble bee, the queens of bumble bee species generally have been found to overwinter on north-facing slopes, in sandy, well-drained soil, and in areas with limited vegetation cover (Liczner & Colla 2019).

Bumble bees are particularly vulnerable to extinction due to their complementary sex determination system and haplodiploid life history (Zayed & Packer 2005), described below in section IIV. CURRENT AND POTENTIAL THREATS – SUMMARY OF FACTORS FOR CONSIDERATION E. Other Natural or Manmade Factors Affecting its Continued Existence; 2. Population Dynamics and Structure; b. Impacts of Genetic Factors on Bumble Bees.

B. Bumble Bee Pollination Ecology

Bumble bees gather pollen and nectar from a wide variety of flowering plants. To meet its nutritional needs, Morrison bumble bee requires a constant supply of flowers that bloom throughout the duration of the colony life cycle, which is from approximately March to October (Koch *et al.* 2012; Williams *et al.* 2014). Nectar provides bumble bees with carbohydrates and pollen provides them with protein. The amount of pollen available to bumble bee colonies directly affects the number of queens that can be produced (Burns 2004). Since queens are the only bumble bees capable of forming new colonies, pollen availability directly impacts future bumble bee population levels.

Morrison bumble bee probably needs floral resources to be located in relatively close proximity to its nest sites, as studies of other bumble bee species indicate that they routinely forage within less than one kilometer from their nests (Knight *et al.* 2005; Wolf & Moritz 2008; Dramstad 1996; Osborne *et al.* 2008), although in some cases nearly two kilometers (Walther-Hellwig & Frankel 2000).

Morrison bumble bee is a short-tongued species (Williams *et al.* 2014) and thus is not able to easily access the nectar in flowers with deep corollas. Short-tongued bees are better suited for pollination of open flowers and those with short corollas (Patten *et al.* 1993). Other bumble bee species have been observed nectar robbing, where a bee chews through the corolla to access nectar. This allows short-tongued species to access nectar in flowers with deep corollas, but this behavior has not been described for Morrison bumble bee.

During collection of pollen and nectar from flowers, bumble bees also transport pollen between flowers, facilitating seed and fruit production. Bumble bees have many qualities that contribute

to their suitability as agricultural pollinators. They are able to fly in cooler temperatures and lower light levels than many other bees, which extends their work day and improves the pollination of crops during inclement weather (Corbet *et al.* 1993). They also possess the ability to “buzz pollinate,” in which a bee grabs the poricidal anthers of the flower in her mandibles and vibrates her wing musculature. This activity causes the flower to vibrate, which in turn dislodges pollen that would have otherwise remained trapped in the flower’s anthers (Buchmann 1983). Not all bee species are capable of buzz pollination. Some important crop plants, including tomatoes, peppers, blueberries, and cranberries require or benefit greatly from buzz pollination.

In addition to commercially important crops, Morrison bumble bee also plays a vital role as a generalist pollinator of native flowering plants, and its loss may have far ranging ecological impacts. An examination of the theoretical effect of removing specialist and generalist pollinators on the extinction of plant species concluded that the loss of generalist pollinators, especially bumble bees, caused the greatest number of plant extinctions (Memmott *et al.* 2004). In Britain and the Netherlands, where multiple pollinators have declined, there is evidence of a parallel decline in the abundance of insect pollinated plants (Biesmeijer *et al.* 2006).

Morrison bumble bee visits a diverse range of plant species. Despite its broad dietary preferences, the species displays selectivity in its foraging choices, and likely favors certain species based on flower shape, color and scent. In observations of Morrison bumble bee and plant associations in Oregon, Washington, and Idaho, the most frequent plant genera associations were: *Securigera*, *Dipsacus*, *Lavandula*, *Lythrum*, *Melilotus*, *Monarda*, *Nepeta*, *Zauschneria*, *Zinnia*, and *Cirsium* (Hatfield *et al.* 2021). Rangewide, the plant genera most commonly associated with Morrison bumble bee include: *Chrysothamnus*, *Helianthus*, *Asclepias*, *Cleome*, *Trifolium*, *Astragalus*, *Penstemon*, *Monarda*, *Cirsium*, and *Melilotus* (Richardson 2022).

Observations of other plant associations include: *Achillea millefolium* (Carril *et al.* 2018), *Arctostaphylos patula* (Carril *et al.* 2018), *Chaenactis douglasii*, *C. stevioides* (Carril *et al.* 2018), *Convolvulus arvensis* (Carril *et al.* 2018), *Cordylanthus wrightii* (Carril *et al.* 2018), *Cryptantha flava*, *C. pterocarya* (Carril *et al.* 2018), *Dalea ornata*, *D. searlsiae* (Cane *et al.* 2012), *Delphinium scopulorum* (Clements & Long 1923), *Elaeagnus angustifolia* (Carril *et al.* 2018), *Ericameria nauseosa*, *E. parryi* (Carril *et al.* 2018), *Erigeron bellidiastrum* (Carril *et al.* 2018), *Eriogonum jamesii*, *Eriogonum ovalifolium* (Carril *et al.* 2018), *Erysimum asperum* (Carril *et al.* 2018), *Heterotheca villosa* (Carril *et al.* 2018), *Hymenopappus filifolius* (Carril *et al.* 2018), *Ibidium strictum* (Clements & Long 1923), *Lupinus argenteus*, *L. duranii*, *L. sericeus* (Carril *et al.* 2018), *Mahonia fremontii* (Carril *et al.* 2018), *Marrubium vulgare* (Carril *et al.* 2018), *Mentha arvensis* (Carril *et al.* 2018), *Phacelia heterophylla* (Carril *et al.* 2018), *Poliomintha incana* (Carril *et al.* 2018), *Psoralea lanceolata* (Carril *et al.* 2018), *Rhus trilobata* (Carril *et al.* 2018), *Rosa woodsii* (Carril *et al.* 2018), *R. acicularis* (Clements & Long 1923), *Rubus deliciosus*, *R. strigosus* (Clements & Long 1923), *Salix exigua* (Carril *et al.* 2018), *Scutellaria resinosa* (Clements & Long 1923), *Senecio multilobatus*, *S. spartioides* (Carril *et al.* 2018), *Sisymbrium altissimum* (Carril *et al.* 2018), *Sphaeralcea coccinea*, *S. grossulariifolia* (Carril *et al.* 2018), *Stanleya pinnata* (Clements & Long 1923; Carril *et al.* 2018), *Thelypodopsis divaricata* (Carril *et al.* 2018), *Thelypodium integrifolium* (Carril *et al.* 2018), *Tradescantia occidentalis* (Carril *et al.* 2018), and *Vicia americana* (Carril *et al.* 2018).

VI. POPULATION STATUS AND DISTRIBUTION

A. Historic Distribution

Historically, Morrison bumble bee was broadly distributed across the United States Intermountain West, north to Southern British Columbia in Canada, south across the eastern Sierra Nevada Range, east across the desert Southwest to New Mexico and West Texas at elevations ranging from sea level to circa 10,000 ft (Fig. 2).

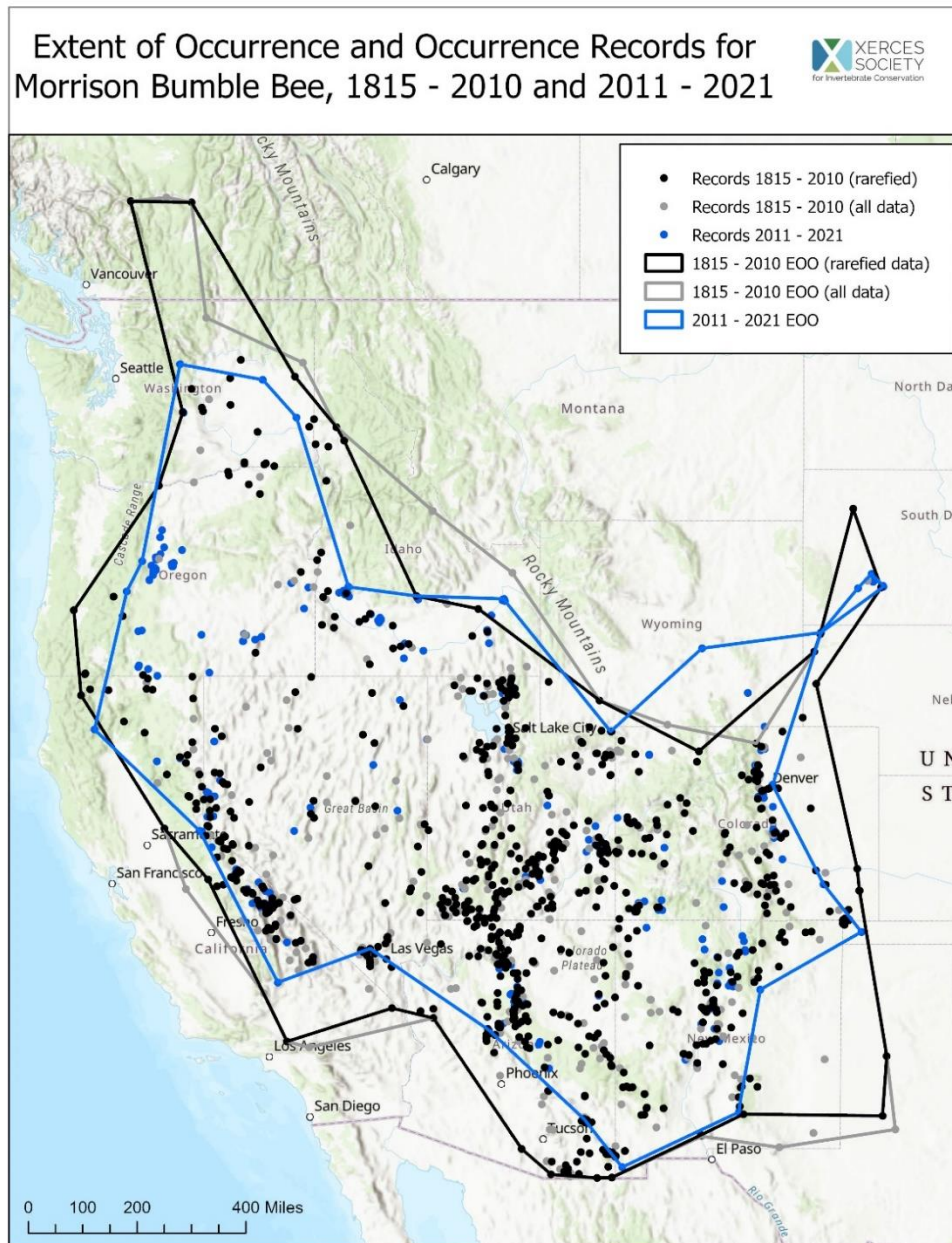


Figure 2. Historic (1815 - 2010) distribution of Morrison bumble bee using rarefied data represented by black circles; historic (1815 - 2010) distribution of Morrison bumble bee using all historic data represented by gray circles; contemporary (2011 - 2021) distribution of Morrison bumble bee represented by blue circles. Extent of occurrence (EOO) constructed using local convex hulls: historic EOO using rarefied data for equal samples for both time periods represented by black line; historic EOO using all historic data represented by gray line; contemporary EOO represented by blue line.

B. Current Distribution and Population Status

To evaluate change in the distribution and population status of Morrison bumble bee, we used a database of over 700,000 North American bumble bee records, assembled from academic, research, private collections, and community scientists using specimen and observation occurrences (Richardson 2022). To conduct analyses, we omitted any observation that did not include a year or a location. Additionally, we removed any duplicates, including observations of the same species from a single collection event, as these are likely members of the same colony, and not a true representation of a species' abundance at a landscape scale. For spatial analysis we omitted any observation with imprecise location information.

Relative abundance

To evaluate change in the relative abundance (RA) of Morrison bumble bee, defined as the abundance of Morrison bumble bee relative to all other species of bumble bee, we used a dataset of over 70,000 bumble bee species occurrence records from within the historic range of Morrison bumble bee. We then divided that dataset into historical (1805 – 2010, n=44,138), and current (2011 – 2021, n=26,590) records, and calculated the RA for both time periods. To estimate change, we divided the current RA by the historic RA. We found that the RA for Morrison bumble bee has declined 74% when compared to historic values. Additionally, we calculated relative abundance for 11 ten-year periods, plus one for all records pre-1910. Relative abundance declined precipitously following 2000, and has remained historically low since then (Fig. 3).

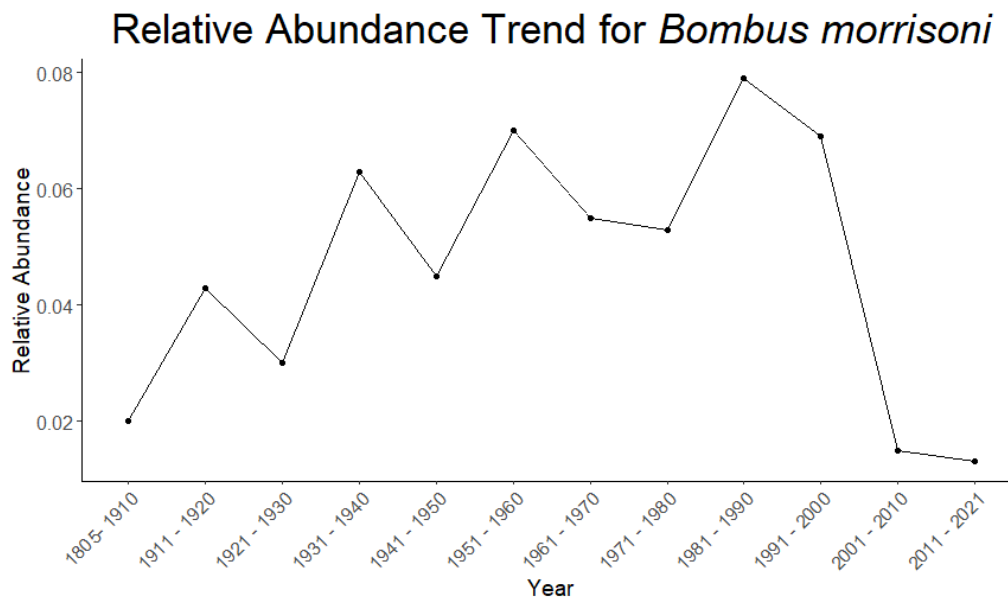


Figure 3. Relative abundance for Morrison bumble bee by decade, constructed using all available and digitized bumble bee occurrences within the range of Morrison bumble bee.

Persistence

To evaluate bumble bee species decline, other studies have used change in persistence to assess to what extent the species in question remains in areas within the range where it once occurred historically (Hatfield *et al.* 2015), akin to the Area of Occupancy metric utilized by the IUCN Red List (IUCN Standards and Petitions Committee 2022). To determine Morrison bumble bee persistence within its range, we divided the continent into a 50 x 50 km grid. To compare the historic time period with the current time period, we divided the number of grid cells occupied by Morrison bumble bee by the number of grid cells occupied by all species within the Morrison bumble bee range for the historic time period, then repeated this process for the current time period. We then divided the value from the current time period by the value from the historic time period to evaluate whether persistence changed over time (Table 1). While this metric does not provide a true measure of range size, it does provide a measure of Morrison bumble bee's persistence within its home range. We found that current persistence has declined 66% compared to the historic time period (Fig. 4). A recent modeling study conducted by Guzman *et al.* (2021) also provides support for range wide declines in occupancy for this species. This study used a multi-species occupancy model that looked at Morrison bumble bee occupancy sites within its range, using time intervals where at least one other bumble bee species was detected, indicating that the area had been sampled. that incorporated species' expected ranges to examine the decline of bumble bees in North America, and reports that the estimate of occupancy (which is similar to persistence) for Morrison bumble bee has declined by 51% (Guzman *et al.* 2021).

Table 1. Values and calculations for analysis of persistence, using records for all bumble bee species that occur within the range of Morrison bumble bee.

