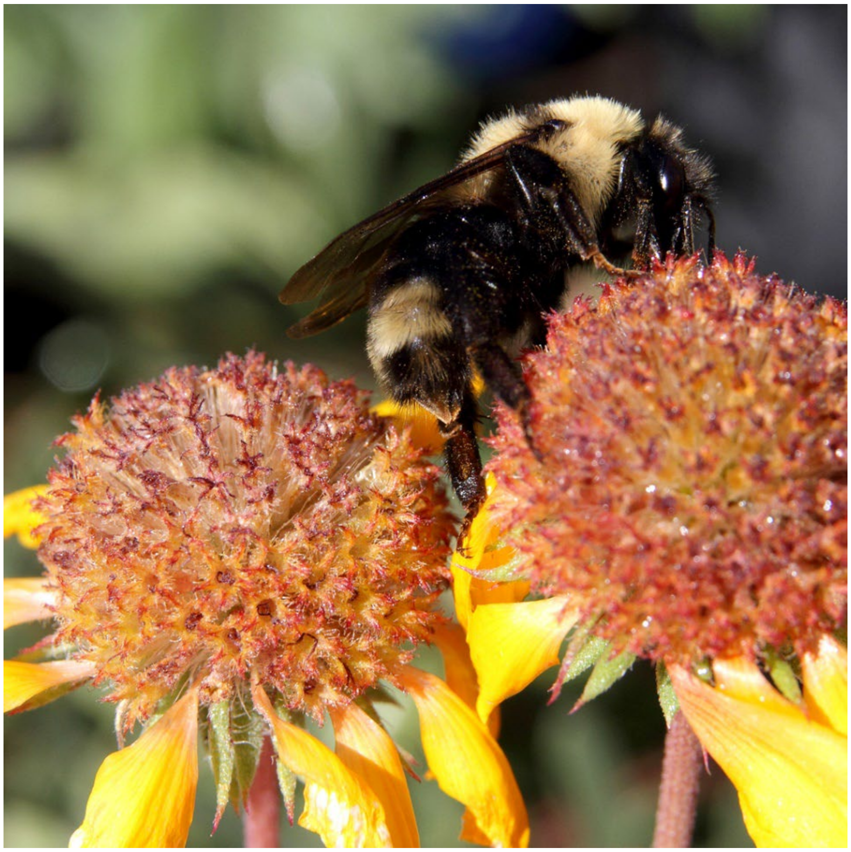
Suckley's Cuckoo Bumble Bee
(*Bombus suckleyi*)
Species Status Assessment
Version 1.0



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Alaska Region

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U.S. Fish and Wildlife Service. 2024. Species status assessment report for the Suckley's Cuckoo Bumble Bee (*Bombus suckleyi*), Version 1.0. August 2024. Alaska Region. 131 pp.

Executive Summary

We conducted a Species Status Assessment (SSA) to assess the viability of Suckley's cuckoo bumble bee (*Bombus suckleyi*). We define viability as the ability of a species to maintain populations in the wild over a biologically meaningful timeframe. We characterized the viability of Suckley's cuckoo bumble bee over time by assessing its ability to withstand environmental, genetic, and demographic stochasticity and disturbances (resiliency), catastrophic events (redundancy), and novel changes in its biological and physical environment (representation). Suckley's cuckoo bumble bee viability is fostered by maintaining the demographic and genetic health of its populations and preserving potential connections between populations.

Biology and habitat requirements: Suckley's cuckoo bumble bee is an obligate social parasite of social bumble bees in the genus *Bombus*. Cuckoo bumble bee females emerge from hibernation in the spring and usurp the nest of a suitable host colony, where host workers provision their young. Suckley's cuckoo bumble bee is described as a semi-specialist parasite and is confirmed to usurp nests of Western bumble bees (*Bombus occidentalis*) and Nevada bumble bees (*Bombus nevadensis),* with other potential hosts in subgenus *Bombus* throughout the extent of its range. The species has a broad distribution across North America, stretching from the Yukon down to Arizona and as far east as Newfoundland. The species has been collected in various habitat types from 2 to 3,200 meters (6 to 10,500 feet) in elevation.

Suckley's cuckoo bumble bees cannot successfully reproduce without the availability of suitable host colonies. Additionally, they require a diversity of native floral resources (pollen and nectar) for nutrition. Little is known about bumble bee overwintering sites, however they are an important habitat need. Bumble bees require temperatures to be within a suitable range throughout their lifecycle, however this number appears to be highly variable both across and within bumble bee species.

Methods: We assessed Suckley's cuckoo bumble bee viability by evaluating the historical and current condition of its populations and identifying the primary influences leading to the species current condition. Climatic influences and land use and management are the key factors driving a multitude of stressors for Suckley's cuckoo bumble bee. We used a published multi-species occupancy model incorporating bumble bee data from across North America to understand trends in both Suckley's cuckoo bumble bee and host species occupancy. Our dataset included 2,317 occurrence records of Suckley's cuckoo bumble bee. We projected future changes in occupancy using Representative Concentration Pathway (RCP) climate projections under lower emissions scenario RCP4.5 without trend momentum (Scenario 1) and higher emissions scenario RCP8.5 with trend momentum (Scenario 2). Trend momentum accounts for changes in occupancy due to historical drivers, for which we do not have data, by modeling temporal changes in occupancy. Key uncertainties and the viability implications are described at the end of the SSA report.

Results: Based on results from a published occupancy model updated for this SSA, we estimate Suckley's cuckoo bumble bee has exhibited a strong, statistically significant temporal decline in probability of occupancy, range wide. This equated to a roughly 85 percent decline in probability of occupancy between 1900 and 2020. From 1900–1960, the median probability of occupancy of Suckley's cuckoo bumble bee range wide was 0.65. By the current time period (2000–2020) all analytical units are estimated to have lower probability of occupancy, with median probability of occupancy less than 0.16 for all analytical units and 0.13 range wide. Suckley's cuckoo bumble bee has not been observed in the current time period (post 2000) in three of four eastern analytical units and three of eleven western analytical units. We considered all analytical units with no current observations of Suckley's cuckoo bumble bee to be quasi-extripated, except for the Taiga Plains where sampling has been more limited. The species has not been observed in the contiguous United States since 2016, despite an expanding survey effort throughout the range through various programs such as the Pacific Northwest Bumble Bee Atlas. Additional analyses of host species occupancy indicates that four of six host species exhibited statistically significant temporal declines in probability of occupancy range wide.

The probability of occupancy of Suckley's cuckoo bumble bee is projected to decline into the future across the range under both scenarios, and in all analytical units, with more marked declines under Scenario 2. In the near term (2020–2040) median probability of occupancy is expected to be less than 0.11 under (Scenario 1) and less than 0.05 under (Scenario 2) across all analytical units. In Scenario 1, median probability of occupancy for all analytical units is estimated to be less than 0.12 beyond mid-century (2040–2060). Under Scenario 1, projected probability of occupancy in northern analytical units levels off relative to historical declines, indicating these northern areas may not be as impacted by climate change in the long term, if in fact declines related to other non-climatic factors are minimal in the future. In Scenario 2, median probability of occupancy is estimated to be at or near zero in all analytical units by the 2060-2080 period indicating that, with a worse climate scenario and trend momentum, nearly all locations across the range are expected to be unoccupied by this time. Confirmed and widely documented host *Bombus occidentalis* was projected to decrease in probability of occupancy under both scenarios, at all time periods, in all analytical units.

Viability synthesis: While there is a paucity of information available about Suckley's cuckoo bumble bee biology, occurrence records and occupancy analysis point to widespread declines. One important uncertainty in the occupancy modeling is it may fail to capture some features that the species' responds to at a smaller scale, due to the coarse spatial resolution of the model. The model also does not account for host species presence, which is required for Suckley's cuckoo bumble bees to reproduce. Nevertheless, we have evidence the species is currently in a low resiliency state because the species has not been observed since 2016 in the contiguous United States and probability of occupancy has declined 85 percent range wide. There has been a loss of redundancy because three of four eastern analytical units and two of eleven western analytical units are currently in a quasi-extirpated state. Representation has also declined as a result of this range contraction and occupancy decline, because the breadth of phenotypic, genetic, and ecological diversity of Suckley's cuckoo bumble bee is likely lower than historically.

Under both future scenarios we projected a range wide decline in Suckley's cuckoo bumble bee, with Scenario 2 indicating range wide quasi-extirpation by 2060–2080. Thus, in the future we expect the species to have low to no resiliency, redundancy, and representation. It is likely that any persisting bees will occur in isolated populations, potentially with low genetic diversity. Finally, it is important to note that the viability of Suckley's cuckoo bumble bee is dependent on its host species, many of which have declined historically and are expected to continue to do so in the future.

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2 Introduction and Analytical Approach

This report synthesizes the results of a Species Status Assessment (SSA) conducted by the U.S. Fish and Wildlife Service (Service) for Suckley's cuckoo bumble bee (*Bombus suckleyi*). Suckley's cuckoo bumble bee has historically been found throughout much of the western North America, as far south as Arizona and in Canada as far north as Yukon and the Northwest Territories. It has also been found into the eastern Canadian provinces from the Hudson Bay, east to the Canadian Maritime provinces.

This SSA report synthesizes available information on Suckley's cuckoo bumble bee's biology and threats and assesses the species' current and future viability. Viability is the ability of a species to maintain populations in the wild over time. To assess viability, we use the conservation biology principles of resiliency, redundancy, and representation (3Rs, adapted from Shaffer & Stein, 2000, pp. 308–311; Wolf et al., 2015, entire). To sustain populations over time, a species must have the capacity to withstand:

(1) environmental and demographic stochasticity and disturbances (resiliency),

(2) catastrophes (redundancy), and

(3) novel changes in its biological and physical environment (representation).

A species with a high degree of resiliency, redundancy, and representation is better able to adapt to novel changes and to tolerate environmental stochasticity and catastrophes. In general, species viability will increase with increases in resiliency, redundancy, and representation.

Resiliency is the ability of a species to withstand environmental stochasticity (normal, year-to-year variations in environmental conditions such as temperature and rainfall), periodic disturbances within the normal range of variation (fire, floods, and storms), and demographic stochasticity (normal variation in demographic rates such as mortality and fecundity). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions.

We can best gauge resiliency by evaluating population level characteristics such as: demography (abundance and the components of population growth rate: survival, reproduction and migration), genetic health (effective population size and heterozygosity), connectivity (gene flow and population rescue), and habitat quantity, quality, configuration, and heterogeneity. Also, for species prone to spatial synchrony (regionally correlated fluctuations among populations), distance between populations and degree of spatial heterogeneity (diversity of habitat types or microclimates) are also important considerations.

Redundancy is the ability of a species to withstand catastrophes. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health and for which adaptation is unlikely.

We can best gauge redundancy by analyzing the number and distribution of populations relative to the scale of anticipated species-relevant catastrophic events. The analysis entails assessing the cumulative risk of catastrophes occurring over time. Redundancy can be analyzed at a population or regional scale, or for narrow-ranged species, at the species level.

Representation is the ability of a species to adapt to both near-term and long-term changes in its physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (pathogens, competitors, predators, etc.) environments. This ability to adapt to new environments–referred to as adaptive capacity–is essential for viability, as species need to continually adapt to their continuously changing environments. Species adapt to novel changes in their environment by either (1) moving to new, suitable environments or (2) by altering their physical or behavioral traits (phenotypes) to match the new environmental conditions through either plasticity or genetic change. The latter occurs via the evolutionary processes of natural selection, gene flow, mutations, and genetic drift.

We can best gauge representation by examining the breadth of genetic, phenotypic, and ecological diversity found within a species and its ability to disperse and colonize new areas. In assessing the breadth of variation, it is important to consider any morphological, behavioral, or life history differences, environmental or ecological variation across the range, and measures of interpopulation genetic diversity. In assessing dispersal ability, it is important to evaluate the ability and likelihood of the species to track suitable habitat and climate over time. Lastly, to evaluate the evolutionary processes that contribute to and maintain adaptive capacity, it is important to assess (1) natural levels and patterns of gene flow, (2) degree of ecological diversity occupied, and (3) effective population size. In our species status assessments, we assess all three facets to the best of our ability based on available data.

2.1 Uncertainties

Suckley's cuckoo bumble bee is only known from approximately 2,317 occurrence records from across North America (the average number of occurrence records for all other North American bumble bee species is 18,756). Cuckoo bumble bees, such as Suckley's cuckoo bumble bee, are naturally lower in abundance than eusocial bumble bees (bees which live in a colony with a social structure) and they do not have a worker caste, so they are inherently more difficult to detect. There are few studies that focus on Suckley's cuckoo bumble bee as the focal species. A literature review found only eight studies including Suckley's cuckoo bumble bee (Dozier et al., 2023, p. 641). Of these eight, none focus on Suckley's cuckoo bumble bee behavior, ecology, biochemistry, conservation, immunology, parasitology, or phenology (Dozier et al., 2023, p. 642). This leaves vast data gaps for conducting a species analysis.

Furthermore, we generally lack an understanding of cuckoo bumble bees, subgenus *Psithyrus*. Lhomme and Hines (2019, p. 136) reviewed the ecology and evolution of cuckoo bumble bees and note that the patterns described are limited by the lack of data on the subgenus *Psithyrus*. Bumble bees in general are increasing in research and conservation attention, but *Psithryus* remain understudied; fewer than 2 percent of bumble bee publications focus on *Psithyrus* despite them making up 11 percent of the genus (Bower et al., 2023, p. 518). Our understanding of *Psithyrus* behavior comes from studies of only a few species (Dozier et al., 2023, p. 647). Additionally, the literature available is skewed toward European species – approximately two thirds of the literature on *Psithryus* is from Europe (Dozier et al., 2023, p. 640).

During expert team meetings for this analysis, it was clear that we lack a nuanced understanding of Suckley's cuckoo bumble bee biology, the species range (see 3.4 Distribution for further discussion), and the effects of various stressors on this species specifically. While conducting this analysis, when we lacked specific information about Suckley's cuckoo bumble bee, we leaned on information about cuckoo bumble bees generally (subgenus *Psithyrus*) or bumble bees (genus *Bombus*) even more generally. We note these applications when they occur to ensure transparency where species-specific uncertainties exist. Acknowledging the paucity of data, we have recommended areas for further study (see 11.1 Suggestions For Future Work).

3 Species Description and Needs

3.1 Taxonomy and Genetics

Bumble bees belong to the genus *Bombus* (Order Hymenoptera, Family Apidae). *Bombus* species are grouped into subgenera based on behavior, morphology, and molecular phylogenies (Cameron et al., 2007, entire; P. H. Williams et al., 2008, entire). There are approximately 250 bumble bee species worldwide, with approximately 50 native bumble bee species in the United States and Canada. These approximately 50 bumble bee species are grouped into eight sub-genera, of which *Psithyrus* is one (P. H. Williams et al., 2014, pp. 10, 45).

Suckley's cuckoo bumble bee was first described by John Greene (Greene, 1860, p. 169), and belongs to the subgenus *Psithyrus*, commonly referred to as cuckoo bumble bees. Cuckoo bumble bees comprise roughly 11 percent of all bumble bee species (Lhomme & Hines, 2019, p. 8), and historically were classified as a separate genus due to their parasitic nature. Recent bumble bee phylogenetic analysis (Cameron et al., 2007, entire; P. H. Williams et al., 2008, entire) recognized *Psithyrus* as a subgenus within the genus *Bombus*. Suckley's cuckoo bumble bee belongs to the bohemicus group of *Psithyrus*. This group includes *Bombus bohemicus*, *Bombus ashtoni*, and two old-world species *Bombus vestalis* and *Bombus coreanus* (Lhomme & Hines, 2019, p. 8). There are no studies on the within species genetic variability and population structure of Suckley's cuckoo bumble bee (COSEWIC, 2019, p. 11).

3.2 Morphology

Cuckoo bumble bees differ morphologically from other *Bombus* subgenera because they have no pollen-collecting apparatus on their hind legs, they do not produce a worker caste and they produce insufficient wax for the construction of a nest. Suckley's cuckoo bumble bee is not eusocial (e.g., there is no division of labor within the species, no caste system exists, and all individuals are reproductive).

Suckley's cuckoo bumble bee is medium-sized and grouped within the "long-faced" clade (Cameron et al., 2007, p. 176; P. H. Williams et al., 2014, p. 163). This species is generally identifiable based on hair coloration and other physical characteristics. In females, the tip of the abdomen is recurved ventrally similar to other female cuckoo bumble bees. The ventral surface is ridged with two strong triangular carinae visible when viewed dorsally (COSEWIC, 2019, p. 163). While females may be identified by color and the recurved abdomen, proper identification of males may require examination of genitalia structures (August 2023 Expert Team Meeting, p. 4–5). Species color patterns vary considerably, descriptions are compiled in part from (P. H. Williams et al., 2014, pp. 163–165):

Females: Suckley's cuckoo bumble bee females are 1.8 to 2.3 centimeters (cm) (0.7 to 0.9 inches (in)) in length. Cuckoo bumble bees, members of the subgenus *Psithyrus* (including Suckley's cuckoo bumble bee), do not have a corbicula (pollen carrying basket on their hind leg), unlike the true bumble bees (pollen collecting, social species). Instead, their hind leg tibia is convex and densely covered in hairs. Suckley's cuckoo bumble bee hair is short and even. The hair of the head (including the vertex – top of the head) is black. The hair of the thorax (including below the wings) is mostly yellow, with a black spot or band between the wings, sometimes with a black triangular notch behind, and between the wings. The first two tergal (T-dorsal plate) segments on the abdomen are black, usually with at least some yellow (laterally and posteriorly) on T3 – no yellow centrally (Figure 1, Figure 2). T4 has predominantly yellow hairs, with a patch of black centrally and anteriorly. T5 is usually black but can have yellow laterally; T6 is black. The tip of the abdomen is recurved ventrally (6th sternite, S6), with two strong triangular carinae (ridges) visible in dorsal view. For photographs highlighting specific features, refer to the Suckley's cuckoo bumble bee COSEWIC report (COSEWIC, 2019, pp. 8–11)

Males: The male is 13 to 16 mm (0.5 to 0.6 in) in length. The color patterns for males of this species are extremely variable. The only consistent features are yellow on all of T1 and T4, with some (or all) yellow on T2, T3, T5 and T6. T7 is black (Figure 2). Examination of genital structures is often required for proper identification.

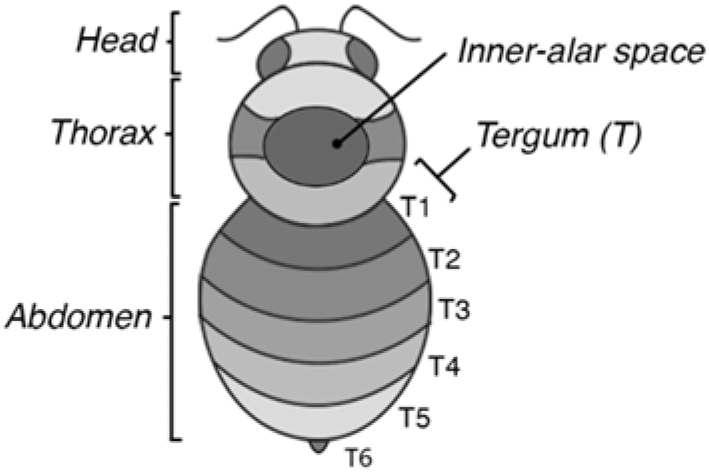

***Figure 1**. Bumble bee anatomy. The basic anatomy of a bumble bee can be described by the head, thorax, and abdomen. The inner-alar space refers to the dorsal plate between the wings on the thorax. A tergum (T) is an abdominal segment; terga are numbered from the anterior (front) to the posterior (back) of the abdomen. Tergum 1 through 5 (T1-T5) can be used to identify species by color pattern. Male bumble bees have a total of seven abdominal segments, which can be used to differentiate them from females, which have a total of six segments (P. H. Williams et al., 2014). Figure produced by Marion Clément, Service.*

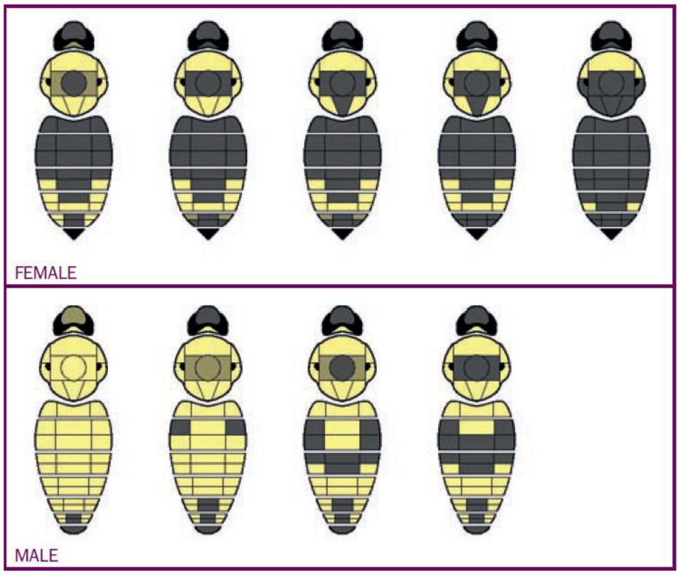

Figure 2. *Suckley's cuckoo bumble bee coloration by sex: female (top) and male (bottom). Suckley's cuckoo bumble bee is a social parasite and does not have workers. Graphic copied from P.H. Williams et al. (2014, p. 164).*

3.3 Life History

Non-parasitic bumble bees are eusocial insects (live in a colony with a social structure) with an annual lifecycle. Queens emerge from hibernation in early spring and search for nesting sites, usually underground. The queen collects pollen and brings it to the nesting site to provision brood cells. Eggs are then laid upon the food store and incubated, with the first workers emerging approximately four to five weeks later. Workers eventually take over brood care and foraging. The queen generally produces workers until the end of the summer when she shifts to producing reproductive males and females (gynes). The new gynes leave the nest, mate, and then hibernate until spring (Lhomme & Hines, 2019, p. 124; P. H. Williams et al., 2014, pp. 12–15). Male reproductive do not overwinter.

Cuckoo bumble bees (subgenus *Psithyrus*) exhibit an altered life cycle from social bumble bees (Figure 3 and Table 1). Female cuckoo bumble bees emerge from hibernation in late spring, typically after their hosts (Lhomme & Hines, 2019, p. 132). The timing of *Psithyrus* emergence compared to their hosts varies across species. Suckley's cuckoo bumble bee, along with *Bombus bohemicus*, *B. fernaldae* (now reclassified as *B. flavidus*), and *B. insularis*, are thought to emerge shortly after their hosts (Hobbs, 1967, p. 1291; Lhomme & Hines, 2019, p. 132). After emergence, females feed on nectar and pollen before invading and usurping host nests (Lhomme & Hines, 2019, p. 132). Host nests are invaded shortly after nest initiation, after the first batch of workers is produced.

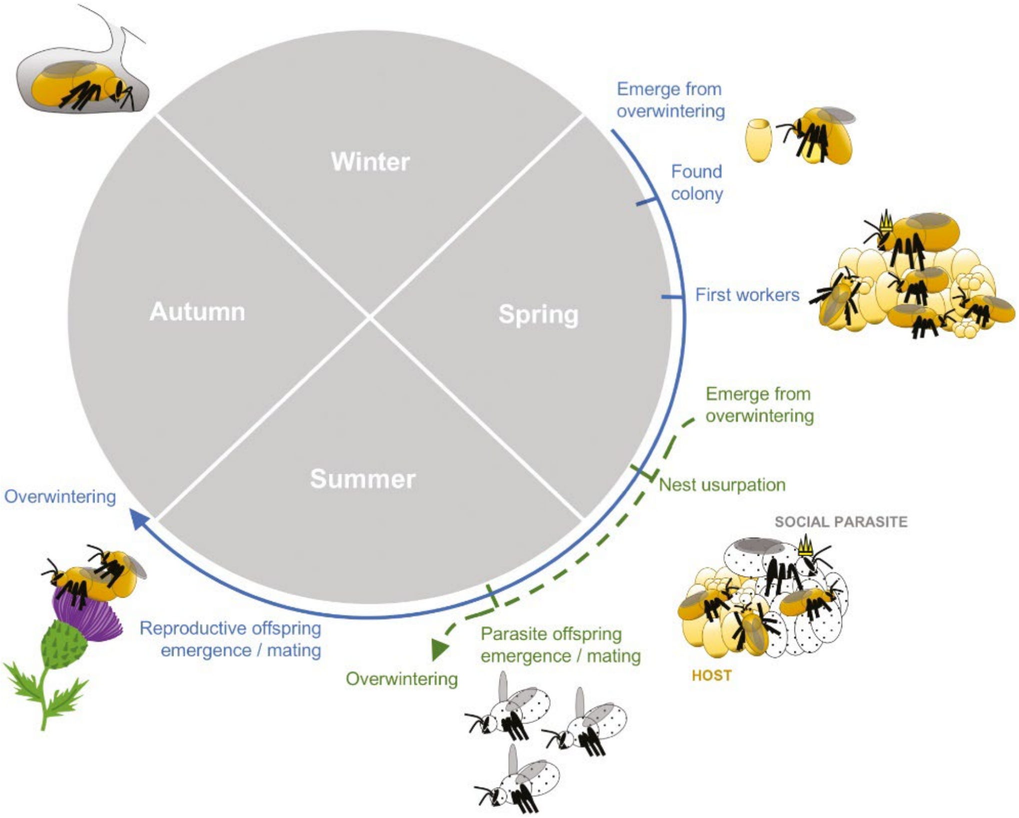

Figure 3. *Diagram depicts the difference in the lifecycle of social bumble bees (blue) and the life cycle of parasitic cuckoo bumble bees (green). Timing is approximate and may vary by species and latitude. Graphic copied from Lhomme and Hines, 2019, p. 124.*

Upon infiltrating a host nest, the female cuckoo bee must overcome being recognized as alien by the hosts. This can be accomplished in various ways. Intruder females may hide under the brood to avoid contact until they acquire the odor of the infiltrated colony, or they may use chemical camouflage (e.g., releasing allomones) to prevent attack from host worker bees (Lhomme & Hines, 2019, p. 134). Once integrated, the female cuckoo bee will often eliminate the host queen, destroy host eggs, and eject host larvae from the nest. This may be driven by the need to create space for parasitic eggs and/or to increase the incubation effort of host workers towards parasitic eggs. Cuckoo bees lack a mechanism to carry pollen and thus depend on social bee hosts to collect the pollen on which they rear their young (Goulson, 2010a, p. 77). Larval development takes approximately five weeks but varies with ecological factors (COSEWIC, 2019, p. 28). Cuckoo bumble bee offpsring typically emerge in late summer and mate through the fall although specific timing differs by species and latitude. Only mated females overwinter. Suckley's cuckoo bumble bees do not produce a worker caste, only reproductive males and females (Lhomme & Hines, 2019, p. 126).

Table 1. *Differences in life history traits between bumble bees and socially parasitic cuckoo bumble bees.*

Life history trait	Social Bumble bees (genus *Bombus)*	Cuckoo bumble bees (subgenus *Psithyrus)*
Emergence	Early spring	Late spring
Produce nest wax	Yes	No
Collect pollen	Yes	No
Consume nectar and pollen	Yes	Yes
Produce workers	Yes	No

As an obligate social parasite, Suckley's cuckoo bumble bees are dependent on host species for much of their life cycle. Only two species have been confirmed as Suckley's cuckoo bumble bee hosts: Western bumble bee (*Bombus occidentalis*) and Nevada bumble bee (*Bombus* nevadensis). However, the distribution of Suckley's cuckoo bumble bee suggests other species may also serve as hosts. See 4.1.4 Suitable Host Colony Availability for further discussion of hosts and Table 3 for common names of each host species.

Like other bumble bees, Suckley's cuckoo bumble bees require suitable foraging, nesting, and overwintering habitat. Host bumble bee nests are often located in abandoned underground holes, such as rodent burrows, in a wide array of habitat types including meadows, fallow fields, croplands, urban areas, and forests (COSEWIC, 2019, p. 26; Liczner & Colla, 2019, p. 795; Martin et al., 2023, p. 24). Suckley's cuckoo bumble bee is a generalist nectar forager with meadows, grasslands, and developed areas providing important foraging habitat (Martin et al., 2023, p. 24). While foraging, cuckoo bumble bee males visit more flowers and perform significantly slower and longer flower visits than eusocial bumble bees (Fisogni et al., 2021, p. 199). This is likely due to differing energy needs, given cuckoo bumble bees consume pollen and nectar, but do not collect pollen for the colony. Therefore, balancing energy intake over expenditure during foraging may be less important for cuckoo bumble bees. Specific requirements for overwintering sites are unknown, but Suckley's cuckoo bumble bee females overwinter underground in areas separate from nesting habitat, likely using mulch or other decomposing vegetation (COSEWIC, 2019, p. 27; Liczner & Colla, 2019, p. 793; Martin et al., 2023, p. 25). For further description of the species needs, see section 4 Species Ecological Needs.

3.4 Distribution

There is significant uncertainty about the range of Suckley's cuckoo bumble bee. Some of this uncertainty stems from misidentification of the species, often due to its similarity in appearance to *Bombus bohemicus*. While some verification work of occurrence records occurred for this assessment, further verification of records across the range is needed (11.1 Suggestions For Future Work). The species has a broad historical distribution across North America, and it has been found in various habitat types including prairies, grasslands, meadows, and woodlands (COSEWIC, 2019, p. 26; Martin et al., 2023, p. 22; Montana Natural Heritage Program, 2023 p. 3; P. H. Williams et al., 2014, pp. 164–165). Occurrence records in the dataset compiled for this assessment were found at elevations ranging from approximately 2 to 3,200 meters (6 to 10,500 feet).

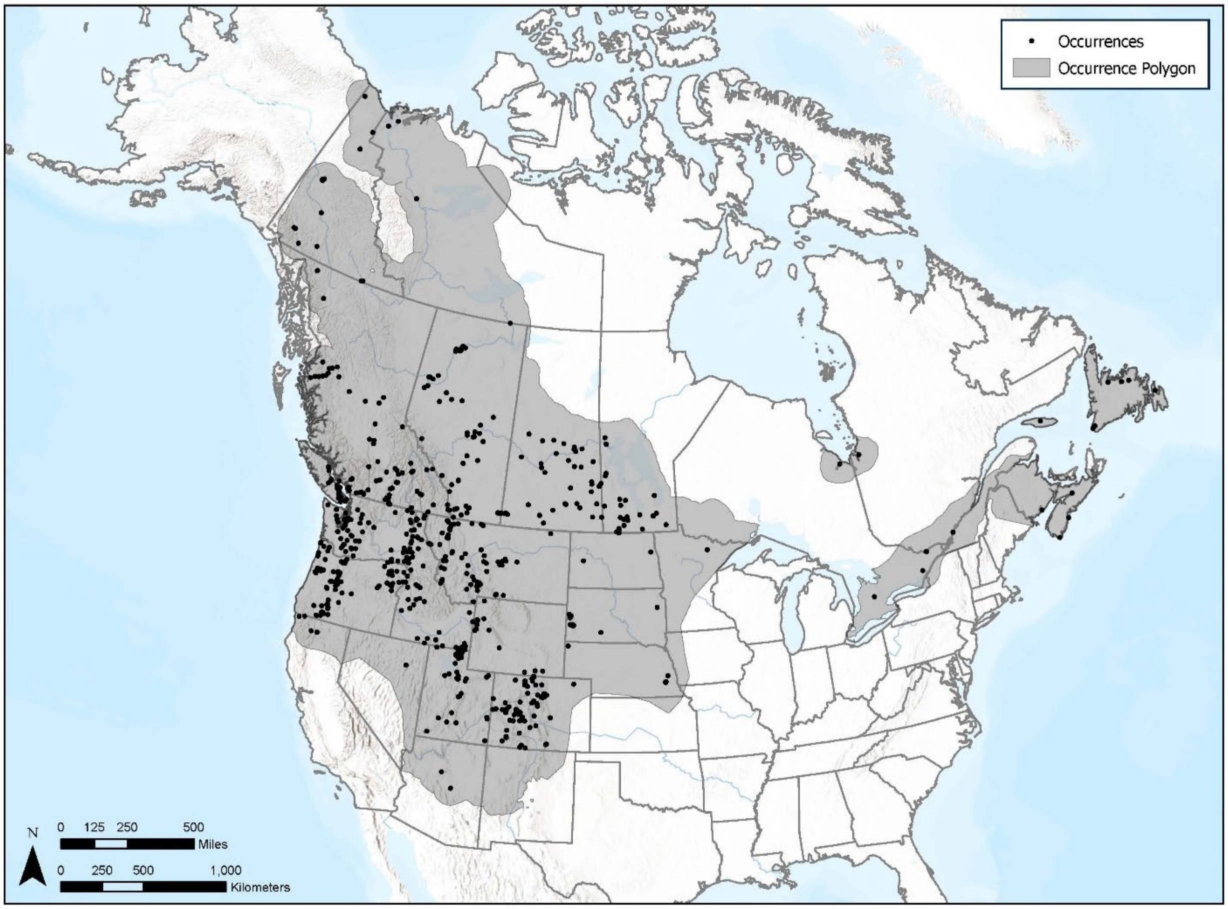

Figure 4. *Occurrence polygon of Suckley's cuckoo bumble bee in grey overlaid with all known occurrence points from 1893-2022 in black. The occurrence polygon was delineated with a combination of level II ecoregions (CEC 1997) and a 100-kilometer (62-mile) buffer around observations of the species. The occurrence polygon is generally inclusive of areas between positive occurrence locations – areas with sparse survey data were included. The methodology used to delineate the occurrence polygon and analytical units is described in 6 Assessment Methodology.*

There are few records of Suckley's cuckoo bumble bee in eastern North America and all of them occur in Canada. While historically there were thought to be some records of Suckley's cuckoo bumble bee in the eastern United States, verification efforts have proven these to be misidentifications (B. Swartz, 2023 pers. Comms; J. Heron and C. Sheffield 2023, pers. Comms). Currently, there is no evidence of Suckley's cuckoo bumble bee in the eastern United States. However, given the proximity of some of the historical records to the United States border in New York, our occurrence polygon does include a small portion of northern New York and Maine (Figure 4).