Time Period	Cells occupied by <i>B. morrisoni</i>	Cells occupied by any bee species	Percent
Historic (1805 - 2010)	480	1129	37.3
Current (2011 - 2021)	143	1125	12.7
	<i>B. morrisoni</i> persistence	$12.7 \div 37.3$	34
	<i>B. morrisoni</i> loss of persistence	$1 - 0.34$	66

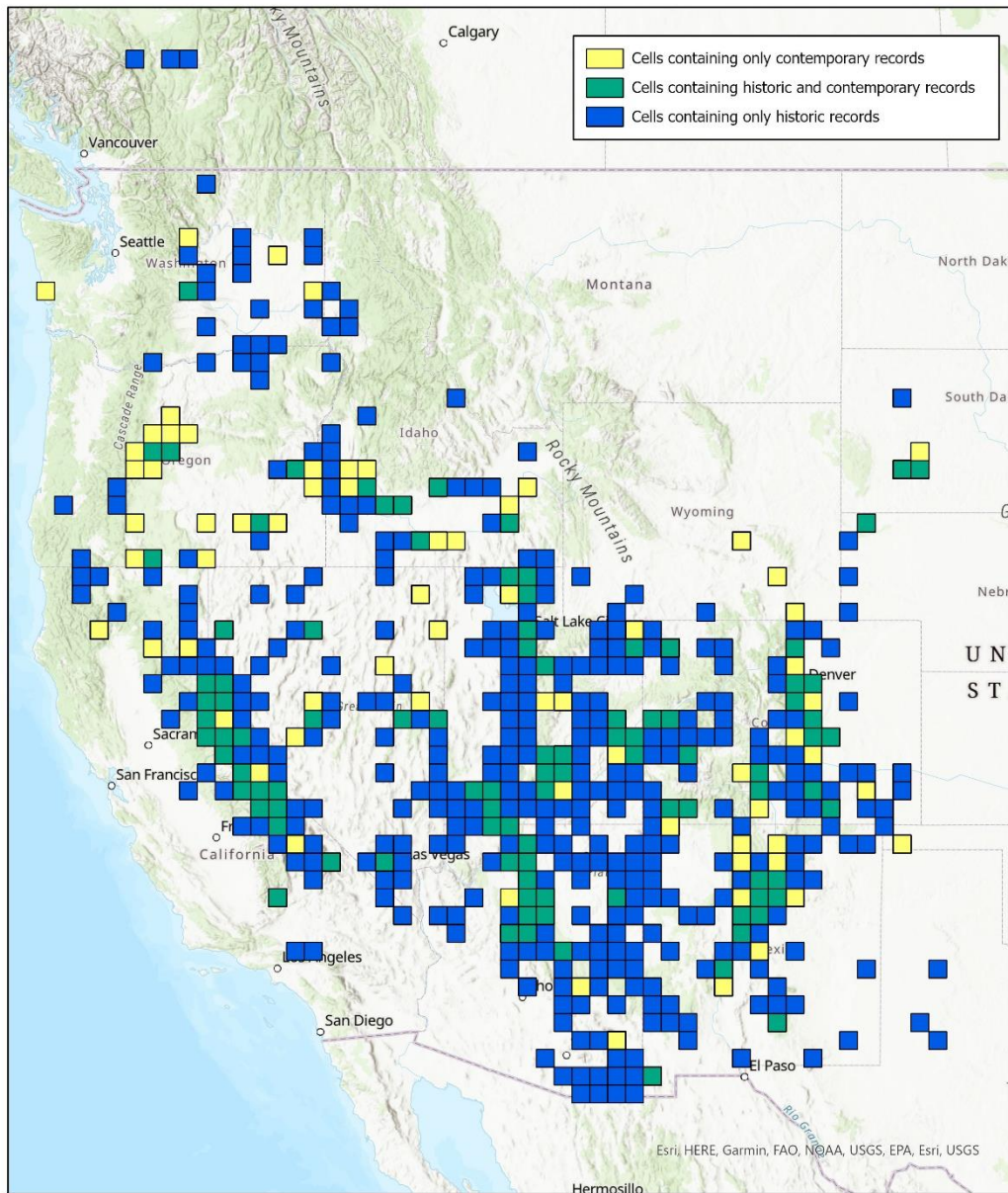


Figure 4. Persistence of Morrison bumble bee in 50 x 50 km grid cells, using data from two time periods: Historic (1805 – 2010) and Current (2011 – 2021). Blue represents cells containing only historic records, yellow represents cells that contained only current records, and green represents cells that had Morrison bumble bee records from both time periods.

Extent of Occurrence

To assess bumble bee decline, previous studies have compared extent of occurrence polygons from two time periods to estimate range loss (Cameron *et al.* 2011a; Hatfield *et al.* 2015). To estimate range loss for Morrison bumble bee, we compared the extent of occurrence (EOO) for the current time period (2011 - 2021) to the historic time period (Pre-2011). Because the historical database of bumble bee observations within the range of Morrison bumble bee had

significantly more records, and could therefore lead to an overestimate of range loss due to an increased chance of including more records near the edge of each species range, we rarefied the historic dataset by randomly selecting 26,590 records from the historic time period to use in the EOO measurement. We then used a z-test to ensure that the relative abundance of Morrison bumble bee was not significantly different in the rarefied dataset from the original dataset. To construct EOOs for each time period, we used a k-nearest neighbors approach to create local convex hulls (Getz *et al.* 2007). Because most historic records available for Morrison bumble bee are from incidental observations or museum specimen records, rather than systematic, equal effort surveys – including lack of detection information – throughout the range, range loss estimates using an EOO approach must be corrected by sampling effort. To address the difference in sampling effort and the potential to overestimate range loss, we constructed sampling density (a proxy for survey effort) raster layers from the presence points of bumble bee occurrences of all species in the data set for both time periods (ArcGIS Pro 3.0). We then calculated the average relative difference in sampling density for the area where the EOO from the historic time period did not overlap with the current time period, and used this value to correct range loss by sampling density (Equation 1). Areas that had been under sampled in the contemporary time period had the area of range loss adjusted down accordingly.

$$RL_A = RL_C \left(\frac{|SD_H - SD_C|}{\frac{SD_H + SD_C}{2}} \right)$$

Equation 1. Equation used to adjust (A) range loss (RL) estimates based on sampling density (SD) in the historical (H) and Current (C) time periods.

Using the area calculated from these polygons, we compared the contemporary area to the historic area to determine change in range size. Before correcting for sampling density, we found a range loss (decline in EOO) of 21.3%. After correction we estimate that this species has experienced a range loss of 13.9%. The EOOs for all historic data, rarefied historic data, and contemporary data are displayed in Figure 2. Range losses at the edges of Morrison bumble bee range are represented by few or no recent observations in the southern Sierra Nevada, south-central Arizona, eastern New Mexico, and northern Washington. Just as concerning, but not represented in the change in EOO, are few or no observations in the center of this bee's range, including the Columbia River Basin, where it was historically more common and has recently received significant recent survey effort (Hatfield *et al.* 2021) The species is also largely absent from Nevada in the recent time period, though the degree of survey effort there is less certain.

Recent Survey Efforts

Since 2000, there has been substantial recent interest in bumble bees among scientists, naturalists, and the general public, and collection effort has dramatically increased (Fig. 5). Our understanding of the current distribution of Morrison bumble bee has been greatly informed by a community science monitoring effort that began tracking the species in 2014, and significantly expanded in 2018. Within the range of Morrison bumble bee, there are four Xerces Society Bumble Bee Atlas projects, covering six states where the species occurs (OR, WA, ID, CA, NE, SD), and an additional project similar to the Atlas projects in Utah. These atlases conduct equal effort, equal area surveys distributed evenly across entire states, to provide baseline data to which future efforts can be compared. Some areas of high recent search effort have detected few Morrison bumble bees, while other parts of the range of the species are generally under surveyed (Fig. 6). While recent incidental observations allow us to identify the current distribution of the Morrison bumble bee, the multiple recent observations of this species should not be interpreted as evidence that this species' populations are stable or recovering. Because survey effort has increased over time in some parts the range, with special attention to rare species of bumble bees, it is important to evaluate changes in relative abundance, persistence, and range size when drawing conclusions about whether a bumble bee species' population is declining, stable, or increasing.

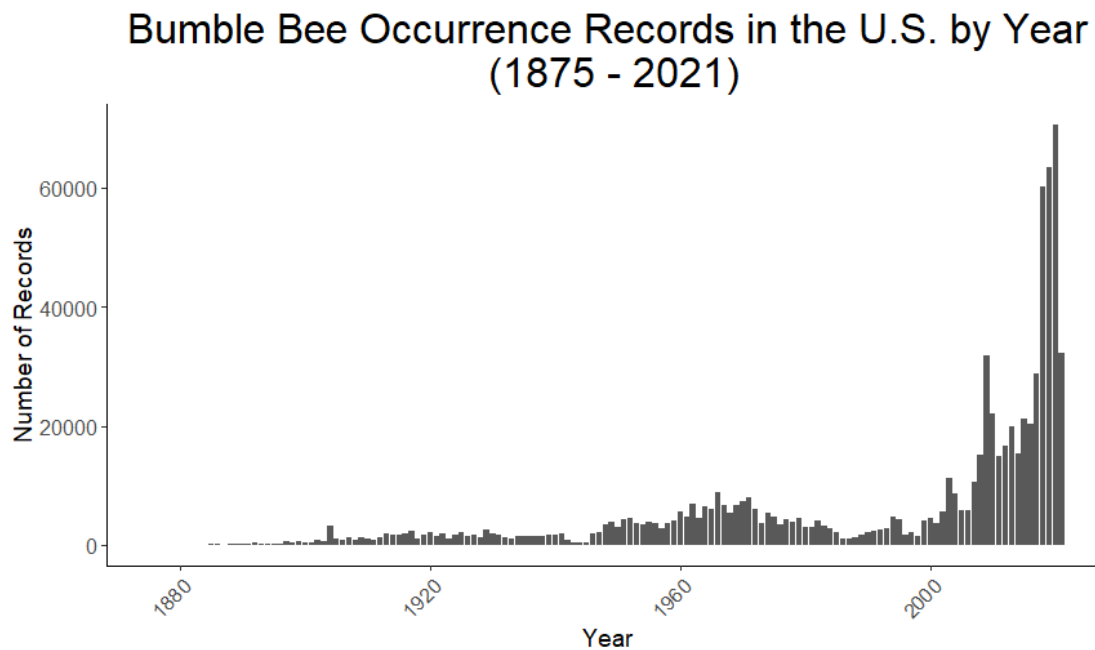


Figure 5. Bumble bee occurrence records over time (Richardson 2022), showing a major increase in collection effort post-2000.

The Pacific Northwest Bumble Bee Atlas covers Oregon, Washington, and Idaho. In bumble bee surveys from this atlas from 2018 - 2021, Morrison bumble bee accounted for less than 0.5% of all bumble bee species observations (Hatfield *et al.* 2021). Using Morrison bumble bee data from multiple sources, records for this species in OR, WA, and ID account for approximately 0.3% of all bumble bee observations in the recent time period (2011 – 2021) (Richardson 2022). From

1902, the first year the species was recorded in the region, to 2010, Morrison bumble bee observations in OR, WA and ID accounted for 1.3% of all bumble bee observations on average (Richardson 2022). Although this species never represented a large portion of the bumble bee fauna of the region, its relative abundance in the Pacific Northwest has decreased by more than half in the recent time period compared to its historical relative abundance.

The California Bumble Bee Atlas, which launched in 2022, detected 5 Morrison bumble bees out of 3,666 total observations in the initial year of surveying, representing approximately 0.1% of all detections. This Relative Abundance measure is consistent with Morrison bumble bee relative abundance in a larger dataset compiled from multiple sources over the recent time period (2011 – 2021) (Richardson 2022). Between 1914, the first year Morrison bumble bee was recorded in the CA, and 2010, the species accounted for 0.6% of the all bumble bee observations in CA on average (Richardson 2022). As such, in California, the relative abundance of Morrison bumble bee has declined by more than 80% between recent and historic time periods.

A 2006 study updating a 1962 survey of the bumble bees of Nebraska (LaBerge & Webb) failed to locate any Morrison bumble bee (Golick & Ellis 2006). Although LaBerge and Webb reported the bee being rare at the edge of its range in Nebraska with 14 individuals detected in the 1962 survey, we report the more recent study here to document its rarity at the edge of its range. The Nebraska Bumble Bee Atlas has surveyed bumble bees throughout the state since 2019, with 6,935 bumble bee observations in 2019, 2020, 2021, and 2022 combined. In that time, there were no Morrison bumble bee observations as part of that search effort. The 2023 Nebraska Bumble Bee Atlas survey effort, still underway at the time of this writing, has detected two Morrison bumble bees out of 250 total bumble bee observations. With the addition of two research grade iNaturalist observations (GBIF 2023), there have only been four observations for this species since 2011.

The Great Plains Bumble Bee Atlas, initiated in 2022, covers Morrison bumble bee range in South Dakota, as well as Kansas and North Dakota, where the species is not known to occur. In 2022, there were 920 bumble bee observations gathered from 106 surveys in South Dakota. Morrison bumble bee was not detected.

In Utah, where Morrison bumble bee has historically represented a larger portion of overall bee observations relative to other states mentioned above, the Utah Pollinator Pursuit has tracked this species since 2020. This project uses both incidental observations and effort standardized searches targeting all bumble bee species that occur within the state. In 2020, 2021, and 2022, Morrison bumble bee accounted for 3.5, 2.3, and 3.5 percent of all bumble bee observations, respectively (A. Barth, pers. comm. with S. Killingsworth, Dec. 2022). From 1902, the first year Morrison bumble bee was recorded in Utah, to 2010, this species represented 18% of all Utah bumble bee observations (Richardson 2022). Using data from multiple sources for Utah, the relative abundance of Morrison bumble bee for the present time period (2011- 2021) is approximately 8%, which is higher than the relative abundance observed by the Utah Pollinator Pursuit, but reduced more than half from historic relative abundance.

In British Columbia, Canada, where Morrison bumble bee is known from several historic records in the southern part of the province, the British Columbia Community Bumble Bee Project has

established long term bumble bee monitoring routes with effort standardized surveys along the routes. These routes were first surveyed in 2022, with 1,198 surveys total across 44 days in July and August, mostly in the southern half of the province. This effort generated 4,175 bumble bee observations, and Morrison bumble bee was not detected (J. Heron and C. Sheffield, British Columbia Community Bumble Bee Project, BC Ministry of Water, Land and Resource Stewardship, Surrey, BC, pers. comm. with S. Killingsworth July 2023).

Systematic bumble bee surveys have not yet been conducted across other states in Morrison bumble bee range.

A study using specimen data and field observations (1998 - 2004) to document bumble bees of the Madrean Archipelago of southeast Arizona described Morrison bumble bee as “the most widespread of the true Sky Island bumble bees” and recorded it in four separate sky island mountain ranges (Schmidt & Jacobson 2005, in Gottfried *et al.* 2005; J. Schmidt, pers. comm. with S. Killingsworth, Sept. 2022). Since 2011, there have only been two records of Morrison bumble bee in any of the sky islands included in the 2005 study, both of which were observed in the same mountain range. Although there has been no coordinated bumble bee survey effort in these mountain ranges, occurrence records for all bumble bee species in the recent time period (2011 – 2021) are well distributed across each mountain range, suggesting that if Morrison bumble bee was still widespread in this region, it would have been more readily detected. Little habitat connectivity between sky islands of the Madrean Archipelago make the chance of Morrison bumble bee naturally returning to this part of its range unlikely if they become extirpated.

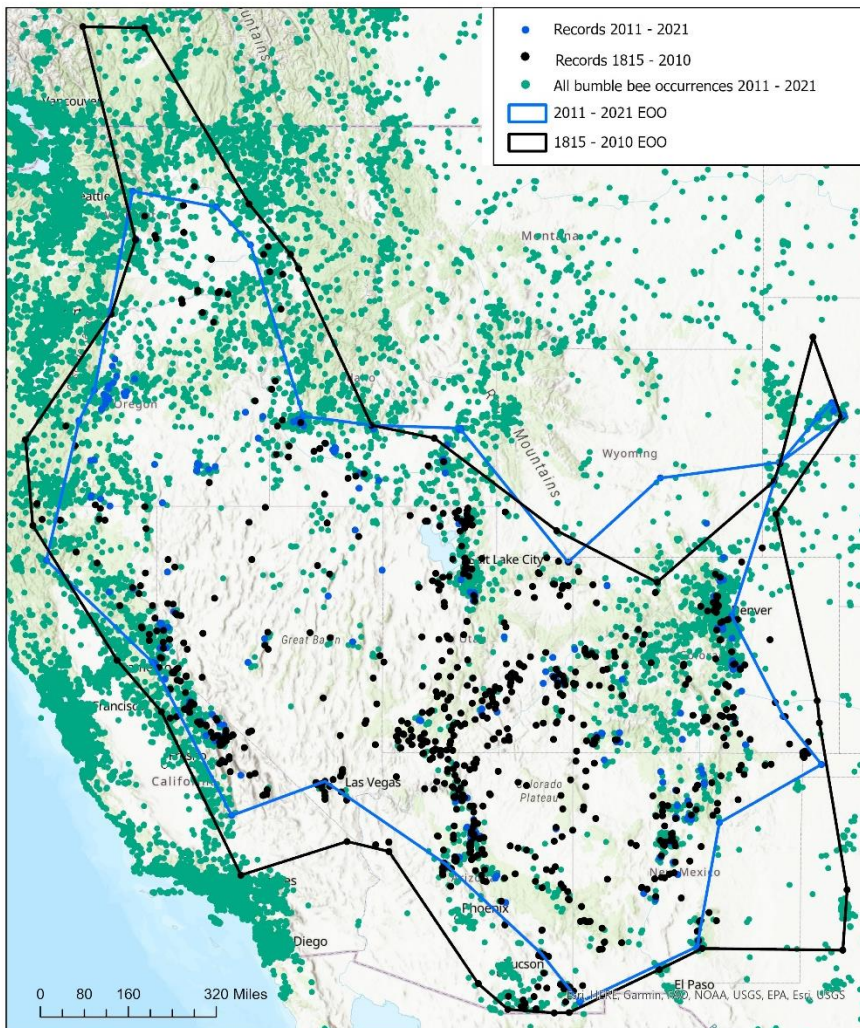


Figure 6. All bumble bee species records within the range of Morrison bumble bee (2011 – 2021), with Morrison bumble bee records from two time periods (1815 – 2010 in black, and 2011 – 2021 in blue).