There is also no evidence of connectivity between the eastern and western portions of the range. Recent efforts in Canada to survey under-sampled areas of habitat that could provide connectivity have failed to find any Suckley’s cuckoo bumble bees; however, based on current search effort their presence cannot be entirely ruled out, particularly in the northern boreal region (S. Cannings, J. Heron, and C. Sheffield, 2023, pers. Comms). The areas of the midwestern United States that could provide connectivity for Suckley’s cuckoo bumble bee are currently very well surveyed without detection of the species and are therefore unlikely to provide connectivity (August 2023 Expert Team Meeting, p. 8). While connectivity could potentially exist, particularly in Canada where further survey efforts should be pursued, the current best available science suggests the species has an eastern population in Canada that is currently disjunct from the core range in the west. The most recent occurrence record from the east is from Newfoundland in 2010.

In the western portion of the range, Suckley’s cuckoo bumble bee observations extend south to Arizona, east to Nebraska and the edge of Minnesota, and north through the Yukon of Canada. The species has not been observed within the United States since 2016. Historically there were records of the species in Alaska, however verification of all known records has indicated they were misidentifications (D. Sikes and J. Rykken 2023, pers. Comms). However, Suckley’s was recorded adjacent to the Alaska border in the Yukon; the species could occur in eastern Alaska, but we have no current evidence of its presence.

The majority of species records are in the western portion of the species range, overlapping the range of confirmed and widely documented host *B. occidentalis*. The most southerly records are two male specimens from 2009 and 2011 in Arizona. Male specimens of this species are challenging to identify, and these records could not be verified during this assessment. As the area of Arizona is within the known range of host species *B. occidentalis,* we chose to include both these specimens, and this southern region, in the range.

4 Species Ecological Needs

4.1 Individual Needs

The needs of Suckley's cuckoo bumble bee individuals vary by sex and life stage, as summarized in Table 2. These needs are further described in the following subsections: 4.1.1 Thermal Suitability, 4.1.2 Diapause Site Availability, 4.1.3 Floral Resource Abundance and Diversity, 4.1.4 Suitable Host Colony Availability.

Table 2. *Ecological needs for survival and reproductive success of Suckley's cuckoo bumble bee individuals by life stage and season.*

Life Stage	Season	Activity	Needs
Adult females	Fall through early spring	Overwintering	• Overwintering sites (loose substrates such as leaf litter, duff, rotting logs) • Thermal suitability
	Late spring	Emergence	• Diversity of native floral resources • Thermal suitability
	Late spring through summer	Usurpation and nest dominance	• Suitable host colony
Eggs & larvae	Summer	Growing and receiving provisions from host workers	• Diversity of native floral resources • Host species workers • Thermal suitability
Male drones & new females	Late summer	Mating	• Diversity of native floral resources • Unrelated mates

4.1.1 Thermal Suitability

Bumble bee species require temperatures to be within a suitable range, bounded by upper (heat) and lower (cold) thermal limits. Morphological, physiological, and behavioral characteristics of bumble bees enable them to thermoregulate to survive in colder temperatures to a greater extent than many other insects, including other bee taxa (Bishop & Armbruster, 1999, p. 717). Research has not determined the specific thermal limits for most species of bumble bees, including Suckley's cuckoo bumble bee.

Thermal suitability and tolerance appear to be highly variable across and within bumble bee species. Recent bumble bee modelling efforts suggest that species-specific occupancy peaks at intermediate temperatures, but these trends are variable and not well characterized by the genus-level trajectories (Jackson et al., 2022, fig. 1b). This indicates ideal temperature ranges are likely species dependent. Bumble bee species at lower elevations (Oyen et al., 2016, p. 55) and those in warmer climates (Martinet et al., 2015, p. 4, 2021, p. 8) may exhibit greater tolerance to increased temperatures and heat stress than species at higher elevations or in cooler climates (i.e., Arctic and boreal species). Heat tolerance can also vary within species by caste and sex. Bumble bee queens are less heat tolerant than males and worker bees, and queen heat tolerance varies with age and reproductive status (Feuerborn et al., 2023, p. 11).

There is also a thermal suitability component to diapause. Temperatures may affect the length of diapause and the energy expenditure during diapause, which influence female survival (see 4.1.2 Diapause Site Availability).

Lower thermal limits

Bumble bees' lower thermal limits are influenced by their ability to increase and maintain their body temperature above ambient temperatures. Bumble bees can increase their thoracic temperature while stationary by contracting their flight muscles (Kammer & Heinrich, 1974, p. 219) and potentially through metabolic activity in their flight muscles (Newsholme et al., 1972, p. entire). Bumble bees' relatively larger body sizes, as well as other morphological characteristics such as long, dense hair reduce heat loss relative to most other bee genera (Bishop & Armbruster, 1999, p. 720). Some researchers have found that larger-bodied bumble bee species and larger individuals within the same species have an increased tolerance to lower temperatures (Bishop & Armbruster, 1999, p. 720; Heinrich, 1975, p. 39; Kenna et al., 2021, p. 9; Lundberg, 1980, p. 106; Oyen et al., 2016, p. 54). Bumble bee species and populations found at higher latitudes and elevations may have a greater adapted tolerance to lower temperatures (Oyen et al., 2016, p. 55; Pimsler et al., 2020, p. 2). This supports that the ideal thermal limits for Suckley's cuckoo bumble bees likely shift by population, depending on habitat type, latitude, and elevation (September 2023 Expert Team Meeting, pp. 2–3).

Generally, a bumble bee's thoracic temperature must be above approximately 30 Celsius (˚C) (86 Fahrenheit (˚F)) to fly, though this likely varies by species (Goulson, 2010a, p. 13; Heinrich, 1975, p. 158). Bumble bees in the arctic maintain similar thoracic temperatures to temperate bumble bees and higher abdominal temperatures, in contrast to some insects in arctic regions that have adapted to the lower ambient temperatures by tolerating lower body temperatures (Heinrich & Vogt, 1993, pp. 260, 265). As bumble bees approach the lower end of their thermal limits, they become lethargic (Oyen et al., 2016, p. 53). The thoracic temperature at which an insect cannot activate its muscles to warm up, entering a "chill coma", ranges from 7.0 to 8.0˚C (45 – 46˚F) for three *Bombus* species (Goller & Esch, 1990, p. 224). Additionally, extreme cold can also affect foraging behavior; exposure to cold narcosis (approximately 4˚C (39˚F) for 5 min.) reduced bumble bee foraging for days after exposure (Wilson et al., 2006, p. 171).

Nest temperatures are important to the maintenance and growth of the colony (Heinrich, 1979, p. 68; Vogt, 1986b, p. 64). Temperatures in underground bumble bee nests fluctuate less than in the surrounding environment, maintaining around 30˚C (86˚F) (Vogt, 1986b, p. 61; Goulson, 2010a, p. 20; Heinrich, 1979, p. 66), due to insulating qualities and colony behavior. Nest temperatures outside of ideal thermal ranges can slow larvae development and colony growth (Heinrich, 1979, p. 68; Vanderplanck et al., 2019, p. 3; Vogt, 1986b, p. 64). The brood is most susceptible to cold temperatures earlier in the season when ambient temperatures are low, and the colony is small. When this occurs, workers will incubate the brood (Vogt, 1986b, p. 63, 1986a, p. 56).

Upper thermal limits

As bumble bees approach the upper end of their thermal limits, they transfer heat from their thorax to their abdomen to cool their bodies (Heinrich, 1975, p. 165). When overheated, they become agitated and begin losing motor coordination, eventually entering into "heat stupor" (Martinet et al., 2021a, p. 3; Oyen et al., 2016, p. 53). Larger bumble bees are generally more susceptible to overheating (Bishop & Armbruster, 1999, p. 716; Goulson, 2010a, p. 55). Researchers have recorded the upper end of bumble bee thermal limits (critical thermal maxima), where loss of muscle control occurs, across species at ambient temperatures ranging from approximately 38 to 53˚C (100 – 129˚F) (Hamblin et al., 2017, p. supplemental dataset; Oyen et al., 2016, p. 54; Oyen & Dillon, 2018, p. 4). Compared to other bee species, bumble bees may be particularly sensitive to increases in temperature because of their relatively low critical thermal maxima (Hamblin et al., 2017, p. 3). Further, bumble bee abundance was observed to decrease following heat waves in Europe (Rasmont & Iserbyt, 2012, p. 276).

Exposure to extreme heat can have various effects on bumble bees. High nest temperatures during development may lead to fewer workers produced and a reduction in the number of workers performing foraging trips and total foraging trips (Gérard et al., 2022, p. 821). When this occurs, workers fan with their wings to cool the nest (Vogt, 1986a, p. 56; Weidenmüller et al., 2002, p. 1068). While Suckley's cuckoo bumble bees do not produce a worker caste, a reduction in host workers and foraging effort could impact suitable host availability and abundance. Additionally, heat shock (40˚C (104˚F)) negatively affected reproductive potential in two cold-adapted bumble bee species by affecting sperm viability, sperm DNA integrity, and production of hormones (Martinet et al., 2020, pp. 744–746). High temperatures may also negatively affect bumble bee behavior and morphology (Gérard et al., 2022, p. 821; White & Dillon, 2023, pp. 3–5).

4.1.2 Diapause Site Availability

Only mated Suckley’s cuckoo bumble bee females overwinter. Observations of females tend to trail off by August or September while males are observed until about October (Koch et al., 2012, p. 134; P. H. Williams et al., 2014, p. 164). There are no studies on Suckley’s cuckoo bumble bee behavior (Dozier et al., 2023, p. 642). Therefore, specific overwintering habitat requirements for Suckley’s cuckoo bumble bee females are unknown. Overwintering is generally understudied and not fully understood in bumble bees, but previous research highlights some patterns (Liczner & Colla, 2019, pp. 787–788). Suitable overwintering sites are important for sustaining diapause (hibernation) in mated females and protecting them from environmental and physical stressors, but extremely limited observations exist for bumble bees overall. Diapause in parasitic and non-parasitic bumble bees appears to be similar. Studies found no difference in hibernaculum location and depth for *Psithyrus* species compared with other non-parasitic bumble bees (D. V. Alford, 1969, pp. 150–158; Lhomme & Hines, 2019, p. 135).

Bumble bee gynes overwinter alone. Most diapause sites reported in the literature for *Bombus* species in North America were underground and most were in shaded areas near trees (Liczner & Colla, 2019, p. 787). In California, bumble bees have been reported to overwinter under tree litter and detritus on the forest floor, where winter temperatures are generally above freezing (N. M. Williams et al., 2019, p. 789). There is some evidence that overwintering gynes tend to avoid areas with dense vegetation in favor of sites with more bare ground or moss, although this interpretation may be skewed because it is easier for researchers to locate hibernacula entrances in sparsely vegetated areas (D. V. Alford, 1969, pp. 151–154). For underground sites, soil type is often described as sandy and well-drained (D. V. Alford, 1969, p. 169), suggesting that maintaining a consistently low moisture level is important (Sladen, 1912, pp. 94–101).

Depth of diapause seems to be variable, ranging between 2 and 15 cm (0.8 and 6 in) (Liczner & Colla, 2019, p. 793). Although factors influencing depth have not been directly studied, depth is hypothesized to vary with ecosystem type and region to best regulate temperature to optimize emergence time (Liczner & Colla, 2019, pp. 787–801). Aspect may also be important because it can influence snowmelt date. A large proportion of overwintering gynes have been recorded to use north-facing slopes (D. V.

Alford, 1969, pp. 149–151; Liczner & Colla, 2019, pp. 787–801), but they have also been found in all other aspects (Hobbs, 1968, pp. 156–164) and occasionally in flat sites (D. V. Alford, 1969, p. 152,154,159). Because soil temperature influences diapause duration and emergence (D. V. Alford, 1969, pp. 161–168; Beekman et al., 1998, p. 207), it has been hypothesized that the apparent preference for north-facing slopes and shaded areas is to prevent the gynes from emerging too early on relatively warm days in the winter or early spring (D. V. Alford, 1969, pp. 149–169), and more generally, it could suggest selection of sites that buffer diapausing bees from both temperature and moisture fluctuations (N. M. Williams et al., 2019, pp. 1–3).

Some reports suggest bumble bee diapause occurs immediately around the natal colony, while others indicate gynes overwinter elsewhere (Plath, 1927, pp. 123–127). Studies of other *Bombu*s species propose that diapause sites must be close to foraging resources because gynes establish colonies near hibernacula (D. V. Alford, 1969, p. 150). It is likely that distances depend on both the bumble bee species and the spatial distribution of suitable overwintering habitat relative to suitable nest locations (Liczner & Colla, 2019, pp. 787–801; Plath, 1927, pp. 123–127; N. M. Williams et al., 2019, pp. 1–3). There is also some evidence that gynes might select diapause sites in meadows with more flowers if suitable overwintering conditions exist (Elliott, 2009, pp. 753–754).

Individual bumble bee weight at the start of diapause may largely determine whether a gyne will be able to survive diapause, because females does not feed during the 6-to-9-month diapause (Beekman et al., 1998, p. 207; Ogilvie & CaraDonna, 2022, p. 2419). Fluctuations in winter temperature are suspected to impact metabolic rates of gynes. Higher temperatures result in higher rates of metabolism, thus gynes exposed to higher temperatures during diapause expend more energy. A European study of *B. lucorum* found that queens in diapause at warmer conditions used more fat compared to queens in diapause at colder conditions (Vesterlund et al., 2014, p. 270). Increased energy expenditure in lower weight gynes can reduce their ability to survive the winter unless their hibernaculum is adequately insulated (Hodek & Hodkova, 1988, pp. 153–165). The duration of diapause is also important for overwinter survival because if the duration is too long, fewer eggs may be produced, and the likelihood of survival will be reduced. For instance, queens are not awakened by rising temperatures if diapause is less than three months, but after five months they can be awoken by increasing temperatures (Larrere et al., 1993, p. 11).

Body condition may also play a role in the timing of emergence. Queens with higher body mass (a proxy for body condition) emerged later than smaller queens (Keaveny & Dillon, 2022, pp. 5–6). This suggests that body condition also plays a role in bumble bee phenology and that emergence from diapause may be determined by multiple factors. Diapause in *Psithyrus* females is likely similar, although they typically emerge from overwintering later than their hosts, which could indicate they have adaptations for longer diapause (Husband et al., 1980, p. 227; Lhomme & Hines, 2019, pp. 132, 135).

Based on the best available data, once suitable substrate is found to create diapause burrows, the two most important characteristics of diapause sites appear to be microsite conditions that regulate temperature and moisture fluctuations during the diapause season, with proximity to natal colonies and spring floral resources being secondary considerations.

4.1.3 Floral Resource Abundance and Diversity

Suckley's cuckoo bumble bees are found in a wide variety of habitats, including montane meadows and prairies, farms, woodlands, boreal forests, active and fallow agricultural fields, and urban areas

(COSEWIC, 2019, pp. v, 26). Within these habitats, Suckley's cuckoo bumble bees have been recorded on a number of genera of plants in the Family Asteraceae (composites), including *Aster*, *Symphyotrichum* (American aster), *Chrysothamnus* (rabbitbrush), *Cirsium* (thistle), *Haplopappus*, *Solidago* (goldenrod), *Rudbeckia* (cone flower), *Centaurea* (e.g., knapweed and star-thistle), and *Helichrysum* (licorice plant) (COSEWIC, 2019, p. 27; Koch et al., 2012, p. 134; P. H. Williams et al., 2014, p. 163). Other food plants are *Melilotus* (sweetclover) and *Trifolium* (clover) in the Family Fabaceae (legumes), *Rubus* (blackberry) and *Cotoneaster* in the Family Rosaceae (roses), *Vaccinium* (e.g., blueberries and huckleberries) in the Family Ericaceae (heath), Penstemon (beardtongue) in the Family Plantaginaceae (plantain), and *Salix* (willow) in the Family Salicaceae. For a full list of known floral associations, see section 10.2 Floral Associations.

Because cuckoo bumble bees are dependent on host workers to raise their offspring, females tend to emerge from hibernation later than their hosts to feed on nectar and pollen in preparation for laying eggs (Lhomme & Hines, 2019, p. 132). In the German Alps, distribution of all cuckoo bumble bee species decreased with increasing elevation, likely in response to the shortened seasonal availability of floral resources for species that have a longer winter diapause compared to their hosts (Sponsler et al., 2022, p. 9). Limited information exists regarding key forage plants for cuckoo bumble bees (Dozier et al., 2023, p. 643), but *Psithyrus* are considered generalist foragers and their short to medium tongue-length limits the flower types they are able to access (The Trustees of the Natural History Museum, n.d.).

Suckley's cuckoo bumble bee females emerge from early April to late May depending on the area, (Koch et al., 2012, p. 134; Thorp et al., 1983, p. 50; P. H. Williams et al., 2014, p. 164) and invade the nests of host species shortly after they are established. The flight period for males is from mid-May to early October. Therefore, as with their hosts, abundant spring floral resources are important to Suckley's cuckoo bumble bee females upon their emergence for ovary development (Lhomme & Hines, 2019, p. 132) and abundant fall floral resources are important to the fitness of the colony (Hatfield & LeBuhn, 2007, pp. 156–157), since this is when new gynes and drones are produced (Goulson, 2010a, pp. 6–8). In addition, as noted in the section above, fall floral resources are important for females who must survive an overwintering diapause without foraging (Beekman et al., 1998, p. 207; Ogilvie & CaraDonna, 2022, p. 2419).

For bumble bees in general, pollen with a high nutritional composition of proteins and amino acids is particularly important for larval development (Somme et al., 2015, p. 93). Different foraging strategies are used to meet colony nutritional needs, sometimes concentrating on a patch of flowers with pollen of high nutritional value, and in other cases broadening diet breadth even when the nutritional value might be lower (Kriesell et al., 2017, p. 113; Somme et al., 2015, p. 93). Therefore, bumble bees probably balance the search for high-quality pollen with the energetics of foraging distance (Kriesell et al., 2017, p. 113).

4.1.4 Suitable Host Colony Availability

Suckley's cuckoo bumble bee is an obligate social parasite (it depends on social hosts for survival and raising young) in the subgenus *Psithyrus*. Bumble bees in the subgenus *Psithyrus* lack a mechanism to carry pollen and thus depend on social bumble bee hosts to collect pollen to rear their young (Goulson, 2010a, p. 77). Suckley's cuckoo bumble bees are therefore entirely dependent on host bumble bee colonies, making host colony availability a critical habitat need for the species survival and overall viability. Cuckoo bumble bees are generally observed in low abundance at the margins of a host species range and cuckoo bumble bee distributions are less than that of the host species (Antonovics & Edwards, 2011, p. 1003).

Psithyrus vary in their degree of host specificity, with both broad generalist *Psithyrus* parasitizing hosts from different bumble bee subgenera to specialist *Psithyrus* parasitizing a single host bumble bee species currently documented (Lhomme & Hines, 2019, pp. 129, 132). Suckley's cuckoo bumble bee is part of the bohemicus group of *Psithyrus* which are thought to parasitize bumble bees in the subgenus *Bombus* (*Bombus* sensu stricto) (Lhomme & Hines, 2019, p. 129). Suckley's cuckoo bumble bee has been described as a semi-specialist (Lhomme & Hines 2019, p. 129), however experts debate the degree of specialization. Bumble bee nests are rare to encounter at a baseline, and usurped nests are even rarer to encounter, making observations supporting host choice limited (Lhomme & Hines (2019, pp. 132-133). Additionally, cuckoo bumble bee females may shelter in nests they do not usurp, leading to inconclusive observations.

Given these challenges, our current understanding is that Suckley's cuckoo bumble bee has two confirmed hosts and numerous potential hosts (Table 3). *B. occidentalis* is the most widely known host of Suckley's cuckoo bumble bee (Hobbs, 1968, p. 164; Lhomme & Hines, 2019, p. 128; P. H. Williams et al., 2014, p. 165). *B. occidentalis* occurs throughout the core of Suckley's cuckoo bumble bee's range in Western North America. There are also three records of Suckley's cuckoo bumble bee nesting successfully (i.e. rearing young) in *B. nevadensis* nests (Hobbs, 1965, p. 127).

Aside from these two confirmed hosts, there are numerous potential hosts (Table 3). Suckley's cuckoo bumble bee has been observed within the nests of *B. terricola*, *B. rufocinctus*, *B. fervidus*, and *B. appositus* (Hobbs, 1968, pp. 157, 164; P. H. Williams et al., 2014, p. 165), however observation in the nest is not confirmation of breeding (Lhomme & Hines, 2019, p. 132). Suckley's cuckoo bumble bees are also observed in locations beyond the range of these confirmed and potential hosts (based on nest observations). Because of this, it is reasonable to assume there are additional hosts not yet confirmed through observations. Thus, through conversations with entomological experts, we have identified the following additional potential hosts for Suckley's cuckoo bumble bee: *Bombus affinis, Bombus mckayi,* and *Bombus cryptarum* (September Expert Team Call 2023, pp. 5–6, (COSEWIC, 2019, p. 17). All three of these species are in the subgenus *Bombus sensu stricto* which is preferred by other *Psithyrus* in the bohemicus group.

For several decades and as recently as 2012, *Bombus occidentalis* (confirmed Suckley's cuckoo bumble bee host) and *Bombus mckayi* were both considered subspecies of *Bombus occidentalis* (*B. o. occidentalis* and *B.o. mckayi*, respectively) (Sheffield et al., 2016, p. 191). Further genetic research using cytochrome c oxidase subunit I (COI) barcode-divergence confirmed them to be two distinct species (P. H. Williams, 2021, entire). In addition to genetic differences, *Bombus occidentalis* has shorter hair and larger variations in color patterns across its range than *Bombus mckayi*. *Bombus occidentalis* occurs in the western contiguous United States and southern British Columbia. *Bombus mckayi* occurs throughout Alaska, Yukon, and northern British Columbia. The geographic ranges of these two species meet around 57 degrees north in Alberta, Canada. It is possible that some of the confirmed records of Suckley's cuckoo bumble bee usurping *B. occidentalis* nests may have been *B. mckayi,* however this has not been confirmed.

Table 3. *Confirmed and potential bumble bee hosts for Suckley's cuckoo bumble bee. Species highlighted in blue belong to subgenus Bombus which are considered the preferred hosts of the bohemicus group of cuckoo bumble bees.*

Host species	Confirmed or potential host	Rationale	Nest descriptions[1]
B. occidentalis (western bumble bee)	Confirmed	Numerous reports of *B. suckleyi* reproducing in *B. occidentalis* nests, subgenus *Bombus* (Hobbs, 1968, p. 164; Lhomme & Hines, 2019, p. 128).	Usually nests underground
B. nevadensis (Nevada bumble bee)	Confirmed	Three records of B. *suckleyi* reproducing in *B. nevadensis* nests (Hobbs, 1965b, p. 127).	Small nests on the ground surface
B. terricola (yellow-banded bumble bee)	Potential	*B. suckleyi* observed in nest, reproduction not confirmed, subgenus *Bombus (*Hobbs, 1968, p. 164).	Nests underground
B. rufocinctus (red-belted bumble bee)	Potential	*B. suckleyi* observed in nest, reproduction not confirmed (Williams et al., 2014, p. 165).	Nests on surface or aboveground
B. fervidus (yellow bumble bee)	Potential	*B. suckleyi* observed in nest, reproduction not confirmed (Williams et al., 2014, p. 165).	Nests on surface or aboveground
B. appositus (white-shouldered bumble bee)	Potential	*B. suckleyi* observed in nest, reproduction not confirmed (Hobbs, 1966, pp. 293–294).	Nests on surface or aboveground
B. affinis (rusty-patched bumble bee)	Potential	Range overlap, subgenus *Bombus* (September Expert Team Call, p. 5–6; COSEWIC 2019, p. 17).	Nests underground
B. mckayi (McKay's bumble bee)	Potential	Range overlap and no confirmed hosts, subgenus *Bombus (September Expert Team Call, p. 5–6;* COSEWIC 2019, p. 17).	Usually nests underground*
B. cryptarum (cryptic bumble bee)	Potential	Range overlap and no confirmed hosts, subgenus *Bombus (*September Expert Team Call, p. 5–*6;* COSEWIC 2019, p. 17).	Nests underground

[1] Williams et al. 2014

*Bombus mckayi and B. occidentalis considered one species in Williams et al. 2014. The subspecies *mckayi* was recently elevated to species status (Williams 2021, p. 278).

For Suckley's cuckoo bumble bee, we lack information about factors leading to successful usurpation of host nests. We also lack information about specific host attributes that may result in some bumble bee host species being preferable to others. Using other *Psithyrus* as proxy, there is information to suggest that there is an optimal size of the host colony for usurpation; colonies too small may not have enough workers to provision parasite larvae while colonies too large can lead to failed usurpation by the parasite (Lhomme & Hines, 2019, pp. 133–134). For example, studies of the European cuckoo bumble bee *Bombus vestalis* indicate highest survival of the parasite in host colonies of about 10-15 workers (Sramkova & Ayasse, 2009, p. 59). The ideal size of a host colony likely varies by cuckoo bumble bee species and is unknown for Suckley's cuckoo bumble bee.

We also lack information on what density or number of host colonies would constitute ideal habitat for Suckley's cuckoo bumble bee. While host availability is a key habitat need for Suckley's cuckoo bumble bee, areas of high host abundance may not necessarily be ideal habitat for the parasite. In the British Isles, regions of high host abundance were not always the regions of highest parasite abundance (Antonovics & Edwards, 2011, p. 1003). Therefore, the constituents of ideal habitat for host and parasite may differ and more research on the specific habitat needs of Suckley's cuckoo bumble bee are needed. The species needs may differ in unknown ways from the host species and other bumble bees in North America.

Host bumble bee nests may be found on the surface or underground, depending on host species. For confirmed host *B. occidentalis*, known nest sites occur in underground cavities (e.g., animal burrows) more often than in aboveground structures (e.g., woody debris, human-made structures, etc.) (Hobbs, 1968, p. 157; Macfarlane et al., 1994, pp. 4, 6; Richards, 1978, p. 304). The transition zone between forest and grassland can be particularly valuable bumble bee nesting habitat, as well as field boundaries, meadow margins, and forest edges due to the presence of abandoned rodent nests and undisturbed habitat with diverse floral resources (Hines & Hendrix, 2005, p. 1483). Nesting habitat availability may be dependent on mammal or other ground burrowing animal abundance (Evans et al., 2008, pp. 6, 28). For instance, within grassland habitat, bumble bees tend to use abandoned rodent nests and areas under dry dense vegetation (Hines & Hendrix, 2005, p. 1481). Ground-nesting species of bumble bees are declining more than aboveground species, likely because ideal underground nest sites are more limited than surface nest sites due to dependence on rodent activity (Higginson, 2017, p. 99).

4.2 Population Needs

There is no population level information available specific to Suckley's cuckoo bumble bee. The species is rare to encounter because it is a naturally less abundant parasite, and it does not have a worker caste active throughout the field season. In this section we rely on the best available science about subgenus *Psithyrus* and genus *Bombus* in the absence of population information specific to Suckley's cuckoo bumble bee.

Healthy populations of Suckley's cuckoo bumble bee must have sufficient survival, abundance, connectivity, and genetic diversity to support viability. Highly resilient populations consist of many genetically diverse individuals that have all their basic resource needs met (i.e., thermal suitability, floral resource abundance and diversity, diapause site availability, and host colony availability) (Figure 5). This translates into a diverse collection of individuals on the landscape with high survival and reproduction success, which ultimately results in population growth and larger populations. Survival and reproduction of Suckley's cuckoo bumble bees depend on the survival and health of the host colony. Host colony workers are paramount to the growth and survival of new generations of Suckley's cuckoo bumble bees because they forage and care for the brood of parasite larva. We have no definitive information on the minimum number of host colonies or minimum habitat patch size for a self-sustaining population of Suckley's cuckoo bumble bee and hosts.

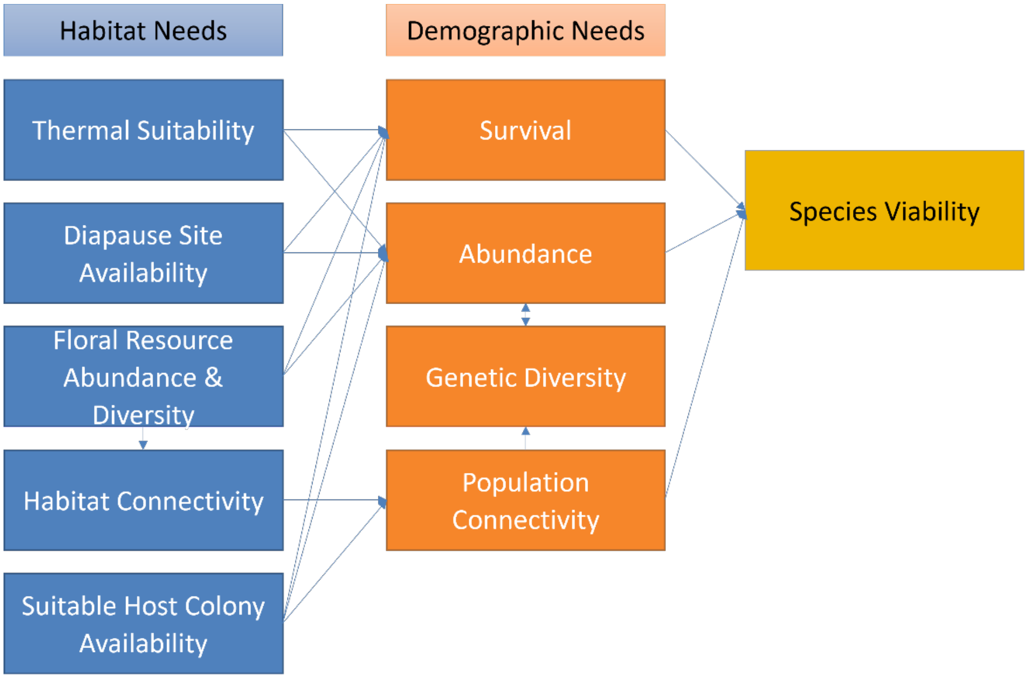

***Figure 5**. The population needs of Suckley's cuckoo bumble bee that support species viability.*

Survival and Abundance

High levels of population resiliency are evidenced by healthy demographics. The size of a population (abundance) protects that population from extirpation resulting from demographic and environmental stochasticity. Historically, the Service has described bumble bee population size in assessments using the number of colonies—not individuals—within a given geographical area. However, Suckley's cuckoo bumble bee does not form colonies made up of workers and instead parasitizes the colonies of other bumble bee hosts. All individuals of Suckley's cuckoo bumble bees are potentially reproductive. Therefore, population size and growth are more aptly described by the number of Suckley's cuckoo bumble bee individuals and any changes in numbers of individuals through time.

The exact number and distribution of individuals required to ensure long-term persistence of a bumble bee population is unknown and likely varies across spatial scales, but small populations are inherently more vulnerable than larger populations and generally have a lower likelihood of persistence over time (Goulson et al., 2008, pp. 197–198). The resiliency of a population is also evidenced by its long-term growth rate. For any population to persist over time, its growth rate must exceed 1.0. Species whose survival and reproduction fluctuate greatly with environmental conditions require high growth rates to avoid extirpation. The minimum growth rate needed to sustain Suckley's cuckoo bumble bee populations over time is unknown, but larger and more stable growth rates are generally an indication of higher levels of resiliency.

Survival and abundance have important implications for the persistence of populations. Invertebrates are particularly susceptible to extinction when populations are small (Gaston & Lawton, 1988). Small populations have lower levels of genetic diversity (heterozygosity) (Colla, 2016, p. 421), which reduces the capacity of the population to respond to environmental change and may lead to reduced population fitness, such as reduced longevity and fecundity, via inbreeding depression (Darvill et al., 2006, p. 608).

Genetic diversity

The genetic diversity of Suckley's cuckoo bumble bee populations has not been studied and further studies of both Suckley's cuckoo bumble bee and *Psithyrus* genetics are needed. It is likely that *Psithyrus* have distinct genetics, as the lineage has a distinct number of chromosomes (26 chromosomes) compared to other species in the genus *Bombus* (16-19 chromosomes) (Lhomme & Hines, 2019, p. 129). Given this, our understanding of genetics and potential constraints for Suckley's cuckoo bumble bee is limited and this section relies on our understanding of genus *Bombus*.

For bumble bees generally, genetic diversity contributes to population health, with one example being that bumble bees with increased genetic diversity have lower prevalence of some pathogens (Parsche & Lattorff, 2018, p. 900). Yet, the reproductive system of bumble bees renders them particularly sensitive to loss of genetic diversity. *Bombus* species are haplodiploid (i.e., males are haploid, and females are diploid) and exhibit a single locus complementary sex determination system (Zayed, 2009, p. 239). Typically, heterozygotes at the sex-determining locus develop into diploid females from fertilized eggs, while hemizygotes develop into haploid males from unfertilized eggs (Zayed, 2009, p. 239). In cases where females mate with haploid males that share a sex-determining allele in common (called "matched mating"), half of the females' progeny will be homozygous at the sex-determining locus and will consequently develop into diploid males instead of females. Diploid males are typically unviable, or if they are viable and mate, produce diploid sperm, which will lead to unviable fertilized eggs or sterile triploid daughters (Zayed, 2009, p. 239). Therefore, diploid males that survive and mate are unable to contribute to next year's cohort.

Matched mating occurs most often when allele diversity at the sex-determining locus is low (Ellis et al., 2006, p. 4376; Zayed, 2009, pp. 239–241). Probability of extinction due to inbreeding depression is an order of magnitude higher in haplodiploid populations than in diploid populations (Zayed & Packer, 2005, pp. 10743–10744). This high extinction risk is attributed to the effects of the "diploid male vortex;" a phenomenon where diploid males initiate a positive feedback cycle that leads to rapid extinction. Several species of bumble bee in England have demonstrated a dynamic consistent with this pattern; for example, despite continued suitability of habitat, *Bombus subterraneous* eventually was extirpated following reduction in population size due to habitat loss (Darvill et al., 2006, p. 608). Thus, maintaining genetic diversity within bumble bee populations is essential. Maintaining genetic diversity within populations requires enough gene flow within and among populations that are sufficiently large. Distribution of individuals across the population area increases viability by dissipating risk and enhancing the success of dispersal and gene flow within a population.

Population and Habitat Connectivity

Population connectivity promotes successful haplodiploid reproduction by increasing the likelihood of individuals finding unrelated mates. Dispersal of bees to find unrelated mates is aided by the proximity of other usurped colonies and the presence of suitable dispersal corridors. Bumble bee reproductive individuals (males and gynes) can disperse up to 10.0 km (6.2 mi) (Darvill et al., 2006, p. 606; Jha & Kremen, 2013, p. 2490; Lepais et al., 2010, p. 287). Dispersal distance can vary widely across species, and it has not yet been described for Suckley's cuckoo bumble bees. The cuckoo bumble bee *B. insularis* was found to disperse up to 7.0 km (4.3 mi) which is comparable to research on other *Bombus* species (Koch et al., 2021, p. 5). For successful dispersal to occur, Suckley's cuckoo bumble bees likely require habitat connectivity; the landscape needs to be permeable and free of hazards for unrelated gynes and males to successfully find enough forage flowers and find unrelated mates. Connectivity is a constraint for cuckoo bumble bees because they live in small, fragmented populations as a result of their dependence on host bumble bee colonies (Suhonen et al., 2016, p. 529).

In well-distributed populations, colonies occupy areas with diverse environmental conditions (spatial heterogeneity). Floral resources vary across space and time (seasonally and year-to-year) due to changes in climate factors, such as temperature and precipitation. Increased spatial heterogeneity across the population reduces the chances of all colonies failing concurrently due to poor environmental conditions.

4.3 Species Needs

4.3.1 Resiliency

Resiliency is the ability of a species to sustain populations in the face of environmental variation (e.g., localized fire and drought; Smith et al., 2018, p. 306). Resiliency of Suckley's cuckoo bumble bee is a function of the number of healthy populations and the distribution of these populations on the landscape. A healthy population is defined above (4.2 Population Needs). Generally, the greater number of healthy populations and the greater spatial heterogeneity occupied by the species, the greater likelihood of sustaining populations through time. Populations with high abundance, survival, connectivity, and genetic diversity are better able to withstand and recover from stochastic events and environmental variation. Thus, the greater the number and diversity of healthy populations, the more resiliency a species possesses. Furthermore, when numerous healthy populations are distributed across diverse environmental gradients, a species is less likely to experience synchronous environmental stochasticity and synchronous population declines; thus, the greater the number and diversity of healthy populations, the greater the overall resiliency at both the population, and species-level, and the lower the extinction risk.

4.3.2 Redundancy

Species-level redundancy reflects the ability of a species to withstand catastrophic events, and it is best achieved by having multiple, widely distributed populations relative to the spatial occurrence of catastrophic events (Smith et al., 2018, p. 306). In addition to guarding against a single or series of catastrophic event(s) extirpating all populations of Suckley's cuckoo bumble bee, redundancy is important to protect against losing irreplaceable sources of adaptive diversity. Having multiple populations distributed across the range of the species will help preserve the breadth of adaptive diversity and, hence, the evolutionary flexibility of the species. Thus, redundancy for Suckley's cuckoo bumble bee is described as having multiple, healthy populations widely distributed across the breadth of adaptive diversity relative to the spatial occurrence of catastrophic events.

4.3.3 Representation

Representation is the ability of the species to adapt to physical (e.g., climate conditions or habitat conditions across large areas) and biological (e.g., novel diseases, pathogens, and predators) changes in its environment now and into the future; it is the evolutionary capacity or flexibility of the species (Smith et al., 2018, p. 306). Representation is the range of variation found in a species, also called adaptive capacity. The adaptive capacity of Suckley's cuckoo bumble bee is a function of the amount and spatial distribution of genetic and phenotypic diversity. By maintaining these two sources of adaptive diversity across a species' range, the responsiveness and adaptability of a species over time is preserved.

Genetic diversity is the raw material upon which natural selection acts and is thus important for adapting to changing environmental conditions (Hendry et al., 2011, pp. 164–165). Species genetic diversity is influenced by the level of gene flow among populations and the rates of genetic drift (changes in the frequency of gene variants) within populations. Gene flow is influenced by the degree of connectivity among populations, and it is affected by a suite of factors including landscape permeability (Lankau et al., 2011, p. 320). No studies of within species genetic diversity have occurred for Suckley's cuckoo bumble bee and thus we recommend future research in this area (See 11.1 Suggestions). One study of the European cuckoo bumble bee *B. vestalis* showed that *Psithyrus* females have significantly lower genetic distances compared to within-group differences of the host species queens (Erler & Lattorff, 2010, p. 375). Another study of American cuckoo bumble bee *B. insularis* showed a lack of within-species diversity or structure across two study sites (Koch et al., 2021, p. 5).

Many bumble bee species exhibit high levels of gene flow at multiple spatial scales (Cameron, Lozier, et al., 2011, pp. 664–665; Lozier et al., 2011, pp. 4880–4882; Woodard et al., 2015, p. 2924) and therefore show little genetic structure at local or regional scales. Although, natural barriers (e.g., large water bodies) and elevation gradients can limit dispersal (Woodard et al., 2015, p. 2924), and hence, lead to genetic structuring. Based on genetic studies of other bumble bee species with similar ecologies, and given the potential dispersal capability, Suckley's cuckoo bumble bee may not exhibit much genetic differentiation across its broad range. Genetic variation can be negatively affected by genetic drift; small populations experience stronger drift (Zayed, 2009, p. 246). Thus, preserving the genetic diversity of Suckley's cuckoo bumble bee may require maintaining relatively large populations and connectivity among them.

Phenotypic diversity (physiological, ecological, and behavioral variation based on genetics/genetic makeup) is also important for adapting to changes in environmental conditions. Phenotypic variation determines how organisms interact with their environment and how they respond to selection pressures (Hendry et al., 2011, p. 161). The degree of phenotypic variation is determined by the diversity of physical and biological pressures to which organisms are exposed, which vary across spatial and temporal scales. As such, species that span environmental gradients are expected to harbor the most phenotypic and genetic variation (Lankau et al., 2011, p. 320). Thus, preserving the breadth of phenotypic diversity of Suckley's cuckoo bumble bee requires maintaining populations across historical latitudinal, longitudinal, and elevational gradients as well as climatic gradients. Doing so increases the likelihood that the species will retain the potential for adaptation over time. Suckley's cuckoo bumble bee representation is, therefore, described as having healthy populations widely distributed across a breadth of ecological conditions.

The adaptive capacity of Suckley's cuckoo bumble bee is complicated by its dependence on host species. The distribution of the parasitic bumble bee is restricted by the geographic distribution and population health of host bumble bees (Antonovics & Edwards, 2011, p. 1003). With changes in the climate, there is potential for phenological mismatch between host emergence and parasite emergence. Theoretically, *Psithyrus* such as Suckley's cuckoo bumble bee may be more vulnerable to changes in climate due to the phenological dependence on host bumble bees, however we currently lack specific studies on the topic (Bower et al., 2023, p. 521).

5 Influencing Factors

In this chapter, we review the factors (both negative and positive) that have influenced Suckley's cuckoo bumble bee historically, are influencing the species currently, and are likely to influence the species in the future. Factors which negatively impact bee individuals or colonies are generally referred to as stressors.

The following sections summarize the factors identified as likely to substantially impact Suckley's cuckoo bumble bee's viability, referred to as stressors. Numerous other stressors were identified by the listing petition, in the literature, or by experts that were concluded to be insubstantial to the overall resiliency of Suckley's cuckoo bumble bee populations. The stressors considered but not carried forward in our analysis are summarized in 11.3 Stressors Excluded from the Analysis.

5.1 Influence Diagram

The following influence diagram summarizes the factors that may substantially impact Suckley's cuckoo bumble bee viability (Figure 6). One of the key complexities to assessing the status of Suckley's cuckoo bumble bee is the compounding impact of stressors on the host species. Suitable host colony availability is a key habitat need of Suckley's cuckoo bumble bee and all the stressors that directly affect the parasite also affect the host species. See section 5.2.1 Host Species Declines for further discussion.

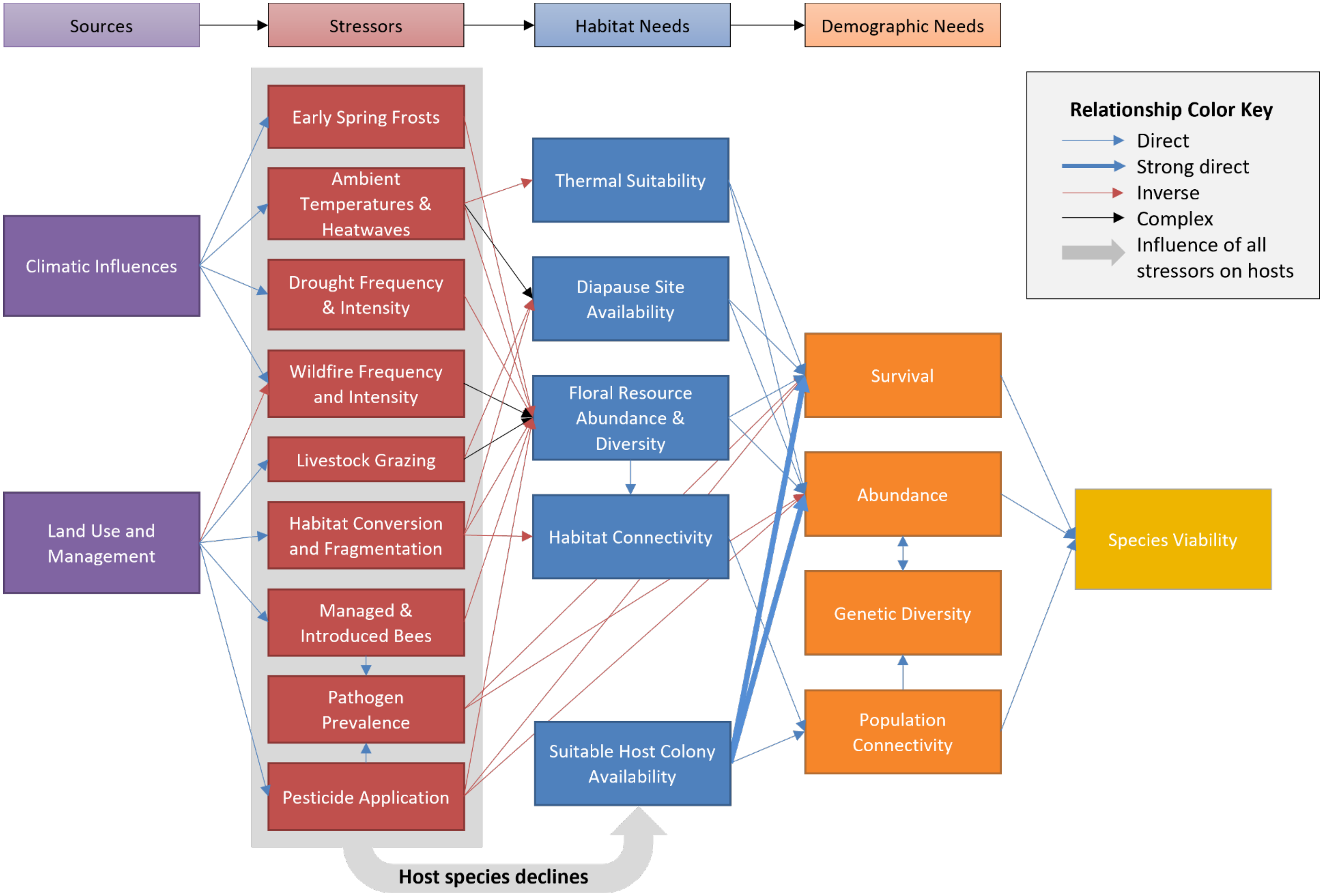

Figure 6. *Influence diagram of how sources and stressors can impact the habitat and demographic needs of Suckley's cuckoo bumble bee. Each box is linked together with colored arrows to show direction of the effect. See stressor descriptions in 5.2 Stressors.*

5.2 Stressors

There are numerous stressors affecting bumble bees but specific data on direct effects to Suckley's cuckoo bumble bee or *Psithyrus* are rarely available. The following sections summarize stressors to bumble bees generally and, when possible, describes direct and indirect effects to Suckley's cuckoo bumble bee.

5.2.1 Host Species Declines

Cuckoo bumble bees are obligate social parasites and thus entirely dependent on host colonies for their survival. Cuckoo bumble bee species have higher threat indices (higher extinction vulnerability) than hosts because they are entirely dependent on hosts (Suhonen et al., 2015, pp. 238–239) and the geographic range areas of cuckoo bumble bees are consistently smaller than the total geographic area of all their hosts combined (Suhonen et al., 2016, p. 532). The presence of parasitic bees depends on the presence of their hosts; any change to resources available to or stressor effects on the host will be reflected in the status of the parasite (Sheffield et al., 2013, p. 508). Because cuckoo bumble bees depend on host species, there is a co-extinction risk for host and parasite species (Suhonen et al., 2015, p. 238). Thus, signs that host species are declining are of major concern to the viability of Suckley's cuckoo bumble bee. The effect of all the stressors impacting Suckley's cuckoo bumble bee are compounded through the additional effects of theses stressors on host species availability (see Figure 6).

Suckley's cuckoo bumble bees parasitize numerous host species, as described in 4.1.4 Suitable Host Colony Availability. In assessments of the status of Suckley's cuckoo bumble bee, the decline of host species availability is cited as a primary threat (COSEWIC, 2019, p. vi; Hatfield et al., 2015, p. 4). The International Union for the Conservation of Nature (IUCN) reported decline of Suckley's cuckoo bumble bee is 77.18 percent and the regression trend mirrors that of the host *B. occidentalis* (Hatfield et al., 2014, p. 46, 2015, p. 3). Many species that Suckley's cuckoo bumble bee parasitizes have documented declines in the past century (Table 4). While one might think that having multiple host species increases redundancy and reduces parasite extinction risk, one study indicates generalist cuckoo bumble bees have a higher risk of extinction than specialist species (Suhonen et al., 2015, p. 238). One potential explanation is that the generalist parasites studied may have stronger compatibilities with some host species than others (Suhonen et al., 2015, p. 240)

Suckley's cuckoo bumble bee is in the bohemicus group of the subgenus *Psithyrus,* which preferentially parasitize bumble bees in the subgenus *Bombus sensu stricto* (Lhomme & Hines, 2019, p. 129). Trends observed across North America suggest subgenus *Bombus* species are experiencing widespread declines (Colla & Packer, 2008, p. 1387; Giles & Ascher, 2006, pp. 217–218; Janousek et al., 2023, p. 2; Schweitzer et al., 2012, p. 7). Host species *Bombus occidentalis* has lower genetic diversity compared with co-occurring populations of stable (non-declining) species (Cameron, Lozier, et al., 2011, pp. 664–666); there was not enough data to do a similar genetic diversity analysis for the other members of subgenus Bombus.

Table 4. *Conservation status and trends of Suckley's cuckoo bumble bee and all confirmed and potential host species. Species highlighted in blue belong to subgenus Bombus, which are considered the preferred hosts of the bohemicus group of cuckoo bumble bees.*

Species	**Global status[1]**	**IUCN red list[2]**	**USA federal listing status[3]**	**Canada federal listing status[4]**	**State statuses[1, **]**	**Other trend information**
B. suckleyi (Suckley's cuckoo bumble bee)	G2 Imperiled	Critically endangered. Average decline = 77.18%	In review	Threatened (2019)	SH – NY S1 – CA, MT, OR, WA S2 – CO, ID	-In Washington state, relative abundance declined 98.36% from historical (pre-2011) to current (2011-2021) time period[8].
B. occidentalis (western bumble bee; subgenus *Bombus*)	G3 Vulnerable	Vulnerable. Average decline = 40.32%*	In review	Threatened (2014)	S1 – CA, Or, UT S2 – AZ, WA S3 – CO, ID	-Detected range area reduction from historical records to 2009 = 28%[5] -Probability of occupancy declined 93% from 1998 to 2019.[6] -Over the past two decades, average occupancy declined 57% in 16 United States ecoregions.[7] -In Washington state, relative abundance declined 81.76% from historical (pre-2011) to current (2011-2021) time period[8].
B. nevadensis (Nevada bumble bee)	G4 Apparently secure	Least concern. Average decline = 15.48%	None	None	SH – NC S4 – CO, WA S5 – ID	
B. terricola (yellow-banded bumble bee; subgenus *Bombus*)	G3 Vulnerable	Vulnerable. Average decline = 49.94%	Not listed (2019)	Special concern (2015)	SX – IL SH – IN S1 – CT, MD, VA, WI S2 – MA, MI, WV S3 – ME, NC, NY, VT	-Detected range area reduction = 31%[5]
B. rufocinctus	G5 Secure	Least concern. No decline.	None	None	SH – NC S2 – AK S3 – ME, MI, NY, VT	

(red-belted bumble bee)					S4 – CO, WA, WI S5 – ID	
B. fervidus (yellow bumble bee)	G3 Vulnerable	Vulnerable. Average decline = 29.71%	None	None	S1 – AR, MO S2 – CT, MA, ME, VT, WI S3 – IN, MI, NC, NY, OR, VA, WV S4 – WA S5 – ID	-In Washington state, relative abundance declined 43.33% from historical (pre-2011) to current (2011-2021) time period[8].
B. appositus (white-shouldered bumble bee)	G3 Vulnerable	Least Concern. Average decline = 28.40%	None	None	S4 – CO, ID, WA	
B. affinis (rusty-patched bumble bee; subgenus *Bombus*)	G2 Imperiled	Critically endangered. Average decline = 69.36%	Endangered (2017)	Endangered (2022)	SX – CT, MA, VT SH – GA, KY, ME, MI ND, NH, NY S1 – IL, IN, MD, NC, OH, PA, TN, VA, WI, WV	-Detected range area reduction = 87%[5]
B. mckayi (McKay's bumble bee; subgenus *Bombus*)	G4 Apparently Secure	Vulnerable. *	In review	Special concern (2014)	S4 – AK	
B. cryptarum (cryptic bumble bee; subgenus *Bombus*)	G5 Secure	Data deficient.	None	None	S4 – AK	

[1]NatureServe 2023, [2] Hatfield et al. 2014, [3] USFWS 2023, [4] COSEWIC 2023, [5]Cameron et al. 2011, [6]Graves et al. 2020, [7]Janousek et al. 2023, [8]Martin et al. 2023.

*Bombus mckayi and B. occidentalis assessed as one species by Hatfield et al. 2014. The subspecies *mckayi* was recently elevated to species status (Williams 2021, p. 278).

** SX = presumed extirpated, SH = possibly extirpated, S1 = critically imperiled, S2 = imperiled, S3 = vulnerable, S4 = apparently secure, S5 = secure

5.2.2 Managed and Introduced Bees

Throughout this assessment, we use the term *managed bees* as a broad term inclusive of a wide variety of bee species and management practices. Managed bees are a threat to Suckley's cuckoo bumble bee in two primary contexts – competition for resources and as a pathway for pathogen introduction and spread (Fürst et al., 2014, p. 365; Goulson et al., 2008, p. 193; Goulson, 2010b, p. 10; Goulson et al., 2015, p. 11; D. M. Thomson, 2004, p. 467; Winter et al., 2006, entire). While individual Suckley's cuckoo bumble bees do need floral resources for nectar and pollen, competition for resources likely more directly affects the hosts for Suckley's cuckoo bumble bees, such as *B. occidentalis*, whose colonies may have hundreds of foraging workers. Also, managed bees may potentially play a role in the spread of other invasive species (particularly the introduction of invasive plants and present a hybridization risk in certain situations.

Generally, the term *managed bees* describes hives or colonies of bees that are used commercially to provide pollination services for a wide variety of crops over the growing season, with some hives or colonies moved within and between states multiple times throughout any one growing season. Managed bees also include hobby or backyard bee keeping, as well as small-scale greenhouse operations. Managed bees include many different species of bees, some of which are crop specialists. European honey bees (*Apis mellifera*) are the main managed bee, but *managed bees* also include alkaline bees (*Nomia melanderi*) and leaf-cutting bees (*Megachile rotundata*), both of which are used to pollinate alfalfa; and blue orchard bees (*Osmia lignaria*), which are used to pollinate early-bloom tree fruit, such as apples (Ratti et al., 2008, pp. 3, 5). *Managed bees* also include several species of native North American bumble bees, including common eastern bumble bee (*B. impatiens*), an eastern species that has recently been moved into the range of Suckley's cuckoo bumble bee for commercial pollination services (Palmier & Sheffield, 2019, p. 9).

Managed beehives and colonies that are moved around the landscape to provide pollination services are most often open, such as hives placed in orchards, and allow managed bees to freely move around the environment. *Bombus* species that are commercially raised and used for greenhouse production commonly escape from greenhouses (Ratti & Colla, 2010, p. 30). Studies have shown that where honey bee density increases, populations of wild *Bombus* decrease as a result of competition, even when controlling for annual variation in other factors (D. M. Thomson, 2016, p. 1251). Another study found that proximity to higher densities of managed *Apis* hives reduced *B. occidentalis* colony reproductive success, as measured by gyne number, gyne ratio, mean gyne size, and male observations (D. M. Thomson, 2004, p. 464). The same study also found reduced pollen foraging effort, suggesting that *Bombus* colonies located closer to *Apis* hives experienced greater nectar scarcity and reallocated foragers from pollen collection for larval rearing to nectar collection for meeting basic colony energy requirements. Increasing the density of honey bees in agricultural fields has various competitive effects on different species of *Bombus*, but short-tongued, generalist species may be able to shift their foraging behavior in response to competition (Walther-Hellwig et al., 2006, pp. 526–527).

Wild bumble bees located more closely to managed honey bee colonies had significantly higher disease rates than wild bumble bees located farther away (Alger et al., 2019, p. 5). Infection rates for several bumble bee pathogens were higher in multiple *Bombus* species near commercial greenhouses where managed bees were used than in areas located far away from commercial greenhouses (Colla et al., 2006, pp. 462–464). Recent research has also found that several RNA viruses, previously thought to be specific to honey bees, have been detected in bumble bees including *B. terrestris* and *B. pascuorum*

(Alger et al., 2019, p. 5; Genersch et al., 2006, entire; Meeus et al., 2011, entire), and that transmission occurs when species forage on the same floral resources (Singh et al., 2010, p. 6). See 5.2.3 Pathogen Prevalence for further discussion of pathogen transmission.

Within the range of Suckley's cuckoo bumble bee, managed bees are used to provide pollination services for a wide variety of crops including, but not limited to, alfalfa, almonds, apples, avocado, canola, cherries, sunflowers, squash, melon, berries, cucumbers, and clover (Bond et al., 2014, entire). The use of managed bees is expanding in some portions of the range of Suckley's, including western Canada (COSEWIC, 2019, p. 49). Nearly all movement of managed bees is unregulated and does not require permits; insufficient data is currently available for us to include a spatial representation of managed bees in our model and analysis. However, managed bees are widespread within our analysis area, and an increasing threat in some areas, posing a threat to Suckley's cuckoo bumble bee and its hosts.

Bombus insularis, another cuckoo bumble bee, has been documented attempting to invade honeybee hives, though it is unclear if this is related to competition among *Psithyrus*, an attempt to steal resources, or some other factor (Strange et al., 2014, p. 556).

5.2.3 Pathogen Prevalence

Bumble bees are susceptible to a variety of pathogens and parasites including fungi, viruses, bacteria, protozoa, tracheal mites, and nematodes. Many of these pathogens and parasites are widespread and cause lethal and sublethal effects for one or more *Bombus* species, including Suckley's cuckoo bumble bee and its known and likely hosts. However, no in-depth studies specific to Suckley's cuckoo bumble bee parasitology currently exist (Dozier et al., 2023, p. 642). We lack spatial data on the spread of various bumble bee diseases (Janousek et al., 2023, p. 7), so we are currently unable to model this stressor.

5.2.3.1 Fungal microsporidians

The fungal pathogen *Vairimorpha bombi* (formerly *Nosema bombi*; Tokarev et al., 2020, p. 11) was historically present and widespread in North America (Cordes et al., 2012, p. 215; Goulson et al., 2015, p. 3; Graystock et al., 2016, p. 71). This pathogen infects the excretory and water-regulating organs (malpighian tubules), fat bodies, midgut, connective tissue, and tracheal system of bumble bees (Macfarlane et al., 1995, p. 135). Bumble bees that are infected may have disabled wings, and queens may have distended abdomens and be incapable of mating (Otti & Schmid-Hempel, 2007, p. 122). *Vairimorpha bombi* is spread by bumble bee workers, larvae, queens, and gynes through fecal-oral contact at flowers that are shared with bees from other nests and within the nest, and through contaminated pollen or nectar fed to the larvae (Rutrecht et al., 2007, pp. 1719–1721). The negative effects of this pathogen on bumble bees ultimately leads to a reduction in colony growth and fitness (Cameron, Lozier, et al., 2011, pp. 664–666; Colla et al., 2006, p. 465; Meeus et al., 2011, p. 666; Otti & Schmid-Hempel, 2007, pp. 118–123; Rutrecht & Brown, 2008, pp. 505–511). It has been documented in Suckley's cuckoo bumble bee, one confirmed host species, *B. occidentalis*, and four suspected host species (*B. affinis, B. fervidus, B. rufocinctus, B. terricola;* (Cordes et al., 2012, p. 212; Kissinger et al., 2011, p. 222).

Vairimorpha ceranae (formerly *Nosema ceranae*; Tokarev et al., 2020, p. 11) is another fungal pathogen that infects honey bees and bumble bees (commercial and wild), is highly virulent, and causes rapid mortality (Graystock, Yates, Darvill, et al., 2013, pp. 116–117). Like *V. bombi*, *V. ceranae* can be present in the midgut and also in the fat body of bumble bees, indicating that the parasite moves between tissues (Graystock, Yates, Darvill, et al., 2013, p. 116). Its introduction, spread, and prevalence has increased over time and it is found everywhere that the European honey bees occur (Colla, 2016, p. 419; Fürst et al., 2014, 2014, p. 364; Goulson et al., 2015, p. 2; Graystock et al., 2016, p. 68). Bumble bees can become infected with V. *ceranae* by ingesting spores left on flowers that have been visited by bees with *V. ceranae* (Graystock, Yates, Darvill, et al., 2013, p. 116). Effects on bumble bees can be both lethal and sublethal, including suppressed immune response and increased mortality (Graystock, Yates, Darvill, et al., 2013, pp. 116–117; Graystock et al., 2014, p. 9, 2015, p. 6, 2016, p. 68; Rotheray et al., 2017, p. 294; U.S. Fish and Wildlife Service, 2018, p. 52). This pathogen has not been documented in Suckley's cuckoo bumble bee.

5.2.3.2 Protozoan parasites

Crithidia bombi is a trypanosome protozoan parasite that affects the intestines of bumble bees. The cells attach to the gut wall and multiply (P. Schmid-Hempel, 2001, p. 150). Similar to *V. bombi*, the parasite is transmitted when ingested through fecal-oral contact and then spread when bees share flowers (Cordes et al., 2012, p. 210; Kissinger et al., 2011, p. 223; Otterstatter & Thomson, 2008, p. 2; P. Schmid-Hempel, 2001, p. 150) or when the next host comes into contact with infected cells on the brood comb or other nest materials (P. Schmid-Hempel, 2001, p. 150). *C. bombi* does not appear to have been introduced by an exotic species, but it is unclear if the parasites prevalence can be attributed to spillover from greenhouses (Cordes et al., 2012, pp. 212–214; Otterstatter & Thomson, 2008, p. 2). Bees from commercial rearing facilities have tested positive for this pathogen upon delivery (Murray et al., 2013, p. 274; Otterstatter et al., 2005, p. 388). The prevalence and intensity of *C. bombi* infections in wild bumble bees has been found to decrease with increasing distance from greenhouses with commercial bumble bee colonies (Colla et al., 2006, pp. 462–466; Graystock et al., 2016, p. 69; Otterstatter & Thomson, 2008, pp. 2–7), although infected bumble bees in the eastern United States have been found in relatively isolated areas that were not near commercial greenhouses (Gillespie, 2010, p. 743; Malfi & Roulston, 2014, p. 24). Nutritionally stressed worker bees infected with *C. bombi* were found to have a 50 percent higher mortality rate than well-fed bees infected with *C. bombi*, which contributed to colony stress and decline, and emergent queens infected with the parasite have delayed colony initiation in the spring (Brown et al., 2000, pp. 424–425, 2003, pp. 997–1000; Kissinger et al., 2011, p. 223). Effects of *C. bombi* infection include reduced colony founding success, lower colony fitness and growth, delayed reproduction, and inadequate fat reserves in hibernating queens (Shykoff and Schmid-Hempel 1991, p. 242; Schmid-Hempel 2001, pp. 148, 150–154; Brown et al. 2003, pp. 995–1000; Graystock et al. 2014;). *C. bombi* has not been documented in Suckley's cuckoo bumble bee.

Crithidia expoeki is protozoan parasite that has been detected in the contiguous United States and Mexico (Gallot-Lavallée et al., 2016, p. 75; Tripodi et al., 2018, entire), and in Alaska and Switzerland (R. Schmid-Hempel & Tognazzo, 2010, p. 337). It may be as widespread as *C. bombi* (Tripodi et al., 2018, p. 151), but additional data is needed to confirm its distribution and hosts (Palmier et al., 2020, p. 17). Of 30 bumble bees collected from two sites in Saskatchewan, 58 percent tested positive for *C. bombi* and

25 percent for *C. expoeki*. *Crithidia* spp. have not been documented in Suckley's cuckoo bumble bee, but have been documented in its known host in the western portion of the range, *B. occidentalis* (Cordes et al. 2012, p. 212).

Apicystis bombi is a protozoan parasite in honey bees, commercial bumble bees, and wild bumble bees in North America, South America, Europe, and Asia (Graystock et al., 2015, p. 1), and it is classified as an emerging infectious disease (Sachman-Ruiz et al., 2015, p. 2044; Wilfert et al., 2016, pp. 595–596). The feeding of commercially-reared bumble bees with pollen that has been collected and processed by honey bees has provided an exposure route for this bee pathogen (Goulson et al., 2015, p. 3; Graystock et al., 2016, p. 71), and commercially-sourced colonies of *B. terrestris* were found to harbor this parasite (Graystock, Yates, Evison, et al., 2013, p. 1208). It is also thought to be spread via fecal-oral transmission and perhaps from infected queens to eggs (Goulson and Hughes 2015, p. 13; Graystock et al. 2015, p. 1). Impacts of this pathogen on bees can include degradation of body fat, neurological effects, reduced overwinter survival, and mortality in adults (Macfarlane et al. 1995, p. 134; Goulson and Hughes 2015, p. 13; Graystock et al. 2015, p. 6). This disease has not been documented in Suckley's cuckoo bumble bee, but it has been documented in one of its known hosts, *B. nevadensis* (Maxfield-Taylor et al., 2011, p. 4).

5.2.3.3 Tracheal mites

Locustacarus buchneri is a widespread bumble bee parasite that has been reported in approximately 25 bumble bee species (Otterstatter & Whidden, 2004, p. 353), and it can be prevalent in both wild and commercial bumble bee colonies (Colla et al., 2006, pp. 462–466). The mites feed on the hemolymph (blood-like fluid) of their hosts (Goka et al., 2006, p. 287; Goulson, 2010a, p. 72; Otterstatter & Whidden, 2004, pp. 352–356). They overwinter in the trachea of young queens, then reproduce inside them in the spring. The larvae from female mites spread to other bees (Goulson, 2010a, p. 72; Husband & Sinha, 1970, p. 1161; Otterstatter & Whidden, 2004, pp. 352–356). A single female mite can be associated with as many as 50 eggs (Goka et al., 2006, p. 286). Mites can also spread from infected bee colonies to uninfected colonies via shared food sources, when infected workers enter colonies other than their own ("drifting workers"), or when infected males mate with uninfected queens (Colla et al., 2006, p. 465; Husband & Sinha, 1970, p. 1161; Otterstatter & Whidden, 2004, pp. 351, 355).

Global redistribution of various strains of *L. buchneri* may have occurred due to the worldwide trade in bumble bees (Goka et al., 2006, p. 285; Goulson, 2010a, p. 188), and there is evidence from Canada that commercial bumble bee populations were spreading *L. buchneri* to wild populations (Colla et al., 2006, pp. 463–465). The negative effects of *L. buchneri* on its hosts include physical damage to the trachea resulting in reduced fitness and colony growth, shortened lifespan of individual bees, and cessation of foraging in worker bees (Husband & Sinha, 1970, pp. 1160–1161; Otterstatter et al., 2005, pp. 383, 385; Otterstatter & Whidden, 2004, p. 354). *L. buchneri* has not been documented in Suckley's cuckoo bumble bee, but it has been documented in one of its known hosts, *B. occidentalis*, and in a suspected host, *B. terricola* (Otterstatter & Whidden, 2004, pp. 353–355).

5.2.3.4 RNA viruses

RNA viruses exhibit high mutation rates and short generation times; as such, they are likely to cross species barriers and adapt rapidly to new host environments (Alger et al., 2019, p. 2). Several honey bee viruses (deformed wing virus, black queen cell virus, sacbrood virus, Kashmire bee virus, Israeli acute paralysis virus, acute bee paralysis virus) are known to infect bumble bees (Robson-Hyska, 2017, pp. 124–125; Singh et al., 2010, p. 8; Tehel et al., 2022, p. 4). These viruses have not been documented in Suckley's cuckoo bumble bee, but they have been documented in several of its likely hosts.