In summary, Morrison bumble bee has declined in relative abundance by 74% across its entire range, with reduced relative abundance or absences of this bee reported in five standardized bumble bee survey efforts within its range. Morrison bumble bee persists in 66% less area than where it was historically observed. It has not been detected in 13.9% of its historic range in the last decade, despite considerable effort to survey many of these locations, and despite increased public interest and survey effort in recent years. Without any intervention these declines will continue, and this species is at risk of extinction.

VII. CURRENT AND POTENTIAL THREATS – SUMMARY OF FACTORS FOR CONSIDERATION

A. The Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range

Morrison bumble bee, like most North American bumble bees, faces general threats from habitat alterations that can interfere with its primary habitat requirements, including: access to sufficient food (nectar and pollen from flowers), nesting sites (such as underground abandoned rodent cavities or above ground in clumps of grasses), and overwintering sites for hibernating queens (undisturbed soil and/or organic material to burrow under). Morrison bumble bee historically occupied the shrublands, grasslands, and woodland edges of the West which have largely been affected by drought, an altered fire regime, livestock grazing and associated habitat changes (e.g. altered hydrology, invasive plant species), and loss or fragmentation by agricultural conversion and urban development. Additionally, this bee's habitat faces new and increased threats from energy development and mining. This species and its habitat are also impacted by drought and climate change, which is detailed in Section *E. Other Natural or Manmade Factors Affecting its Continued Existence* Subsection 1. *Global Climate Change*. Several ecosystem types that overlap with Morrison bumble bee range are considered vulnerable or imperiled. These include the Palouse prairies of Idaho, Oregon, and Washington, the warm desert riparian bosque of the Southwest, two Chihuahuan grassland types of Arizona, New Mexico, and Texas, the intermountain semi-desert grassland, and the Snake-Columbia shrub steppe (Lichthardt and Moseley 1997; NatureServe 2022b-e; Noss *et al.* 1995).

Bumble bee species richness, abundance, and genetic diversity are influenced by the quality of habitat on a landscape level. Isolated patches of habitat may not be sufficient to support bumble bee populations (Hatfield & LeBuhn 2007; Öckinger & Smith 2007), and populations of bumble bees existing in fragmented habitats can also face problems with inbreeding depression exacerbated by their unique method of sex determination (Zayed & Packer 2005; Darvill *et al.* 2006, 2012; Ellis *et al.* 2006). For example, Darvill *et al.* (2012) found that bumble bee populations limited to less than 15 km² of habitat were more likely to show signs of inbreeding. Goulson (2010) suggests that a viable population of bumble bees probably requires approximately 3.3-10 km² of suitable habitat. Habitat fragments smaller than 10 – 15 km² may not be large enough to support genetically diverse bumble bee populations. Habitat fragmentation and degradation has reduced the total contiguous habitat within the range of Morrison bumble bee. Not all remaining patches of wild habitat may be large enough to support robust, genetically diverse Morrison bumble bee populations (Goulson *et al.* 2008). Although this species can use suburban and agricultural lands, populations occurring there can be impacted by dangerous levels of pesticides present in these landscapes, and the lack of connectivity between these sites in the Intermountain West will make persistence difficult.

1. Livestock Overgrazing

Land degradation from livestock overgrazing threatens Morrison bumble bee. Much of the range of the species includes the Intermountain West, a region that has not evolved with heavy grazing pressure (Mack & Thompson 1982; Grayson 2006). This region is now grazed heavily by cattle, sheep, and feral horses (Beever *et al.* 2008). In the western US, livestock grazing is the most widespread use of federal public lands, and the domestic cow is the most widely distributed

mammalian herbivore (Fleischner 1994). Approximately 230 million acres of Forest Service and Bureau of Land Management (BLM) lands are grazed in the western U.S. (U.S. GAO 2005; Kauffman *et al.* 2022). Additionally, in 2017 there were over 220 million acres of non-federal grazing lands in use in 10 western states that make up the bulk of Morrison bumble bee's range (USDA 2022). On BLM land, over 54 million acres of grazed lands fail to meet the agency's own standards for land health. This represents over half of the BLM grazing allotments that have had their health assessed (BLM 2022a).

Ungulate grazing can impact ecosystems in several direct ways, with cascading effects relevant to bumble bee health. Grazing removes vegetation, reducing the available forage for wildlife species, including bees. Livestock can trample streambanks and compact soils, which increases soil erosion, streambank incision, and lowers the water holding capacity of soils (Dwire *et al.* 1999; Kauffman and Pyke 2001). Compaction of the soils from cattle can also limit the abundance of suitable bee nesting sites (Kimoto *et al.* 2012; Thapa-Magar *et al.* 2022). Finally, grazing contributes to exotic species invasion by dispersing plant propagules or creating favorable conditions for invasion (Reisner *et al.* 2013; Olf & Ritchie 1998; Chambers *et al.* 2014).

In addition to direct effects, ungulate grazing can have indirect impacts on landscapes. The primary indirect effect of ungulate grazing is from the removal of floral resources bumble bees rely on (Hatfield & LeBuhn 2007; Black *et al.* 2011; Cane 2011, Kimoto *et al.* 2012; DeBano *et al.* 2016; Davidson *et al.* 2020). In the Intermountain West specifically, livestock grazing has been associated with the invasion of the European annual grass species *Bromus tectorum* (cheatgrass). On millions of acres of western rangeland, this has become the dominant species (Mack 1981). The impact of this invasion on bumble bees is twofold. First, cheatgrass outcompetes many native forbs (Mack 1981), reducing the availability of pollen and nectar resources for Morrison bumble bee. Second, its ability to germinate and grow in the harsh interstitial spaces between shrubs has connected fuels together and contributed to a much more frequent fire interval in an ecosystem that had a historic fire interval as infrequent as every 100 years (See Section VII. *Current and Potential Threats A. Factor 2. Fire and Fire Suppression*; Wright and Bailey 1982; Brooks *et al.* 2004). Together, these factors combine to contribute to desertification, reduced animal and plant diversity, vegetation compositional shifts, and local extirpations of sensitive species (Fleischner 1994; Manley *et al.* 1997; Kauffman *et al.* 2022).

Generally, as the intensity of livestock grazing increases, pollinators, including bees, 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 Tschardt 2002a, 2002b; Vulliamy *et al.* 2006; Pöyry *et al.* 2006; Littlewood 2008; Jerrentrup *et al.* 2014; Elwell *et al.* 2016). Bee pollen and nectar plants can be reduced by grazing. Even if plants that bees use for forage are unpalatable to livestock, they may still be impacted by trampling (Waterbury *et al.* 2019). Although grazing has the potential to 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; Ellingsen *et al.* 1997; Smallidge and Leopold 1997; Weiss 1999; Pöyry *et al.* 2004, 2005; Saarinen *et al.* 2005; Rundlöf *et al.* 2008; Potts *et al.* 2009; Vanbergen *et al.* 2014), these effects

are largely seen only in low to moderately grazed grasslands that evolved with megafaunal herbivores (Bussan 2022).

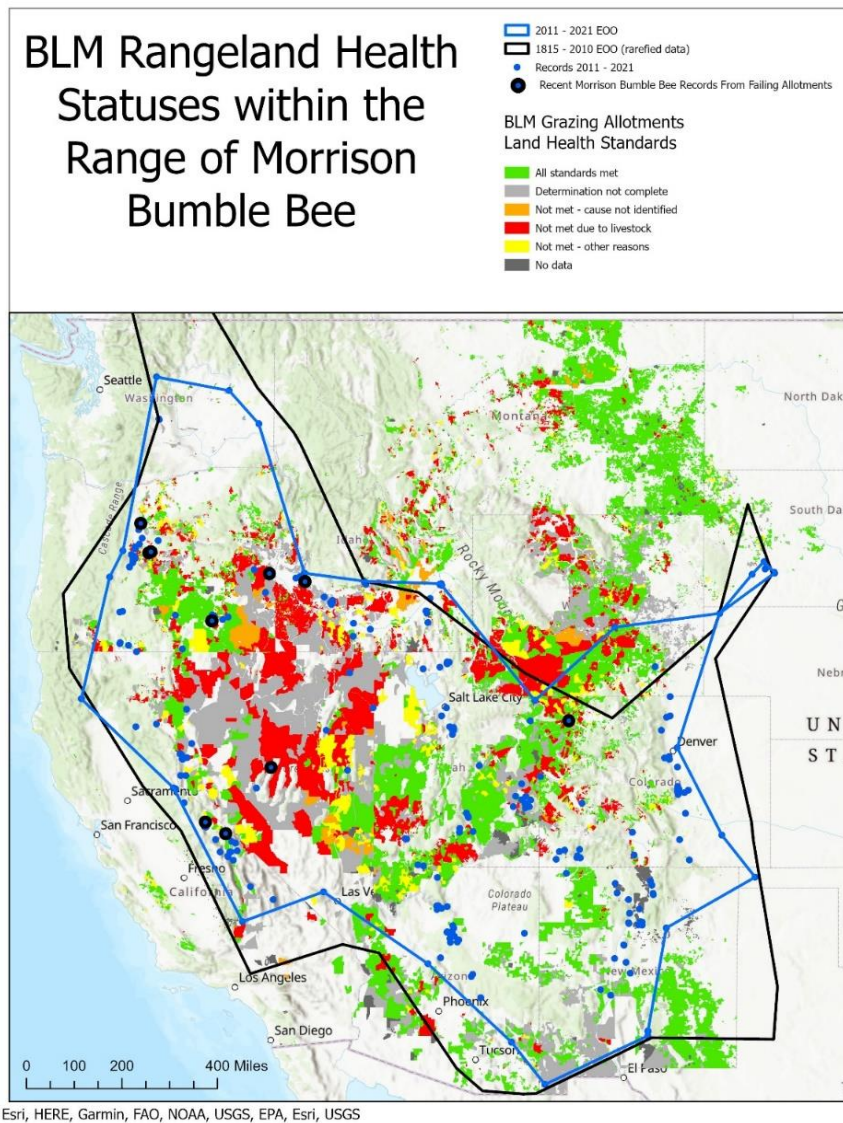


Figure 7. Land Health Standards statuses of BLM grazing allotments showing the overlap of overgrazed public lands with the historic and current extent of occurrence for Morrison bumble bee. Data from BLM assessments from 1997 – 2019 (BLM 2022a).

Currently, 54 million acres of BLM land fail to meet the agency's own land-health standards, and of the allotments that are failing, 72% of them have their failure attributed to livestock grazing. Within the historic range of Morrison bumble bee, 50 million acres of BLM range land are failing to meet health standards (Fig. 7). These land-health standards include minimum benchmarks to ensure sustainable landscape function, and are based on factors including water quality, soil health, species diversity, and habitat quality. Of the 246 million acres of land that the BLM oversees, over half of its acreage, or 155 million acres, has been leased as grazing allotments for cattle, sheep and other livestock (BLM 2022a). Many allotments within the 155

million acres of grazing land have never had their health status assessed, so the failure rate reported here may be an underestimate.

The degradation of habitat due to overgrazing directly contributes to the risk of extinction for Morrison bumble bee in several ways. First, hydrological changes caused by streambank erosion, streambank incision, and compaction of soils can contribute to the desiccation of meadows and floodplains and increased aridity within the range of Morrison bumble bee (Beschta *et al.* 2013; Kauffman *et al.* 2022), which may limit the quality and quantity of pollen and nectar resources for this species. Second, grazing activity can contribute to the alteration of the plant community, which has trended towards an exotic annual grassland throughout much of the range of the species, which contributes little to the dietary needs of the bumble bee. Lastly, grazing cattle are in direct competition with Morrison bumble bee, foraging on plant species that the bee requires for successful reproduction and colony growth. Impacts of overgrazing contribute to a landscape with reduced foraging opportunities for Morrison bumble bee.

2. Fire and Fire Suppression

Wildfire and prescribed fire can have both negative and positive effects on bumble bees. In parts of Morrison bumble bee range where ecosystems are fire adapted, such as the eastern Sierra of California and the pine forests of the Colorado Plateau, wildfire may support Morrison bumble bee. Frequent, low severity wildfire in fire-adapted landscapes has been shown to contribute to higher densities of forbs (Mola & Williams 2018), longer within-species flowering times (Wroblewski & Kauffman 2003), and higher nectar concentrations (Potts *et al.* 2003). Fire suppression in these areas threatens Morrison bumble bee by both reducing the forb cover and increasing the chance of catastrophic wildfire, both now (due to a history of fire suppression) and in the future.

When fire suppressed landscapes do eventually burn, fire behavior can be much more extreme compared to historical conditions, due to an accumulation of unburned fuels. Certain fire conditions resulting from fire suppression may not support bumble bees, even in fire adapted ecosystems. Accumulated fuels can lead to more severe wildfires, causing increased mortality for standing vegetation, and burning into soil layers, destroying the seed bank (Miller *et al.* 2009). Although fire has been shown unlikely to kill ground-nesting solitary bees (Cane & Neff 2011), some bumble bee species overwinter in the flammable duff layer (Williams *et al.* 2019), which may be thoroughly burned in severe wildfires. Fires with accumulated fuels may burn more homogeneously, leaving few patches of unburned habitat as refugia for bumble bees (Cassell *et al.* 2019). The effects of fire suppression and subsequent increased fire intensity is expected to become more severe with climate change (Cassell *et al.* 2019), leading to more complete burning of nesting sites, and fewer refugia for bumble bees. Finally, wildfires in fire suppressed landscapes may be larger, with the potential to destroy bee populations and habitat over a wider area (Keane *et al.* 2008; Miller *et al.* 2009). In most of the West, the frequency of large wildfires, defined as burning over 405 hectares, is increasing (Fig. 8; Dennison *et al.* 2014). Larger wildfire footprints, especially if wildfire is severe and there are no unburned patches of vegetation, have the potential to negatively influence or destroy wildlife populations (Ponisio *et al.* 2016; Steenvoorden *et al.* 2019), including those of Morrison bumble bee.

Impacts from wildfire, as a result of drought, climate change, exotic species invasion, and the legacy of fire suppression, are expected to be ongoing throughout much of the range of Morrison bumble bee. Wildfire hazard potential across much of the west is predicted to be high, including areas of high habitat suitability for the species. This metric is based on wildfire likelihood and intensity, fuels data, and point locations of past wildfires, and describes fires that are likely to be difficult to contain or suppress with the given conditions (Dillon and Gilbertson-Day 2020). For Morrison bumble bee, this means larger, more severe, and more homogenous fires predicted within its range, reducing the available nectar and pollen resources over larger areas, and limiting the available habitat.

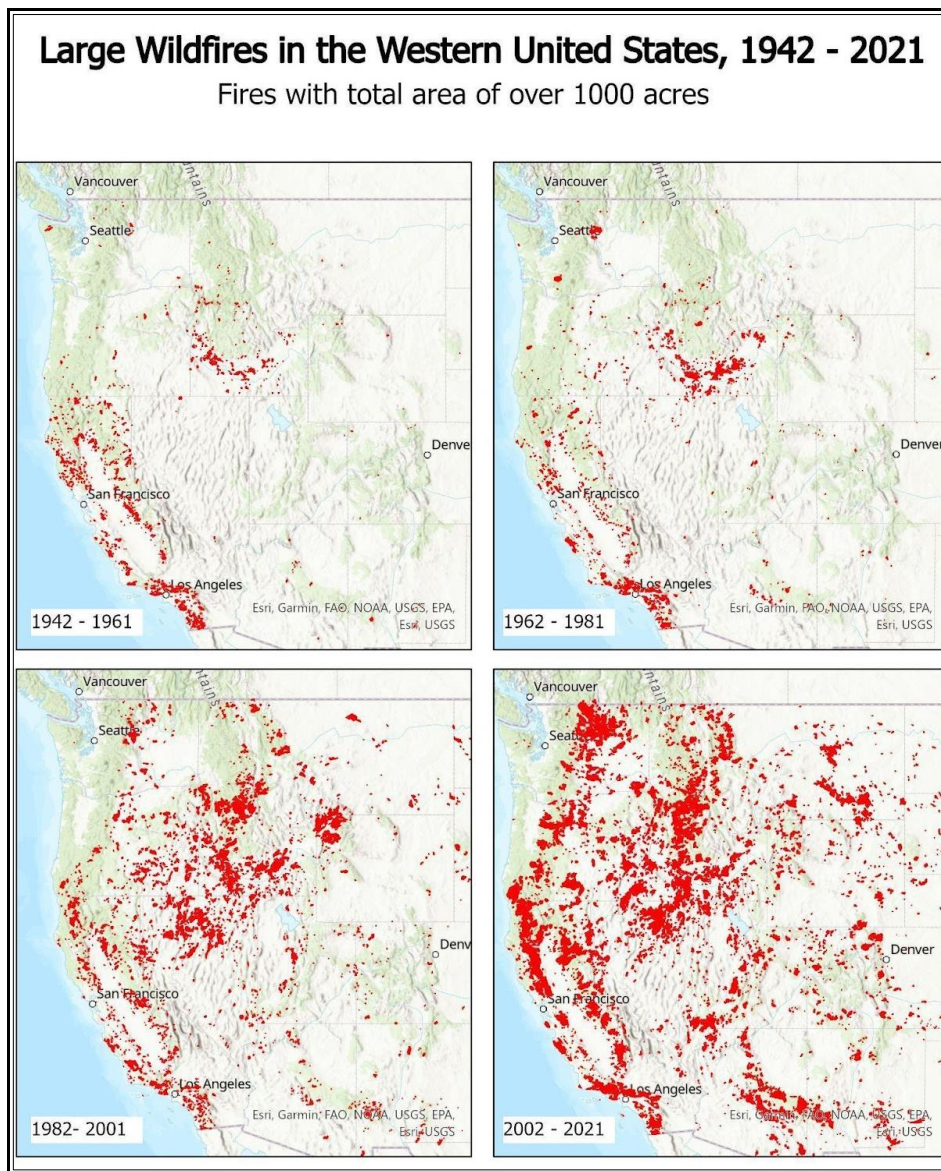


Figure 8. Perimeters of large fires (> 1000 acres) in four 20-year periods. Data from National Interagency Fire Center, available at https://data-nifc.opendata.arcgis.com/search?tags=Category%2Chistoric_wildlandfire_opendata

Some portions of the range of Morrison bumble bee have been impacted by changes in fire regime, like Great Basin big sagebrush (*Artemisia tridentata*) ecosystems, which have seen the encroachment of woody species (Pinyon – primarily *Pinus monophylla*, and Juniper – primarily *Juniperus osteosperma*) as a result (Miller & Rose 1999; Miller *et al.* 2008). Where these woody species have encroached, they can compete with the herbaceous perennials required by Morrison bumble bee and suppress their growth (Chambers *et al.* 2014). When fire suppressed pinyon-juniper woodlands burn, the accumulated fuel can promote cheatgrass (*Bromus tectorum*) invasion in some of the warmer parts of big sagebrush ecosystems (Chambers *et al.* 2007). Other ecoregions within the range of Morrison bumble bee have seen their fire regimes significantly altered by the invasion of nonnative grasses, with some of the most common including *Bromus rubens*, *B. tectorum*, *Cenchrus ciliaris*, and *Schismus arabicus* (Brooks *et al.* 2001; Abella *et al.* 2015). These grasses add fine fuels to wildfires and create unbroken tracts of fuel by occupying interstitial space between vegetation. This has increased the fire frequency and severity in certain landscapes where fire was historically infrequent and less severe, especially the Mojave, Sonoran, and southern Great Basin Deserts (Brooks and Chambers 2011; Dennison *et al.* 2014). In some of these invaded landscapes, the fire interval can now be as frequent as every 3 - 5 years (Whisenant 1990; Chambers *et al.* 2007), compared to a historic fire regime of every 30 - 100 years (Wright and Bailey 1982; Brooks *et al.* 2004). These more frequent fires destroy biotic soil crusts that would normally encourage the germination and growth of native forbs. The more frequent fire intervals also kill the historically dominant vegetation that is not fire adapted, and reduce the cover of forbs that burn before they reach reproductive age (Whisenant 1990; Brooks *et al.* 2004). This altered fire regime effectively replaces stands of diverse shrubland or desert with homogeneous grassland (Young and Evans 1978), reducing food sources for Morrison bumble bee.

These changes, through altered fire behavior in ecosystems where fire has been suppressed, or from increased fire frequency where fire was historically rare, can lead to degradation of plant communities and seed banks, and endanger Morrison bumble bee populations at a particular site, contributing to its risk of extinction.

3. Urban Development

Morrison bumble bee requires a diverse landscape with flowering plants available in abundance throughout the active season (March – October), open ground for nest construction, and overwintering sites with minimal soil disturbance. Although Morrison bumble bee can persist in some urban environments, urban development can have negative impacts on this species. Urban development directly removes habitat through the construction of buildings and roads and often with the replacement of natural vegetation with turf grass. Habitat loss and plant community compositional changes as a result of urbanization directly threaten this species with extinction.

The conversion of wildlands to urban and suburban uses alters and fragments habitat, which has likely had a negative impact on populations of many bumble bee species, including Morrison bumble bee. Roads, railroads, and urban centers fragment plant populations, and thus restrict the movement of bumble bees (Bhattacharya *et al.* 2003). Fragmentation and urbanization also decrease plant diversity, contributing to reduced bee abundance and diversity (Winfrey *et al.* 2009; Glaum *et al.* 2017; Theodorou *et al.* 2020). Research in northern California found that the

overall area of the landscape covered by pavement had a negative effect on the density of bumble bee nests (Jha & Kremen 2013b). In addition, human land use and the associated development of impervious surfaces have been shown to limit bumble bee dispersal and gene flow (Jha & Kremen 2013a; Jha 2015).

Human populations in western states across the range of Morrison bumble bee are also expected to increase between 2020 and 2040, including a 21% increase in Oregon, 25% in Idaho, 27% in Washington, 30% in Nevada, 32% in Colorado, and 35% in Utah (Weldon Cooper Center for Public Service, University of Virginia 2022). Housing development needed to meet these increases will undoubtedly threaten currently open habitats used by Morrison bumble bee, contributing to further declines of this species.

While increased urbanization is likely to further threaten Morrison bumble bee with extinction, it is important to note that some residential gardens and urban parks can provide valuable floral, and in some cases, nesting and overwintering resources, and may serve as important habitat refuges for bumble bees (Frankie *et al.* 2005; McFrederick & LeBuhn 2006; Goulson *et al.* 2010; Daniels *et al.* 2020), even though they may not support the species richness that was found historically (McFrederick & LeBuhn 2006). Morrison bumble bee has been found in some natural areas within urban environments, such as residential yards, parks, and schools within the urban centers of Boise, ID; Albuquerque, NM; Denver, CO; and Salt Lake City, UT. Although Morrison bumble bee can use plant species that occur in disturbed environments (Hatfield *et al.* 2021), including residential settings, agricultural margins, and urban edges, and many recent observations of this species are from urban environments, Morrison bumble bee may become less abundant as these settings become more urban. The fact that Morrison bumble bee can be found in urban areas may not necessarily indicate that this is the preferred habitat for the species, but rather may be an effect of a decades long drought across most of the wildland range of the species. Its occurrence in urban lands may have more to do with the floral resource availability in these irrigated areas (Hung *et al.* 2021), and may come with both increased exposure to pesticides and less availability of nesting sites due to roads and other impervious surfaces.

4. Agricultural Intensification

Agricultural intensification is implicated in the decline of bumble bees in Europe (Diekötter *et al.* 2006; Goulson *et al.* 2008; Vray *et al.* 2019), and may also pose a significant threat to bumble bees in North America (Grixti *et al.* 2009). In general, increases in farm size have led to increased habitat homogeneity, with loss of features crucial to the success of bumble bees, including hedgerows, slopes, and field margins (Rundlöf *et al.* 2008; Vray *et al.* 2019). Changes in farming technology and increased efficiency have led to several practices that can be harmful to bumble bees, including widespread use of herbicide, which reduces availability of wildflower forage in field margins. Additionally, direct exposure to herbicide surfactants and co-formulants has been shown to cause mortality in bumble bees (Straw *et al.* 2021). Broad use of pesticides, including pesticide-coated seeds, poses a significant threat to Morrison bumble bee, which is further discussed in Section E. *Other Natural or Manmade Factors Affecting its Continued Existence* Subsection 2. *Pesticides*. Morrison bumble bee nests, which are typically underground, may be destroyed by farm practices such as tilling and mowing. Although some flowering crops may provide an abundance of nectar and pollen for bumble bees, large monocultures and the

resources they provide are typically only available for a short period of time. Bumble bees require foraging habitat with a diverse assemblage of plants that support continuous foraging from early spring through fall (Winfree *et al.* 2011; Cameron and Sadd 2020). The amount of pollen available to foragers affects the number of new queens that a bumble bee colony can produce, and therefore, along with a variety of other factors, impacts the size of the future bumble bee population (Burns 2004). Temporally limited nectar and pollen from flowering crops in agricultural settings may limit the number of queens a colony can produce.

Although agriculture is not the primary use of most land within Morrison bumble bee range, several regions where the species occurs have seen major habitat loss because of agricultural intensification. The Columbia River Basin in Washington lost over 60% of its original shrub-steppe habitat by 1986 (Dobler *et al.* 1996), and the Palouse grasslands of Washington and Idaho have been almost entirely converted to agriculture or otherwise altered, with only remnant patches remaining, mostly in steep canyons (Lichthardt and Moseley 1997). These are regions with few recent observations of Morrison bumble bee, despite concerted and systematic survey efforts (Hatfield *et al.* 2021). Regions with recent agricultural expansion include large sections of this species' range in central and northern Utah, western Colorado, central Washington, and north-central Oregon (Lark *et al.* 2015). This expansion has likely contributed to the decline of Morrison bumble bee through the conversion of once suitable habitat, and by increasing the exposure of the species to harmful pesticides used in agricultural settings. Within the range of Morrison bumble bee, the top ten agricultural uses include winter wheat, alfalfa, cotton, corn, non-alfalfa hay, sorghum, spring wheat, millet, potatoes, and barley production, totaling over 29 million acres (USDA 2022). Most of these crops are grains that rely on wind pollination, and as such produce little or no nectar, and thus have little nectar forage value for bumble bees. Of the top ten agricultural crops grown within the range of Morrison bumble bee, corn, cotton, wheat, and hay farming are among the highest pesticide uses for any US crop (Fernandez-Cornejo *et al.* 2014; USGS 2022; Fig. 11). Commonly used pesticides applied to these crops within the range of the species include some of the most harmful classes of insecticides, such as neonicotinoids, organophosphates, and carbamates (Wieben 2021). The effects of these pesticides are detailed in Section E. *Other Natural or Manmade Factors Affecting its Continued Existence*. Subsection 2. *Pesticides*.

5. Energy Development and Mining

The interior West supports much of the nation's on-shore oil and gas production. Infrastructure to support this industry, including pads for wells and compressor stations, storage tanks, and roads can directly impact bumble bees through loss of habitat. Disturbance created by the development and use of new roads can also facilitate the spread of exotic species. Many states where Morrison bumble bee occurs are likely to see increasing lithium mine development in the coming years. Currently, there are 72 proposed lithium extraction sites in the US, with projects focused in Arizona, California, New Mexico, Nevada, Oregon, Utah, and Wyoming (Fig. 9; Parker *et al.* 2022). Together, these states make up the bulk of the range for Morrison bumble bee, and are likely to overlap with habitats used by the species. Lithium mines, their

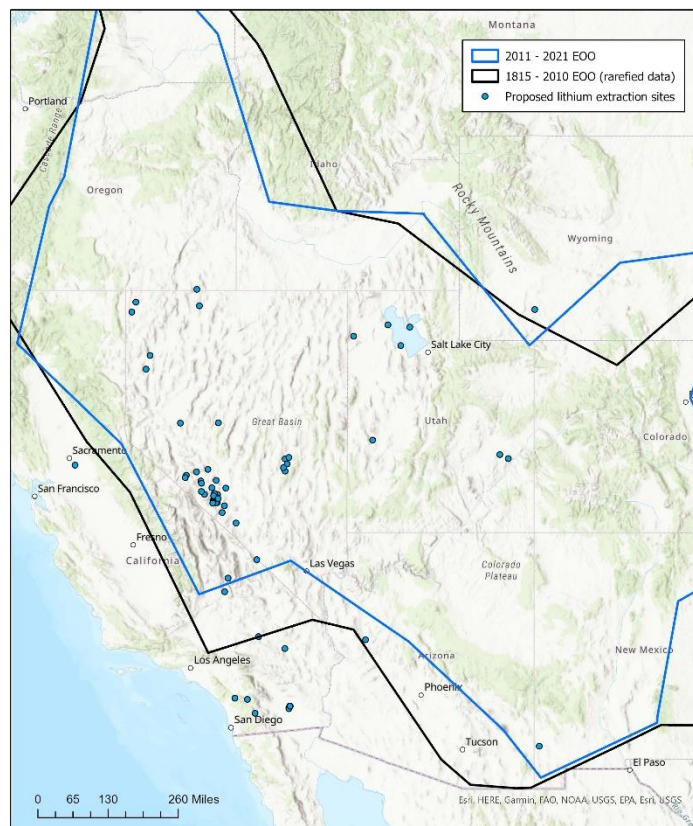


Figure 9. Proposed lithium extraction sites within the current and historic Extent of Occurrence (EEO) or range of Morrison bumble bee. Lithium site data from Parker *et al.* (2022).

infrastructure, and roads to access them can directly impact bumble bees through habitat loss. Hydrological impacts from water use at mines can indirectly impact bumble bees by depleting

and polluting ground water, which can alter the abundance and composition of flowering plants used by foraging bumble bees.

Existing solar energy farms and solar development threaten this species. Solar development on public lands limits the habitat potential for Morrison bumble bee. BLM has prioritized 870,000 acres of BLM land for solar development. As of 2021, the BLM had over 13,000 acres of active solar production within the range of Morrison bumble bee, with many more thousands of acres approved for solar use but pending construction (BLM 2022b). Construction of solar facilities typically begins by removing vegetation and grading the land, leading to changes in soil density and water infiltration capacity, which can then facilitate invasion by exotic species, and make surrounding areas more vulnerable to erosion (Lovich & Ennen 2011; Hernandez *et al.* 2014). Together these factors can alter plant productivity, reduces forage available to wildlife, and destroys nesting and overwintering sites for bumble bees. In addition to the limited habitat value of solar farms, there has also been documented mortality for insects flying over them. Although no direct observations of bee mortality have been recorded at solar facilities, one study documented “hundreds upon hundreds” of butterflies found dead at an Ivanpah, CA solar facility, many with signs of singeing (Kagan *et al.* 2014). Researchers deduced that the solar facility acts as a “mega-trap” for insects by attracting them to the brightly lit area near the boiler. The threat of mortality to bees by solar facilities may be significant, especially as more solar energy projects are developed.

B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

To the best of the petitioners’ knowledge, overutilization for commercial, recreational, scientific, or educational purposes does not pose a threat to Morrison bumble bee because this species is not produced or sold commercially. While specimens of female workers or males may still occasionally be collected for research purposes, this activity probably does not pose a threat to the overall survival of the species. However, if a Morrison bumble bee queen is collected, an entire colony is effectively eliminated. Collection of queens or larger numbers of workers or males from isolated parts of its range could threaten Morrison bumble bee, though we have no evidence this is occurring. Other historic scientific use includes collection of Morrison bumble bee for studies on the activity of its venom (Hussein *et al.* 1999; Zalat *et al.* 1999; Hussein *et al.* 2001). These studies do not detail the methods, collection locations, or quantities of Morrison bumble bee collection. Large scale collection of Morrison bumble bee queen, workers, or males for scientific study could threaten this species, although we have no evidence that this has occurred at a large scale, or is ongoing.

C. Disease or Predation

1. Pathogens and Parasites of Bumble Bees

Pathogens likely pose a substantial threat to the continued survival of Morrison bumble bee. Worldwide, reported pathogens and parasites of bumble bees include: viruses, bacteria, fungi, protozoa, nematodes, hymenopteran and dipteran parasitoids, one lepidopteran parasite, and mites (Acari) (summarized in Schmid-Hempel 2001; Evans *et al.* 2023; Figueroa *et al.* 2023). Pathogen prevalence and fitness effects in wild North American bumble bees are generally not

well understood, though pathogens are the leading hypothesis for the declines seen in several other bumble bees throughout North America (Cameron *et al.* 2011a; Cameron and Sadd 2020).

Micro- and macroparasites are abundant and widespread in bumble bee species that have been studied across the U.S. Although parasite loads have not been comprehensively studied in Morrison bumble bee, *Apicystis*, *Crithidia bombi*, *Vairimorpha bombi* (formerly *Nosema bombi*), Deformed wing virus, sacbrood virus, and black queen cell virus have all been detected in this species (J. Strange, pers. comm. with S. Killingsworth, Jan. 2023). A nationwide study on wild *Bombus* pathogens included 20 Morrison bumble bee individuals from Colorado, New Mexico, and Utah (J. Strange, unpublished results). For these individuals, observed infection levels were 15% for *Vairimorpha bombi*, 10% for an unknown *Vairimorpha* species, 15% for *Crithidia bombi*, 10% for *Apicystis*, 5% for deformed wing virus, 35% for sacbrood virus, 60% for Black Queen Cell virus, and 5% for conopid parasitism (J. Strange, pers. comm. with S. Killingsworth, Jan. 2023). Although the rates of parasitism reported here are within the normally observed range for bumble bee species in North America, and the small sample size of this study does not allow us to draw specific conclusions without further study, these pathogens can cause considerable reductions in fitness in bumble bees and pose a threat to Morrison bumble bee. The micro- and macroparasites that have been identified as pathogens of concern to wild North American bumble bees and implicated in the decline of some species (Cameron *et al.* 2011b; Cameron & Sadd 2020) are discussed below.

a. Microparasites

Vairimorpha

This genus of microsporidian pathogen is represented by two species that occur in the U.S, *Vairimorpha bombi* and *V. ceranae* (formerly classified as *Nosema*). *Vairimorpha bombi* is prevalent in wild bumblebees throughout North America (Colla *et al.* 2006; Gillespie *et al.* 2010; Kissinger *et al.* 2011; Cameron *et al.* 2011a; Cordes *et al.* 2012; Cameron *et al.* 2016). *V. ceranae*, considered an emerging pathogen of the honey bee, is known to spillover to bumble bees (Fürst *et al.* 2014; Graystock *et al.* 2016), and has been detected in many bumble bee communities worldwide (Li *et al.* 2012; Fürst *et al.* 2014; Gamboa *et al.* 2015; Grupe & Quandt 2020). Cameron *et al.* (2011a) found a significantly higher prevalence of *V. bombi* in declining North American bumble bee species (*B. occidentalis* and *B. pensylvanicus*), although Morrison bumble bee was not included in this study. *Vairimorpha bombi* infection was significantly lower in species that have not exhibited recent declines in range and relative abundance. Further work by Cameron *et al.* (2016), shows that this pathogen is likely amplified in commercial bumble bee colonies, and the subsequent spread to wild *Bombus* is implicated in the decline of bumble bee species.