Deformed wing virus has been cataloged as an emerging infectious disease (Fürst et al., 2014, p. 365; Wilfert et al., 2016, p. 596). It has been detected in wild and commercially-sourced bumble bees (Amiri et al., 2018, p. 2; Barroso-Arévalo et al., 2019, p. 1). Honey bees can test positive for deformed wing virus and be asymptomatic, whereas bumble bees that test positive for this virus are symptomatic (i.e., manifest deformed wings), suggesting that deformed wing virus is more virulent in bumble bees than in honey bees (Genersch et al., 2006, p. 63). Infection with deformed wing virus can also result in non-viable offspring and reduced longevity in bumble bees (Fürst et al., 2014, p. 364). This virus has been documented in a potential Suckley's cuckoo bumble bee host, *B. rufocintus* (Robson-Hyska, 2017, pp. 63–65).

Black queen cell virus has been detected in at least 10 bumble bee species (Alger et al., 2019, p. 5; Manley et al., 2015, p. 333; McMahon et al., 2015, p. 619; Peng et al., 2011, p. 650). It causes mortality of queen larvae and pupae, and it can infect worker bees, but worker bees do not normally exhibit outward disease symptoms (Chen & Siede, 2007, p. 38). As with deformed wing virus, its infectivity and pathology are not well understood (Chen & Siede, 2007, pp. 38–39; McMahon et al., 2015, p. 622). This virus has been documented in a potential Suckley's cuckoo bumble bee host, *B. rufocintus* (Robson-Hyska, 2017, pp. 64–65).

As with the other honey bee viruses, *sacbrood, Kashmire bee virus, Israeli acute paralysis virus, and acute bee paralysis virus* result in various negative effects, including death in honeybees (Brutscher et al., 2016, p. 2; Grabensteiner et al., 2001, p. 1). The effects of these viruses to bumble bees are unknown, but these diseases have been documented in several potential hosts of Suckley's cuckoo bumble bee including *B. fervidus, B. rufocintus*, and *B. terricola* (Robson-Hyska, 2017, pp. 64–65, 124–125).

5.2.3.5 Nematodes

Sphaerularia bombi is a parasitic nematode. Infection by this species can result in multiple negative effects to bumble bee queens such as a change in gene expression affecting energy usage, translation, and circadian rhythm (Colgan et al., 2020, p. 170), and in endocrine gland function involved in growth and development in the larva and pupa (Maxfield-Taylor et al., 2011, p. 134). Further, queens infected with *S. bombi* have ovaries that are undeveloped (Maxfield-Taylor et al., 2011, p. 134) inhibiting them from establishing nests and laying eggs (Poinar & Van Der Laan, 1972, p. 239). *S. bombi* has been reported in 15 *Bombus* species in North America (Maxfield-Taylor et al., 2011, pp. 134–135). In Nova Scotia, the prevalence of *S. bombi* infections for seven species of *Bombus* was 22 percent (McCorquodale et al., 1998, pp. 879–881). *S. bombi* has not been documented in Suckley's cuckoo bumble bee, but it has been documented in one of its known hosts, *B. occidentalis* (Poinar, 1974, p. 305).

5.2.4 Pesticide Application

Throughout this assessment, we use the term *pesticides* as a broad term inclusive to a wide variety of human-made chemicals that are generally used to control plant and animal pests. Pesticides include herbicides, insecticides, fungicides, miticides, rodenticides, and adjuvants. Adjuvants are compounds added to many pesticides at the time of application to enhance the penetration, spread, and duration of effect of the active ingredient(s) (Fine et al., 2017, p. 1).

Lethal and sub-lethal effects on bumble bee eggs, larvae, and adults have been well documented for many different pesticides under various scenarios (Table 5). Bumble bees, which nest underground, are exposed to pesticide residues in the soil which can accumulate higher toxicity levels when treated with pesticides multiple times (DiBartolomeis et al., 2019, p. 21). Bumble bee larvae consume large amounts of unprocessed pollen, which pose another source of exposure to pesticide residues (Arena & Sgolastra, 2014, p. 333). Studies have also found evidence of adverse impacts to bumble bee habitat associated with herbicide use. Specifically, they found changes in vegetation and the loss or reduction of flowers needed to provide consistent sources of pollen, nectar, and nesting material (Johansen, 1977, p. 188; Kearns et al., 1998, pp. 91–92; Kearns & Inouye, 1997, p. 302; Plowright et al., 1978, p. 1145; Smallidge & Leopold, 1997, p. 264). Rodenticides such as aluminum phosphide, zinc phosphide, diphacinone, chlorophacinone, and strychnine may reduce rodent populations in some habitats, which in turn could limit nesting habitat for the hosts on which Suckley's cuckoo bumble bee depends (Witmer et al., 2012, p. 210).

Table 5. *Summary of lethal and sub-lethal effects of pesticides on bumble bees. Pesticide effects are not limited to those listed here.*

Pesticide effect	Reference(s)
Reduced or lack of male production	(Fauser-Misslin et al., 2014, pp. 453–454; Feltham et al., 2014, p. 320; Gill et al., 2012, p. 107; Mommaerts et al., 2006, pp. 754–755; Scholer & Krischik, 2014, p. 7)
Reduced or lack of egg hatching	(Elston et al., 2013, pp. 6–7; Mommaerts et al., 2006, pp. 754–755)
Reduced queen production	(Fauser-Misslin et al., 2014, pp. 453–454; Feltham et al., 2014, p. 320; Whitehorn et al., 2012, p. 352)
Reduced queen longevity	(Fauser-Misslin et al., 2014, pp. 453–454)
Loss of workers and reduced worker biomass	(Gels et al., 2002, p. 726)
Reduced colony weight	(Feltham et al., 2014, p. 320; Gels et al., 2002, p. 726; Scholer & Krischik, 2014, p. 6; Whitehorn et al., 2012, p. 351)
Reduced brood size/production	(Elston et al., 2013, p. 6; Feltham et al., 2014, p. 320; Gill et al., 2012, p. 107; Laycock et al., 2012, p. 3; Laycock & Cresswell, 2013, p. 4; Mommaerts et al., 2006, p. 756)
Reduced feeding/foraging	(Fauser-Misslin et al., 2014, pp. 453–454; Feltham et al., 2014, p. 320; Gill et al., 2012, p. 107; Gill & Raine, 2014, pp. 211–212; Morandin et al., 2005, pp. 623–624; Scholer & Krischik, 2014, p. 5)
Impaired ovary development	(Baron et al., 2017, p. 4; Laycock et al., 2012, pp. 4–5)

Impaired sperm viability and feeding gland function	(Minnameyer et al., 2021, p. 7)
Reduced ability to learn and remember	(Siviter et al., 2018, p. 2816)
Reduced foraging motivation	(Lämsä et al., 2018, p. 4)
Weakened resistance to disease	(Manley et al., 2015 entire)

Bumble bee exposure to pesticides can also occur from direct spray or drift (Johansen and Mayer 1990, entire), or from gathering or consuming contaminated nectar or pollen (Morandin et al., 2005, p. 619). Systemic pesticides are one source of contamination because the chemicals are absorbed by the plant and can be transported to the pollen and nectar. The timing of exposure to pesticides affects the magnitude of their potential impacts. Most pesticides are applied to the landscape during the bumble bee active season. Bumble bees are especially susceptible to pesticides applied early in the year because, for approximately one month every year, the entire bumble bee population depends on the success of the queens to forage and establish new colonies. However, the loss of worker bees is also concerning. Bumble bees have smaller colonies relative to honey bees, which increases the relative impact of individual worker losses to colony survival (Thompson & Hunt, 1999, p. 155). As an obligate parasite that relies on the workers of the parasitized host colony, the loss of individual workers would also have cascading impacts on Suckley's cuckoo bumble bee.

5.2.4.1 Neonicotinoids

Neonicotinoid pesticides have been strongly implicated in the decline of several *Bombus* species due to the correlated introduction of neonicotinoid insecticides and the precipitous decline of those species (Colla & Packer, 2008, p. 10; Goulson, 2013, pp. 7–8; Pisa et al., 2015, p. 69). Most studies examining the effect of neonicotinoids on bees have been conducted using the European honey bee (*Apis mellifera*) and a handful of *Bombus* species (Camp & Lehmann, 2021, p. entire; Lundin et al., 2015, p. 7).

Neonicotinoids are based on nicotine compounds; they are systemic insecticides that act as a neurotoxin at varying levels of toxicity, affecting the central nervous system of insects. The neonicotinoid family of insecticides includes: acetamiprid, clothianidin, imidacloprid, nitenpyram, nithiazine, thiacloprid, and thiamethoxam. Neonicotinoids kill insects by interfering with the receptors of their nervous systems, causing overstimulation, paralysis, and death (Buszewski et al., 2019, p. 34728). Sub-lethal effects of neonicotinoids to bumble bees can include impairments to reproduction. Thiamethoxam impairs ovary development (Baron et al., 2017, p. 4), and imidacloprid reduces reproductive success and the production of reproductive females (Raine, 2018, p. 40; Whitehorn et al., 2012, pp. 351–352; Wu-Smart & Spivak, 2018, pp. 4–5). Because Suckley's cuckoo bumble bees produce only reproductive offspring, they may be especially vulnerable to the effects of neonicotinoids which impair reproductive function.

Neonicotinoids are used in a wide variety of agricultural applications, including common use as seed coatings in corn, wheat, soybeans, and cotton (A. Alford & Krupke, 2017, p. 1). These prophylactic seed treatments can increase toxic loads in soils, increasing risk of bumble bee exposure (DiBartolomeis et al., 2019, p. 22). Suckley's cuckoo bumble bee have been observed in and around regions of agricultural production, including those involved in the production of crops commonly treated with neonicotinoids.

5.2.4.2 Adjuvants

Pesticide formulations typically contain less than 50 percent active ingredients with the remainder being surfactants (surface active agent that reduces the surface tension of water) and solvents, collectively referred to as adjuvants. As bees forage, they are exposed to many pesticide adjuvants as well as the active ingredients (Mullin et al., 2015, p. 7). Adjuvants, however, are not typically included in risk assessments that are required for pesticide registration (Mullin et al., 2015, p. 2) and are generally considered by the Environmental Protection Agency to be biologically inert (Fine et al., 2017, p. 1). Therefore, adjuvants are less studied, but can be as or more toxic to bees as the active ingredients (Mullin et al., 2015, p. 4). For example, bumble bees are highly susceptible to emulsifiers such as perfluorooctane sulfonic acid (Mommaerts et al., 2011, pp. 450–452). Organosilicones are a nonionic class of surfactant that has been shown to have a toxic effect on honey bees (Fine et al., 2017, p. 1). Four of 11 commercially available spray adjuvants were toxic to honey bees when applied at field rates (Goodwin & McBrydie, 2000, p. 232). Pesticide active ingredients and adjuvants may also interact synergistically, causing impacts that would not occur by exposure to the active ingredients alone (Mullin et al., 2015, p. 3).

5.2.5 Habitat Conversion and Fragmentation

Throughout North America, habitat historically occupied by bumble bees has been lost, degraded, or fragmented. The conversion of natural habitat to agricultural and urban areas is the primary cause of bumble bee habitat loss (Goulson et al., 2015, p. 2). Other factors contributing to the loss or degradation of forested habitat include increased parcelization and fragmentation of land; deterioration of forests from introduced pests and pathogens; and unsustainable land management practices in some areas (Mola et al., 2021).

Suckley's cuckoo bumble bee is associated with a wide variety of habitats including prairies, grasslands, meadows, and woodlands as well as urban and agricultural areas (COSEWIC, 2019, p. 26; Martin et al., 2023, p. 22; Montana Natural Heritage Program, 2023, p. 3). They require host nests and overwintering, mating, and foraging habitat areas that connect, overlap, and are located relatively near to each other. Habitat conversion and fragmentation from intensive agriculture (generally, agriculture which is designed for increased yield, less frequent fallow, and which relies at least in part on technology) and urbanization are recognized as having significant impacts to many *Bombus* species and likely contribute to current bee declines (Colla, 2016, p. 419; Duchenne et al., 2020, p. 6762; Goulson et al., 2015, p. 1; Grixti et al., 2009, pp. 79–81; Hatfield & LeBuhn, 2007, pp. 154–157; Hines & Hendrix, 2005, pp. 1481–1483). Habitat loss is commonly cited as a long-term contributor to bee declines through the 20th century, and it may continue to contribute to current declines, at least for some species (Brown & Paxton, 2009, pp. 411–412; Goulson et al., 2008, pp. 191–198, 2015, p. 2). Suckley's cuckoo bumble bee and the confirmed host species, being generalist foragers, may not be as severely affected by habitat loss compared to habitat specialists, such as native prairie endemics, but loss or degradation of habitat reduces bee diversity and abundance (Potts et al., 2010, pp. 348–349), and small, isolated patches of habitat may not be sufficient to support healthy bee populations (Hatfield & LeBuhn, 2007, pp. 154–156; Öckinger & Smith, 2006, pp. 55–56).

Habitat conversion and fragmentation reduce the amount or accessibility of suitable host nests and foraging and overwintering habitat. They also reduce the connectivity required for healthy populations to expand in response to environmental or demographic changes and to maintain genetic diversity. High bumble bee species richness and abundance are associated with diverse floral resources, particularly when surrounded by a complexity of natural habitats across the landscape (Hatfield & LeBuhn, 2007, pp. 154–157; Hines & Hendrix, 2005, pp. 1481–1483). Due to their dispersal characteristics, long colony cycle, and nesting and overwintering requirements, bumble bees are sensitive to the negative effects of habitat fragmentation (Grixti et al. 2009, p. 79).

Habitat conversion leads to the reduction of abundant and diverse floral resources. We lack information specific to Suckley's cuckoo bumble bees, however adverse effects are well documented for bumble bees in general. The lack of sufficient nutritional resources can reduce colony growth, health, and reproduction, and negatively influence long-term bee populations (Vaudo et al., 2015, p. 4040). Food shortfalls because of habitat loss can induce longer larval development, production of smaller and fewer individuals, and an early shift to male production (Beekman & van Stratum, 1998, entire; Sutcliffe & Plowright, 1990, pp. 1056–1057). Nutritional stress caused by habitat loss can affect learning and memory that can lead to reduced foraging efficiency, increased competition, and overall decline in colony fitness (Colla, 2016, p. 413; Townsend-Mehler & Dyer, 2011, pp. 275–286). Larval and colony growth can be significantly affected by pollen type (plant species), pollen diversity, and the varying nutritional quality and quantity. Differences in colony development, growth, and fitness between bumble bee species have been observed when they are fed the same diet and under the same controlled rearing conditions (Colla, 2016, pp. 416–417; Tasei & Aupinel, 2008, pp. 397–409).

Monoculture farming is another factor that impacts plant community changes and thus reduces nesting opportunities for host colonies (Kearns & Inouye, 1997, p. 298). Agricultural manipulation and changes across various landscape types have likely impacted the availability of host nest sites in North America in the 20th century due to habitat degradation, modification, conversion, and loss (Goulson, 2003, p. 142). These are important changes because diet breadth and coexistence in bumble bees can become limited due to habitat loss (Goulson et al., 2008, pp. 193–200) and coexistence is important for parasitic species like Suckley's cuckoo bumble bee that rely on host colonies to raise their young. Decreases in foraging habitat increases competition, because there is overlap in resources that bumble bee species use (Goulson et al., 2008, p. 196).

While habitat conversion and fragmentation are well documented throughout the range of Suckley's cuckoo bumble bee, recent observations of Suckley's cuckoo bumble bee, *B. occidentalis,* and *B. affinis* do occur in urban and agricultural settings. As outlined above, these areas may not represent high-quality habitat, and records of these species in these habitats may represent range contraction (U.S. Fish and Wildlife Service, in review).

5.2.6 Changes in Ambient Temperatures and Heatwaves

Global annual surface temperatures have risen an average of 0.09˚C (0.17˚F) each decade from 1901 to 2020 (U.S. Environmental Protection Agency, 2021b, n.p.). Temperature increases in the contiguous

United States since the late 1970s have surpassed the global rates for that period, increasing from 0.17˚C to 0.30˚C (0.31˚F to 0.54˚F) each decade. The northern and western parts of the United States and Alaska have experienced the greatest increases (U.S. Environmental Protection Agency, 2021b, n.p.); this represents the bulk of the range of Suckley's cuckoo bumble bee. Temperatures in the arctic have increased at twice the global rate and they are expected to continue to increase at a higher rate than the global average (Hayhoe et al., 2018, pp. 91–92; IPCC, 2014, p. 60). Climate change is contracting temperate, arctic, and alpine zones (Staten et al., 2018, p. 770) in which bumble bees are distributed and to which they are adapted (Goulson, 2010a, p. 2). In addition to increasing average temperatures, heatwaves in the United States have become more frequent, more intense, and longer in duration and have occurred over a longer season (U.S. Environmental Protection Agency, 2021a, n.p.).

Temperature influences events in bumble bees' lifecycles such as diapause, emergence, foraging, and colony maintenance (see 4 Species Ecological Needs). Rising ambient temperatures and heatwaves can negatively affect bumble bee individuals and colonies by reducing survival, increasing energy expenditures, reducing flight and foraging, and reducing reproduction (see 'upper thermal limits' in 4.1.1 Thermal Suitability). A study of ten bumble bee species in North America found that queens emerged from diapause on average 10.4 days earlier per decade from 1880 to 2010, with the biggest shift occurring since 1970 (Bartomeus et al., 2011, p. 20645). Entering diapause may also be affected by temperature changes due to climate change (Maebe et al., 2021, p. 4229). Intrinsic characteristics determine the extent to which Suckley's cuckoo bumble bee can adapt physiologically to tolerate warming temperatures (see 4.1.1 Thermal Suitability). Responding to the changes could require physiological adaptations to persist in habitats despite warming temperatures and behavioral ability to disperse to and colonize new areas that become suitable (Maebe et al., 2021, pp. 4230–4231).

Bumble bees have low variation in heat stress resistance and therefore may have low capacity to adapt physiologically to warming temperatures (Martinet et al., 2021, p. 7). Bumble bee species that occur across a relatively broad climatic range, such as Suckley's cuckoo bumble bee, likely have a greater capacity to adapt to warming temperatures than species with narrow ranges. While we do not know how well Suckley's cuckoo bumble bee will adapt to rising temperatures, their main host species appears sensitive to temperature. Between 1998 and 2020, the largest contributor to the decline in occupancy of *B. occidentalis* was temperature of the warmest quarter (synonymous with heat wave events) (Janousek et al., 2023, p. 2). This climate variable had twice the negative effect on occupancy as the second largest influencing variable, consecutive years of drought. Though this study may not model the precise climatic niche of *B. occidentalis*, due to the coarse macroclimate data used, it does suggest that broad scale temperature patterns are important drivers of where an important Suckley's cuckoo bumble bee host species is found on the landscape. Heat waves are projected to increase, particularly in the western portions of North America (Hicke et al., 2022, p. 1937), which represents the bulk of the range of Suckley's cuckoo bumble bee.

Temperature changes could make southern portions of Suckley's cuckoo bumble bee range less suitable for the species, while additional habitat may become more suitable in the northern portions of their ranges and at higher elevations. Colonizing new areas may be dependent on dispersal ability (see sections 2.3 Life history and 3.2 Population needs for resiliency) and they may require adapting to novel communities where Suckley's cuckoo bumble bees could be exposed to new conditions and potential hosts (Cameron, Jepson, et al., 2011, pp. 39–40; Pradervand et al., 2014, p. 5). Additionally, warming ambient temperatures and heatwaves commonly co-occur with drought, another influence that could compound the effects of any one of the influences alone (Cameron, Jepson, et al., 2011, pp. 39–40; Pradervand et al., 2014, p. 5).

Trends in bumble bee species' ranges indicate that the taxon does not have a high capacity for adapting to increasing temperatures (Kerr et al., 2015, p. entire; Sirois-Delisle & Kerr, 2018, p. entire). Ranges of bumble bee species in North America and Europe have contracted in their southern-most portions (Kerr et al., 2015, p. 178). This contrasts with dragonfly and butterfly taxa that evolved in the tropics and have generally maintained their southern range limits despite rising temperatures (Sunday et al., 2012, p. 687). While their ranges are contracting in the south, bumble bees generally have not expanded to track warming temperatures north of their ranges as dragonfly and butterflies have; this lag may be due to insufficient capacity to colonize new areas (Fourcade et al., 2018, p. 648; Kerr et al., 2015, p. 178; Sunday et al., 2012, p. 687). In mountainous areas, bumble bee species have shifted their ranges up elevational gradients (Bartomeus et al., 2011, p. 20647; Fourcade et al., 2018, p. 649; Kerr et al., 2015, p. 179; Pyke et al., 2016, p. 15). *B. occidentalis* has been documented shifting to higher elevation subalpine habitat in Colorado (Pyke et al., 2016, p. 15). Another study of *B. vancouverensis* using genetic markers indicated that populations of montane bumble bees may be able to rapidly adapt to climate-associated range changes (Lozier et al., 2023, p. 10). However, opportunity to move up in elevation is constrained by the height of the mountains and sometimes results in decreasing area with increasing elevation. Analyses of long-term data of bumble bee species distribution, including cuckoo bumble bee and its hosts, reveal widespread declines and predict extirpations in the southern portions of their ranges that exceed colonization rates in the northern portions of their ranges, resulting in recent and projected shrinking bumble bee ranges in the future (Sirois-Delisle & Kerr, 2018, pp. 4–5; Soroye et al., 2020, pp. 685, 687).

Warming temperatures could additionally affect Suckley’s cuckoo bumble bee and its hosts by affecting their floral resources. Studies evaluating multiple decades of data have found shifts in flowering phenology correlated with warming temperatures. Shifts in flowering times could result in phenological mismatch between pollinators and their foraging resources (see section 4.2.7, Drought Frequency and Intensity). Heatwave conditions directly reduce bumble bee foraging, as well as indirect effects on bumble bee foraging from heat-stressed flowering resources (N. M. Williams & Hemberger, 2023, p. 597). Decreased pollination due to phenological mismatch could reduce plant reproduction and further affect floral resource availability (Forrest et al., 2010, p. 438; Pyke et al., 2016; J. D. Thomson, 2010, p. 3197).

5.2.7 Drought Frequency and Intensity

Droughts are prolonged periods of reduced precipitation, within- or across-seasons, that reduce soil moisture, surface and groundwaters, and snow levels in an area. Drought negatively impacts floral resources and the pollinators that depend on them. A large portion of Suckley’ cuckoo bumble bee range is within water-limited areas of western North America where drought is frequent and has major implications for floral resources. The frequency of biologically-significant drought events is projected to increase within the range of Suckley’s cuckoo bumble bee and its host species due to changes in climate and resource impacts (Swain & Hayhoe, 2015, pp. 2737–2750).

Droughts could also amplify impacts to pollinators from increased temperatures, grazing, invasive species, altered surface and groundwater levels, and competition, which directly and indirectly effects pollinators. Drought indirectly impacts bumble bees by altering or reducing floral resources, including reductions to the quality, quantity, and availability of pollen and nectar (Waser and Price 2016, p. 1405; Phillips et al. 2018, pp. 3226–3235; Carroll et al. 2001, p. 443). Shifts in the spatial and temporal patterns of flowering resulting in mid-summer floral gaps (Aldridge et al., 2011, entire), highlight that drought-stress impacts on a plant community could reduce pollen and nectar resources needed for successful pollinator reproduction. For example, as a result of drought stress, fireweed (*Chamaenerion angustifolium*) flower size and nectar volume are measurably reduced (Carroll et al., 2001, p. 443). Further, drought can cause within-season resource shortages that may impact foraging activity, growth (Crone & Williams, 2016, p. 6; Westphal et al., 2006, p. 393), and colony health (McNeil et al., 2020, p. 5). Since droughts have a direct effect on floral resources, this in turn has an effect on pollinators at the population level (Roulston & Goodell, 2011, p. 305). Drought may also lead to increased competition with honey bees (see section 5.2.2 Managed and Introduced Bees) in areas where their resources overlap. When drought conditions impact preferred floral resources, bumble bees will forage on less preferred, drought-resistant species used by the more numerous honey bees (D. M. Thomson, 2016, pp. 1247–1255).

Warming temperatures increase evaporation and decrease transpiration in plants, yet when temperature is compounded with low precipitation it can alter plant transpiration and survival strategies during important stages of the growing season(Hanson, 1991, pp. 99–104). Biologically severe droughts are intensified when combined with higher temperatures (see section 5.2.6 Changes in Ambient Temperatures and Heatwaves). A model of cumulative effects of climate changes, landcover, and pesticide-use on *B. occidentalis* occupancy found that consecutive years of severe drought was the second most influential cause of occupancy declines (Janousek et al., 2023, pp. 2–3). In addition, suitable habitats may be restricted with rising temperatures, reducing the range of some species, especially at range edges where abundance may decline when floral resources decline in response to drought (D. M. Thomson, 2016, pp. 1247–1255). Though drought may be locally and temporarilly alleviated by precipitation, the impacts to the growing season may persist and reduced floral resources impact Suckley's cuckoo bumble bees that require pollen and nectar in order to overwinter with adequate body mass (see section 4.1.2 Diapause Site Availability). Droughts can increase within-season stress on these resources (flowers, pollen, and nectar) in turn causing gaps in their availability for pollinators later within the same season.

One study argues that declining availability of floral resources due to warmer summers may be favoring bumble bees that are generalists, and thus may be prompting evolution of bumble bees toward features that allow for more generalist foraging(Miller-Struttmann et al., 2015, pp. 1541–1544). However, other models of climate and drought effects over a few successive seasons reduced *B. occidentalis* occupancy (Janousek et al., 2023). Apparent declines in pollinator populations suggest that negative impacts of increasingly frequent and intense droughts may outpace adaptation opportunities over time.

Droughts, especially in consecutive years (Janousek et al., 2023, p. 2), will likely amplify biologically-significant negative effects on Suckley's cuckoo bumble bee resources and host species, though the direct drought impacts on Suckley's cuckoo bumble bee individuals and populations are unstudied. Droughts are expected to negatively affect resources, alter floral communities to less preferred flowering conditions or timing, increase competition between pollinators, and negatively impact Suckley's cuckoo bumble bee and its hosts throughout much of the species' range.

5.2.8 Wildfire Frequency and Intensity

Wildfires pose complex effects to bumble bees, from decreasing resource availability (Galbraith et al., 2019, p. 15; Mola et al., 2020, p. 1807; Mola & Williams, 2018, p. 7) to increasing floral and bee abundance and diversity (Carbone et al., 2019, entire; Galbraith et al., 2019, entire; Mola et al., 2020, pp. 1804–1808; Mola & Williams, 2018, pp. 4–8). Fire disturbance can temporarily increase floral resources, thereby enhancing bee body size, reproductive output, genetic diversity, and population size (Carbone et al., 2019; Mola et al., 2020, pp. 1804–1808). Higher floral abundance following fire has been shown to increase forager abundance, the subsequent number of colonies, and increased mean worker size (Mola et al., 2020, pp. 1804–1807). The relative effect of high intensity fires varies based on the ecological conditions in which it took place. Since Suckley's cuckoo bumble bee is a broad ranging species with multiple hosts, fire is likely to have variable and diverse effects throughout the range.

While there is evidence to suggest that fires increase resources post-fire, these effects may be temporary and vary by habitat type (i.e, forested vs. grassland habitats). While increased queen abundance across burned and unburned sites was observed for two years post-fire, after two years worker abundance was reduced across all sites, and body sizes were similar between sites (Mola et al., 2020, pp. 1805–1808). Other studies show the effects of fire and grazing disturbance to floral communities have similar short-term (two to three year) impacts on bee communities in tallgrass prairie (Bruninga-Socolar et al., 2022, p. 6).

Regulating factors that impact bumble bees through habitat management, such as habitat restoration or prescribed fire, may be an effective approach, as evidenced by the findings of a study conducted by (Mola et al., 2020, p. 1808), but fire regimes would also need to be monitored and managed to inform biologists about bumble bee population dynamics (Carbone et al., 2019, p. 1487). Given the current fire regime changes at the global scale, it is important to monitor post-fire pollinators across many ecosystems, as studies suggest that fire regime is critical in determining the dynamics of pollinator communities (Carbone et al., 2019, p. 1487). Another consideration is the introduction of fire tolerant weeds, such as cheatgrass, which reduces floral resources (Stark & Norton, 2015, p. 799). Opportunities to manage habitat components are limited over space, time, and resources. Regardless, wildfire frequency and intensity are expected to increase throughout the range of Suckley's cuckoo bumble bee, particularly in the western United States (Gao et al., 2021, p. 6; Parks et al., 2018, p. 919).

5.2.9 Livestock Grazing

Livestock grazing occurs throughout much of the historical range of Suckley's cuckoo bumble bee, primarily by cattle, sheep and wild horses, and it can have mixed effects on bumble bees. Bee researchers typically find reduced floral resources and lower bumble bee diversity and abundance at grazed sites (Hatfield & LeBuhn, 2007, p. 150; Sjödin, 2007, pp. 2110–2113; Sjödin et al., 2008, p. 763). Although grazing can be a useful management tool for maintaining early successional habitat, benefits are dictated by frequency, intensity, species (i.e. sheep, cattle, horses, etc.) and timing (Carvell, 2002, p. 44; Kimoto, DeBano, Thorp, Taylor, et al., 2012, pp. 9–13). Low intensity grazing preserves floral resources benefiting bumble bees (Scohier et al., 2012, pp. 287–292), while increased intensity of grazing can reduce species richness of bees as a result of altered floral composition, including invasive species establishment and soil compaction (Hatfield & LeBuhn, 2007, p. 156; Vázquez & Simberloff, 2003, p. 1080). Bumble bees are sensitive to grazing intensity early in the season, potentially because of altered foraging behavior (Kimoto, DeBano, Thorp, Rao, et al., 2012, pp. 9–13). Intense summer grazing, compared to areas only grazed in winter, reduces vegetation height and floral resources, leading to fewer bee visits to preferred food plants and decreased bumble bee diversity (Xie et al., 2008, pp. 699–700). The reduction of vegetation height and structure from high intensity grazing in the Pacific Northwest bunchgrass prairies has been linked to declines in bumble bee richness and abundance (Kimoto, DeBano, Thorp, Taylor, et al., 2012, pp. 12–13).

Grazing may compact soil and change plant communities (M. Connors, 2016, pers. comms), thus impacting nesting habitat for Suckley's cuckoo bumble bee hosts (Defenders of Wildlife, 2015, p. 14). Livestock may also trample nesting sites (Kearns et al., 1998, p. 90) and negatively impact ground-nesting rodents (Johnson & Horn, 2008, p. 444), which in turn may reduce the number of nest sites available for bumble bees (The Xerces Society & Thorp, 2010, p. 13). Livestock grazing also impacts hydrology through compaction of soils and degraded riparian areas and may lead to increased fire cycles through the introduction of exotic species (Dwire et al., 1999, pp. 319–321).

In summary, grazing can assist in maintaining open habitat, and low intensity grazing can preserve floral resources. However, high intensity grazing can have a negative impact on floral resources and negatively impact nest site availability. Thus, grazing has varied and complex effects on bumble bees, which makes the impacts difficult to analyze.

5.2.10 Early Spring Frosts

Early spring frosts pose a risk to bumble bees, specifically to new queens and newly established colonies, by damaging floral resources; negative effects to bees are of growing concern with the changing climate. Floral resources are important throughout the lifecycles of both Suckley's cuckoo bumble bee and its hosts, but they are particularly important for host colony establishment in early spring (see section 4.1.3 Floral Resource Abundance and Diversity). Negative effects of early spring frost on host colony queens, workers, and overall colony size may reduce their ability to persist within season, and to successfully support *Psithyrus* such as Suckley's cuckoo bumble bee (see 4.1.4 Suitable Host Colony Availability).

Spring frosts can damage early blooming floral resources (CaraDonna & Bain, 2015, pp. 61–62). Since 1974, there has been an increase in frequency of frost damage to plants in a subalpine, long-term study (Inouye, 2008, pp. 358–359). Flowers, compared to vegetative parts, are generally more sensitive to damage from frost events (CaraDonna & Bain, 2015, pp. 61–62), and spring frosts reduce the overall availability of floral resource abundance across the subsequent summer (CaraDonna et al., 2014, p. 4919).

Despite some plants with early phenology exhibiting some tolerance to freezing temperatures (CaraDonna & Bain, 2015, pp. 61–63), advancing bloom date for plants triggered by climate change (CaraDonna et al., 2014, p. 4919) exposes early floral growth to additional risk of frost damage (Willmer, 2012, p. R131). Climate change was found to result in earlier snowmelt, and the reduced snowpack exposes plants to spring frosts and damage to floral resources (Inouye, 2008, p. 357). The increased frost damage to flowering plants could also contribute to an observed change in flowering dates across small geographic and altitudinal scales (Inouye, 2008, pp. 357, 361).

Loss of floral resources at a critical time in the bumble bee life cycle is the main concern for early spring frosts. Physiological constraints during early spring frost events are less likely to be an issue for Suckley's cuckoo bumble bee and its hosts because bumble bees are somewhat adapted to tolerate cold temperatures. For example, *B. mckayi* queens in Alaska were still capable of flight at 9° (C) (49° F) and workers flew at 18°C (64°F) (Bishop & Armbruster, 1999, p. 715). Bumble bee queens may return to torpor if temperatures fall below their lower thermal threshold (Oyen & Dillon, 2018). High-altitude bees may be more likely to recover from unexpected cold snaps, including early spring frosts, by using rapid-cold hardening (Oyen et al., 2016, p. 56). However, there is some evidence that exposure to sudden cold temperatures decreases foraging activity of *B. occidentalis* (Wilson et al., 2006, pp. 170–172).

Spring frost damage to floral resources has been linked to within-season declines in wild bee populations (Graham et al., 2021, p. 6). Spring frosts could have a negative impact on the success of local bumble bee colonies (Inouye, 2008, p. 361) by reducing existing and future within-season availability of pollen and nectar resources on which pollinators, including host colonies for Suckley's cuckoo bumble bee, rely. Inadequate floral resources could especially affect colony establishment and growth if they occur during critical times, such as early spring when new queens emerge from diapause and are establishing their colonies. Some bumble bee species may depend on a specific floral resource during a specific time period. For example, (Vogt et al., 1994, p. 1552) subarctic bumble bee queen emergence coincided with the first blooming willows, and their early emergence allowed them time to establish a colony in the shortened growing season in the tundra (Vogt et al., 1994, p. 1552). Delays in emergence or colony initiation after emergence may hinder the ability of bees to complete their life cycle before the end of the relatively short subarctic season (Vogt et al., 1994, p. 1554).