Vairimorpha spp. infect bumble bees primarily in the malpighian tubules, but also in fat bodies, nerve cells, and sometimes the trachea (Macfarlane *et al.* 1995). *Vairimorpha bombi* can reduce colony fitness, colony size (Otti & Schmid-Hempel 2008), reduce individual reproduction rate and life span of bumble bees (Schmid-Hempel & Loosli 1998; Schmid-Hempel 2001; Colla *et al.* 2006; Otti & Schmid-Hempel 2008).

Crithidia

Crithidia bombi is a gut trypanosome protozoan found in bumble bees. It can be passed from and potentially amplified by commercial bumble bees and flower flies to bumble bees at flowers (Davis *et al.* 2021). Its effects on bumble bees include reduced longevity and colony fitness, impaired foraging ability, interference in learning among workers (Otterstatter *et al.* 2005; Gegear *et al.* 2006), decreased pollen loads, increased ovary development in workers (Shykoff & Schmid-Hempel 1991), and reduced queen hibernation success (Fauser *et al.* 2017).

Crithidia expoeki is a less-studied trypanosome protozoan characterized from bumble bees collected in Canada (Palmier *et al.* 2020), the U.S. and Switzerland (Schmid-Hempel & Tognazzo 2010; Tripodi *et al.* 2018) This pathogen has been detected in *Bombus caliginosus*, *B. fervidus*, and *B. vosnesenskii* in the West (Tripodi *et al.* 2018). Although it has not been detected in Morrison bumble bee, the species in which it has been detected co-occur with Morrison bumble bee, or have abutting ranges. The specific effects of this pathogen on its host are not well understood, but are presumed to be similar to *Crithidia bombi*, and thus are likely to reduce fitness in infected bumble bees (Palmier *et al.* 2020).

Apicystis bombi

Apicystis bombi is a neogregarine protozoan that is associated with rapid death of infected bumble bee queens early in the season (Macfarlane *et al.* 1995; Rutrecht & Brown 2008). It has also been shown to inhibit ovary development and reduce queen longevity (Rutrecht & Brown 2008). More research is needed to understand causal effects of this parasite in bumble bees, and its virulence. This parasite has been found in commercial bumble bee colonies (Meeus *et al.* 2011), and researchers suggest that this pathogen may have been introduced from Europe to NW Patagonia, Argentina on commercial bumble bees, where it is believed to be a causal factor in an observed population decline in the native South American bumble bee *Bombus dahlbomii* (Arbetman *et al.* 2012; Aizen *et al.* 2019).

RNA viruses

RNA viruses that have historically been considered to be specific to honey bees (*Apis mellifera*), including Israeli acute paralysis virus, black queen cell virus, sacbrood virus, deformed wing virus, and Kashmir bee virus, have been detected in wild North American bumble bees foraging near apiaries (Singh *et al.* 2010). A study on virus spillover from managed honey bees in Vermont found that prevalence of deformed wing virus and black queen cell virus was higher in bumble bees collected near apiaries, and presence of one or both viruses was detected on 19% of flowers within apiaries (Alger *et al.* 2019). Deformed wing virus, which is associated with severe winter losses in honey bees (Highfield *et al.* 2009), was also detected in bumble bees in Germany, and the infected bumble bees displayed the same deformities that are typical of infected honey bees (Genersch *et al.* 2006). The threat that RNA viruses pose to Morrison bumble bee is discussed below in Section 2. *Pathogen spillover, b. honey bees*. The prevalence of RNA viruses in wild populations of bumble bees, as well as their effects on bumble bee fitness, are in urgent need of further study.

b. Macroparasites

Locustacarus buchneri

Bumble bees are infected by mites, including *Locustacarus buchneri*, a species that parasitizes the trachea of bumble bees (Husband & Shina 1970). *Locustacarus buchneri* is associated with reduced foraging and lethargic behavior (Husband & Shina 1970) and a significantly reduced lifespan in male bumble bees (Otterstatter & Whidden 2004). In the west, *L. buchneri* has been documented in *Bombus mixtus* and *B. sitkensis* (Kissinger *et al.* 2011). Little is known about how this parasite is affecting Morrison bumble bee.

Sphaerularia bombi

Sphaerularia bombi is an entomopathogenic nematode that infects hibernating bumble bee queens and sterilizes them (Schmid-Hempel 2001). In a literature review, Macfarlane *et al.* (1995) notes that bumble bee queens infected with this parasite in New Zealand colonized new areas at a rate of less than 1% of that of healthy queens. This nematode has been documented in California (Poinar 1974) and Oregon (Maxfield-Taylor *et al.* 2011), so could plausibly affect Morrison bumble bee.

2. Pathogen Spillover

The potential for the spread of pathogens to Morrison bumble bee from the domesticated common eastern bumble bee (*Bombus impatiens*) and other species of bumble bees that have been developed for commercial use (including the western species *Bombus huntii* and *Bombus vosnesenskii*) threatens Morrison bumble bee with considerable reduction in fitness. In addition, RNA viruses from the domesticated honey bee (*Apis mellifera*) can be transmitted to bumble bees at shared flowers (Singh *et al.* 2010), and pose a novel threat to Morrison bumble bee, which have the potential to cause population declines absent any protection for this species.

a. Commercial Bumble Bees

Commercial bumble bees are used primarily to pollinate greenhouse tomatoes, and increasingly to pollinate a wide variety of other greenhouse and open field vegetable and fruit crops in the U.S. and worldwide (Velthuis & van Doorn 2006). The commercial bumble bee industry has grown dramatically in the past several decades (Velthuis & van Doorn 2006), coincident with the growth of the greenhouse tomato industry. From 1985-2005, there was a 30% increase in fresh tomato consumption in the U.S., with more than one-third of the fresh tomatoes in stores coming from hothouses (compared to a negligible amount in the early 1990s) (Calvin & Cook 2005), which typically use bumble bees to achieve pollination. Commercial bumble bees often escape greenhouses to forage on nearby plants (Morandin *et al.* 2001; Whittington *et al.* 2004; Strange *et al.* 2023), where they interact with wild bumble bees and have the opportunity to transmit pathogens at shared flowers. Commercially raised bumble bees frequently harbor high pathogen loads (Goka *et al.* 2000; Whittington & Winston 2003; Niwa *et al.* 2004; Colla *et al.* 2006; Murray *et al.* 2013) and the spillover of pathogens from commercial bumble bees in greenhouses to wild, native bumble bees foraging near greenhouses has been documented (Colla *et al.* 2006; Goka *et al.* 2006; Otterstatter & Thomson 2008; Murray *et al.* 2013, Graystock *et al.* 2013a).

There are some limited regulations on commercial bumble bee use in the states of CA, OR, and WA. In the rest of Morrison bumble bee range, commercial bumble bee use is not regulated, and information about commercial interstate movement, use, and pathogen load of commercial bumble bees is not public. This makes assessing and managing pathogen risk to wild bumble bees nearly impossible. Greenhouse tomato production, which relies heavily on commercial bumble bees, occurs in many of the states within the species range, including CA, OR, WA, ID, UT, WY, NM, NE and SD (Baskins *et al.* 2019), although OR and WA no longer permit the commercial use of non-native bumble bees. Additionally, open field use of commercial bumble bees, which provides even more opportunity for disease transmission, occurs within the range of Morrison bumble bee, and is not well regulated (Strange *et al.* 2023), though this practice is not allowed in OR, WA, and CA. Nevertheless, this presents a nearly range-wide possibility of pathogen transmission between wild Morrison bumble bees and commercial bumble bees.

Meeus *et al.* (2011) reviewed the effects of invasive parasites on bumble bee declines. They report that the commercial production of bumble bees has the potential to lead to bumble bee declines in three ways: commercial colonies may have high parasite loads, which could then infect wild bumble bee populations; commercial production may allow higher parasite virulence to evolve, leading to the introduction of parasites that are potentially more harmful to wild bumble bees than naturally occurring parasites; and the global transport of commercial bumble bees can introduce novel parasites to which resident, native bumble bees have not adapted. Pathogens reported from commercial bumble bee colonies worldwide include: *Apicystis bombi*, *Crithidia bombi*, *Locustacarus buchneri*, *Vairimorpha bombi*, black queen cell virus, deformed wing virus, Israeli acute paralysis virus, and Kashmir bee virus (Meeus *et al.* 2011). Commercial bumble bee colonies in North America have tested positive for *Crithidia bombi*, *Vairimorpha bombi*, *Locustacarus buchneri*, deformed wing virus, black queen cell virus, sacbrood virus (Morkeski & Averill 2012; Graystock *et al.* 2013b) and Israeli acute paralysis virus (Singh *et al.* 2010).

The role that pathogens play in widespread bumble bee declines has been an active area of research for over a decade. The spillover of the microsporidian parasite *Vairimorpha bombi* from commercial to wild bumble bees has been hypothesized as a cause of the sudden, rapid decline of four closely related North American bumble bee species - rusty patched bumble bee (*Bombus affinis*), Franklin's bumble bee (*Bombus franklini*), the western bumble bee (*Bombus occidentalis*) and the yellow banded bumble bee (*Bombus terricola*) (Thorp & Shepherd 2005; Evans *et al.* 2008). This hypothesis is supported by the timing, speed, and severity of the population declines. Additionally, research has shown that certain declining bumble bee species harbor significantly higher levels of *V. bombi* than stable species (Colla *et al.* 2008; Cameron *et al.* 2011a). This pathogen was present in low levels in wild bumble bees in North America prior to the establishment of the commercial bumble bee trade, but increased significantly in prevalence in the mid-1990s, concurrent with the *V. bombi*-induced collapse of commercial *Bombus occidentalis* production (Cameron *et al.* 2016; Cameron and Sadd 2020). It is thus surmised that naturally occurring *V. bombi* was amplified in commercial bumble bee production and escaped to wild bumble bee populations prior to some of the major declines of North American bumble bees (Cameron *et al.* 2016). Coincident with the crash in commercial colonies of the western bumble bee, researchers noticed that the western bumble bee and its relatives in the subgenus *Bombus* s. str. began disappearing from the wild in the late 1990s (Thorp &

Shepherd 2005; Evans *et al.* 2008; Thorp *et al.* 2010). Morrison bumble bee is expected to be similarly impacted by *V. bombi* amplification based on the likelihood of coming into contact with commercial bumble bees.

Analysis by Szabo *et al.* (2012) found a significant correlation between vegetable greenhouse density, which was used as a proxy for commercial bumble bee use, and the decline of the yellow banded (*Bombus terricola*) and American (*B. pensylvanicus*) bumble bees. However, this analysis did not address the possibility of an acute pathogen spillover event in which a rapid disease spread through wild populations. Furthermore, the analysis did not include areas where bumble bees are used in open field settings.

In Canada, higher levels of the protozoan parasite *Crithidia bombi* were detected in wild bumble bees foraging near greenhouses that used commercial bumble bees (Colla *et al.* 2006; Otterstatter & Thomson 2008), and it was suggested that this pathogen may be implicated in the sudden, widespread decline observed in North American bumble bees in the subgenus *Bombus sensu stricto* (Otterstatter & Thomson 2008). However, another analysis of pathogen prevalence in wild bumble bees did not find evidence that *Crithidia* infections are involved in the decline of U.S. bumble bee species (Cordes *et al.* 2012).

In Japan, where both Japanese and European bumble bee species are imported from the Netherlands for commercial use, researchers found that commercially raised bumble bees had a higher rate of infestation by the tracheal mite *Locustacarus buchneri* than wild bees. Their findings also suggested that a European strain of this mite has likely invaded native Japanese bumble bee populations. (Goka *et al.* 2000, 2006).

In NW Patagonia, Argentina, the commercial buff-tailed bumble bee (*Bombus terrestris*) was introduced from Europe in 2006. Researchers suggest that the highly pathogenic *Apicystis bombi* hitchhiked on the commercial bumble bees and spread to wild bumble bees, potentially causing the observed population collapse in the world's largest native bumble bee – *Bombus dahlbomii* (Arbetman *et al.* 2012). The introduction of non-native, commercial bumble bees has also introduced or amplified *C. bombi*, *V. bombi*, and *L. buchneri* to populations of wild *Bombus dahlbomii* (Schmid-Hempel *et al.* 2014; Morales *et al.* 2016).

In summary, the amplification and spillover of pathogens from commercial to wild bumble bees has been documented and is plausibly ongoing, and needed protections are not in place (Strange *et al.* 2023). Threats presented by commercial bumble bees are expected to contribute to the extinction risk for Morrison bumble bee, as the species has documented infection by several pathogens with high prevalence in commercial bumble bees. Morrison bumble bee also likely comes into contact with commercial bumble bees at foraging sites throughout its range, wherever commercial bumble bees are used.

b. Pathogen Spillover from Honey Bees

The spillover of RNA viruses from honey bees to bumble bees is a growing threat to wild bumble bees (Figuroa *et al.* 2023). A number of RNA viruses that were formerly thought to be specific to honey bees have now been reported to infect bumble bees (Genersch *et al.* 2006;

Meeus *et al.* 2010; Singh *et al.* 2010; Morkeski & Averill 2012; Alger *et al.* 2019). The virulence of many of these RNA viruses in bumble bees has not yet been evaluated. RNA viruses can be transmitted from honey bees to wild bumble bees when they interact at shared flowers (Singh *et al.* 2010; and see section *E. Other Natural or Manmade Factors Affecting its Continued Survival*, subsection 3. *Competition with Honey Bees* for a description of overlap between wild bumble bees and honey bees), where infected pollen grains left by honey bees are collected by bumble bees and brought back to the nest. Bumble bees may also be infected by RNA viruses when commercial bumble bee producers use honey bee pollen to rear bumble bee colonies (if the pollen is not treated with radiation). Alger *et al.* (2019) found higher prevalence and active infection with deformed wing virus in bumble bees collected near infected honey bee apiaries, and Singh *et al.* (2010) found Israeli acute paralysis virus in colonies from one North American commercial bumble bee production facility.

D. The Inadequacy of Existing Regulatory Mechanisms

Existing regulations fail to protect Morrison bumble bee from threats it faces from habitat loss or modification, pathogen infection, and exposure to pesticides.

1. Existing Regulations are Inadequate to Protect this Species' Habitat

Because Morrison bumble bee is not listed under the Endangered Species Act, the habitat essential to its survival is not protected from destruction or adverse modification throughout its range in the US. Morrison bumble bee is listed as a Species of Greatest Conservation Need (SGCN) in California, Colorado, Idaho, and Washington, USFS Sensitive for Region 6, and BLM Sensitive in Oregon and Washington, although these designations do not provide habitat protection in these states.

a. Lack of State Wildlife Agency Regulatory Authority

There are several states within the range of this species that do not have existing state mechanisms or protective measures in place to protect Morrison bumble bee. State wildlife agencies in more than half of the states where Morrison bumble bee occurs, including the states that represent the core of this species' current range, lack regulatory authority to conserve insects. These states include: Nevada, Utah, Colorado, Wyoming, Arizona, New Mexico, and Oregon. Only the wildlife agencies in California, Washington, Idaho, Nebraska, and South Dakota have this authority. Arizona's State Wildlife Action Plan notes "Although the AZGFD [Arizona Game and Fish Department] does not have statutory authority over...insect species, we recognize the crucial role these taxa play in the ecosystems and wildlife communities of the state" (Arizona Game and Fish Department 2022). For Colorado, the State Wildlife Action Plan states that "Colorado is also home to many hundreds of plant and invertebrate animal species that fall outside of CPW's authority" but includes several insect species, including Morrison bumble bee as SGCN, noting "because CPW does not have legislative authority over these species groups, we rely upon our conservation partners to fill this gap" (Colorado Parks and Wildlife Department 2015). The lack of authority of state wildlife agencies to work to conserve imperiled insects represents a major impediment in preventing further decline of pollinators like Morrison bumble bee and emphasizes the need for protecting this species under the federal Endangered Species Act.