During the early stage of the colony's cycle, the queen, larvae, and the small number of workers are especially vulnerable to floral resource gaps since the colony initially relies on small pollen stores and a nectar pot during establishment (see sections 4.1.3 Floral Resource Abundance and Diversity). While host colonies for Suckley's cuckoo bumble bee may be able to physiologically survive the impacts of early spring frost events, resource gaps that result from damaging spring frosts have the potential to negatively impact host queens and colonies necessary to support the lifecycle of Suckley's cuckoo bumble bees.

5.3 Voluntary and Regulatory Conservation Measures

Suckley's cuckoo bumble bee is listed as threatened by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) (COSEWIC, 2019, p. iii) and is listed as critically endangered by the IUCN (Hatfield et al., 2015, p. 1). In the United States, Suckley's cuckoo bumble bee is listed on the sensitive species list for the U.S. Forest Service and Bureau of Land Management Interagency Special Status /Sensitive Species Program (ISSSSP) in the Pacific Northwest (ISSSSP, 2021). It is also listed as a species of greatest conservation need in Idaho, Washington, Colorado, and California, where it is also a candidate for listing under the California Endangered Species Act (CNDDB, 2023, p. 6; Colorado Parks and Wildlife, 2015, p. B-1; Idaho Department of Fish and Game, 2017, p. xvii; Washington Department of Fish and Wildlife, 2015, pp. 3–39). These states generally outline research and conservation needs for species of greatest conservation need in State Wildlife Action Plans, but these plans do not offer regulatory protection.

Some states regulate the import of nonnative bee species, which can help protect native species, including Suckley's cuckoo bumble bee (see 5.2.2 Managed and Introduced Bees). The Oregon Department of Agriculture restricts some potential sources of the parasite *V. bombi* from entering the state, including commercially produced colonies of *Bombus impatiens*; only *Bombus* species native to Oregon are permitted for commercial pollination purposes (Oregon Department of Agriculture, 2017, p. 5). California requires permits to import *B. impatiens* and other species such *O. lignaria* for pollination services (California Department of Food and Agriculture, 2023, p. 2).

The U.S. Forest Service (USFS) has taken steps to reduce impacts of nonnative bee species. The Pacific Northwest Region of the USFS is working on finalizing apiary guidelines, which include recommendations for management practices and considerations to protect native pollinators and minimize negative effects from managed bees and apiaries on USFS managed lands (J. Everett 2023, pers. comms). Additionally, at least one USFS National Forest in the Pacific Northwest already restricts commercial and privately owned bees on their managed lands (J. Everett 2023, pers. comms). For example, the Colville National Forest plan directs that apiaries should not be placed where they would pose a risk to native pollinators, butterflies, or rare bee species (U.S. Forest Service, 2019, p. 67). These measures highlight that the USFS takes the threat of managed bees seriously and is actively working in some regions to protect native bees.

States have also begun implementing strategies to limit pesticide effects on bees. The Oregon Bee Project was initiated in 2017 as a collaboration between the Oregon Department of Agriculture, the Oregon State University Extension Service, and the Oregon Department of Forestry. One project goal is to mitigate bee exposure to pesticides through increased pesticide label comprehension, adoption of new application practices, and increasing coordination between beekeepers and pesticide applicators (Oregon Department of Agriculture et al., 2022, pp. 2–3). Washington State has established similar goals, as stated in their Managed Pollinator Protection Plan, which offers best management practices for beekeepers, growers, and pesticide applicators to help protect pollinators from pesticides (Washington State Department of Agriculture, 2018, p. 5). Washington State also recently adopted additional recommendations and funding for pollinator health (State Bill SB5253).

Neither the Managed Pollinator Protection Plan nor the State Bill restrict the use of pesticides in Washington, but several other states have begun passing regulations on pesticides that are harmful to bees and other pollinators. For example, California recently developed regulations that will implement mitigation measures to protect pollinators by limiting the agricultural uses of certain neonicotinoid pesticides, which went into effect January 1, 2024 (California Department of Pesticide Regulation, 2023). Other states that have enacted various regulations on neonicotinoid pesticides to reduce impacts to bees and other pollinators include Colorado, Connecticut, Maine, Minnesota, Massachusetts, Maryland, Nevada, New York, New Jersey, Rhode Island, and Vermont (Malfi, 2023).

Other recent efforts aim to better understand bees at risk and implement broader protections. The Pacific Northwest Bumble Bee Atlas (a collaborative effort between the Washington Department of Fish and Wildlife, the Idaho Department of Fish and Game, the Oregon Department of Fish and Wildlife, and the Xerces Society for Invertebrate Conservation) seeks to increase understanding of bumble bee distributions and their habitats (Washington Department of Fish and Wildlife et al., 2023). This effort has contributed to the creation of a statewide strategy to protect bumble bee species of conservation concern in Washington, with one of the focal species being Suckley's cuckoo bumble bee (Martin et al., 2023, p. 12). In Alaska, the the Alaska Center for Conservation Science at the University of Alaska Anchorage and the Bureau of Land Management have developed the Alaska Bee Atlas. The program aims to collect data on bee biodiversity within the state which can eventually be used to inform sensitive species lists and management (Fulkerson et al., 2023, p. 18). The number of bumble bee atlases are increasing across the United States and increasingly covering the range of Suckley’s cuckoo bumble bee (see bumblebeeatlas.org for up-to-date coverage). The expanding coverage of these atlases will increase understanding of bumble bee status and distribution in North America.

On a broader scale, the Colla Laboratory at York University in Toronto has released a national pollinator strategy for Canada. This national strategy identifies specific goals and actions to protect pollinators and needed research to fill knowledge gaps (Colla & Nalepa, 2023, p. 5). The Service is also working on a nationwide plan through a bumble bee Conservation Benefit Agreement, known as The Nationwide Conservation Benefit Agreement for Bumble Bees on Energy and Transportation Lands. This Conservation Benefit Agreement is modeled after the nationwide monarch butterfly Candidate Conservation Agreement with Assurances, which is a voluntary agreement with transportation and utility landowners that provides incentives for monarch conservation measures on their lands. The current monarch agreement includes 19 transportation and 27 utility applicants across 39 states, amounting to approximately 6.3 million acres (2.5 million hectares) (J. Everett 2023, pers. comms). The bumble bee Conservation Benefit Agreement will likely include many of the same acres enrolled for the protection of several bumble bee species, including Suckley's cuckoo bumble bee and some of its host species (i.e., *B. occidentalis*, *B. affinis*, and *B. terricola*). The Nationwide Conservation Benefit Agreement for Bumble Bees on Energy and Transportation Lands is expected to be completed in early 2024 (J. Everett 2023, pers. comms).

Together, these voluntary and regulatory measures highlight an increase in effort to protect native bumble bee species across North America in recent years. Some of these measures specifically target Suckley's cuckoo bumble bee, and/or its host species, for conservation. Broad efforts to protect and conserve native pollinators and bees will also likely benefit Suckley's cuckoo bumble bee.

6 Assessment Methodology

6.1 Bumble Bee Occurrence Data

We compiled a dataset comprised of all known Suckley's cuckoo bumble bee occurrence records in North America, herein referred to as the Suckley's cuckoo bumble bee master dataset. The master dataset includes all Suckley's cuckoo bumble bee occurrence records within the Bumble Bees of North America dataset. The Bumble Bees of North America dataset is curated by Leif Richardson with contributions from over 300 individuals and institutions around the world. It includes over 275,000 records of bumble bee specimen collected in North America since 1805. The version used for this assessment reflects data contributions as of August 2023. Please visit https://www.leifrichardson.org/bbna.html to learn more. We received additional contributions of Suckley's cuckoo bumble bee occurrence records from partners across the continent in response to our August 2023 Dear Interested Party letter. Additionally, we funded the verification and digitization of 350 occurrence records from the Oregon State Arthropod Collection at Oregon State University[1] and added these records to the master dataset. The Suckley's cuckoo bumble bee master dataset used in this analysis is comprised of 2,317 occurrence records from 1986-2022.

Suckley's cuckoo bumble bees, particularly males, are difficult to identify (August 2023 Expert Team Meeting, p. 4-5) and misidentifications may exist in the master dataset. Verification work of specimens on the margins of the range occurred for this assessment but further work is needed (see 11.1 Suggestions For Future Work). All Suckley's cuckoo bumble bee specimens collected in Alaska were reviewed for this assessment and determined to be misidentifications (D. Sikes and J. Rykken 2023, pers. comms). Specimens historically collected in New York and Maine were also concluded to be misidentifications (B. Swartz, 2023 pers. comms; J. Heron and C. Sheffield 2023, pers. comms). Given the history of misidentification of the species, other misidentified occurrence records may exist in the master dataset used for this analysis.

The location accuracy varies for documented occurrences of Suckley's cuckoo bumble bee, especially for older observations. Occurrence data collected before the Global Positioning System (GPS) was publicly available were digitized based on noted landmarks, municipalities, or geopolitical boundaries (e.g., state, province, county), and may not reflect the actual observation location.

Survey effort for bumble bees in North America has been variable throughout history and across ecoregions. Much of the range of Suckley's cuckoo bumble bee has not been systematically surveyed for bumble bees, and where surveys have taken place, they may not follow the same methodologies and lack consistency in timing. For example, we lack sufficient survey data to confirm connectivity between populations across Canada (S. Cannings, 2023, pers. comms). Similarly, most of the range of the known and suspected host species has not been systematically surveyed (Xerces Society Wesern bumble bee website profile; Fiedler, C. 2018, pg. 3,).

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[1] Best, L. & Price, B. (2023) Dataset: 350 Bombus suckleyi species occurrence records based on specimens archived in the Oregon State Arthropod Collection (OSAC) at Oregon State University. Corvallis, Oregon.

Bumble bee surveys include non-lethal methods such as traditional net-capture surveys, as well as lethal trapping. Most lethal trapping in the contiguous United States does not target bumble bees and is focused on surveillance for silvicultural pests and other invasive species, however some accidental by-catch of bumble bees has been reported. Surveys require taxonomic verification of identified species which, as noted above, is difficult for Suckley's cuckoo bumble bee. The recent expansion of bumble bee atlas projects across a portion of the range of Suckley's cuckoo bumble bee, such as the Pacific Northwest Bumble Bee Atlas managed by The Xerces Socity, has increased survey effort and uses non-lethal survey methods with taxonomic support. The continued expansion of standardized surveys would improve our understanding of Suckley's cuckoo bumble bee, its hosts, and changes in distribution over time, particularly in higher elevation sites which are poorly studied (see 10.1 Suggestions for Future Work).

6.2 Analytical Units

The species occurrence polygon was delineated based on the Commission for Environmental Cooperation level II ecoregions (Commission for Environmental Cooperation, 1997 entire) and Suckley's cuckoo bumble bee occurrence data (described in 6.1 Bumble Bee Occurrence Data). Because of minor drawing errors with the geospatial data along shorelines, we applied a 1.5 km (1.0 mi) buffer around occurrence points so each would be associated with an ecoregion. Any ecoregion with a known occurrence was retained, and those ecoregions without known occurrences were excluded from the occurrence polygon.

Several large ecoregions extended far beyond known occurrences of Suckley's cuckoo bumble bee. In some cases, occurrence points were along the shared border of an ecoregion with many known occurrences. This likely occurred because of small scale topographic variation between ecoregions that could provide suitable habitat within the dispersal capacity of the species. To account for this small topographic variation, we applied a 50 km (31 mi) buffer to the ecoregions, eliminated any large ecoregions with occurrences only in the buffer zone, and grouped those occurrence points with the neighboring ecoregion. Where some large ecoregions remained that extended far beyond known occurrences of Suckley's cuckoo bumble bee, we constrained the occurrence polygon to within 100 km (62.1 mi) of a known occurrence and did not include those entire ecoregions. This distance has been used in bumble bee mapping efforts because it is generally believed to help describe a bumble bee species' range (Colla et al., 2011, p. 13; Koch et al., 2012, p. 14). We did not restrict the Taiga Plains analytical unit because the region is less surveyed.

The resulting level II ecoregions, which were buffered or constrained during the delineation of the occurrence polygon, became the analytical units. There are 15 analytical units for Suckley's cuckoo bumble bee, 4 in eastern North America and 11 in western North America (Figure 7).

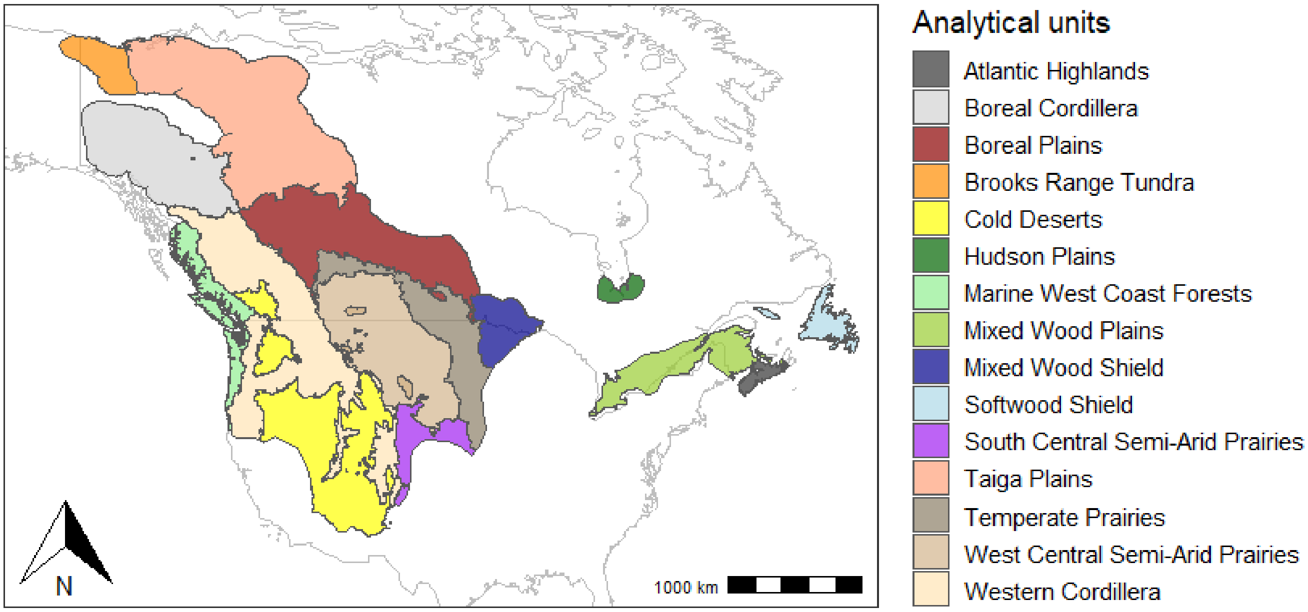

Figure 7. *Map of analytical units for Suckley's cuckoo bumble. The analytical units were delineated using modified ecoregion level II boundaries (CEC 1997) and bounded by the species occurrence polygon (the gray region in Figure 4).*

6.3 Multi-Species Occupancy Model

6.3.1 Historical Trends in Occupancy

We characterized the historical trends of Suckley's cuckoo bumble bee, and its host species, using published models developed by (Jackson et al. 2022, entire). This analysis generally followed the methods outlined in Jackson et al. (2022, entire), with some minor modifications described in 11.4 Additional Details on Occupancy Modelling Approach. Briefly, the analysis used the Bumble Bees of North America occurrence records database and the Suckley's cuckoo bumble bee master dataset (see 6.1 Bumble Bee Occurrence Data), along with data on floral resources and climate, to assess how the probability of occupancy for Suckley's cuckoo bumble bee, and its host species has changed over time. The analysis used 796,985 records of 46 bumble bee species, and a method that specifically accounted for imperfect detection, since Suckley's cuckoo bumble bee is a cryptic species. For the purposes of the study, sites were defined as 50 km (19.3 mi) grids.

To assess historical change in occupancy over time, the study binned bumble bee detections from 1900 to present into 10-year bins and modeled temporal changes in probability of occupancy over time. These temporal trends were estimated range-wide, for the purpose of model predictions. Their model also included average maximum temperature of the warmest month in each year, decadal monthly mean precipitation and spatially explicit floral resources that varied through time as a function of land cover change. All climate data were from the ACCESS1-3 climate model statistically downscaled to a finer spatial resolution using the CHELSA algorithm, and validated as performing well, using observed historical climate data (Karger et al., 2020, entire). Calculation of floral resources is described in Jackson et al. (2022).

In addition to the models described above, a separate model was run (without climate variables) to specifically assess whether *temporal* trends were similar across analytical units. There was enough data to assess regionally specific historical trend in the Mixed Wood Plains, Temperate Prairies, West-Central Prairies, Cold Deserts, Western Cordillera, Marine West Coast Forest, Softwood Shield, Boreal Plain, and Boreal Cordillera (79 percent of the occurrence polygon). All these analytical units exhibited a substantial historical decline in probability of occupancy, though the confidence bands around the trend in the Boreal Plain were fairly large. Due to small sample size, regionally specific temporal trends could not be estimated for the Brooks Range Tundra, Taiga Plains, Mixed Wood Shield, and South Central Semi-Arid Prairies in the western portion of the range, or for the Hudson Plains, and Atlantic Highlands in the eastern portion. Given these results, we are confident that the overall rangewide magnitude of temporal decline is reasonable to use for projections in much of the species' range. However, we do not know if the range wide temporal trends are representative in the analytical units with fewer observations of Suckley's cuckoo bumble bee.

6.3.2 Current and Future Occupancy

We used the modified Jackson et al. (2022) multi-species occupancy models to spatially project probability of occupancy for Suckley's cuckoo bumble bee across its range, currently, and into the future at multiple time steps (2020–2040; 2040–2060; 2060–2080; 2080–2100). "Current" projections reflect estimated occupancy for the 2000–2020 time period. Future occupancy projections were made under two different scenarios, described further in 8.1 Future Condition Scenarios. We use the results for the 2000–2020 period to document historical trends, and to provide a snapshot of current occupancy, while we use the near-term occupancy results (2020–2040) to assess current risk of extinction, and results from the mid- and late-century projections (2040–2060, and 2080–2100) to assess future risk of extinction.

For each time step, and each scenario, we summarized occupancy in each analytical unit by taking the median of the estimated occupancy across all the 50 km (19.3 mi) grids in each analytical unit. Occupancy trend for an analytical unit was estimated by calculating the percent change in median probability of occupancy from one time period to another. Specifically, for each analytical unit, we subtracted the median probability of occupancy from the earlier time period from the median at the later time period and divided by the median of the earlier time period. Occupancy trend likely reflects population trends on the landscape, such as changes in abundance and density across populations, so this metric relates directly to population growth. In addition, localized extirpation is likely implied when occupancy declines to zero.

We used the same occupancy modeling framework to spatially project current and future probability of occupancy for the confirmed and potential hosts of Suckley's cuckoo bumble bee. For each confirmed and potential host species, we followed all the same procedures as outlined for Suckley's cuckoo bumble bee. However, for each host, we limited projections to only those analytical units in which the host species has been observed at least once since 2000. We specifically used any confirmed records from the Bumble Bees of North America occurrence records database for this analysis. We present occupancy information on the confirmed hosts and the likely preferred hosts from the subgenus *Bombus.*

6.3.3 Assumptions and Uncertainties of Occupancy Model

Strengths and limitations of the occupancy modeling framework

The analytical approach used in this SSA helps to address several key factors that typically cause uncertainty in assessments of cryptic species for which poor detection can lead to false absences, and incorrect inferences. Suckley's cuckoo bumble bee is a prime example of a cryptic species for which occupancy modeling is an apt approach. The species does not have workers to detect during surveys, and adults are only detectable outside nests in early and late summer. Detectabilty may have also declined over time, if people have increasingly avoided lethal collection of females, over concern for the species. Similarly, aggregation of data from multiple sources can lead to detection biases across time, and space. Simple inferences based on raw collection data could therefore infer spurious changes in occupancy due to these detection biases. However, since the analysis is based on binary occupancy data (detection or non-detection) rather than abundance data, any potential biases due to changes in sampling effort are likely minimized (Jackson et al., 2022, p. 7). In addition, the occupancy modeling framework explicitly accounts for potential detection biases by leveraging repeated surveys at each location to estimate detection probability and sampling effort, and to adjust occupancy estimates, accordingly (Kery et al., 2010, p. 1388).

To account for sampling intensity that varies over time, the occupancy models used here also allowed detection probability to vary spatially and temporally (Jackson et al., 2022, p. 7). While it is possible that some temporal heterogeneity in sampling intensity was not accounted for by the model, the total number of worldwide bumble bee specimen records has increased over time in the Global Biodiversity Information Facility (GBIF) database (Zattara & Aizen, 2021, p. 116). In addition, the number of surveys in the Bumble Bees of North America dataset within the occurrence polygon of Suckley's cuckoo bumble bee has increased over time (see section 11.5 Supplementary Tables and Figures, Figure A3). Thus, any unmodeled variation in sampling intensity would likely bias the results to show population increases. Finally, the SSA also specifically uses a multi-species occupancy model, which is an apt approach for 1) estimating species-specific trends over large timescales, and 2) increasing statistical power for species with smaller sample size (Jackson et al., 2022, p. 2).

While the multi-species occupancy modeling framework addresses many of the biases and challenges of assessing occupancy and trends for cryptic and data limited species, there are several important tradeoffs involved with using this approach. The variables used in the occupancy modeling (maximum monthly temperature, and floral resources), reflect the species' need for suitable thermal environments and their need for diverse pollen and nectar resources to support survival and reproduction (4 Species Ecological Needs), and thus reflect key aspects of fitness for Suckley's cuckoo bumble bee. However, the climate data is aggregated over a large spatial scale and does not likely match the spatial scale at which most small-bodied organisms experience climate (Potter et al., 2013, p. 1). Since microclimatic features Suckley's cuckoo bumble bee and its hosts likely respond to are not well represented in the climate data, the model may not reflect the actual climate niche experienced by these species well. Failure to account for habitat selection at the correct scale can lead to spurious understanding of habitat relationships and could cause probability of occupancy to be underestimated (Potter et al., 2013, p. 1). Consequently, assessment across multiple spatial scales is recommended (McGarigal et al., 2016, p. 1). Although the limited occurrence records precluded an analysis using fine resolution climate data, the results of the published occupancy model used in the SSA were similar when run at multiple spatial scales (Jackson et al. 2022, p. 4).

The coarse spatial scale of the climate data used in the analysis also means that within a grid deemed otherwise unsuitable to the species, small pockets of suitable habitat may remain that are not predicted by the model. For instance, some locations on the landscape may be ***temporarily*** buffered from climate change in the future due to local topographic or geological conditions that decouple local scale climatic processes from broader, regional conditions. Consequently, the models may overestimate decline in occupancy and range contraction in the nearterm for both Suckley's cuckoo bumble bee, and its hosts, particularly in topographically complex areas. However, ***long-term*** microrefugia are expected to be rare under anthropogenic climate change due to the sustained trajectory of increasing temperatures (Hannah et al., 2014, pp. 392–393, 395). Moreover, populations that persist in microrefugia will typically become more isolated, and in turn, more vulnerable to stochastic processes like genetic drift.

On the other hand, other limitations of the modeling approach could lead to potential overestimates of occupancy. For instance, availability of overwintering habitat – which we did not have data to model – may be limited in a grid otherwise deemed suitable in terms of climate and floral resources. Similarly, because the occupancy models did not explicitly model host presence as a variable influencing occupancy of Suckley's cuckoo bumble bee, the models are limited in their ability to directly demonstrate relationships between availability of host colonies and the focal species. The models and the resultant projections do not directly account for small-scale variation in some key threats like land cover change, invasive species, pathogens, etc. due to lack of data, and the spatial scale of the analysis.

Indeed, the lack of available spatial and temporal data and explicit information about stressors at the scale of the species range continues to limit our approach to assessing bee population trends (Janousek et al., 2023, p. 7). However, the modeling framework used here incorporated a temporal trend (e.g., change in occupancy over time), that can help to capture the impacts of important influences that were not explicitly included in the model (like land cover change, invasive species, pathogens, pesticides, wildfire, grazing, or host availability). Thus, some of these small-scale factors for which we do not have data are indirectly accounted for, though we cannot precisely say which is driving the historical temporal trends.

One final limitation of this study is that we used an existing published occupancy model that relied on climate data from a single statistically downscale climate model. A single climate model does not typically encapsulate as much variability in climate projections as a multimodel ensemble. However, it is unlikely that this had a large influence on the overall model results, or conclusions in the SSA, because the ACCESS 1-3 model is fairly representative of climate conditions that would be derived from a multimodel ensemble. The climatic conditions simulated by the ACCESS 1-3 model are quite similar to the averaged climate output from a large suite of 36 climate models from the CMIP5 (Sanderson et al., 2015, p. 5181). Compared to other climate change models, the ACCESS models tend to project temperature changes in North America that are slightly higher, but quite near the average of an ensemble of 13 other highly skilled climate models for the region, and precipitation increases that are slightly lower (Mahony et al., 2022, p. entire)[2].

Assumptions of occupancy modeling framework

There are several assumptions of the occupancy modeling framework used for this assessment. First, occupancy modeling assumes that species are correctly identified, as misidentification can result in an overestimate of the species' occupancy and range (Louvrier et al., 2018, p. 62). Since some Suckley's cuckoo bumble bee occurrence records may be misidentified, this assumption may be modestly violated in our analysis. Second, the model used here infers nondetection for a species for each site survey, if any other species of bumble bee was detected at a site during a given survey. This inherently assumes that conditions were favorable for detection of all species at the site during the survey, even if the survey was not targetingting the focal species specifically (Kery et al., 2010, p. 1390). Lack of detection at a site is not however assumed to imply absence; by leveraging multiple temporally repeated surveys across the landscape the model subsequently teases apart nondetection from nonoccurrence (Kery et al., 2010, p. 1392). Simulation studies have demonstrated that this assumption is generally robust and does not bias results (Guzman et al., 2021, p. 1). Another assumption is that the occupancy state does not change in each time interval; that is, the sites do not become extirpated or become newly colonized during the interval. Violation of this assumption typically leads to an overestimation of occupancy, especially if detection probability is low (Devarajan et al., 2020, p. 1614). However, this assumption is likely met reasonably well in this study; though the sampling era is relatively long, since the sites are so large, complete turnover in terms of extirpation or colonization is unlikely. Finally, the multi-species model used here also assumes that surveys are looking for all species in common, which is likely reasonable, given that most survey efforts are focused on bumble bee community sampling. In addition, the multi-species model allows for species-specific responses to temperature, and species-specific detection probabilities, but to simplify the model structure, assumed that the niche breadth and detection probability both followed a normal distribution across the modeled species; normaility assumptions were verified during analysis.

There are also assumptions involved with making spatial projections from the occupancy models. For one, spatial projections assume that rangewide relationships between climate and occupancy, and rangewide temporal trends are fairly consistent across the range. For instance, the model used occurrence data from across the range to describe how climate influences the occupancy of Suckley's cuckoo bumble bee. This relationship is then used to estimate occupancy in un-surveyed areas, assuming that the relationship between climate and occupancy is similar across the range. This assumption is widely applied in the literature, and is likely reasonable here, since the wide geographic

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[2] This study specifically assessed the CMIP6 version of the ACCESS model (ACCESS-ESM1.5); however, updates from ACCESS 1-3 to ACCESS ESM 1.5 resulted in negligible alteration of the simulated climate (Ziehn et al., 2020, pp. 11–12).

sample of the occurrence records (see section 11.5 Supplementary Tables and Figures, Figure A3) means that we likely captured the full range of climate conditions experienced today and expected into the future for the species. However, it is less certain whether *temporal* trends in occupancy due to historical threats like land use change, or pesticides are similar across analytical units. As noted earlier, a separate model run suggested that most of the analytical units (79 percent of the occurrence polygon) have exhibited historical declines, and that it is therefore likely reasonable to apply rangewide estimates of temporal decline for projections throughout much of the species' range. We are less confident that the rangewide temporal trends are representative in Eastern portion of the range, and sections of the western range where there are fewer observations of Suckley's cuckoo bumble bee (Brooks Range tundra, Taiga Plains, Mixed Wood Shield, South Central Semi-Arid Prairies, Hudson Plains, and Atlantic Highlands; 21 percent of the occurrence polygon). Implications of this uncertainty are discussed throughout the SSA results.

Projecting results of the occupancy model into the future requires assuming niche conservatism, or that a species will remain within their current climate niche into the future (Pearman et al., 2008 entire). If a species shifts to novel thermal environments in the future, the relationship between climate and occupancy may not hold into the future, making future projections inaccurate. The assumption of niche conservativism is likely reasonable for Suckley's cuckoo bumble bee. For one, rising temperatures and temperature extremes are key factors driving thermal suitability for the species into the future, and there is mounting evidence that strong physiological constraints on the evolution of heat tolerance is present in most terrestrial life forms, causing a high degree of conservatism of upper thermal tolerances (Araújo et al., 2013, entire). Indeed, there is growing evidence that insects have limited capacity to change their thermal limits (Hoffmann et al., 2013, pp. 938, 944; Weaving et al., 2022, p. 7), and recent work has demonstrated that an adaptive response to high temperature is also likely constrained in bumble bees (Pimsler et al., 2020, p. 1). Furthermore, an adaptive shift to new thermal requirements (e.g., violation of niche conservativism) is more likely in species with relatively stable historical distributions and population trends, and a high degree of connectivity and genetic diversity (Maebe et al., 2021, p. 4231), none of which is likely true for Suckley's cuckoo bumble bee.

6.4 Analyzing Resiliency, Redundancy, and Representation

The factors influencing the species needs of resiliency, redundancy, and representation of Suckley's cuckoo bumble bee are described in 4.3 Species Needs. We will use the following methodology (Table 6) to analyze these species needs:

Table 6. *Methodology for assessing Suckley's cuckoo bumble bee resiliency, redundancy, and representation.*

Species Needs	Assessment Methodology
Resiliency	We analyze resiliency by assessing the median occupancy and occupancy trend of Suckley's cuckoo bumble bee across analytical units. We do not have reliable information on what constitutes a Suckley's cuckoo bumble bee population, so we based our resiliency analysis on the analytical units described in 6.2 Analytical Units. Highly resilient populations of Suckley's cuckoo bumble bee should have robust median occupancy levels, exhibit

	stable or growing occupancy trends compared to earlier time periods, and modern occurrence records.
Redundancy	Redundancy for Suckley's cuckoo bumble bee is described as having multiple, healthy populations widely distributed across the breadth of adaptive diversity relative to the spatial occurrence of catastrophic events. Since we lack the information needed to delineate populations for the species, the spatial distribution of extant analytical units will be used to describe redundancy. Contraction of the species range will be considered a loss in redundancy.
Representation	Suckley's cuckoo bumble bee representation is described as having healthy populations widely distributed across a breadth of ecological conditions. The analytical units are large areas across the continent and will also be used as representative units in this assessment. These units delineate broad ecotypes the species occupies, and the number of representative units and the resiliency condition (based on occupancy) of the analytical units will be used to describe representation. A loss of representation at the species level occurs when the number of extant representative units decreases over a period of time.

We will also use the occupancy results to analyze host availability, a key resource need for Suckley's cuckoo bumble bee.

7 Historical Context and Current Condition

7.1 Model Results for Suckley’s Cuckoo Bumble Bee

Occupancy of Suckley’s cuckoo bumble bee is positively related to floral resources, and negatively related to monthly mean precipitation (Figure 8a, 8c). The model indicates that Suckley’s cuckoo bumble bee occupancy peaks when the average maximum temperature is near 20 °C (68 °F) and declines at very low and high temperatures (Figure 8b).

After accounting for changes in environmental covariates, the model results also demonstrate that occupancy of Suckley’s cuckoo bumble bee has decreased over time, indicating declines related to additional factors that were not explicitly incorporated into the models. Moreover, while changes in temperature, precipitation and floral resources drove part of the observed historical decline, much of the decline is accounted for by unmodeled variables. These temporal trends (e.g., declines) may be in response to stressors like Host Species Declines, Pathogen Prevalence, and Pesticide Application, that were not directly incorporated into the statistical models, but are believed to impact the species (see 5 Influencing Factors).

Suckley’s cuckoo bumble bee has exhibited a strong, statistically significant temporal decline in occupancy, range wide (Figure 8d). This equated to a roughly 85 percent decline in occupancy between the 1900–1910 and 2010–2020 eras across the range (Figure 8d), which closely matches the 77 percent decline from 1912 to 2012 estimated by the IUCN for the species (Hatfield et al., 2015, p. 3). In this multi-species occupancy model, individual species are pulled modestly toward the mean trend of all other species. The mean trend in occupancy for all bumble bee species in the multi-species occupancy model is not declining, so the serious decline observed for Suckley’s cuckoo bumble bee is even more notable.