2. Existing Regulations are Inadequate to Protect this Species from Disease

Existing regulatory mechanisms fail to protect Morrison bumble bee from pathogens that can cause disease. Although the U.S. Department of Agriculture's Animal and Plant Health Inspection Service (APHIS) prohibits the importation of exotic commercial bumble bee species from other countries, such as the European buff-tailed bumble bee (*Bombus terrestris*), it does not regulate the movement of commercial common eastern bumble bees (*Bombus impatiens*), native to the eastern US, to western states, leaving Morrison bumble bee vulnerable to exposure to diseases from commercial bumble bees, and as such there are no requirements in place that common eastern bumble bees be tested for pathogens prior to being shipped to customers. At the state level, only Oregon Department of Agriculture prohibits the importation of commercial bumble bees, whereas Washington State Department of Agriculture and California Department of Food and Agriculture only prohibit the importation of commercial bumble bees into their states for open field use, allowing them to be imported for greenhouse use. Furthermore, there are no regulations requiring commercial bumble bee producers to irradiate honey bee pollen before feeding it to commercial bumble bees (which is part of the bumble bee rearing process), and research has demonstrated that deformed wing virus, present in honey bee pollen can remain virulent after six months of storage (Singh *et al.* 2010). In January of 2010, the Xerces Society, Dr. Robbin Thorp, Defenders of Wildlife, and the Natural Resources Defense Council petitioned the Secretary of Agriculture and APHIS with a request that they require that all commercial bumble bees transported across state lines be certified as disease-free, citing their authority under the Plant Protection Act, the Honeybee Act and the Animal Health Protection Act (Xerces Society 2010), but APHIS has not reinitiated regulation of the movement of commercial bumble bees. Recently, a group of bee researchers developed a set of recommended components of a clean stock program for commercial bumble bees, in an effort to mitigate disease risk and the risk of pathogen transmission from commercial bumble bees to wild bees (Strange *et al.* 2023). To date, there has been no significant nation-wide action that would prevent the transmission of harmful pathogens from commercial bee colonies to wild bees. Additionally, there are no state or federal regulatory mechanisms that govern the placement of honey bee colonies; the risks associated with this practice include the potential transmission of honey bee diseases (discussed above in section C. *Disease or Predation*) and competition for floral resources (discussed below in section E. *Other natural or manmade factors affecting its continued existence*).

3. Existing Regulations are Inadequate to Protect this Species from Pesticides

Existing regulations regarding the approval of new pesticides and the use of existing pesticides fail to protect bumble bees from exposure to harmful pesticides. The Environmental Protection Agency regulates the approval of new pesticides, and this agency currently does not require that research be done to evaluate the lethal or sublethal effects of insecticides, herbicides or fungicides on bumble bees before those chemicals are approved for use. Additionally, there is limited regulation of adjuvants and co-formulants in pesticides, in spite of evidence that these substances can produce lethal and sublethal effects in bees (Ciarlo *et al.* 2012; Mesnage & Antoniou 2018; Straw & Brown 2022; Straw *et al.* 2021, Wernecke *et al.* 2022). Although acute toxicity to honey bees (*Apis mellifera*) is evaluated in the pesticide approval process, honey bees are not adequate surrogates for bumble bees in this process. Because bumble bees have different behaviors and life histories than honey bees (for example, they have smaller colonies that are

founded each spring, they forage at different times of the day, and they do not process pollen before feeding it to immature bees), they will have different exposure scenarios and may be more vulnerable to certain pesticides than honey bees (Thompson & Hunt 1999; Fischer & Moriarty 2011; Osborne 2012, Stoner 2016, Gradish *et al.* 2019; Schmolke *et al.* 2021). The pesticide regulatory process assesses toxicity on a single chemical basis, which fails to account for interactive or synergistic toxicity in the complex mixtures that bees encounter in the environment, which can include dozens of active ingredients as detected in pollen, nectar, and soils (Raimets *et al.* 2018; Siviter *et al.* 2021; Rondeau *et al.* 2022). Interactive toxicity between multiple chemicals encountered in the field, alongside interactions with other stressors such as poor nutrition and pathogens, can contribute to a variety of lethal and sublethal effects in bees that are not well studied and not accounted for in pesticide risk assessment frameworks (Sanchez-Bayo & Goka, 2014; Siviter *et al.* 2021).

The current regulations governing the use of pesticide-treated seed in the United States fail to provide adequate protection for vulnerable species. A critical shortfall lies in the 'treated article exemption' within the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), which exempts pesticidal seed coatings from regulation and allows for the widespread use of pesticide coatings without adequate environmental assessment and mitigation measures. Neonicotinoid seed coatings account for the majority of neonicotinoid use and more than 10% of the total insecticide applied in the United States (Douglas & Tooker 2015); these insecticides pose concerns to Morrison bumble discussed further in Section *E. Other Natural or Manmade Factors Affecting its Continued Existence* Subsection 2. *Pesticides*.

Lastly, federal pest programs fail to appropriately assess pesticide risk to protected species. The U.S. Department of Agriculture's Animal and Plant Health Inspection Service (APHIS) does not adequately consider harm to endangered species caused by insecticide spraying across millions of acres of western grasslands (Xerces Society & Center for Biological Diversity 2022). These widespread insecticide applications are approved across the range of Morrison bumble bee and have the potential to dramatically impact the species.

E. Other Natural or Manmade Factors Affecting its Continued Existence

1. Global Climate Change

Climate change across the range of Morrison bumble bee threatens this species with extinction. Observed ongoing changes in climate within the range of Morrison bumble bee include decreased precipitation, reduced snowpack and snow extent, snow droughts in mountains, increased aridity, and a multi-decade drought in the species' range that is the most extreme in the past 1000 years (Hicke *et al.* 2022; Williams *et al.* 2022). Additionally, there has been reduced perennial vegetation cover in the Mojave and Sonoran deserts due to increased temperatures, decreased rainfall, and wildfires, while non-native grass invasion and woody plant encroachment has increased in the Sagebrush Steppe (Mirzabaev *et al.* 2022). These vegetation community changes as a result of climate change threaten Morrison bumble bee by altering the abundance and composition of the food plants it relies on. Predicted continued impacts of climate change in the range of Morrison bumble bee include intensified extreme heatwaves, decreased precipitation, increased aridity, more intense and prolonged droughts, increased wildfire severity,

increased snow drought years, and declines in snow extent, duration of snow cover, and snowpack (Hicke *et al.* 2022).

Drought, heatwaves, and increased aridity have been correlated with declines in insect populations across habitats and climates (Forister *et al.* 2010, 2021; Martay *et al.* 2017; Warren *et al.* 2021), and can impact bees directly by causing physiological stress and mortality, and indirectly by altering and degrading the habitat and food plants on which Morrison bumble bee is dependent. Hotter temperatures are correlated with reduced relative abundance of bumble bees worldwide (Cameron & Sadd 2020). These changes may lead to increased pathogen pressure, decreased resource availability (both floral resources and hibernacula), and a decrease in nesting habitat availability due to changes in rodent abundance or distribution (Cameron *et al.* 2011b). Warmer temperatures during winter may also impact bumble bee queens in winter diapause, which in lab studies used more energy reserves under warmer conditions, risking starvation later in their lifecycle (Vesterlund *et al.* 2014).

Climate change directly affects bumble bee populations by altering floral resource availability (Thomson 2016; Ogilvie *et al.* 2017). Variability in climate can lead to phenological asynchrony between bumble bees and the plants they use (Memmott *et al.* 2007; Thomson 2010; Pyke *et al.* 2016), with evidence of mismatch between early blooming plants and their bumble bee pollinators (Kudo *et al.* 2004). Early spring is a critical time for bumble bees since that is the time when the foundresses emerge from hibernation and initiate nests. Since bumble bees can have a flexible diet, they do not require synchrony with a specific plant, but climate change-related variability in floral resources may reduce bumble bee colony performance, especially if resource gaps occur at times that are critical to bumble bee colony success (Hemberger *et al.* 2022). One example is in the Rocky Mountains, where researchers in the last 30 years have observed a mid-season period of low floral resources, a change which can negatively impact pollinators (Aldridge *et al.* 2011). Furthermore, changes in the distributions of plants visited by bumble bees have been correlated with a changing climate (Inouye 2008; Forrest *et al.* 2010). Drought and warming in the west have been linked to shifts in plant distribution and altered phenology (Cayan *et al.* 2001; Kelly & Goulde 2008; Brusca *et al.* 2013; CaraDonna *et al.* 2014; Bloom *et al.* 2022). This variability in floral resource availability is expected to impact Morrison bumble bee populations across its range, contributing to reduced colony performance for the species.

Climate change can also affect the nutritional quality of pollen and nectar. Pumpkin flowers grown under experimental conditions mimicking predicted climate futures were altered in attractiveness and nutritional quality (Hoover *et al.* 2012). Bumble bees foraging on these plants suffered a 22% reduction in survival. Although this study was based on predicted future conditions, similar effects may be occurring presently at levels that are undetected but may still affect bumble bee populations. Flowering plants grown under experimental drought conditions had decreased nectar quality and quantity, and lower protein quality in pollen, and bumble bee colonies housed in these experimental arrays had reduced survival and productivity relative to control conditions (Wilson Rankin *et al.* 2020). In research on the tripartite interactions between soil microbes, flowering plants and pollinators, Keeler *et al.* (2021) predict smaller floral displays, lower quality floral rewards, and decreased foraging efficiency for bees under climate change predictions. Across the West, which is experiencing the most extreme drought in 1,000

years (Hicke *et al.* 2022), Morrison bumble bee populations are expected to be impacted by reduced quality and quantity of floral resources, impacting the species productivity and survival across its range.

A recent study on the potential mechanisms driving the decline of the western bumble bee (*Bombus occidentalis*) found that climate conditions, especially increasing temperature and drought, were most strongly linked to range-wide declines of this species (Janousek *et al.* 2023). Although this study did not include Morrison bumble bee, these results can be expected to be applicable, as Morrison bumble bee has high range overlap with the western bumble bee. In a modeling study, Sirois-Delisle and Kerr (2018) predicted that climate change will contribute to significant range losses for Morrison bumble bee, even if its dispersal rate is as high as the highest recorded rate for any bumble bee species. In a study of North American and European bumble bees, researchers found that there were widespread losses in southern range across species, and a general failure for bumble bees to shift their ranges northward (Kerr *et al.* 2015). A recent study reports the estimate of occupancy for Morrison bumble bee has declined by 51% (Guzman *et al.* 2021). Morrison bumble bee is one of the southern-most occurring bumble bees in North America, and also occurs in some of the most arid habitats. The inability for this species to shift northward as its range becomes hotter and drier will have significant impacts. Fewer observations in the southern-most part of its range, and in the most arid parts of its range may suggest this pattern is contributing to range losses already (Fig. 2). Additionally, populations of Morrison bumble bee that persist in the most arid parts of the range may be forced upwards in elevation, leading to patchy distributions of the bee in non-contiguous high elevation zones, and thus increased genetic isolation.

2. Pesticides

Pesticides are used widely in agricultural, urban, and natural areas and can exert both lethal and sublethal toxic effects on bumble bees. Foraging bumble bees can be poisoned by pesticides when they absorb toxins directly through their exoskeleton, drink contaminated nectar, gather contaminated pollen, or when larvae consume contaminated pollen. Because bumble bees nest in the ground, they may also be exposed to pesticides in the nest or overwintering hibernacula from residues in the soil (Gradish *et al.* 2019, Rondeau *et al.* 2022). Pesticides applied in the spring, when bumble bee queens are foraging and colonies are small, are likely to be most detrimental to bumble bee populations (Goulson *et al.* 2008). Since males and queens are produced at the end of the colony cycle, sublethal doses of pesticides applied at any time during the bumble bee lifecycle that affect brood production or growth and development could have substantial adverse effects on fecundity and subsequent generations (e.g., Laycock *et al.* 2012). Pesticide risk to Morrison bumble bee is dependent on the toxicity of the pesticides and the level and route of exposure. Any application of pesticides can threaten bumble bees, but aerial spraying can be particularly harmful, as pesticide drift from aerial applications can lead to contamination of habitat well outside of the target area. In Europe, the recent declines in bumble bees have been partially attributed to the use of pesticides (Williams 1986; Thompson & Hunt 1999; Rasmont *et al.* 2005).

Morrison bumble bee is threatened by the widespread use of pesticides across its range. Insecticides are designed to kill insects directly, and have the greatest potential for lethal and

sublethal harm to Morrison bumble bee and other species of bees. Herbicides can directly affect bumble bees lethally and sublethally depending on formula, concentration, and application method, and indirectly affect bumble bees by removing floral resources. Research on fungicides show that certain active ingredients and formulations can have direct sublethal effects on bumblebees (Bernauer *et al.* 2015, Straw & Brown 2021). Some fungicides can synergistically increase the toxicity of certain insecticides, including in mixtures commonly found in flowering plants and soils in agricultural landscapes (Sgolastra *et al.* 2017; Raimets *et al.* 2018; Rondeau *et al.* 2022). Fungicide contamination can also indirectly affect bumble bees by altering the fungal microbiota in their nectar and pollen provision. Bumble bees are exposed to multiple classes of pesticides, and the synergistic effects may present additional risks as well.

a. Insecticides

Insecticide exposure poses threats to bumble bees at an individual, colony, and population level. Many insecticides used in agricultural, urban, forestry, rangeland, and other applications can be toxic to bumble bee larvae and/or adults, causing harm to survival and reproduction via impacts to feeding and navigation, growth and development, brood production, and other endpoints.

Across its range, Morrison bumble bee could encounter a wide range of different insecticides in yards and parks, crop fields, rangeland, and forests. On public lands in the west, a range of insecticides have been used to control timber pests such as tussock moth, gypsy moth, bark beetles, and spruce budworm. Broad-spectrum insecticides, including carbaryl, and diflubenzuron, have been and continue to be used across millions of acres of western rangeland to control grasshoppers and Mormon crickets.

Many types of insecticides have toxicity to larval or adult bees, including effects on behavior, physiology, and reproduction that are not well captured by the standardized toxicity testing required for pesticide registration. Organophosphate, carbamate, and pyrethroid insecticides - all of which target the insect nervous system and thus have broad activity against many groups of insects - have been associated with bee poisonings in food crops (Johansen 1977; Kearns *et al.* 1998). Bumble bee deaths have been reported after application of a pyrethroid insecticide to oilseed rape (Thompson 2001). In New Brunswick, Canada, bumble bee populations declined drastically when exposed to the organophosphate insecticide fenitrothion, resulting in reduced pollination of nearby commercial blueberries and other plants such as orchids and clovers (Kevan & Plowright 1995; Kevan 1975; Plowright *et al.* 1978, 1980).

Effects of insecticide exposure are not always well captured by studies or incidents involving bee mortality; bees exposed to insecticides at different life stages may exhibit a variety of symptoms that affect behavior, growth/development, and reproduction even much later in their life cycles. Spinosad, a commonly used insect neurotoxin, has resulted in reduced worker foraging efficiency when bumble bee larvae are fed with pollen containing this pesticide (Morandin *et al.* 2005). Skyrn (2011) observed significant queen mortality when exposed to low doses of spinosad. In an examination of the effect of chitin synthesis inhibitors on *Bombus*, Mommaerts *et al.* (2006) found that even at very low concentrations, diflubenzuron and teflubenzuron increased egg mortality and removal of larvae. Novel sulfoximine-based insecticides, proposed as a replacement for neonicotinoid-resistant crop pests, had severe sublethal effects when fed to

bumble bees at field relevant levels, including reduction of male and queen production (Siviter *et al.* 2018). Insecticide exposure that does not reach lethal levels can still affect individual health in key ways that could reduce fitness and population health over time.

Within the range of Morrison bumble bee, some of the most common agricultural land uses include wheat, alfalfa and cotton production (USDA 2022) and rangeland. Wheat and cotton crops contribute to a large share of the total national use of imidacloprid (Fig. 12), as well as other insecticides (Wieben 2021). Alfalfa production, another main agricultural land use within the species range, is a major use of a number of organophosphate insecticides (Wieben 2021). The widespread use of insecticides across the range of Morrison bumble bee threatens this species with extinction.

Rangeland Insecticide Applications

Across the West, insecticides have been administered aerially by the U.S. Department of Agriculture's Animal and Plant Health Inspection Service (APHIS) to prevent grasshoppers and Mormon Crickets from competing with livestock for forage (Fig. 10). APHIS does not provide the public with final spray maps, although advertised spray contract solicitations are publicly available. FOIA requests to APHIS to learn the locations of final treatment areas were submitted in 2021 but in most cases, final maps were withheld. On average between 2006 and 2017, APHIS solicited treatments across 500,000 acres each year (USDA-APHIS 2020).




These treatments include carbaryl and diflubenzuron, which are pesticides that contribute to a variety of lethal and sublethal effects for bumble bees, depending on the concentration. In APHIS sprays, carbaryl has a modeled application rate of deposition between 37,300 and 55,000 ppb (USDA-APHIS 2019). This range may present acute oral toxicity for bumble bees (Sanchez-Bayo and Goka 2014). Diflubenzuron has a modeled application rate of deposition on host plants is between 980 and 1,760 ppb, and it can take weeks for the pesticide to degrade (USDA-APHIS 2019). Diflubenzuron is an insect growth regulator impacting larval insects, and does not present much risk to adult bees. However, if worker bees collect pollen that has been contaminated by diflubenzuron to feed to larval bees, larval mortality can occur (Tasei 2001). Mommaerts *et al.* (2006), investigating the effect of diflubenzuron on *B. terrestris* brood production, found drastic reproductive failure with larval bees exposed to diflubenzuron in pollen at levels of 288 ppb and above, well below the deposition concentrations expected from APHIS applications. When diflubenzuron was experimentally fed to bumble bees at a concentration of 1000 ppb, nest building activities were dramatically reduced (Camp *et al.* 2020), suggesting that the modeled application rate for rangeland treatments may reduce nest productivity if pollen and nectar are contaminated by the pesticide.

Thousands of acres of rangeland within the range of Morrison bumble bee have been treated with carbaryl and diflubenzuron under this program in the last several years, with additional treatments planned. For example, in 2019, over 30,000 acres of range land were treated with carbaryl in southwest Idaho, within the range of Morrison bumble bee. In early June of 2023, Oregon solicited bids for diflubenzuron treatments across a block of 120,000 acres near McDermitt, Oregon, as well as treatments across 31,100 acres of rangeland in Harney County, Oregon. Just across the border in Nevada, also in 2023, APHIS advertised diflubenzuron

treatments across a footprint of 75,000 acres in north and central Nevada. All of these treatments occurred within the range of Morrison bumble bee. This practice has the potential to cause population impacts, including potential range wide reductions in Morrison bumble bee populations.

Planned Insecticide Sprays for Federal Grasshopper Suppression, 2008 - 2021

Treatment areas likely an underestimate due to lack of final treatment maps from APHIS

-  APHIS Treatment Areas
-  1815 - 2010 EOO
-  2011 - 2021 EOO

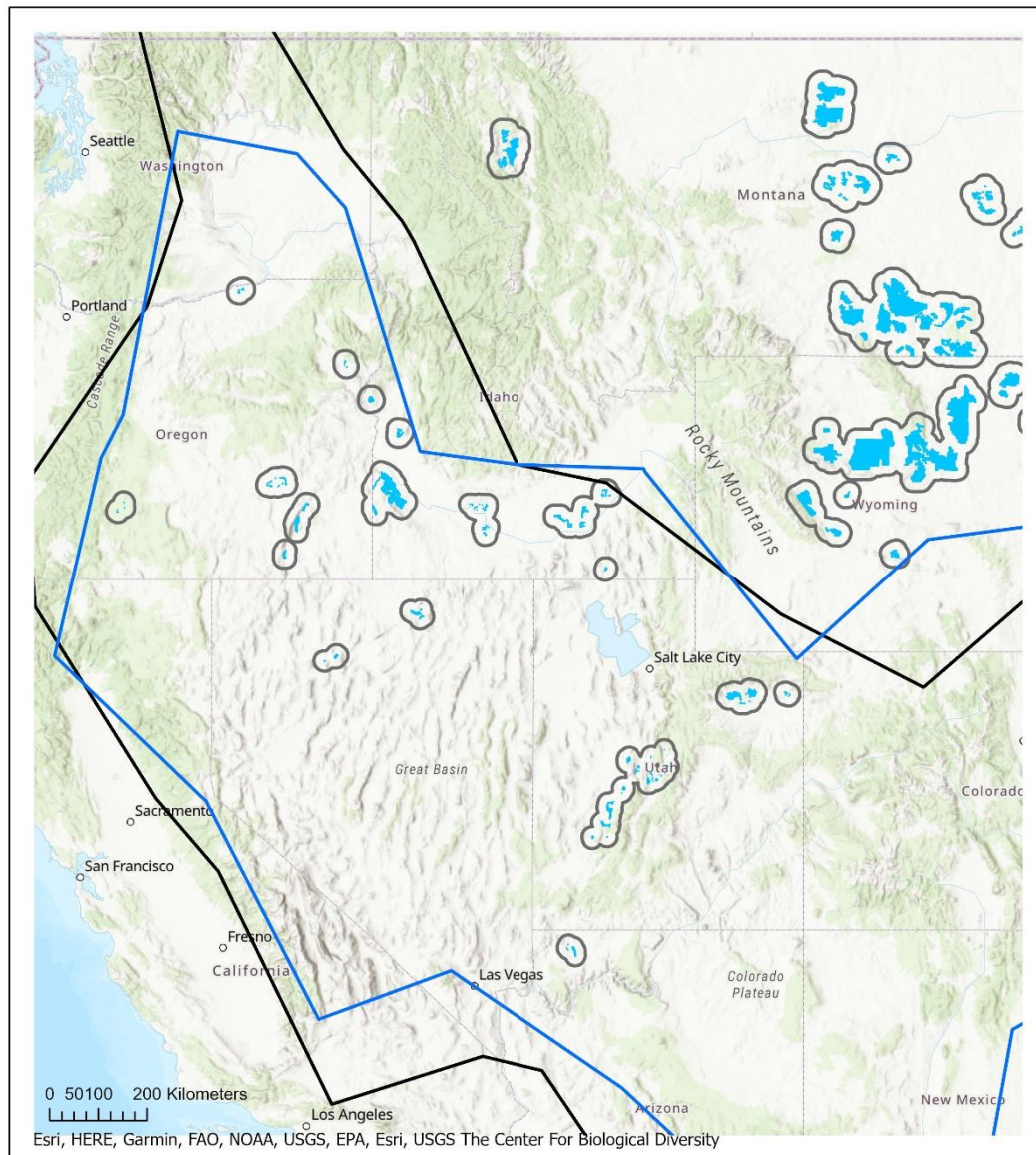


Figure 10. U.S. Department of Agriculture's Animal and Plant Health Inspection Service (APHIS) planned treatment areas from 2008 to 2021 and the range of Morrison bumble bee. Final treatment maps are unavailable despite requests to APHIS. Treatment data displayed here are likely an underestimate of actual acreage treated.

Neonicotinoids

Neonicotinoids are a class of systemic insecticides that are used widely to combat insect pests of agricultural crops, turfgrass, gardens and pets (Cox 2001). While many insecticides pose risk to bees, neonicotinoids are often highlighted as a particular threat due to their high toxicity to bees, multiple exposure routes, persistence, and widespread use. They are the most widely used insecticide class globally, mostly used as seed dressings, where typically only 5% of the active ingredient is taken up by the plant (Sur and Stork 2003; Wood and Goulson 2017), making the remaining chemical available for non-target organisms. As systemic, water-soluble insecticides, these chemicals can be taken up by plants and transported into pollen, nectar, and guttation droplets, where they can be consumed or collected by foraging adult bees. Combined with high oral toxicity to bees, this potential route of exposure is particularly risky. Despite being identified by the Environmental Protection Agency as likely harming three quarters of endangered species (EPA 2021a, 2021b, 2021c), and being banned and restricted in Europe and Canada, respectively, there are limited U.S. regulations on neonicotinoids, and their use became widespread and increased dramatically after their introduction (Fig. 11). The large-scale increase after 2003 was driven by use as crop seed coatings, which are currently not regulated as pesticides (Douglas & Tooker 2015). Colla & Packer (2008) suggested that neonicotinoids may be one of the factors responsible for the decline of eastern bumble bees since the use of this class of insecticides began in the U.S. in the early 1990s, shortly before the bumble bee declines were documented. A recent study on mechanisms driving the decline of the western bumble bee (*Bombus occidentalis*) found that neonicotinoid use was associated with range wide declines in occupancy for this species (Janousek *et al.* 2023). Although Morrison bumble bee was not included in this study, neonicotinoid exposure and impact can be expected to be similar, as there is high range overlap and similar life histories for these two species.

A study exposing bumble bees to field-realistic levels of the neonicotinoid imidacloprid found an 85% reduction in the production of new queens and significantly reduced colony growth rates compared to control colonies (Whitehorn *et al.* 2012). The authors suggest that neonicotinoids “may be having a considerable negative impact on wild bumble bee populations across the developed world” (Whitehorn *et al.* 2012). Another study of bumble bees exposed to varying levels of imidacloprid found a dose-dependent decline in fecundity and documented that field realistic levels of this pesticide were capable of reducing brood production by one-third (Laycock *et al.* 2012). The authors speculate that this decline in fecundity is a result of individual bumble bees failing to feed, which raises concerns about the impact of this pesticide on wild bumble bees (Laycock *et al.* 2012). Other toxicity studies have demonstrated that contact exposure of imidacloprid and clothianidin to bumble bees can be very harmful (Marletto *et al.* 2003; Gradish *et al.* 2010; Scott-Dupree *et al.* 2009), and an acute oral dose of imidacloprid is highly toxic to bumble bees (Marletto *et al.* 2003, In Hopwood *et al.* 2016). Mommaerts *et al.* (2010) found that chronic exposure of three neonicotinoids to bumble bees was dose dependent, and another study by Incerti *et al.* (2003) found that one third of bumble bees in a flight cage exposed to blooming cucumbers treated with a “field dose” of imidacloprid died within 48 hours (In Hopwood *et al.* 2016). A study by Gill *et al.* (2012) examining the effects of the combined exposure of bumble bees to field realistic levels of two pesticides – an imidacloprid and a pyrethroid – found that foraging behavior was impaired, worker mortality increased, and both brood development and colony success were significantly reduced.

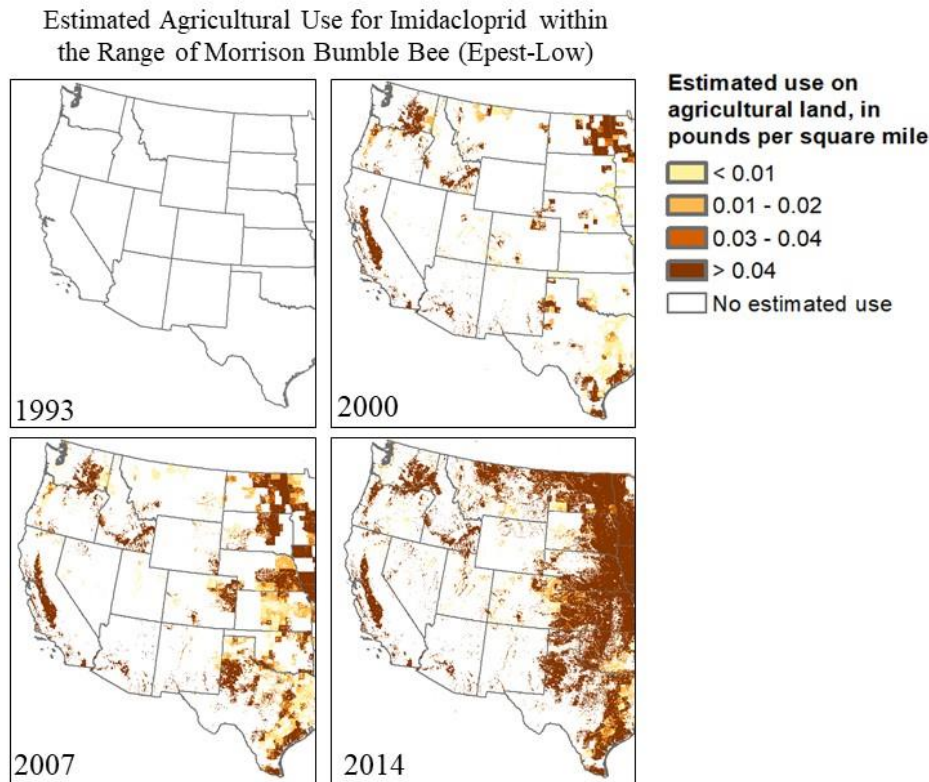


Figure 11. Increasing use of neonicotinoid insecticide imidacloprid in the United States in 1993, 2000, 2007, and 2014 (most recent year that includes seed coatings in the total use). Data from USGS Pesticide National Synthesis project, available at: https://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=2013&map=IMIDACLOPRID&hilo=L

Other studies have also documented sublethal effects of neonicotinoids on bumble bees, including: reduced foraging ability (Morandin & Winston 2003); reduced drone production and longer foraging times (Mommaerts *et al.* 2010); reduced foraging activity, reduced food storage and reduced adult survival (Al-Jabr 1999); impaired crop pollination services (Stanley *et al.* 2015); and lower worker survival and reduced brood production (Tasei *et al.* 2000). In a field study using neonicotinoid treated seeds, researchers found that bumble bee colony growth was reduced in treatment sites versus the control (Rundlöf *et al.* 2015).

Neonicotinoids are widely used on agricultural crops that are attractive to pollinators, as well as on horticultural plants and lawns in urban and suburban areas, within Morrison bumble bee's range. Thus, this class of insecticide is likely to affect Morrison bumble bee. Of particular concern is a finding in a review of the impact of neonicotinoid pesticides on pollinating insects which found that products approved for home and garden use may be applied to ornamental and landscape plants and turf grass at significantly higher concentrations (potentially 32 times higher) than the allowable concentration of the same products applied on agricultural crops (Hopwood *et al.* 2016).

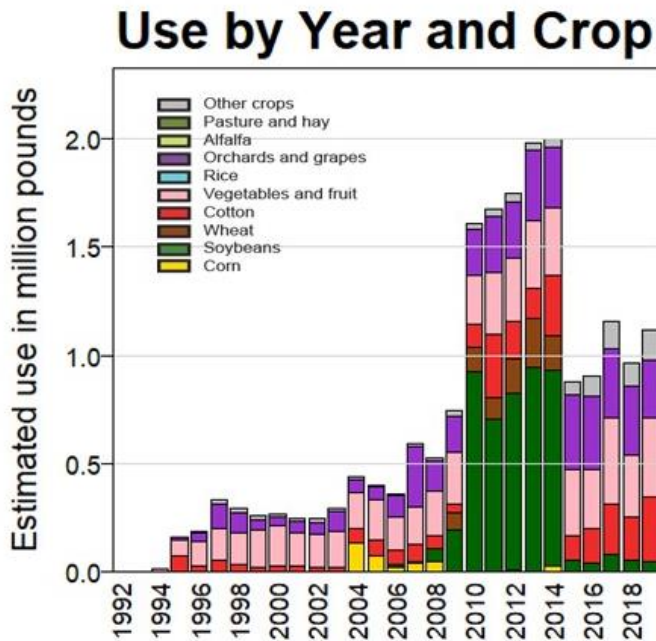


Figure 12. Imidicloprid use by crop from 1992 - 2018 in the United States. Alfalfa, wheat, cotton, corn, and pasture/hay are among the top crops grown within the range of Morrison bumble bee. Note that in 2015, use in seed treatments was no longer tracked, leading to perceived declines in use. Use is likely grossly underreported following 2014. Data from the USGS Pesticide National Synthesis Project:

https://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=2013&map=IMIDACLOPRID&hilo=L

b. Herbicides

Herbicides can be a valuable tool for the control of invasive weed species. However, the use of broad-spectrum herbicides to control weeds can indirectly harm pollinators by decreasing the habitat quality for pollinators through removal of flowers that provide pollen and nectar for existing populations (Williams 1986; Pleasants & Oberhauser 2013).

Just as pollinators can influence the plant community, changes in vegetation can have an impact on pollinators (Kearns & Inouye 1997). The broadcast application of a non-selective herbicide can indiscriminately reduce floral resources, host plants, and nesting habitat (Smallidge & Leopold 1997). Morrison bumble bee requires consistent sources of nectar, pollen, and nesting material during times adults are active, typically from mid-March to mid-October. Morrison bumble bee has been shown to be sensitive to changes in floral resources (Jackson *et al.* 2022). The reduction in resources caused by non-selective herbicide use could cause a decline in bumble bee reproductive success and/or survival rates. Kevan (1999) found that herbicides reduced Asteraceae and Lamiaceae flowers in France, contributing to a decline in bumble bee populations. This study also found that herbicide applications have reduced the reproductive success of blueberry pollinators by limiting alternative food sources that can sustain the insects when the blueberries are not in bloom. Kearns *et al.* (1998) state “herbicide use affects pollinators by reducing the availability of nectar plants. In some circumstances, herbicides

appear to have a greater effect than insecticides on wild bee populations... Some of these bee populations show massive declines due to the lack of suitable nesting sites and alternative food plants.”

The use of the herbicide glyphosate (Roundup™) has dramatically increased with the widespread planting of genetically modified glyphosate-tolerant corn and soybeans, which were introduced in 1998 and 1996, respectively (Fig. 13; Pleasants & Oberhauser 2013). Increased use of glyphosate in agricultural areas has likely led to the reduced availability of wildflowers in field margins – which otherwise would have been an important resource for Morrison bumble bee. Pleasants and Oberhauser (2013) estimate a 58% reduction in milkweed, an important nectar plant for bumble bees, in the Midwestern U.S. from 1999-2010, and suggest that this decline is due to the increased use of glyphosate in corn and soybean fields.