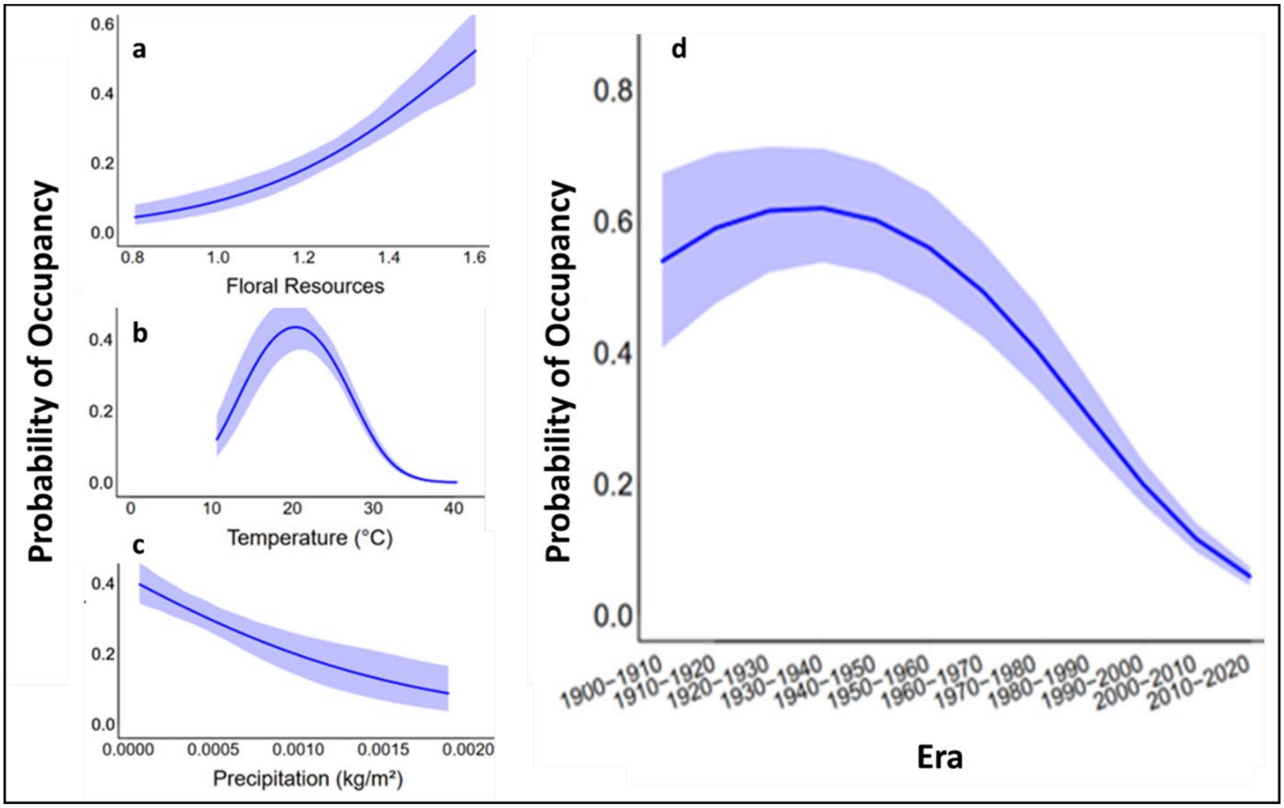


Figure 8b. *Modeled relationship between Suckley's cuckoo bumble bee probability of occupancy and (a) floral resources, (b) the average maximum temperature of the warmest month in each year, and (c) decadal monthly mean precipitation. Panel (d) shows the estimated temporal trend in probability of occupancy over time. All results are based on modified models from Jackson et al. (2022). Here, a larger value for the floral resource metric equates to greater resource availability.*

7.2 Historical Context and Current Condition of Suckley's Cuckoo Bumble Bee

We analyze the current condition (2000–2020) of Suckley's cuckoo bumble bee by evaluating the current occupancy of the species per each analytical unit. We then use the historical occupancy from 1900-1960 to put the current occupancy into perspective. By comparing the two, we can present a trend in Suckley's cuckoo bumble bee occupancy for each analytical unit. As noted above, the species has experienced a range wide decline of 85 percent from the 1900 to 2020 eras. We also provide the most recent occurrence record of each analytical unit.

Suckley's cuckoo bumble bee has exhibited a statistically significant decline, resulting in lower occupancy relative to historical. Historically, the median occupancy of Suckley's cuckoo bumble bee range wide was 0.653. Median occupancy in each analytical unit ranged in the middle probabilities (0.485 up to 0.706), except for the South Central Semi-Arid Prairies (0.118; Table 7). By the current time period, all analytical units exhibit a lower probability of occupancy, with median occupancy probabilities less than 0.16 for all analytical units, and range wide (0.128). This means that on average there was a less than 16 percent probability of occupancy in each of the 50 km (19.3 mi) grids in the last two decades. Though we are representing each analytical unit by its median occupancy score, the maximum value of any of the 50 km

(19.3 mi) grids in any of the analytical units during the current time period ranges from 0.15 to 0.35 across analytical units (with 0.35 in the Western Cordillera), meaning that even in the most prevalent analytical unit there was at most a 35 percent probability of occupancy in any location, in the last two decades.

Comparing current to historical occupancy, the Taiga Plains had the smallest observed decrease in occupancy of 77 percent, while the greatest declines were in the Temperate Prairies and South Central Semi-Arid Prairies where occupancy declined 89 percent. This shift in probability of occupancy from historical to current is visualized across the range in the maps displayed in Figure 10. Across the analytical units, current median occupancy of Suckley's cuckoo bumble bee is estimated to currently be the highest in the Taiga Plains (0.161) and the lowest in the South Central Semi-Arid Prairies (0.100; Table 7). The species has not been detected in the United States since 2016 and has only been detected since 2000 in 9 of the 15 analytical units (60 percent of the total analytical units; Table 7 and Figure 9).

***Figure 9**. Analytical units and all occurrence records of Suckley's cuckoo bumble bee. Occurrence points shown from time periods pre-2000 (black), post-2000 (blue), and post-2018 (red). The date of the most recent occurrence point per analytical unit is also labelled.*

In the east, three of four eastern analytical units lack detections of Suckley's cuckoo bumble bee in the last 50 years. Suckley's cuckoo bumble bee has not been detected since 2000 in the Atlantic Highlands (last detection 1924), Hudson Plains (last detection 1949), Mixed Wood Plains (last detection 1971). The Softwood Shield (Newfoundland) is the only location in the east where we have modern records of this species persisting; the last detection was in 2010 (Table 7 and Figure 9).

In the west, three of the eleven analytical units lack modern detections of Suckley's cuckoo bumble bee. Notably, Marine West Coast Forest has 515 historical occurrence records (23 percent of total occurrences), but no occurrence records post 1982. The Mixed Wood Shield has two historical records of the species, with the most recent in 1995. The species has not been observed in the Taiga Plains since 1962, however this is a much less sampled area. Almost half (49 percent) of total occurrence records for Suckley's cuckoo bumble bee are in the Western Cordillera, where occupancy declines from 0.692 historically to 0.140 in the current time period, an 80 percent decline (Table 7).

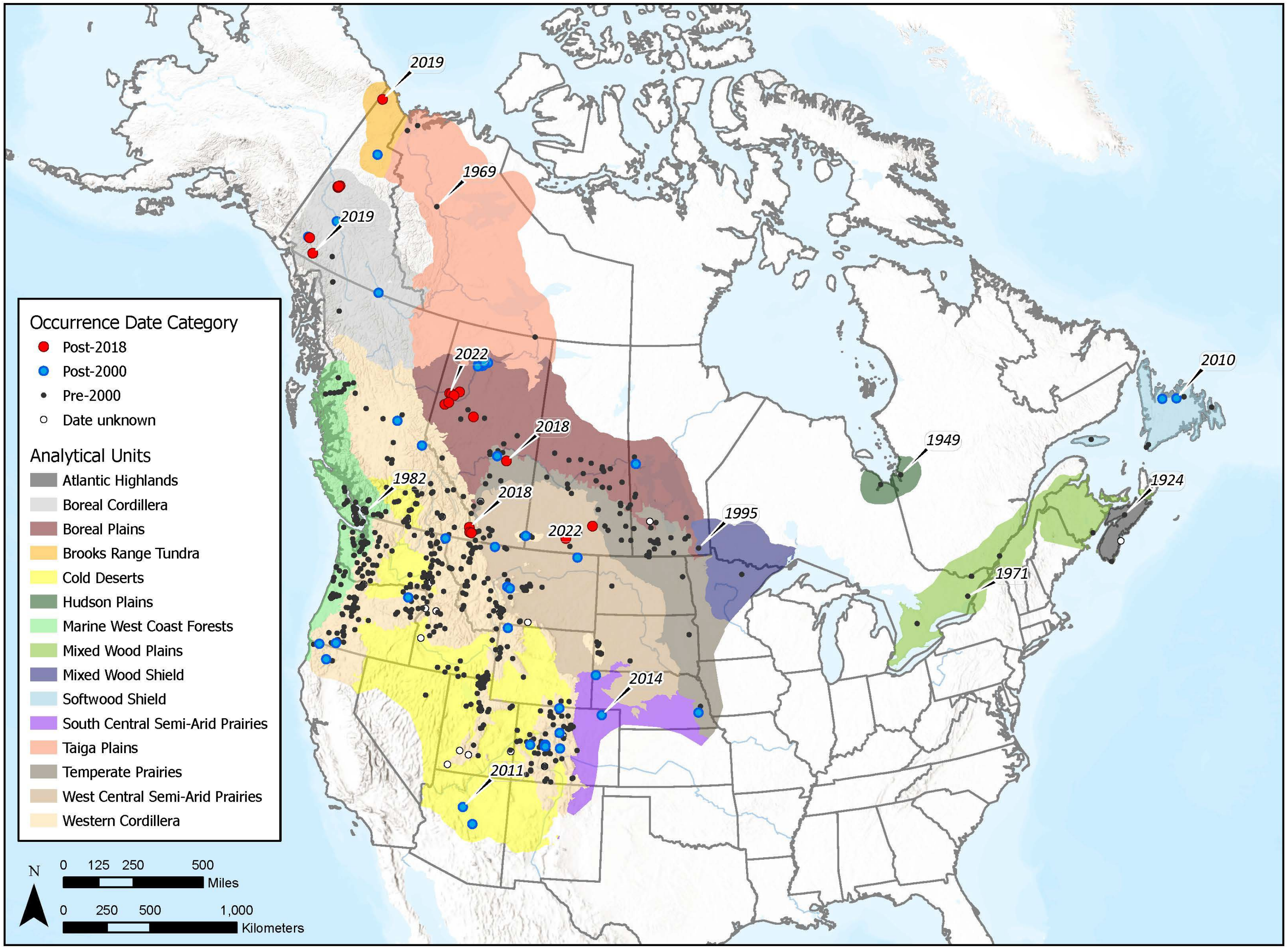


***Figure 9**. Analytical units and all occurrence records of Suckley's cuckoo bumble bee. Occurrence points shown from time periods pre-2000 (black), post-2000 (blue), and post-2018 (red). The date of the most recent occurrence point per analytical unit is also labelled.*

Table 7. *Median estimated probability of occupancy for Suckley's cuckoo bumble bee in each analytical unit during the historical period (average from 1900–1960) and the current (2000–2020) period. The percent change in median occupancy from historical to current period is also noted for each analytical unit, as is the maximum current occupancy value currently estimated across any of the 50 km (19.3 mi) grid cells in each analytical unit, the total land area of each analytical unit (km^2) and the percent of the range the unit represents in terms of land area.*

Analytical unit	Area (km^2)	Percent of range	Last detection	Median historical occupancy	Median current occupancy	Maximum current occupancy	Percent change
Atlantic Highlands (East)	44,482	0.6	1924	0.712	0.149	0.184	-79%
Boreal Cordillera	532,782	7.4	2019	0.706	0.147	0.180	-79%
Boreal Plains	772,369	10.8	2022	0.661	0.145	0.181	-78%
Brooks Range Tundra	99,755	1.4	2019	0.705	0.147	0.188	-79%
Cold Deserts	1,047,895	14.7	2011	0.485	0.061	0.276	-87%
Hudson Plains (East)	55,863	0.8	1949	0.682	0.133	0.153	-80%
Marine West Coast Forests	250,206	3.5	1982	0.580	0.095	0.177	-84%
Mixed Wood Plains (East)	360,958	5.0	1971	0.640	0.106	0.176	-83%
Mixed Wood Shield	205,107	2.9	1995	0.590	0.098	0.181	-83%
Softwood Shield (East)	119,152	1.7	2010	0.685	0.142	0.182	-79%
South Central Semi-Arid Prairies	208,917	2.9	2014	0.118	0.013	0.190	-89%
Taiga Plains	905,619	12.7	1969	0.708	0.161	0.248	-77%
Temperate Prairies	501,088	7.0	2018	0.263	0.028	0.182	-89%
West Central Semi-Arid Prairies	832,871	11.6	2022	0.577	0.100	0.245	-83%
Western Cordillera	1,214,900	17.0	2018	0.692	0.140	0 .353	-80%
Range wide	**7,151,965**	**100**	**2022**[1]	**0.653**	**0.128**	**0.353**	**-85%**[2]

[1] The last detection of Suckley's cuckoo bumble bee in our dataset was 2022. However, field data from across the country from the 2023 field season or beyond had not been fully curated in time to include in this SSA.

[2] Range wide decline from 1900 to present, based on an analysis of all Suckley's cuckoo bumble bee occurrence records, from 7.1 Model Results for Suckley's Cuckoo Bumble Bee Figure 8. Decline from the 1900-1960 historical period (as represented in the table) to current time period (2000-2020) is 80 percent.

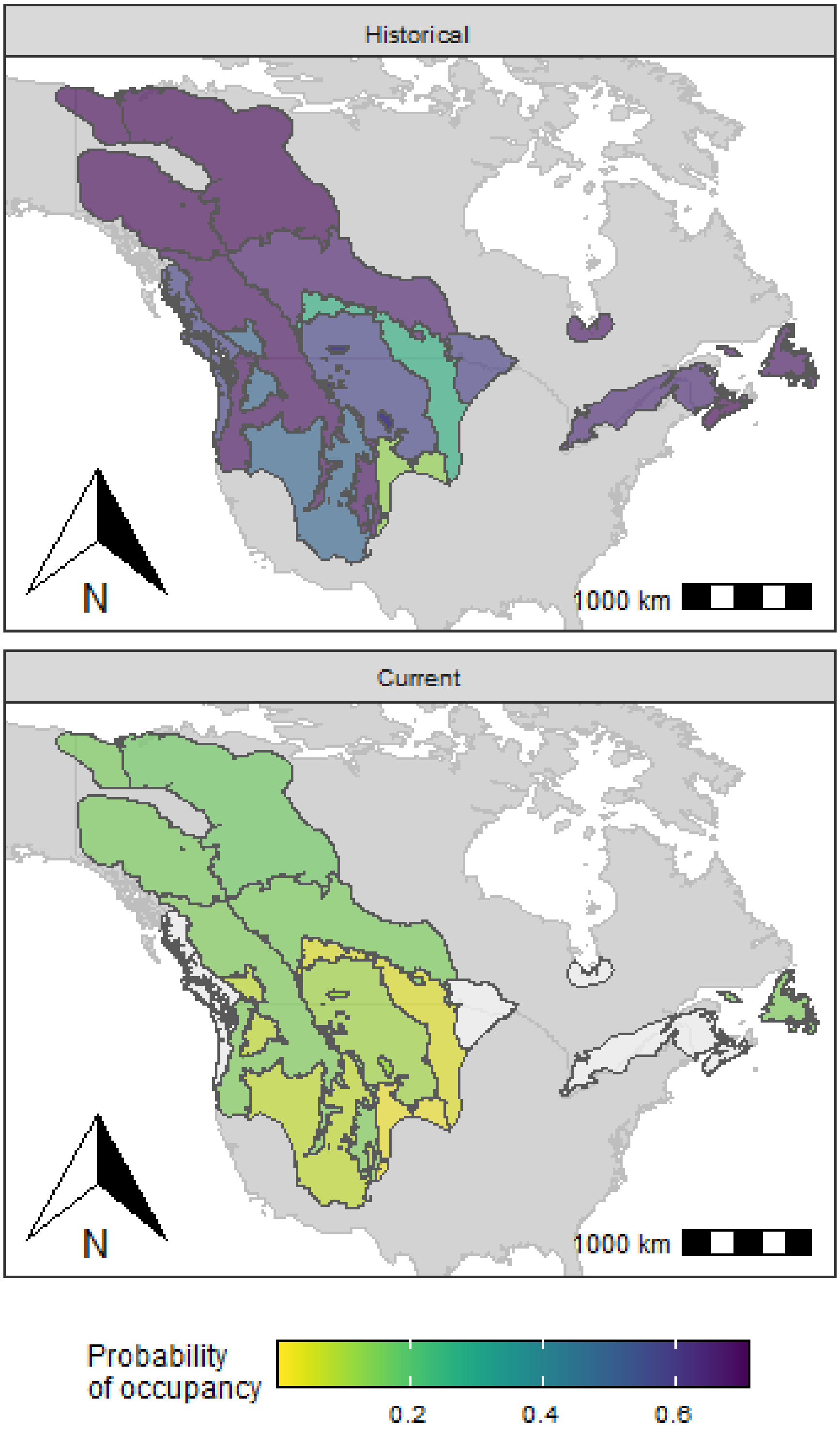


Figure 10. *Median probability of occupancy for Suckley's cuckoo bumble bee across analytical units for the historical (1900–1960) and current (2000–2020) period. Analytical units colored in white are likely extirpated, given the lack of detections within the current condition time period, and the low estimated probability of occupancy.*

7.3 Current Condition of Host Species

We assessed host species availability primarily based on whether the confirmed hosts (*B. occidentalis* and *B. nevadensis)* and likely preferred host from subgenus *Bombus* (*B. terricola, B. affinis, B. cryptarum, B. mckayi)* have been detected in each analytical unit since 2000. Suckley's cuckoo bumble bee is most likely to overlap spatially with a host species in locations where the probability of occupancy is high for both Suckley's cuckoo bumble bee, and the host. We therefore also assessed host species availability based on the probability of occupancy of the confirmed host species. The greater the spatial overlap with hosts, the greater the potential for demographic connectivity of Suckley’s cuckoo bumble bee individuals for breeding.

Of the confirmed host species, *B. occidentalis* has been detected at least once in seven of the 15 analytical units since 2000 (47 percent of the total analytical units) and *B. nevadensis* in nine (60 percent of the total analytical units). Within the dataset of all Suckley’s cuckoo bumble bee occurrence records, 97 percent of Suckley’s cuckoo bumble bee occurrence points are located within analytical units where *B. occidentalis* and *B. nevadensis* occur. *B. occidentalis* is the most widely documented and recognized host species.

Four of six host species exhibited statistically significant temporal declines in occupancy range wide (Figure 11). All declining species were in the subgenus *Bombus,* the subgenus documented to be experiencing range wide declines (Colla & Packer, 2008, p. 1387; Giles & Ascher, 2006, pp. 217–218; Janousek et al., 2023, p. 2; Schweitzer et al., 2012, p. 7). Confirmed host *B. nevadensis* and potential host *B. cryptarum* (also in subgenus *Bombus*) have an increased probability of occupancy. If host declines are the driver of Suckley’s cuckoo bumble bee declines, then the results of this model suggest *B. occidentalis, B. mckayi, B. affinis, and B. terricola* are the host species driving Suckley’s cuckoo bumble bees declining trend. Lastly, host redundancy, or the availability of multiple hosts, is lowest in the east where the declining *B. terricola* is the only available and preferred host.

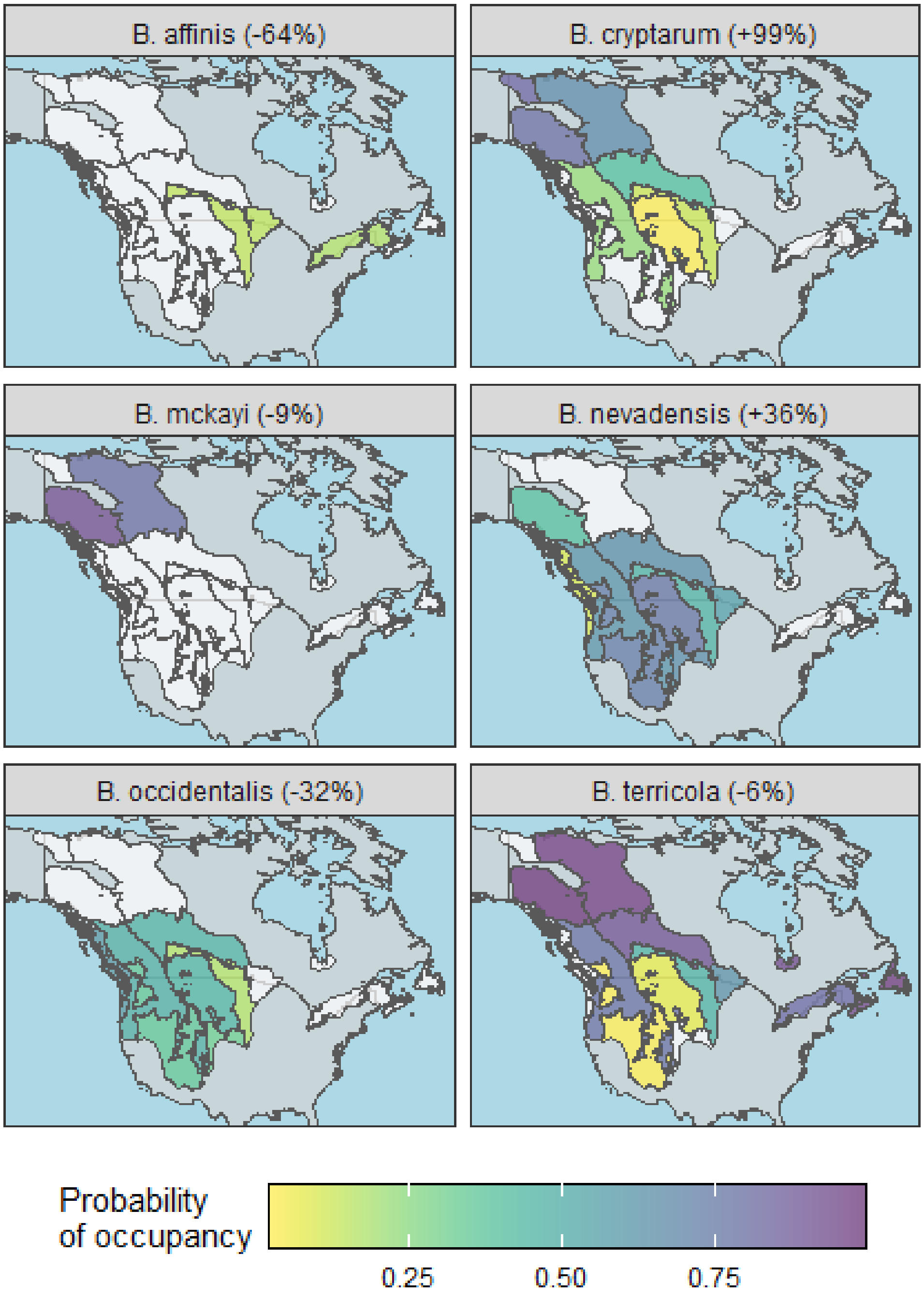


***Figure 11**. Median probability of occupancy for Suckley's cuckoo bumble bee hosts across analytical units for the current (2000–2020) period. Results are shown for two confirmed Suckley's cuckoo bumble bee host species (B. occidentalis and B. nevadensis), and four other likely preferred hosts in the Bombus sub-genus. Analytical units in white have not had a confirmed detection of the host species since 2000. The percent change in occupancy from historical period (1900–1910) to current is shown for each species, in parentheses; negative numbers indicate a decline, positive numbers an increase.*

7.4 Summary of Current Condition

Resiliency

The current condition analysis indicates a statistically significant, range wide 85 percent decline in Suckley's cuckoo bumble bee occupancy since the historical period. Median occupancy in all analytical units is less than 0.16 in the 2000–2020 current period. Additionally, the species has not been observed in the contiguous United States since 2016 despite widespread historical occurrence records and increased sampling effort for bumble bees. These results suggest that the species is currently found in fewer locations across its range than it was historically. High abundance and survival are demographic needs of healthy Suckley's cuckoo bumble bee populations. While there are some areas in Canada where the species is still regularly observed, these results suggest that the resiliency is generally lower across all analytical units, compared to historical conditions. As noted earlier, there were too fewer occurrence records to assess if overall rangewide historical declines were exhibited locally in the Brooks Range Tundra, Taiga Plains, Mixed Woods Shield, South Central Semi-Arid Prairies, Hudson plains, and Atlantic highlands (roughly 21 percent of the occurrence polygon). However, since Suckley's cuckoo bumble bee has not been detected in the Hudson Plains since 1949, the Atlantic Highlands since 1924, and the Mixed Wood Shield since 1995, it is likely that declines have also occurred in these regions. Therefore, our estimates of current probability of occupancy are likely more certain and representative in the analytical units with more occurrence records (79 percent of the of the occurrence polygon) but are less certain in regions with fewer occurrence records, and especially in the Brooks Range Tundra, Taiga Plains, and South Central Semi-Arid Prairies.

Redundancy

Redundancy buffers the species against catastrophic events and can be summarized based on the spatial distribution of resilient populations relative to catastrophic events. Historically, Suckley's cuckoo bumble bee was observed in four ecoregions (analytical units) in the East. The only current records (post 2000) of the species from the eastern portion of the range are in the Softwood Shield (Newfoundland). Probability of occupancy is estimated to be low across the other three eastern analytical units and we have no evidence of the species persisting in those areas within the current time period. Thus, three eastern analytical units are likely quasi-extirpated (when the density of reproductive individuals in a population becomes so small that it is unable to sustain a growing or even stable population). This apparent contraction of the range in the east results in a loss of redundancy for the species.

In the west, we have also seen range contraction with quasi-extirpation (no records in the current time period) in Marine West Coast Forests and Mixed Wood Shield analytical units, and thus a loss in redundancy. The Taiga Plains analytical unit also does not have modern records of Suckley's cuckoo bumble bee; however, we did not consider it quasi-extirpated because the region is less sampled. The Marine West Coast Forest historically had over 500 records of Suckley's cuckoo bumble bee and, while isolated occupied areas may exist, we have not observed the species in this analytical unit since 1982 despite increased search effort by Canadian colleagues since the COSEWIC listing decision in 2019 (S. Cannings, J. Heron, and C. Sheffield 2023, pers. Comms). Additionally, the reduced occupancy across the entire range indicates additional areas could be at risk of extirpation, which would further reduce redundancy.

Finally, given that four of the six confirmed and potential host species are in decline, redundancy in terms of host species is potentially currently lower than it was historically. For instance, if a catastrophic event wipes out one host species in an area, then there is less likely to be an alternative host species available.

Representation

Currently, Suckley's cuckoo bumble bee likely has substantially lower connectivity, representation, and adaptive capacity than historically, due to estimated declines in occupancy.

Population connectivity is important for Suckley's cuckoo bumble bee's viability as it increases the likelihood of genetic diversity and promotes successful haplodiploid reproduction. However, population connectivity is already a constraint for cuckoo bumble bees because they live in small, fragmented populations because of their dependence on host bumble bee colonies (Suhonen et al., 2016, p. 529). Dispersal of bees to find unrelated mates is aided by the proximity of other usurped colonies. Consequently, the sharp historical decrease in the prevalence of both Suckley's cuckoo bumble bee, and many of its confirmed and potential host species, has likely reduced population connectivity relative to historical conditions. Reduced gene flow may have consequences on the genetic diversity of Suckley's cuckoo bumble bee, because small populations can experience stronger genetic drift (Zayed, 2009, p. 246). This is important because high genetic diversity reduces prevalence of some pathogens (Parsche & Lattorff, 2018, p. 900), and the risk of "matched mating", which produces diploid males that do not contribute to population growth (Zayed, 2009, p. 239). Loss of connectivity, genetic drift, and inbreeding may be particularly consequential for bumble bees due to their low effective population size and their haplodiploid sex determination (Goulson et al., 2008, p. 205). The research on these phenomena in *Psithyrus* is limited.

Preserving the breadth of phenotypic diversity of Suckley's cuckoo bumble bee requires maintaining populations across a breadth of ecological conditions. Given the observed 85 percent decrease in the species' occupancy relative to historical conditions, the low current occupancy across all analytical units, and potential that the species is extirpated or quasi-extirpated in three of four analytical units in the eastern portion of the range, the species has lost representation across longitudinal, and ecological gradients. This is important as this variation is the raw material for adaptive evolution, and behavioral plasticity, and determines how organisms respond to selection pressures (Hendry et al., 2011, p. 161). Consequently, species that span environmental gradients are expected to harbor the most phenotypic and genetic variation (Lankau et al., 2011, p. 320).

Moreover, the representation and adaptive capacity of Suckley's cuckoo bumble bee is also dependent on host species, as the distribution of the parasitic bee is restricted by the geographic distribution and population health of host bees, and parasite abundance is low where host abundance is low (Antonovics & Edwards, 2011, p. 1003). Availability of host species may also therefore restrict the adaptive capacity of Suckley's cuckoo bumble bee, given that four of the six confirmed or potential host species are in decline.

8 Future Condition

8.1 Future Condition Scenarios

To analyze the future condition of Suckley's cuckoo bumble bee, future occupancy projections were made under two different scenarios (Table 8). Both scenarios hold floral resources constant at their 2020 levels, as data are not available to project this variable into the future. Both scenarios use future projections of average decadal maximum temperature of the warmest month in each year and decadal monthly mean precipitation. Both scenarios were projected in 20-year increments, out to 2100.

Scenario 1 uses climate projections under RCP4.5, which represents a moderate climate change warming scenario, while Scenario 2 uses projections under RCP8.5, which represents a high warming scenario. Scenario 2 also assumes that average, observed, historical rates of decline due to other, non-climatic factors continue into the future (i.e., trend momentum), while Scenario 1 does not project these historical trends into the future (i.e., no trend momentum). Thus, under Scenario 1, any changes in future occupancy are based solely on changes in climate (assuming RCP4.5), while under Scenario 2, any changes in future occupancy are based on both climate change (assuming RCP8.5), as well as historical rates of change due to non-climatic factors.

We used the same occupancy modeling framework and scenarios to spatially project future probability of occupancy for the confirmed and potential hosts of Suckley's cuckoo bumble bee.

Table 8. *Scenarios for analyzing the future condition of Suckley's cuckoo bumble bee.*

Factor	Scenario 1	Scenario 2
Climate change (temperature and precipitation)	RCP4.5	RCP8.5
Floral resources	Held constant at 2020 levels.	Held constant at 2020 levels.
Trend Momentum. Temporal declines observed after accounting for changes in climate and floral resources (e.g., temporal declines driven by other factors not explicitly included in models, like pesticides, etc.).	No trend momentum: Historical rate of temporal decline (driven by non-climatic factors) not projected into the future.	Continued trend momentum: Historical rate of temporal decline (driven by non-climatic factors) projected into the future.

While we would ideally incorporate additional stressors (such as pesticides) into the occupancy model and future projections, we are limited by a lack of data on many non-climatic stressors. Other bumble bee assessments acknowledge the limited availability of spatial data about key stressors and resources for bees (Janousek et al., 2023, p. 7). The model used in this SSA reflects current best available science, and specifically addresses the data limitation by including a temporal variable that accounts for changes in occupancy due to important historical drivers for which we do not have data. This modeling approach does not specifically identify *which* non-climate related factors drove historical trends but allows us to account for their influence on occupancy. In turn, by incorporating these unmodeled drivers as "trend momentum" in Scenario 2, we can model the continued effects of these non-climatic stressors, albeit in a less direct way. Furthermore, as noted earlier in the SSA we are uncertain whether the historical temporal decline driven by non-climatic factors are representative in portions of the range with fewer occurrence records (Brooks Range Tundra, Mixed Wood Shield, South central Semi-Arid Prairie, Taiga Plains, Hudson Plains, and Atlantic Highlands). We account in part for this uncertainty by including separate scenarios with and a scenario without this trend momentum projected into the future. Despite these data limitations, and our indirect method for incorporating non-climatic stressors, our approach directly models occupancy in relation to precipitation and temperature, which are key drivers of bumble bee occupancy and fitness. Indeed, occupancy modelling for the *B. occidentalis* SSA incorporated more variables (such as land cover) and found that temperature had the largest effect on occupancy, with twice the negative effect as the second largest climate effect (Janousek et al., 2023, p. 2).

Both scenarios are somewhat optimistic. Though occupancy of Suckley's cuckoo bumble bee is entirely dependent on presence of hosts species, the occupancy model does not necessitate host presence as a parameter for parasite presence. Thus, future projections may be optimistic under both scenarios, as they may project occupancy at sites where climate conditions are likely suitable for Suckley's cuckoo bumble bee, but host species are not likely present. Moreover, these scenarios both merely capture occupancy probability, so neither captures the potential for synergistic threats, nor potential impacts to connectivity, resiliency, redundancy, or representation as occupancy declines, and as populations become smaller, and more isolated.

In addition, Scenario 1 is likely optimistic (e.g., likely over-estimates occupancy and in turn resiliency), because it only reflects future changes in occupancy of Suckley's cuckoo bumble bee due to climate change, and does not project changes in occupancy due to any of the other factors that have historically impacted the species, and are likely to continue to do so into the future (see 5.2 StressorsStressors). While Scenario 2 projects more severe threats into the future, it too may be somewhat optimistic ***in part of the range*** because it assumes that the declining trend of Suckley's cuckoo bumble bee remains the same as observed historically. However, where declines have already occurred, it is possible that the influence of stressors, such as pesticides application and disease, could intensify in the future. In contrast, as noted earlier, the rangewide historical declines in occupancy may or may not be representative in analytical units with few occurrence records (Brooks Range Tundra, Mixed Wood Shield, South central Semi-Arid Prairie, Taiga Plains, Hudson Plains, and Atlantic Highlands; roughly 21 percent of the occurrence polygon). Consequently, future projections that incorporate trend momentum, under Scenario 2 could be biased low in these analytical units.

8.2 Future Condition of Suckley's Cuckoo Bumble Bee

Occupancy of Suckley's cuckoo bumble bee is projected to decline into the future across the range, and in all analytical units under both scenarios, with more marked declines under Scenario 2 (Figure 12).

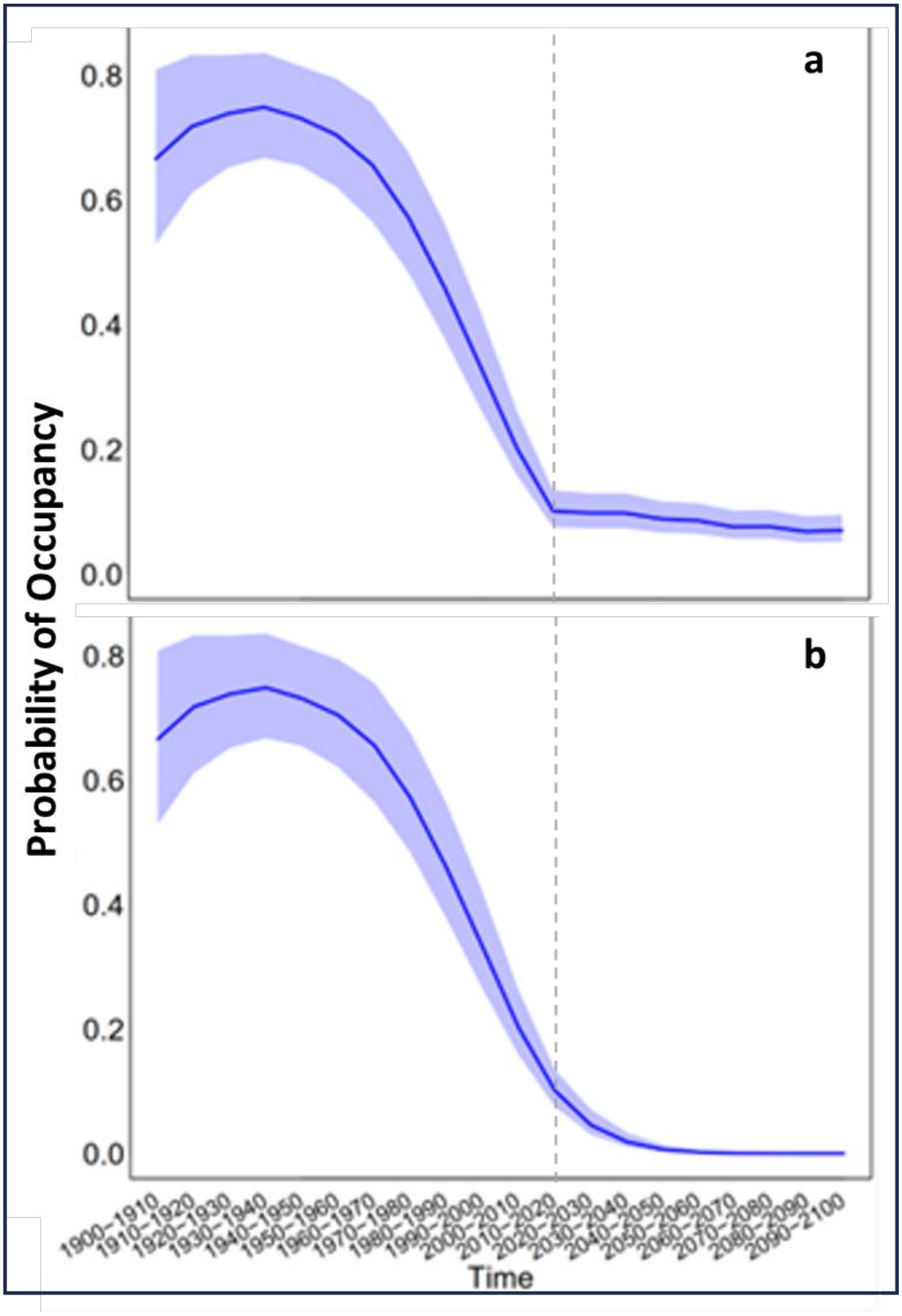


Figure 12. *Range-wide temporal trend in occupancy of Suckley's cuckoo bumble bee over time, including declines from the historical to current period, and projected occupancy into the future under (**a**) **Scenario 1**, which assumes RCP4.5, and projects changes in future occupancy based solely on climate change (e.g., historical rates of declines due to other factors are not projected), and (**b**) **Scenario 2**, which assumes RCP8.5, and projects changes in future occupancy based on both climate change, and historical rates of decline due to non-climactic factors.*

Near-term projections (2020–2040)

As noted earlier, Suckley's cuckoo bumble bee is already potentially quasi-extirpated in the Atlantic Highlands, Hudson Plains, Marine West Coast Forests, Mixed Wood Plains, and Mixed Wood Shield analytical units, where it is estimated to currently have low probability of occupancy and lacks modern detections. Our analyses indicate that occupancy is expected to continue to decline in the near term (2020–2040), across all analytical units, under both scenarios. Thus, all analytical units are projected to have relatively low occupancy, in the next two decades, while the South Central Semi-Arid Prairie analytical unit may also approach quasi-extirpation, particularly under Scenario 2.

Under Scenario 1, median probability of occupancy is estimated to be less than 0.11 across all analytical units over the next two decades (Table 9). This represents a 26 to 77 percent decline relative to 2000–2020 estimates, depending on analytical unit. Median occupancy is estimated to be highest in the Brooks Range Tundra (0.109), and lowest in the South Central Semi-Arid Prairies (0.003).

Under Scenario 2, median probability of occupancy is estimated to be less than 0.05 across all analytical units over the next two decades (Table 10). This represents a 73 to 92 percent decline relative to 2000–2020 estimates, depending on analytical unit. Median occupancy is estimated to be highest in the Brooks Range Tundra (0.04), and lowest in the South Central Semi-Arid Prairies (0.001).

Mid- and late-century projections (2040–2060 and 2080–2100)

Under scenario 1, median probability of occupancy is estimated to be less than 0.1 in all but two analytical units by mid-century (2040–2060). At mid-century, the highest probability of occupancy (0.124) is projected to occur in the Brooks Range Tundra, in the northwestern portion of the range (Table 9). Relative to current conditions, occupancy is expected to decline from 16 to 79 percent by mid-century and by 17 to 92 percent late-century depending on analytical unit. All but four analytical units are expected to experience a greater than 50 percent decline in occupancy, relative to current, by late century. The steepest future declines are projected in the Atlantic highlands, Boreal Plains, Hudson Plains, Mixed Wood Shield, West Central Semi-Arid Prairies, and Western Cordillera (Figure 13). While occupancy in the Brooks Range Tundra and Boreal Cordillera has exhibited a marked historical decline and is expected to be low at the 2020–2040 time step, occupancy is projected to subsequently level off from the 2020–2040 period into the future under this scenario. This indicates that future occupancy in these far northern analytical units may not be sharply impacted by climate change in the long term, if in fact declines related to other non-climactic factors are minimal in the future.

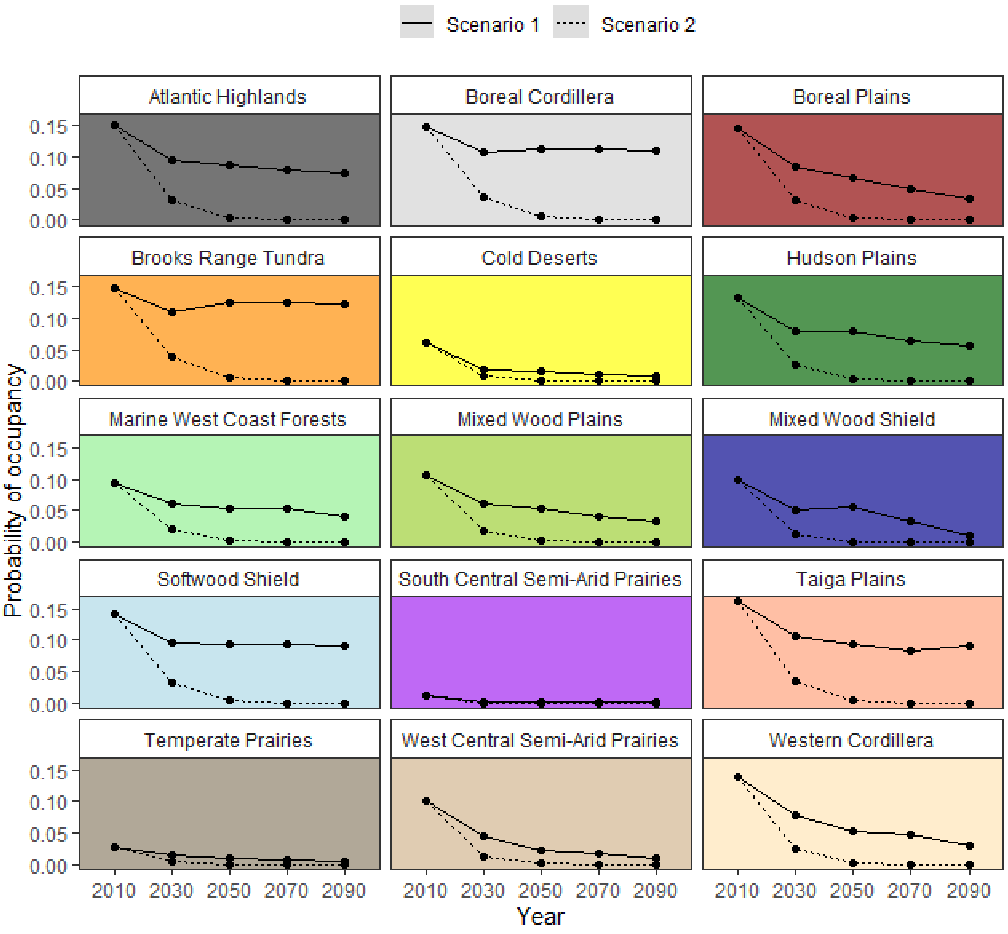
Under scenario 2, median probability of occupancy is projected to be less than 0.1 across all analytical units as early as the 2020-2040 period (Table 10). Compared to current conditions, median occupancy is expected to decline by 96 to 100 percent across the analytical units by mid-century. Median probability of occupancy is expected to be at or near zero by the 2040-2060 period in the Cold Desert, South Central Semi-Arid Prairies, and Temperate Prairies, indicating quasi-extirpation in these regions. Median occupancy is estimated to be at or near zero in all analytical units by the 2060-2080 period, indicating that nearly all locations across the range are expected to be unoccupied by this time. Steep declines are evident across all analytical units except the South Central Semi-Arid Prairies and the Temperate Prairies, where current occupancy is already low enough that steep future declines in occupancy are simply not possible (Figure 13). In Scenario 2, all analytical units are projected to be unoccupied by 2100, and all but one by the 2060-2080 period, suggesting near extirpation across the range by as early as 2060-2080 (Table 10 and Figure 14).

Table 9. Scenario 1*. Median estimated probability of occupancy for Suckley's cuckoo bumble bee in each analytical unit during the current (2000–2020) period, near term (2020–2040), and future time periods. Scenario 1 assumes RCP4.5, and projects changes in future occupancy based solely on climate change (e.g., historical rates of declines due to other factors are not projected). The percent change in median occupancy from the current period to near term (2020–2040), mid-century (2040-2060), and late century (2080–2100) is also noted for each analytical unit. The 25th, 50th and 75th quantile for historical, current, and all future estimates of probability of occupancy are presented in Table A6 in 11.5 Supplementary Tables and Figures.*

Analytical unit	**Historical**	**Current (2000-2020)**	**Near-term (2020-2040)**	**2040-2060**	**2060-2080**	**2080-2100**	**Percent change by 2020-2040**	**Percent change by 2040-2060**	**Percent change by 2080-2100**
Atlantic Highlands (East)	0.712	0.149	0.096	0.087	0.079	0.075	-36%	*-42%*	*-50%*
Boreal Cordillera	0.706	0.147	0.108	0.112	0.111	0.110	-27%	*-24%*	*-25%*
Boreal Plains	0.661	0.145	0.084	0.067	0.048	0.033	-42%	*-54%*	*-77%*
Brooks Range Tundra	0.705	0.147	0.109	0.124	0.125	0.122	-26%	*-16%*	*-17%*
Cold Deserts	0.485	0.061	0.019	0.016	0.010	0.009	-69%	*-74%*	*-85%*
Hudson Plains (East)	0.682	0.133	0.079	0.078	0.064	0.056	-41%	*-41%*	*-58%*
Marine West Coast Forests	0.580	0.095	0.060	0.052	0.052	0.041	-37%	*-45%*	*-57%*
Mixed Wood Plains (East)	0.640	0.106	0.060	0.054	0.041	0.034	-43%	*-49%*	*-68%*
Mixed Wood Shield	0.590	0.098	0.051	0.056	0.033	0.011	-48%	*-43%*	*-89%*
Softwood Shield (East)	0.685	0.142	0.096	0.093	0.094	0.009	-32%	*-35%*	*-37%*
South Central Semi-Arid Prairies	0.118	0.013	0.003	0.003	0.002	0.001	-77%	*-77%*	*-92%*
Taiga Plains	0.708	0.161	0.106	0.092	0.084	0.090	-34%	*-43%*	*-44%*
Temperate Prairies	0.263	0.028	0.014	0.010	0.006	0.004	-50%	*-64%*	*-86%*
West Central Semi-Arid Prairies	0.577	0.100	0.046	0.021	0.016	0.009	-54%	*-79%*	*-91%*
Western Cordillera	0.692	0.140	0.078	0.052	0.046	0.030	-44%	*-63%*	*-79%*

Table 10. Scenario 2. *Median estimated probability of occupancy for Suckley's cuckoo bumble bee in each analytical unit during the current (2000–-2020) period, near-term (2020–2040) and future time periods. Scenario 2 assumes RCP8.5, and projects changes in future occupancy based on both climate change, and historical rates of decline due to non-climactic factors. The percent change in median occupancy from the current period to near-term (2020–2040), mid-century (2040–2060), and late century (2080–2100) is also noted for each analytical unit. The 25th, 50th and 75th quantile for historical, current, and all future estimates of probability of occupancy are presented in Table A7 in 11.5 Supplementary Tables and Figures.*

Analytical unit	Historical	Current (2000-2020)	Near-term (2020-2040)	2040-2060	2060-2080	2080-2100	Percent change by 2020-2040	Percent change by 2040-2060	Percent change by 2080-2100
Atlantic Highlands (East)	0.712	0.149	0.031	0.004	0	0	-79%	*-97%*	*-100%*
Boreal Cordillera	0.706	0.147	0.037	0.005	0	0	-75%	*-97%*	*-100%*
Boreal Plains	0.661	0.145	0.031	0.003	0	0	-79%	*-98%*	*-100%*
Brooks Range Tundra	0.705	0.147	0.040	0.006	0.001	0	-73%	*-96%*	*-100%*
Cold Deserts	0.485	0.061	0.007	0	0	0	-89%	*-100%*	*-100%*
Hudson Plains (East)	0.682	0.133	0.025	0.003	0	0	-81%	*-98%*	*-100%*
Marine West Coast Forests	0.580	0.095	0.020	0.002	0	0	-79%	*-98%*	*-100%*
Mixed Wood Plains (East)	0.640	0.106	0.019	0.002	0	0	-82%	*-98%*	*-100%*
Mixed Wood Shield	0.590	0.098	0.014	0.001	0	0	-86%	*-99%*	*-100%*
Softwood Shield (East)	0.685	0.142	0.032	0.004	0	0	-77%	*-97%*	*-100%*
South Central Semi-Arid Prairies	0.118	0.013	0.001	0	0	0	-92%	*-100%*	*-100%*
Taiga Plains	0.708	0.161	0.035	0.004	0	0	-78%	*-98%*	*-100%*
Temperate Prairies	0.263	0.028	0.004	0	0	0	-86%	*-100%*	*-100%*
West Central Semi-Arid Prairies	0.577	0.100	0.013	0.001	0	0	-87%	*-99%*	*-100%*
Western Cordillera	0.692	0.140	0.024	0.002	0	0	-83%	*-99%*	*-100%*



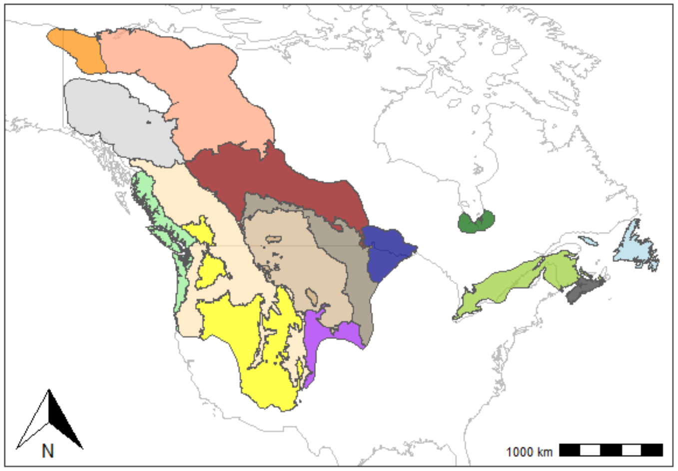


Figure 13. *Projected changes in median probability of occupancy for each analytical unit under Scenario 1 (solid lines), which assumes RCP4.5, and projects future changes in occupancy based solely on climate change (e.g., historical rates of declines due to other factors are not projected), and Scenario 2 (dashed lines), which assumes RCP8.5, and projects future changes in occupancy based on both climate change, and historical rates of decline due to non-climactic factors. A map of the analytical units is shown for reference. Note these declines are above and beyond the 85 percent decline that has occurred since the 1900s.*

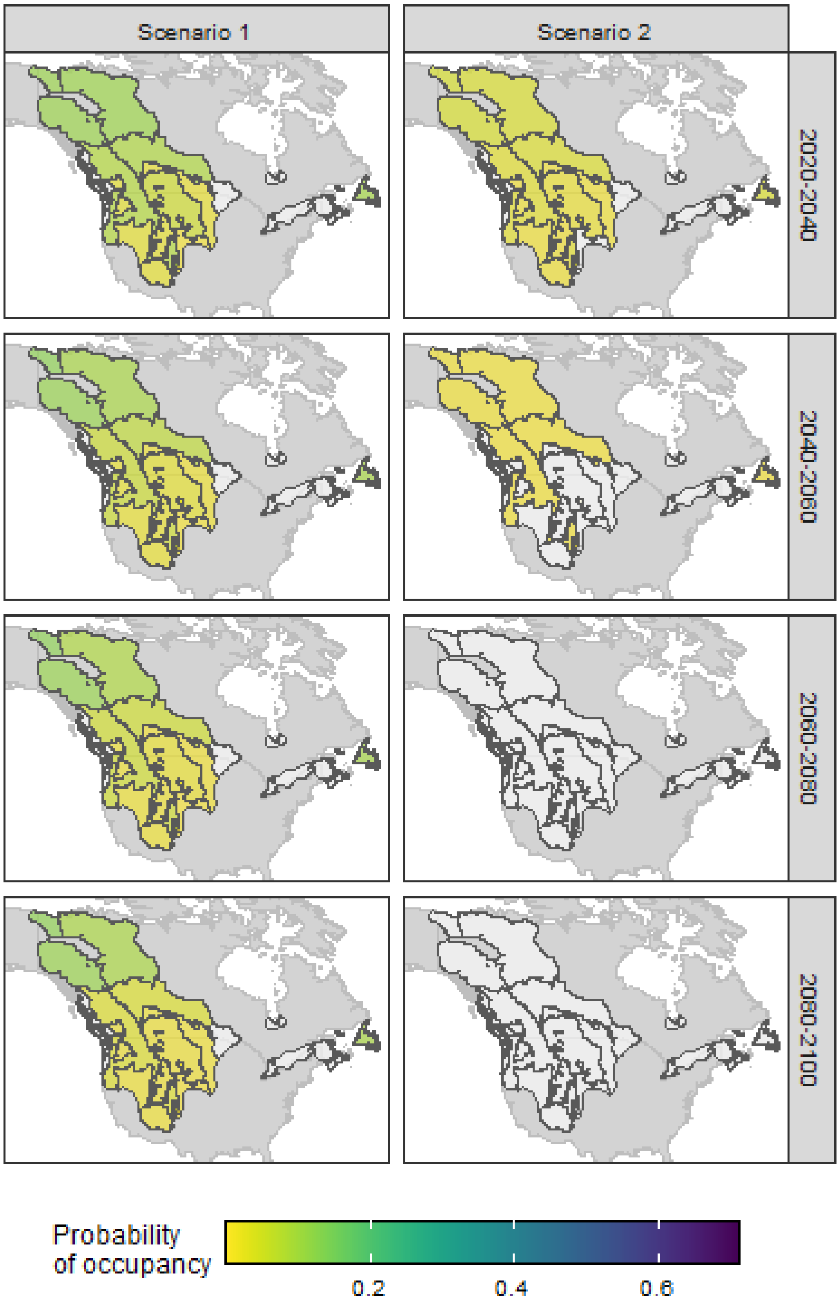


Figure 14. *Predicted median probability of occupancy for Suckley's cuckoo bumble bee in each analytical unit under a low impact future scenario (Scenario 1; left) and a higher impact future scenario (Scenario 2; right). Scenario 1 assumes RCP4.5 and projects changes in future occupancy based solely on climate change (e.g., historical rates of declines due to other factors are not projected), while Scenario 2 assumes RCP8.5, and projects changes in future occupancy based on both climate change, and historical rates of decline due to non-climactic factors. Analytical units are colored white if the median probability of occupancy is predicted to be less than 0.001, or if Suckley's cuckoo bumble bee has not been detected in the analytical unit during the current condition period (2000–-2020), with the exception of the Taiga Plains.*

8.3 Future Condition of Host Species

Under both scenarios, confirmed and widely documented host *B. occidentalis* decreases in occupancy in all analytical units (Figure 15). Under Scenario 1, *B. occidentalis* has less than a 50 percent probability of occupancy in six of seven analytical units by mid-century and all but one has less than 30 percent probability of occupancy by late century. Interestingly, the one analytical unit with relatively higher occupancy is the Marine West Coast Forest, where Suckley's cuckoo bumble bee have not been observed since 1982. This result supports the notion that ideal host and parasite habitat may not be the same. Meanwhile, under Scenario 2, *B. occidentalis* has less than a 40 percent probability of occupancy in all analytical units by mid-century and less than six percent probability of occupancy in all analytical units by late century.

Under Scenario 1, probability of occupancy of all confirmed and preferred host species is projected to decline into the future range wide, with the sharpest declines projected for *B. occidentalis*, *B. cryptarum*, *B. mckayi*, and *B. terricola* (Figure 15**Figure 14**). Thus, host availability and redundancy are both expected to decrease in the future across the range. As both Suckley's cuckoo bumble bee and its hosts become less prevalent, the probability of them overlapping spatially decreases, as does demographic connectivity between Suckley's cuckoo bumble bees during the breeding period.

Under Scenario 2, occupancy of *B. occidentalis*, *B. affinis*, *B. mckayi*, and *B. terricola* is expected to decline, while occupancy of *B. nevadensis* and *B. cryptarum* are expected to increase (Figure 15). Host availability is also expected to generally decrease into the future across the range. Though the prevalence of *B. cryptarum* and *B. nevadensis* is expected to increase, prevalence of all other host species is projected to decline markedly. We do not have a nuanced understanding of host preference and if certain hosts are more biologically suitable to support the parasite. Thus, it is difficult to presume that the increased occupancy of *B. nevadensis* and *B. cryptarum* will widely support Suckley's cuckoo bumble into the future if other hosts, such as *B. occidentalis*, are absent.

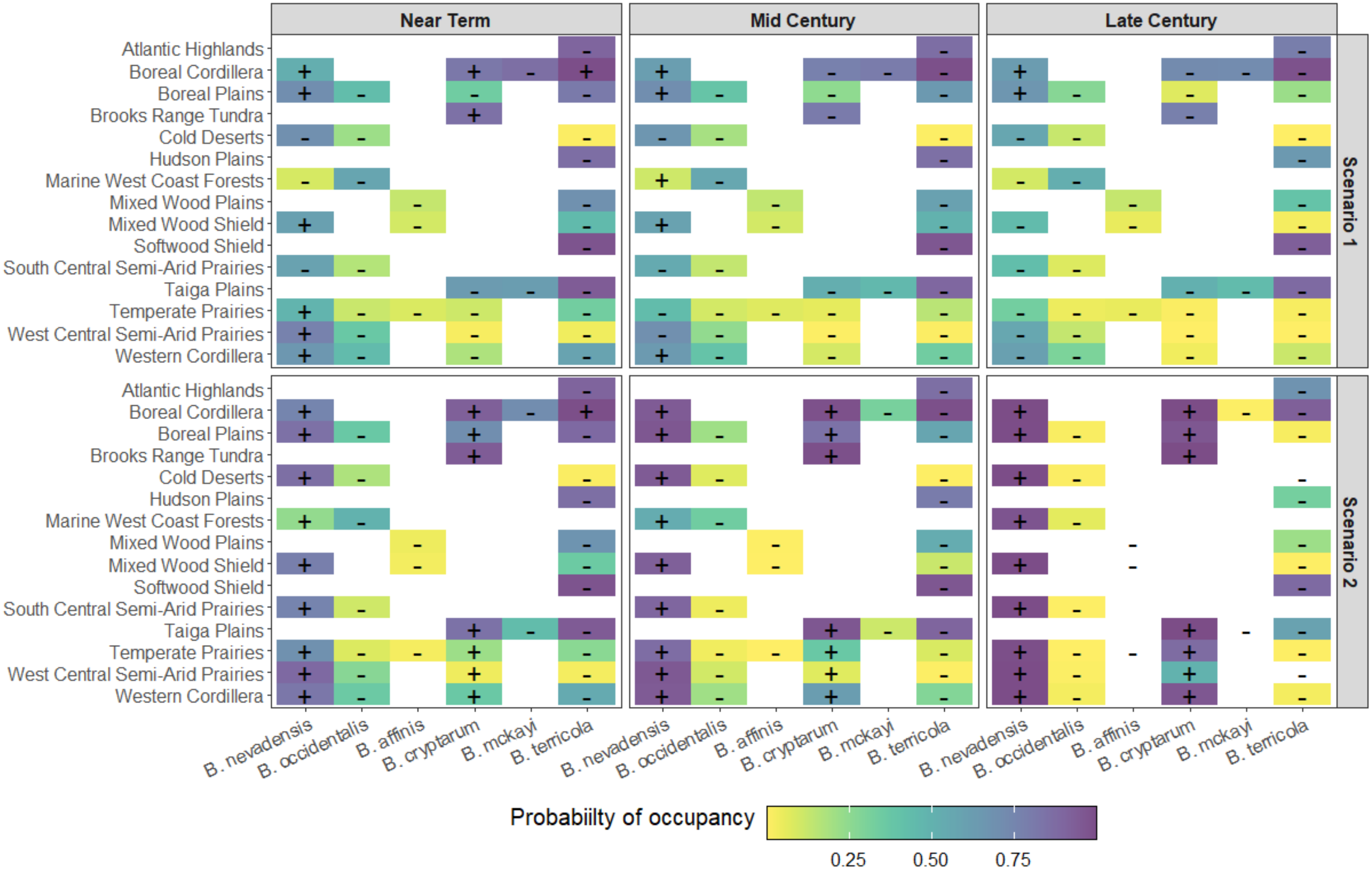


Figure 15. *Estimated median probability of occupancy and change from current conditions for Suckley's cuckoo bumble bee hosts. Occupancy depicted in the near term (2020–2040), mid-century (2040–2060), and late century (2080–2100) under Scenario 1 (top) and Scenario 2 (bottom). Probability of occupancy at each time step is depicted by color, and the trend relative to current conditions is shown as increasing (+) or decreasing (-). Species for which occupancy is projected to decline to less than 0.001 are marked in white with a (-) symbol.*

8.4 Summary of Future Condition

Resiliency

Both future condition scenarios project a continued, range wide decline of Suckley's cuckoo bumble bee. By the 2020–2040 period, median probaillity of occupancy is expected to be less than 0.11 across analytical units under Scenario 1, and less than 0.05 across all analytical units under Scenario 2. These scenarios both represent potentially optimistic estimates, as they do not account for the presence of host species. Our results collectivity suggest that Suckley's cuckoo bumble bee may already be quasi-extirpated in the Atlantic Highlands, Hudson Plains, Marine West Coast Forests, Mixed Wood Plains, and Mixed Wood Shield, and may also approach quasi-extirpation in the near-term in the South Central Semi-Arid Prairie analytical unit.

Median probability of occupancy in all analytical units is projected to be less than 0.13 by mid-century in Scenario 1, which represents a best-case scenario with optimistic estimates. In Scenario 2, median occupancy probability of all analytical units is projected to be less than 0.05 by mid-century and zero by late-century. While Scenario 2 is a high threat plausible future, the stressors driving the trend in historical Suckley's declines could intensify in the future, resulting in an even worse outcome for the species. As noted earlier, we are less confident that historical trends have matched overall range declines in several analytical units due to low data availability, particularly those with modern occurrence records. In specific, we are less certain about projecting historical declines due to non-climatic factors into the future under Scenario 2 in the Brooks Range Tundra, or South Central Semi-Arid Prairie, and the Taiga Plains. This uncertainty is inconsequential in the South Central Semi-Arid Prairie, where both future scenarios project extremely low probability of occupancy, due to high projected temperatures along the species' southern range boundary.

These projections indicate that there will be a large decline in occupancy, with potentially few to no Suckley's cuckoo bumble bees on the landscape by 2100. Even if the species persists in some small pockets of suitable habitat, these remnant populations are likely to be small and isolated. There are no indicators the species will have healthy demographics (survival and abundance) in the future compared to the historical condition, and the already less resilient current condition. Without healthy demographics, the species will not be able to withstand stochastic events. Resiliency in the future is expected to be poor at best and non-existent at worst.

Redundancy

Redundancy buffers the species against catastrophic events and can be summarized based on the spatial distribution of resilient populations relative to catastrophic events. Currently we estimate three of four eastern analytical units and two of 11 western analytical units are quasi-extirpated, with another likely approaching quasi-extirpation in the west in the near-future. Under both future scenarios we project continued range contraction, albeit much more extreme in Scenario 2. In Scenario 1, probability of occupancy is expected to decline for all analytical units through late-century. We project probability of occupancy at less than 0.10 across the entire range, except for the northern portion in the Boral Cordillera (0.11) and Brooks Range Tundra (0.122). In Scenario 2, the species is expected to be quasi-extirpated across the entire range. While there may be isolated, fragmented regions where the species persists in the future, they likely will not provide redundancy in the face of catastrophic events such as pathogen outbreak or wildfire. Additionally, we project a loss in host redundancy as at least four of six hosts are expected to decline in occupancy into the future.

Representation

Under both future scenarios Suckley's cuckoo bumble bee is expected to have lower representation and adaptive capacity in the future compared to current conditions. This is notable given the species has already experienced a statistically significant decline in probability of occupancy from historical to current condition. The fewer Suckley's cuckoo bumble bees on the landscape, the lower the likelihood of population connectivity; population connectivity is important for genetic diversity and successful reproduction of bumble bees. Under both future scenarios, occupancy probability is expected to be low (less than 0.15) for all analytical units. Low abundance and connectivity will likely make it difficult for bees to find unrelated mates.

Preserving the breadth of phenotypic diversity of Suckley's cuckoo bumble bee requires maintaining populations across a breadth of ecological conditions. Projections from mid-century to late-century for both scenarios demonstrate a declining trend in occupancy, with lowest occupancy probability and earliest quasi-extirpation in the southern portion of the range (i.e., the contiguous United States). Suckley's cuckoo bumble bees in this region are likely more adapted to warm temperatures and their extirpation will reduce phenotypic diversity.

Moreover, representation and adaptive capacity of Suckley's cuckoo bumble bee is also dependent on host species, as the distribution of the parasitic bee is restricted by the geographic distribution and population health of host bees, and parasite abundance is low where host abundance is low (Antonovics & Edwards, 2011, p. 1003). Therefore, availability of host species may also restrict the adaptive capacity of Suckley's cuckoo bumble bee, given that under both future scenarios four of the six confirmed and preferred host species are in decline. Ideal habitat requisites for the parasite may differ or be more specific than that of the host (Antonovics & Edwards, 2011, p. 1003), so future changes in habitat due to climate change could further restrict areas of ide
dspial parasite habitat which also supports hosts. Lastly, there is potential for phenological mismatch between host and parasite emergence with changing climate (Bower et al., 2023, p. 520).

Suckley's cuckoo bumble bees may be limited in their ability to respond to changes in climate by shifting in space. There is evidence suggesting that while bumble bee species have some capacity to shift in elevation to track changing climatic conditions, most have a limited capacity to shift in space latitudinally. For instance, using a database of approximately 423,000 georeferenced observations of 67 European and North American bumble bee species from 1975 to 2010, researchers found little evidence of northward range expansion, despite substantial concurrent warming, and evidence of rapid and marked range losses along southern range limits, independent of patterns of human land-use and development (Kerr et al., 2015, p. 178). The same study found that elevational shifts occurred, but varied by species, and latitude (Kerr et al., 2015, p. 179). Our understanding of these patterns for Suckley's cukoo bumble bee and other bumble bees in North America are limited by a lack of data in the far northern regions of the continent.

Suckley's cuckoo bumble bee also potentially limited in their ability to respond to changes in climate by adjusting their upper thermal limits. As noted earlier, most terrestrial species are thought to have strong physiological constraints on the evolution of greater heat tolerance (Araújo et al., 2013, entire). Insects have an especially limited capacity to change their thermal limits (Hoffmann et al., 2013, pp. 938–934; Weaving et al., 2022, p. 7), including bumble bees (Pimsler et al., 2020, p. 1). Furthermore, the ability of Suckley's cuckoo bumble bee to adaptively respond to climate change through evolution or behavioral plasticity will likely be limited in the future due to declines in occupancy, and concurrent declines in connectivity, population sizes, representation across ecological conditions, and genetic diversity.

9 Viability Synthesis

While there is a paucity of information available about Suckley's cuckoo bumble bee biology, both occurrence records and occupancy analysis point to widespread declines.

Suckley's cuckoo bumble bee was historically widespread across western North America and somewhat prevalent in the east, with some records in Canada. We have evidence the species is currently in a low resiliency state throughout much of its range because 1) there are no records of the bee in the continental United States since 2016, despite historical persistence, and 2) the species has exhibited an average 85 percent decline in probability of occupancy range wide from historical to current. This historical decline was evident in all analytical units for which there was enough data to assess local temporal declines in occupancy (79 percent of the occurrence polygon).