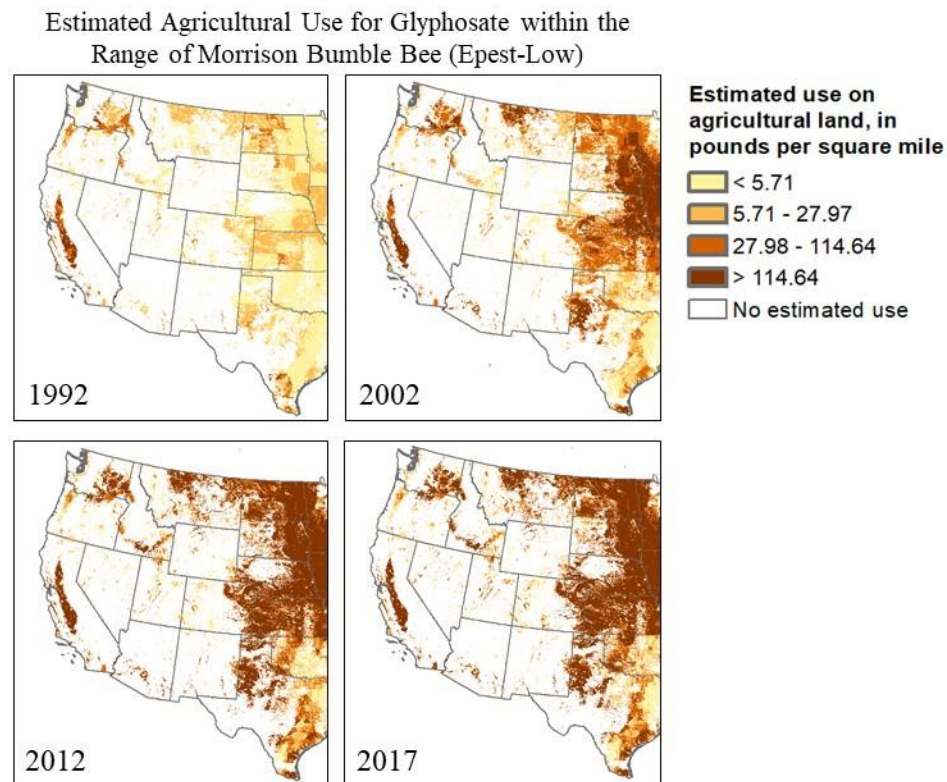


Figure 13. Increasing glyphosate use in the United States in 1992, 2002, 2012, and 2017 (the most recent year complete data are available). Data from the USGS Pesticide National Synthesis Project, available from: https://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=2017&map=GLYPHOSATE&hilo=L

There is growing evidence of direct effects of herbicide on bees. Because bumble bees will forage indiscriminately on herbicide-treated plants before they die, they can be directly exposed (Thompson *et al.* 2022). Straw *et al.* (2021) report that in an experiment to test lethality of Roundup™ formulations on bumble bees, a ready to use glyphosate product, diluted formulations, and a glyphosate-free product all contributed to significantly higher mortality than their control. This suggests that co-formulants, which are often non-disclosed trade secret ingredients, may be responsible for mortality in bumble bees. In another study, Weidenmüller *et*

al. (2022) examined the thermoregulation ability of bumble bees under scarce food conditions when exposed to glyphosate. They found that the ability of glyphosate exposed colonies to maintain necessary brood temperatures was decreased by 25%. This study replicates a realistic field condition where herbicide application both indirectly affects bumble bees by reducing their forage, and through direct exposure, impacting their optimal in-nest temperatures.

Within the range of Morrison bumble bee, major crops alfalfa, wheat, and cotton are treated with considerable quantities of herbicides including glyphosate, hexazinone, and metribuzin (Wieben 2021). Herbicide use directly contributes to the extinction risk of Morrison bumble bee. Its use is widespread across the range of the species, reduces the available forage in agricultural, suburban, and some wildland areas, and contributes to mortality and sublethal effects for bumble bees.

c. Fungicides

Fungicides are widely used across the range of Morrison bumble bee (Fig. 14). They are typically the most abundant pesticide found in bumble bee and honey bee hive material (Sanchez-Bayo & Goka 2014; Botias *et al.* 2017). Although most research on pesticides focuses on the direct impact of insecticides on bees, there is growing research on the lethal and sublethal effects of fungicide use. One study that exposed bumble bees to field relevant levels of the fungicide chlorothalonil showed reduced production of workers, lower bee biomass, and smaller queens, contributing to significant colony losses overall (Bernauer *et al.* 2015). Chlorothalonil is one of the most common fungicides in the US and is used on a variety of crops in the Morrison bumble bee range, including heavy use in potatoes and tomatoes, as well as on golf courses and lawns. A recent study assessing the presence of pesticides on milkweed (*Asclepias* spp.) across land use types found chlorothalonil in 91% of milkweed leaf samples, at times in concentrations above a lepidopteran LD₅₀ (Halsch *et al.* 2020). Several studies have found direct impacts of exposure to strobilurin fungicides on non-*Apis* bees. A semi-field experiment exposing bumble bees to the fungicide Amistar (azoxystrobin) found that bumble bee foraging performance was negatively affected (Tamburini *et al.* 2021).

Some fungicides also pose risk to *Bombus* by synergizing the toxicity of certain insecticides. Demethylation inhibitor (DMI) fungicides are a subset of sterol biosynthesis inhibitor (SBI) fungicides that specifically interact with a demethylation enzyme in the sterol biosynthesis pathway, and can synergize the toxicity of a variety of insecticide classes to bees by inhibiting detoxification, including pyrethroids, neonicotinoids, flupyradifurone, diflubenzuron, and novaluron (UC IPM 2022). Bees exposed to a mixture of these fungicides and insecticides experience significantly increased toxicity relative to bees exposed only to the insecticide.

Estimated Agricultural Use for Chlorothalonil within the Range of Morrison Bumble Bee (Epest-Low)

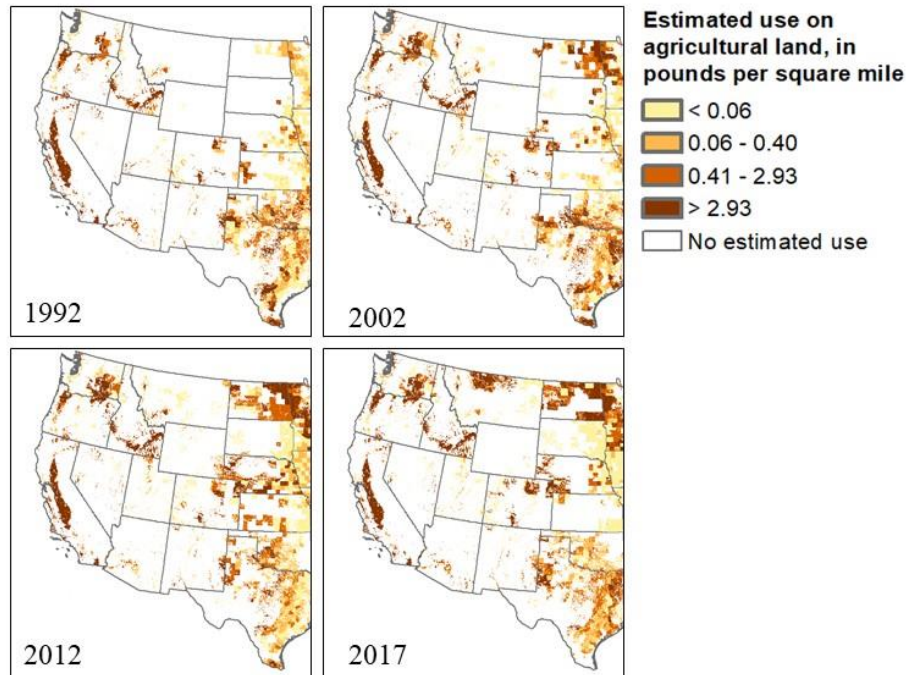


Figure 14. Increasing chlorothalonil use in the United States in 1992, 2002, 2012, and 2017 (the most recent year complete data are available). Data from the USGS Pesticide National Synthesis Project, available from: <https://water.usgs.gov/nawqa/pnsp/us>

Formulated fungicide products often have greater toxicity to bees than the technical grade active ingredients, as co-formulants can synergize the toxicity of active ingredients and/or have direct impacts on insects. Bumble bees exposed to the fungicide co-formulant alcohol exothylate in a lab setting showed 23% mortality, a 45% reduction in sucrose consumption, and a marked increase in gut damage (Straw & Brown 2021).

3. Population Dynamics and Structure

a. Declining North American Bumble Bees have lost Genetic Diversity

Research indicates that populations of other bumble bee species whose ranges overlap with Morrison bumble bee – the declining western bumble bee (*Bombus occidentalis*) and American bumble bee (*B. pensylvanicus*) – have lower genetic diversity compared to populations of co-occurring stable species (Cameron *et al.* 2011a; Lozier *et al.* 2011). Another genetic study of the declining American bumble bee found an increase in this species' population structure, suggesting that the American bumble bee has become increasingly isolated over the past four decades (Lozier & Cameron 2009). It is reasonable to expect that the Morrison bumble bee may have suffered a similar loss of genetic diversity and increase in population structure, although this has not been examined.

b. Impacts of Genetic Factors on Bumble Bees

Loss of genetic diversity, which is frequently the result of inbreeding or random drift, can pose significant threats to small, isolated populations of bumble bees (Whitehorn *et al.* 2009). A loss of genetic diversity limits the ability of a population to adapt and reproduce when the environment changes and can lead to an increased susceptibility to pathogens (Altizer *et al.* 2003). Bumble bees have a single locus complementary sex determination system, meaning that the sex of an individual bee is determined by the number of unique alleles at the sex-determining locus (van Wilgenburg 2006). Normally this sex determination comes through a haplodiploid genetic structure in which female bees are diploids and are produced from fertilized eggs with two different copies of an allele at the sex-determining locus. Most male bees are haploid, and they are produced from unfertilized eggs (with only a single copy of an allele at the sex-determining locus). However, when closely related bumble bees mate, the offspring can have two copies of the exact same allele (or be homozygous) at the sex-determining locus, which causes a diploid male to be produced instead of a diploid female. These diploid males may have reduced viability or may be sterile (van Wilgenburg 2006). When diploid males are able to mate, they produce sterile triploid offspring, which has been found to be negatively correlated with surrogates of bumble bee population size (Darvill *et al.* 2012). Diploid males are produced at the expense of female workers and new queens, and the production of diploid males can reduce colony fitness (including slower growth rates, lower survival, and colonies that produce fewer offspring) in bumble bees (Whitehorn *et al.* 2009). It has been suggested that diploid male production in inbred populations substantially increases the risk of extinction in bumble bee populations compared to other animal taxa (Zayed & Packer 2005). Inbreeding and loss of genetic diversity can increase parasite prevalence in populations and parasite susceptibility in individuals (Frankham *et al.* 2010 in Whitehorn *et al.* 2011). Populations of bumble bees with low genetic diversity have been found to have a higher prevalence of pathogens (Whitehorn *et al.* 2011; Cameron *et al.* 2011a), suggesting that as populations lose genetic diversity, the impact of parasitism will increase and threatened populations will become more prone to extinction.

4. Competition with Commercial Honey Bees

Honey bees (*Apis mellifera*) were introduced to eastern North America in the early 1620s. They compete with bumble bees for floral resources. A single honey bee hive annually consumes between 20-130 lbs. of pollen and 45-330 lbs. of honey – representing the removal of between 120-900 lbs. of nectar from the surrounding environment, per year (Goulson 2003, and references therein). Cane and Tepedino (2016) estimate that a 40-hive apiary pastured on wildlands collects as much pollen and nectar in a three-month window as is required to produce four million wild, native bees. Depending on the density of hive placement, year-to-year conditions of the site, or degree to which flowering plants may have been locally reduced by other types of degradation, this can represent a significant portion of the resources available for any nectivore.

In addition to honey bee presence reducing the availability of food resources for native bees (Paton 1996; Wills *et al.* 1990; Horskins & Turner 1999; Dafni & Shmida 1996; Dupont *et al.* 2004; Cane & Tepedino 2016, Page & Williams 2023), honey bee presence changes the competitive dynamics of bee communities. Certain bees may be more at risk of competitive

effects with honey bees due to high niche overlap (Ropars *et al.* 2019; Prendergast *et al.* 2021, Page & Williams 2023). Because bumble bees share some traits with honey bees, most notably their larger size, wide dietary preference, and opportunistic foraging behavior, their high niche overlap makes them especially susceptible to competitive effects (Ropars *et al.* 2019; Weaver *et al.* 2022). Through competitive exclusion by honey bees, native bees, are forced to switch to other, less abundant, and less rewarding plant species (Wratt 1968; Eickwort & Ginsberg 1980; Pleasants 1981; Ginsberg 1983; Paton 1993; 1996; Buchmann 1996; Horskins & Turner 1999; Dupont *et al.* 2004; Thomson 2004; Walther-Hellwig *et al.* 2006; Tepedino *et al.* 2007; Roubik 2009; Shavit *et al.* 2009; Hudewenz & Klein 2013; Rogers *et al.* 2013; but see Butz-Huryn 1997; Steffan-Dewenter & Tschardt 2000; Minckley *et al.* 2003) – but none of these studies have addressed population level effects on native bees.

The long-term implications of this shift in resource use are not entirely clear, although there is a growing body of research on bumble bees that demonstrates negative competitive effects of honey bees on bumble bees, including lower reproductive success, smaller body size, and changes in bumble bee foraging behavior—and most notably, a reduction in pollen gathering (Evans 2001; Goulson *et al.* 2003; Thomson 2004, 2006; Walther-Hellwig *et al.* 2006; Goulson & Sparrow 2009; Ropars *et al.* 2019).

As of 2020, there were at least seven permits for over 964 honey bee hives active on US Forest Service land in Arizona and Utah (Grand Canyon Trust *et al.* 2020), contributing significant competitive effects in habitat occupied by Morrison bumble bee. Apiaries have been placed on BLM and Forest Service land across the west without compliance with the National Environmental Policy Act, which requires agencies to assess the environmental impacts of their decisions. High density placement of honey bee hives in areas where Morrison bumble bee occurs contributes to considerable competition and could threaten the continued survival of this species.

VIII. CONCLUSION

Bumble bees are essential pollinators of crops and wildflowers in agricultural, urban and natural ecosystems. They play an important role in the reproduction of tomato, blueberry, pepper, cranberry, clover, and many other crops. This charismatic bee also contributes to the pollination of countless native plant species, which are essential to a functional ecosystem.

Although the Morrison bumble bee was historically distributed throughout the Western US, recent analyses show that its relative abundance has declined by 74%, and its persistence has declined by 66%. Despite increasing efforts to monitor bumble bees, and efforts to specifically locate this species, it has not been detected in 13.9% of its historic range in the last decade, with a considerable reduction in observations throughout its entire range. Other models show this species has declined in occupancy by 51%.

Morrison bumble bee faces a wide variety of threats. Habitat loss and degradation through overgrazing, invasive species introduction, habitat conversion, altered fire regime, the development of mines, and the conversion of wild lands to urban and agricultural lands threaten this species with extinction. Increased pressure from pesticide exposure, pathogens and

competition with honey bees contribute to the risk for Morrison bumble bee. Impacts from climate change, especially increased temperature and drought, alter the habitat required by this species, and present physiological impacts and multiple life stages. This species is not expected to shift its range northward fast enough to account for climate change impacts, which makes the need to conserve it all the greater. When considered individually, each of these factors pose a significant potential threat to Morrison bumble bee. However, when considered together, they present a daunting case for the recovery of this animal. In addition, existing regulations are inadequate to protect Morrison bumble bee from disease and pesticides, and to protect its habitat. Without intervention, this bee is likely to go extinct. Morrison bumble bee should be listed as an Endangered Species under the U.S. Endangered Species Act to prevent global extinction of this once common bumble bee.

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X. PERSONAL COMMUNICATION

A. Barth, pers. comm with S. Killingsworth Dec. 2022

J. Heron and C. Sheffield, British Columbia Community Bumble Bee Project, BC Ministry of Water, Land and Resource Stewardship, Surrey, BC, pers. comm. with S. Killingsworth July 2023.

J. Schmidt, pers. comm. with S. Killingsworth, Sept. 2022

J. Strange, pers. comm. with S. Killingsworth, Sept. 2022

XI. APPENDIX

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Museum National d'Histoire Naturelle
Museum of Northern Arizona
Museum of Southwestern Biology, Division of Arthropods
National Museum of Natural History, Washington, DC
Natural History Museum of Rotterdam, Netherlands
Natural History Museum of Utah
Natural History Museum, London
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University of Arizona, Insect Collection
University of Arkansas
University of Bern
University of Calgary
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University of California, Riverside Entomology Museum
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University of Central Florida Collection of Arthropods
University of Central Oklahoma
University of Colorado Museum of Natural History, Boulder
University of Colorado, Boulder, Museum of Natural History
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University of Delaware Insect Collection
University of Georgia
University of Guelph
University of Guelph, Centre for Biodiversity Genomics
University of Helsinki
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University of Kansas Snow Entomological Collections
University of Kentucky Insect Collection
University of Massachusetts, Amherst
University of Michigan
University of Minnesota Duluth Insect Collection
University of Mississippi
University of Nevada Reno
University of New Hampshire, Durham
University of Northern Iowa - Wen Research Collection
University of Oklahoma
University of Oklahoma, Sam Noble Museum Department of Recent Invertebrates
University of Prince Edward Island Museum
University of Tartu Natural History Museum, Estonia
University of Texas Insect Collection
University of Texas, El Paso, Biodiversity Collections Entomology Collection
University of Toronto
University of Vermont, Zadock Thompson Invertebrates Collection
University of Wisconsin, Madison
University of Wyoming
University of Wyoming Dillon Lab Insect Collection
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