However, for several analytical units (21 percent of the occurrence polygon), there were too few occurrence records to estimate a local historical trend; consequently, current estimates in those analytical units are driven in part by range wide temporal declines that may or may not be representative. These historical declines are plausible in the in the Hudson Plains, Atlantic Highlands, and Mixed Wood Shield, where the species has not been detected since 1949, 1924, and 1995, respectively, but may or may not be representative in the Brooks Range Tundra, Mixed Wood Shield, and South Central Semi-Arid Prairie, where more recent detections have occurred, or the Taiga Plains where the species has not been detected since 1969, but survey effort is low. Consequently, current occupancy in the Brooks Range Tundra, Mixed Wood Shield, Taiga Plains, and South Central Semi-Arid Prairie could be biased low, if Suckley's cuckoo bumble bee has not been exposed to a similar degree of non-climatic threats in these portions of the range as it has elsewhere. Similary, future projections in these analytical units could be biased low in Scenario 2, as the rangewide historical temporal trends are projected into the future under this scenario.

Acknowledging these uncertainties, our results suggest that the species has likely lost redundancy because three of four eastern analytical units and two of eleven western analytical units are currently in a quasi-extirpated state. The Marine West Coast Forest is a particularly concerning loss because Suckley's cuckoo bumble bee was historically widespread and well-documented in this analytical unit. Representation has also declined as a result of this range contraction and occupancy decline, because the breadth of phenotypic, genetic, and ecological diversity of Suckley's cuckoo bumble bee is lower than historically.

The observed historical declines in occupancy are expected to continue in the near-term, and consequently, six analytical units are expected to be quasi-extirpated in the next two decades. The near-term decline in prevalence can have important consequences for the extinction risk of Suckley's cuckoo bumble bee. First, as usurped colonies become less common across the landscape, Suckley's cuckoo bumble bees will likely have a harder time finding unrelated mates. This decline in demographic connectivity can increase the risk of "matched mating", which produces diploid males that do not contribute to population growth (Zayed, 2009, p. 239). Population fragmentation, genetic drift, and inbreeding may all become more likely in the near term as the species becomes even less prevalent. These processes can be particularly consequential for bumble bees due to their low effective population size and their haplodiploid sex determination (Goulson et al., 2008, p. 14). Furthermore, our results suggest that the species will continue to lose representation across longitudinal, and ecological gradients in the near-term. This is important as this variation is the raw material for adaptive evolution, and behavioral plasticity (Hendry et al., 2011, p. 161). Finally, it is important to note that the viability of Suckley's cuckoo bumble bee is also highly dependent on its host species, many of which have declined historically, and are expected to continue to do so in the near term.

Under both future scenarios we projected substantial subsequent declines in probability of occupancy for Suckley's cuckoo bumble bee. Scenario 2 suggests range wide quasi-extirpation by 2060-2080. Though results are less certain in the Brooks Range Tundra, Mixed Wood Shield, Taiga Plains, and South Central Semi-Arid Prairie under Scenario 2, all these analytical units also are projected to have low probability of occupancy under Scenario 1, which does not project historical declines from non-climatic threats into the future. Thus, in the future we expect the species to have very low resiliency, redundancy, and representation. It is likely that any persisting bees will occur in isolated populations, which can pose a threat to gene flow and genetic diversity.

As noted earlier in the SSA, there are several sources of uncertainty, and model assumptions that must be considered carefully when interpreting these results. First, the occupancy modeling may fail to capture some features that Suckley's cuckoo bumble bee and its hosts respond to at a smaller spatial scale, due to the coarse spatial resolution of the climate data. Reflecting this, current occupancy may be underestimated by coarse-resolution climate models, especially in regions with high topographic variation. Second, though some locations on the landscape may be temporarily buffered from climate change due to local topographic or geological conditions, the coarse scale climate data used in the analysis does not represent these potential micro-refugia, and the models may consequently overestimate decline in occupancy and range contraction in the near-term for both Suckley's cuckoo bumble bee, and its hosts in topographically complex areas. However, long-term microrefugia are expected to be rare due to the sustained trajectory of increasing temperatures (Hannah et al., 2014, pp. 392–393, 395). Third, Suckley's cuckoo bumble bee is an obligate social parasite that is entirely dependent on host species. The occupancy model does not necessitate host presence as a parameter for parasite presence, and therefore may overestimate occupancy of Suckley's cuckoo bumble bee. Four of six host species are experiencing declines that are projected to continue into the future. There is potential for host/parasite spatial and/or phenological mismatch with changes to the climate; host and parasite emergence and/or preferred habitat could be decoupled in a catastrophic way to the parasite.

10 References

10.1 Personal Communications

Expert Team Meetings

August 2023 Expert Team Meeting. 2023. Personal communications (virtual meeting) between Sabrina Farmer (U.S. Fish and Wildlife Service) and Sam Droege (U.S. Geological Survey), Rich Hatfield (Xerces Society), John Mola (Colorado State University), Leif Richardson (Xerces Society), Hillary Sardinas (California Department of Fish and Wildlife), Derek Sikes (University of Alaska Fairbanks), James Strange (Ohio State University), Kimiora Ward (California Department of Fish and Wildlife), and Hollis Woodard (University of California Riverside) on August 30, 2023, regarding the species range.

September 2023 Expert Team Meeting. 2023. Personal communications (virtual meeting) between Sabrina Farmer (U.S. Fish and Wildlife Service) and Lincoln Best (Oregon State University), Syd Cannings (Canadian Wildlife Service), Michael Dillon (University of Wyoming), Rich Hatfield (Xerces Society), Leif Richardson (Xerces Society), Jessica Rykken (University of Alaska Anchorage), Hillary Sardinas (California Department of Fish and Wildlife), Cory Sheffield (Royal Saskatchewan Museum), Derek Sikes (University of Alaska Fairbanks), James Strange (Ohio State University), and Kimiora Ward (California Department of Fish and Wildlife), on September 27, 2023, regarding the species needs.

November 2023 Expert Team Meeting. 2023. Personal communications (virtual meeting) between Sabrina Farmer (U.S. Fish and Wildlife Service) and Lincoln Best (Oregon State University), Syd Cannings (Canadian Wildlife Service), Rich Hatfield (Xerces Society), Jonathan Koch (U.S. Department of Agriculture), Leif Richardson (Xerces Society), Jessica Rykken (University of Alaska Fairbanks), Hillary Sardinas (California Department of Fish and Wildlife), Cory Sheffield (Royal Saskatchewan Museum), Derek Sikes (University of Alaska Fairbanks), James Strange (Ohio State University), and Kimiora Ward (California Department of Fish and Wildlife) on November 1, 2023, regarding threats to the species.

Other Communications

Cannings, S., J. Heron, and C. Sheffield. 2023. Personal communication (virtual meeting) between Sabrina Farmer (U.S. Fish and Wildlife Service), Syd Cannings (Canadian Wildlife Service), Jennifer Heron (British Columbia Ministry of Water Land and Resource Stewardship), and Cory Sheffield (Royal Saskatchewan Museum) on September 12, 2023, regarding the range of Suckley's cuckoo bumble bee in Canada.

Connors, M.J. 2016. Personal communications (comment through Federal eRulemaking Portal) between U.S. Fish and Wildlife Service and Michael Connor on May 16, 2016, regarding the positive 90-day finding on the petition to list the Western bumble bee.

Everett, J. 2023. Personal communication (email) between Kaitlyn Howell (U.S. Fish and Wildlife Service) and Jeff Everett (U.S. Fish and Wildlife Service) on December 11, 2023, regarding conservation actions for Suckley's cuckoo bumble bee.

Heron, J. and C. Sheffield. 2023. Personal communication (email) between Sabrina Farmer (U.S. Fish and Wildlife Service), Jennifer Heron (British Columbia Ministry of Water Land and Resource Stewardship), and Cory Sheffield (Royal Saskatchewan Museum) on December 6, 2023, regarding historical records of Suckley's cuckoo bumble bee in New York.

Sikes, D. and J. Rykken. 2023. Personal communication (email) between Sabrina Farmer (U.S. Fish and Wildlife Service), Derek Sikes (University of Alaska Fairbanks), and Jessica Rykken (University of Alaska Fairbanks) on December 19, 2023, regarding the range of Suckley's cuckoo bumble bee in Alaska.

Swartz, B. 2023. Personal communication (email) between Sabrina Farmer (U.S. Fish and Wildlife Service) and Beth Swartz (Maine Department of Inland Fisheries and Wildlife) on August 8, 2023, regarding historical records of Suckley's cuckoo bumble bee in Maine.

10.2 Literature Cited

Abu-Asab, M. S., Peterson, P. M., Shetler, S. G., & Orli, S. S. (2001). Earlier plant flowering in spring as a response to global warming in the Washington, DC, area. *Biodiversity and Conservation*, *10*(4), 597–612. https://doi.org/10.1023/a:1016667125469

Aldridge, G., Inouye, D. W., Forrest, J. R. K., Barr, W. A., & Miller-Rushing, A. J. (2011). Emergence of a mid-season period of low floral resources in a montane meadow ecosystem associated with climate change. *Journal of Ecology*, *99*(4), 905–913. https://doi.org/10.1111/j.1365-2745.2011.01826.x

Alford, A., & Krupke, C. H. (2017). Translocation of the neonicotinoid seed treatment clothianidin in maize. *PLOS ONE*, *12*(3), e0173836. https://doi.org/10.1371/journal.pone.0173836

Alford, D. V. (1969). A Study of the Hibernation of Bumblebees (Hymenoptera:Bombidae) in Southern England. *The Journal of Animal Ecology*, *38*(1), 149. https://doi.org/10.2307/2743

Alger, S. A., Burnham, P. A., Boncristiani, H. F., & Brody, A. K. (2019). RNA virus spillover from managed honeybees (Apis mellifera) to wild bumblebees (Bombus spp.). *PLOS ONE*, *14*(6), e0217822. https://doi.org/10.1371/journal.pone.0217822

Amiri, E., Kryger, P., Meixner, M. D., Strand, M. K., Tarpy, D. R., & Rueppell, O. (2018). Quantitative patterns of vertical transmission of deformed wing virus in honey bees. *PLOS ONE*, *13*(3), e0195283. https://doi.org/10.1371/journal.pone.0195283

Antonovics, J., & Edwards, M. (2011). Spatio-temporal dynamics of bumblebee nest parasites (Bombus subgenus Psythirus ssp.) and their hosts (Bombus spp.). *Journal of Animal Ecology*, *80*(5), Article 5. https://doi.org/10.1111/j.1365-2656.2011.01846.x

Araújo, M. B., Ferri-Yáñez, F., Bozinovic, F., Marquet, P. A., Valladares, F., & Chown, S. L. (2013). Heat freezes niche evolution. *Ecology Letters*, *16*(9), 1206–1219. https://doi.org/10.1111/ele.12155

Arena, M., & Sgolastra, F. (2014). A meta-analysis comparing the sensitivity of bees to pesticides. *Ecotoxicology*, *23*, 324–334. https://doi.org/DOI 10.1007/s10646-014-1190-1

Baron, G. L., Raine, N. E., & Brown, M. J. F. (2017). General and species-specific impacts of a neonicotinoid insecticide on the ovary development and feeding of wild bumblebee queens. *Proceedings of the Royal Society B: Biological Sciences*, *284*(1854), 20170123. https://doi.org/10.1098/rspb.2017.0123

Barroso-Arévalo, S., Fernández-Carrión, E., Goyache, J., Molero, F., Puerta, F., & Sánchez-Vizcaíno, J. M. (2019). High Load of Deformed Wing Virus and Varroa destructor Infestation Are Related to Weakness of Honey Bee Colonies in Southern Spain. *Frontiers in Microbiology*, *10*, 1331. https://doi.org/10.3389/fmicb.2019.01331

Bartomeus, I., Ascher, J. S., Wagner, D., Danforth, B. N., Colla, S., Kornbluth, S., & Winfree, R. (2011). Climate-associated phenological advances in bee pollinators and bee-pollinated plants. *Proceedings of the National Academy of Sciences*, *108*(51), 20645–20649. https://doi.org/10.1073/pnas.1115559108

Beekman, M., & van Stratum, P. (1998). Bumblebee sex ratios: Why do bumblebees produce so many males? *Proceedings of the Royal Society of London. Series B: Biological Sciences*, *265*(1405), 1535–1543. https://doi.org/10.1098/rspb.1998.0469

Beekman, M., van Stratum, P., & Lingeman, R. (1998). Diapause survival and post-diapause performance in bumblebee queens (Bombus terrestris). *Entomologia Experimentalis et Applicata*, *89*(3), 207–214. https://doi.org/10.1046/j.1570-7458.1998.00401.x

Bishop, J. A., & Armbruster, W. S. (1999). Thermoregulatory abilities of Alaskan bees: Effects of size, phylogeny and ecology. *Functional Ecology*, *13*(5), 711–724. https://doi.org/10.1046/j.1365-2435.1999.00351.x

Bloom, T. D. S., O'Leary, D. S., & Riginos, C. (2022). Flowering time advances since the 1970s in a sagebrush steppe community: Implications for management and restoration. *Ecological Applications*. https://doi.org/10.1002/eap.2583

Bond, J., Plattner, K., & Hunt, K. (2014). *Fruit and Tree Nuts Outlook: Economic Insight* (Situation and Outlook FTS-357SA; p. 6). U.S. Department of Agriculture Economic Research Service.

Bower, R., Bulling, M., & Norton, B. (2023). Concern for cuckoo bumblebees (Bombus subgenus Psithyrus): Addressing our lack of knowledge. *Journal of Insect Conservation*, *27*(3), Article 3. https://doi.org/10.1007/s10841-023-00462-2

Brown, M. J. F., Loosli, R., & Schmid-Hempel, P. (2000). Condition-dependent expression of virulence in a trypanosome infecting bumblebees. *Oikos*, *91*(3), 421–427. https://doi.org/10.1034/j.1600-0706.2000.910302.x

Brown, M. J. F., & Paxton, R. J. (2009). The conservation of bees: A global perspective. *Apidologie*, *40*(3), 410–416. https://doi.org/10.1051/apido/2009019

Brown, M. J. F., Schmid-Hempel, R., & Schmid-Hempel, P. (2003). Strong context-dependent virulence in a host-parasite system: Reconciling genetic evidence with theory. *Journal of Animal Ecology*, *72*(6), 994–1002. https://doi.org/10.1046/j.1365-2656.2003.00770.x

Bruninga-Socolar, B., Griffin, S. R., Portman, Z. M., & Gibbs, J. (2022). Variation in prescribedfire and bison grazing supportsmultiple bee nesting groups in tallgrass prairie. *Restoration Ecology*, *30*(3), 1–10. https://doi.org/10.1111/rec.13507

Brutscher, L. M., McMenamin, A. J., & Flenniken, M. L. (2016). The Buzz about Honey Bee Viruses. *PLOS Pathogens*, *12*(8), e1005757. https://doi.org/10.1371/journal.ppat.1005757

Buszewski, B., Bukowska, M., Ligor, M., & Staneczko-Baranowska, I. (2019). A holistic study of neonicotinoids neuroactive insecticides—Properties, applications, occurrence, and analysis. *Environmental Science and Pollution Research*, *26*(34), 34723–34740. https://doi.org/10.1007/s11356-019-06114-w

California Department of Food and Agriculture. (2023). *Pollinator Protection*. California Department of Food and Agriculture. https://www.cdfa.ca.gov/plant/pollinators/

California Department of Pesticide Regulation. (2023). *Reevaluations—Neonicotinoids*. California Department of Pesticide Regulation. https://www.cdpr.ca.gov/docs/registration/reevaluation/chemicals/neonicotinoids.htm

California Natural Diversity Database (CNDDB). 2023. State and Federally Listed Endangered and Threatened Animals of California. California Department of Fish and Wildlife. Sacramento, CA. 39 pp.

Cameron, S. A., Hines, H. M., & Williams, P. H. (2007). A comprehensive phylogeny of the bumble bees (Bombus). *Biological Journal of the Linnean Society*, *91*(1), 161–188. https://doi.org/10.1111/j.1095-8312.2007.00784.x

Cameron, S. A., Jepson, S., Spevak, E., Strange, J., Vaughan, M., Engler, J., & Byers, O. (2011). *North American Blumble Bee Species Conservation Workshop Final Report*. IUCN/SSC Conservation Breeding Specialist Group.

Cameron, S. A., Lozier, J. D., Strange, J. P., Koch, J. B., Cordes, N., Solter, L. F., & Griswold, T. L. (2011). Patterns of widespread decline in North American bumble bees. *Proceedings of the National Academy of Sciences*, *108*(2), 662–667. https://doi.org/10.1073/pnas.1014743108

Camp, A. A., & Lehmann, D. M. (2021). Impacts of Neonicotinoids on the Bumble Bees Bombus terrestris and Bombus impatiens Examined through the Lens of an Adverse Outcome Pathway Framework. *Environmental Toxicology and Chemistry*, *40*(2), 309–322. https://doi.org/10.1002/etc.4939

CaraDonna, P. J., & Bain, J. A. (2015). Frost sensitivity of leaves and flowers of subalpine plants is related to tissue type and phenology. *Journal of Ecology*, *104*(1), 55–64. https://doi.org/10.1111/1365-2745.12482

CaraDonna, P. J., Iler, A. M., & Inouye, D. W. (2014). Shifts in flowering phenology reshape a subalpine plant community. *Proceedings of the National Academy of Sciences*, *111*(13), 4916–4921. https://doi.org/10.1073/pnas.1323073111

Carbone, L. M., Tavella, J., Pausas, J. G., & Aguilar, R. (2019). A global synthesis of fire effects on pollinators. *Global Ecology and Biogeography*, *28*(10), 1487–1498. https://doi.org/10.1111/geb.12939

Carroll, A. B., Pallardy, S. G., & Galen, C. (2001). Drought stress, plant water status, and floral trait expression in fireweed (Epilobium angustifolium) (Onagraceae). *American Journal of Botany*, *88*, 438–446.

Carvell, C. (2002). Habitat use and conservation of bumblebees (Bombus spp.) under different grassland management regimes. *Biological Conservation*, *103*(1), 33–49. https://doi.org/10.1016/s0006-3207(01)00114-8

Chen, Y. P., & Siede, R. (2007). Honey Bee Viruses. In *Advances in Virus Research* (Vol. 70, pp. 33–80). Elsevier. https://doi.org/10.1016/S0065-3527(07)70002-7

Colgan, T. J., Carolan, J. C., Sumner, S., Blaxter, M. L., & Brown, M. J. F. (2020). Infection by the castrating parasitic nematode *Sphaerularia bombi* changes gene expression in *Bombus terrestris* bumblebee queens. *Insect Molecular Biology*, *29*(2), 170–182. https://doi.org/10.1111/imb.12618

Colla, S. R. (2016). Status, Threats and Conservation Recommendations for Wild Bumble Bees (Bombusspp.) in Ontario, Canada: A Review for Policymakers and Practitioners. *Natural Areas Journal*, *36*(4), 412–426. https://doi.org/10.3375/043.036.0408

Colla, S. R., & Nalepa, R. (2023). *Conserving Canada's Wild Pollinators: National Strategy Recommendations*. 38 pp. https://www.savethebumblebees.ca/plan/

Colla, S. R., Otterstatter, M. C., Gegear, R. J., & Thomson, J. D. (2006). Plight of the bumble bee: Pathogen spillover from commercial to wild populations. *Biological Conservation*, *129*(4), 461–467. https://doi.org/10.1016/j.biocon.2005.11.013

Colla, S. R., & Packer, L. (2008). Evidence for decline in eastern North American bumblebees (Hymenoptera: Apidae), with special focus on Bombus affinis Cresson. *Biodiversity and Conservation*, *17*(6), 1379–1391. https://doi.org/10.1007/s10531-008-9340-5

Colla, S. R., Richardson, L., & Williams, P. (2011). *Bumblebees of the eastern United States*. USDA Forest Service and the Pollinator Partnership.

Colorado Parks and Wildlife. (2015). *State Wildlife Action Plan*. Colorado Parks and Wildlife. 865 pp. https://cpw.state.co.us/aboutus/Pages/StateWildlifeActionPlan.aspx

Commission for Environmental Cooperation. (1997). *Ecological regions of North America: Toward a common perspective*. Commission for Environmental Cooperation.

Cordes, N., Huang, W.-F., Strange, J. P., Cameron, S. A., Griswold, T. L., Lozier, J. D., & Solter, L. F. (2012). Interspecific geographic distribution and variation of the pathogens Nosema bombi and Crithidia species in United States bumble bee populations. *Journal of Invertebrate Pathology*, *109*(2), 209–216. https://doi.org/10.1016/j.jip.2011.11.005

COSEWIC. (2019). *COSEWIC assessment and status report on the Suckley's Cuckoo Bumble Bee Bombus suckleyi in Canada.* (p. 70). Committee on the Status of Endangered Wildlife in Canada.

Crone, E. E., & Williams, N. M. (2016). Bumble bee colony dynamics: Quantifying the importance of land use and floral resources for colony growth and queen production. *Ecology Letters*, *19*, 460–468.

Darvill, B., Ellis, J. S., Lye, G. C., & Goulson, D. (2006). Population structure and inbreeding in a rare and declining bumblebee, Bombus muscorum (Hymenoptera: Apidae): INBREEDING IN A RARE BUMBLEBEE. *Molecular Ecology*, *15*(3), 601–611. https://doi.org/10.1111/j.1365-294X.2006.02797.x

Defenders of Wildlife. (2015). *A petition to list the Western bumble bee (Bombus occidentalis) as an endangered, or alternatively as a threatened, species pursuant to the Endangered Species Act and for the designation of critical habiat for this species.*

Devarajan, K., Morelli, T. L., & Tenan, S. (2020). Multi-species occupancy models: Review, roadmap, and recommendations. *Ecography*, *43*(11), 1612–1624.

DiBartolomeis, M., Kegley, S., Mineau, P., Radford, R., & Klein, K. (2019). An assessment of acute insecticide toxicity loading (AITL) of chemical pesticides used on agricultural land in the United States. *PLOS ONE*, *14*(8), e0220029. https://doi.org/10.1371/journal.pone.0220029

Dozier, A. E., Straw, E. A., & Stanley, D. A. (2023). Systematic review of cuckoo bumblebee research reveals data gaps and understudied species. *Ecological Entomology*, *48*(6), 636–649. https://doi.org/10.1111/een.13260

Duchenne, F., Thébault, E., Michez, D., Gérard, M., Devaux, C., Rasmont, P., Vereecken, N. J., & Fontaine, C. (2020). Long-term effects of global change on occupancy and flight period of wild bees in Belgium. *Global Change Biology*, *26*(12), 6753–6766. https://doi.org/10.1111/gcb.15379

Dwire, K. A., Kauffman, J. B., & Mcintosh, B. A. (1999). Ecological Influences of the introduction of livestock on Pacific Northwest ecosystems. In *Northwest Lands, Northwest Peoples. Readings in Environmental History* (pp. 313–355). University of Washington Press.

Elliott, S. E. (2009). Subalpine Bumble Bee Foraging Distances and Densities in Relation to Flower Availability. *Environmental Entomology*, *38*(3), 748–756. https://doi.org/10.1603/022.038.0327

Ellis, J. S., Knight, M. E., Darvill, B., & Goulson, D. (2006). Extremely low effective population sizes, genetic structuring and reduced genetic diversity in a threatened bumblebee species, Bombus sylvarum (Hymenoptera: Apidae): GENETIC STRUCTURING IN DECLINING BUMBLEBEES. *Molecular Ecology*, *15*(14), 4375–4386. https://doi.org/10.1111/j.1365-294X.2006.03121.x

Elston, C., Thompson, H., & Walters, K. (2013). Sub-lethal effects of thiamethoxam, a neonicotinoid pesticide, and propiconazole, a DMI fungicide, on colony initiation in bumblebee (Bombus terrestris) micro-colonies. *Apidologie*, *44*, 563–574. http://dx.doi.org/10.1007/s13592-013-0206-9

Erler, S., & Lattorff, H. M. G. (2010). The degree of parasitism of the bumblebee (Bombus terrestris) by cuckoo bumblebees (Bombus (Psithyrus) vestalis). *Insectes Sociaux*, *57*(4), 371–377. https://doi.org/10.1007/s00040-010-0093-2

Evans, E., Thorp, R., Jepsen, S., & Black, S. H. (2008). *Status review of three formerly common species of bumble bee in the subgenus Bombus [Report]*. Xerces Society. https://xerces.org/publications/scientific-reports/status-review-of-three-formerly-common-species-of-bumble-bee

Fauser-Misslin, A., Sadd, B., Neumann, P., & Sandrock, C. (2014). Influence of combined pesticide and parasite exposure on bumblebee colony traits in the laboratory. *Journal of Applied Ecology*, *51*, 450–459.

Feltham, H., Park, K., & Goulson, D. (2014). Field realistic doses of pesticide imidacloprid reduce bumblebee pollen foraging efficiency. *Ecotoxicology*, *23*(3), 317–323. https://doi.org/10.1007/s10646-014-1189-7

Feuerborn, C., Quinlan, G., Shippee, R., Strausser, T. L., Terranova, T., Grozinger, C. M., & Hines, H. M. (2023). Variance in heat tolerance in bumble bees correlates with species geographic range and is associated with several environmental and biological factors. *Ecology and Evolution*, *13*(11), e10730. https://doi.org/10.1002/ece3.10730

Fine, J. D., Cox-Foster, D. L., & Mullin, C. A. (2017). An Inert Pesticide Adjuvant Synergizes Viral Pathogenicity and Mortality in Honey Bee Larvae. *Scientific Reports*, *7*(1). https://doi.org/10.1038/srep40499

Fisher, R. M. (1985). Evolution and host specificity: Dichotomous invasion success of Psithyrus citrinus (Hymenoptera: Apidae), a bumblebee social parasite in colonies of its two hosts. *Canadian Journal of Zoology*, *63*(4), Article 4. https://doi.org/10.1139/z85-144

Fisogni, A., Bogo, G., Massol, F., Bortolotti, L., & Galloni, M. (2021). Cuckoo male bumblebees perform slower and longer flower visits than free-living male and worker bumblebees. *Belgian Journal of Zoology*, *151*. https://doi.org/10.26496/bjz.2021.93

Forrest, J., Inouye, D. W., & Thomson, J. D. (2010). Flowering phenology in subalpine meadows: Does climate variation influence community co-flowering patterns? *Ecology*, *91*(2), 431–440. https://doi.org/10.1890/09-0099.1

Fourcade, Y., Åström, S., & Öckinger, E. (2018). Climate and land-cover change alter bumblebee species richness and community composition in subalpine areas. *Biodiversity and Conservation*, *28*(3), 639–653. https://doi.org/10.1007/s10531-018-1680-1

Fulkerson, J. R., Carlson, M. L., & Burns, C. T. (2023). *Alaska Bee Atlas: Bee Inventory and Monitoring Plan for Alaska* Alaska Center for Conservation Science, University of Alaska Anchorage. 33 pp.

Fürst, M. A., McMahon, D. P., Osborne, J. L., Paxton, R. J., & Brown, M. J. F. (2014). Disease associations between honeybees and bumblebees as a threat to wild pollinators. *Nature*, *506*(7488), 364–366. https://doi.org/10.1038/nature12977

Galbraith, S. M., Cane, J. H., Moldenke, A. R., & Rivers, J. W. (2019). Wild bee diversity increases with local fire severity in a fire-prone landscape. *Ecosphere*, *10*(4), e02668. https://doi.org/10.1002/ecs2.2668

Gallot-Lavallée, M., Schmid-Hempel, R., Vandame, R., Vergara, C. H., & Schmid-Hempel, P. (2016). Large scale patterns of abundance and distribution of parasites in Mexican bumblebees. *Journal of Invertebrate Pathology*, *133*, 73–82. https://doi.org/10.1016/j.jip.2015.12.004

Gao, P., Terando, A. J., Kupfer, J. A., Varner, J. M., Stambaugh, M. C., Lei, T. L., & Hiers, J. K. (2021). Robust projections of future fire probability for the conterminous United States. *Science of The Total Environment*, *789*, 1–13. https://doi.org/10.1016/j.scitotenv2021.147872

Gaston, K. J., & Lawton, J. H. (1988). Patterns in the distribution and abundance of insect populations. *Nature*, *331*(6158), 709–712. https://doi.org/10.1038/331709a0

Gels, J. A., Held, D. W., & Potter, D. A. (2002). Hazards of Insecticides to the Bumble Bees Bombus impatiens (Hymenoptera: Apidae) Foraging on Flowering White Clover in Turf. *Journal of Economic Entomology*, *95*(4), 722–728. https://doi.org/10.1603/0022-0493-95.4.722

Genersch, E., Yue, C., Fries, I., & de Miranda, J. R. (2006). Detection of Deformed wing virus, a honey bee viral pathogen, in bumble bees (Bombus terrestris and Bombus pascuorum) with wing deformities. *Journal of Invertebrate Pathology*, *91*(1), 61–63. https://doi.org/10.1016/j.jip.2005.10.002

Gérard, M., Cariou, B., Henrion, M., Descamps, C., & Baird, E. (2022). Exposure to elevated temperature during development affects bumblebee foraging behavior. *Behavioral Ecology*, *33*(4), 816–824. https://doi.org/10.1093/beheco/arac045

Giles, V., & Ascher, J. S. (2006). A survey of the bees of the Black Rock Forest preserve, New York (Hymenoptera: Apoidea). *Journal of Hymenoptera Research*, *15*(2), 208–231.

Gill, R. J., & Raine, N. E. (2014). Chronic impairment of bumblebee natural foraging behaviour induced by sublethal pesticide exposure. *Functional Ecology*, *28*(6), 1459–1471. https://doi.org/10.1111/1365-2435.12292

Gill, R. J., Ramos-Rodriguez, O., & Raine, N. E. (2012). Combined pesticide exposure severely affects individual- and colony-level traits in bees. *Nature*, *491*(7422), 105–108. https://doi.org/10.1038/nature11585

Gillespie, S. (2010). Factors affecting parasite prevalence among wild bumblebees. *Ecological Entomology*, *35*(6), 737–747. https://doi.org/10.1111/j.1365-2311.2010.01234.x

Goka, K., Okabe, K., & Yoneda, M. (2006). Worldwide migration of parasitic mites as a result of bumblebee commercialization: Special feature: population ecology of biological invasion. *Population Ecology*, *48*(4), 285–291. https://doi.org/10.1007/s10144-006-0010-8

Goller, F., & Esch, H. (1990). Comparative Study of Chill-Coma Temperatures and Muscle Potentials in Insect Flight Muscles. *Journal of Experimental Biology*, *150*(1), 221–231. https://doi.org/10.1242/jeb.150.1.221

Goodwin, R., & McBrydie, H. (2000). EFFECT OF SURFACTANTS ON HONEY BEE SURVIVAL. *New Zealand Plant Protection*, *53*, 230–234.

Goulson, D. (2003). Conserving wild bees for crop pollination. *Food, Agriculture, and Environment*, *1*, 142–144.

Goulson, D. (2010a). *Bumblebees: Behaviour, ecology, and conservation* (2nd ed.). Oxford University Press.

Goulson, D. (2010b). Impacts of non-native bumblebees in Western Europe and North America. *Applied Entomology and Zoology*, *45*(1), 7–12. https://doi.org/10.1303/aez.2010.7

Goulson, D. (2013). An overview of the environmental risks posed by neonicotinoid insecticides. *Journal of Applied Ecology*, *50*, 977–987.

Goulson, D., & Hughes, W. O. H. (2015). Mitigating the anthropogenic spread of bee parasites to protect wild pollinators. *Biological Conservation*, *191*, 10–19. https://doi.org/10.1016/j.biocon.2015.06.023

Goulson, D., Lye, G. C., & Darvill, B. (2008). Decline and Conservation of Bumble Bees. *Annual Review of Entomology*, *53*(1), 191–208. https://doi.org/10.1146/annurev.ento.53.103106.093454

Goulson, D., Nicholls, E., Botias, C., & Rotheray, E. L. (2015). Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. *Science*, *347*(6229), 1255957–1255957. https://doi.org/10.1126/science.1255957

Grabensteiner, E., Ritter, W., Carter, M. J., Davison, S., Pechhacker, H., Kolodziejek, J., Boecking, O., Derakhshifar, I., Moosbeckhofer, R., Licek, E., & Nowotny, N. (2001). Sacbrood Virus of the Honeybee (Apis mellifera): Rapid Identification and Phylogenetic Analysis Using Reverse Transcription-PCR. *Clinical Diagnostic Laboratory Immunology*, *8*(1), 93–104. https://doi.org/10.1128/cdli.8.1.93-104.2001

Graham, K. K., Gibbs, J., Wilson, J., May, E., & Isaacs, R. (2021). Resampling of wild bees across fifteen years reveals variable species declines and recoveries after extreme weather. *Agriculture, Ecosystems & Environment*, *317*, 107470. https://doi.org/10.1016/j.agee.2021.107470

Graystock, P., Blane, E. J., McFrederick, Q. S., Goulson, D., & Hughes, W. O. H. (2016). Do managed bees drive parasite spread and emergence in wild bees? *International Journal for Parasitology: Parasites and Wildlife*, *5*(1), 64–75. https://doi.org/10.1016/j.ijppaw.2015.10.001

Graystock, P., Goulson, D., & Hughes, W. O. H. (2014). The relationship between managed bees and the prevalence of parasites in bumblebees. *PeerJ*, *2*, e522. https://doi.org/10.7717/peerj.522

Graystock, P., Goulson, D., & Hughes, W. O. H. (2015). Parasites in bloom: Flowers aid dispersal and transmission of pollinator parasites within and between bee species. *Proceedings of the Royal Society B: Biological Sciences*, *282*(1813), 20151371. https://doi.org/10.1098/rspb.2015.1371

Graystock, P., Yates, K., Darvill, B., Goulson, D., & Hughes, W. O. H. (2013). Emerging dangers: Deadly effects of an emergent parasite in a new pollinator host. *Journal of Invertebrate Pathology*, *114*(2), 114–119. https://doi.org/10.1016/j.jip.2013.06.005

Graystock, P., Yates, K., Evison, S. E. F., Darvill, B., Goulson, D., & Hughes, W. O. H. (2013). The Trojan hives: Pollinator pathogens, imported and distributed in bumblebee colonies. *Journal of Applied Ecology*, *50*(5), 1207–1215. https://doi.org/10.1111/1365-2664.12134

Greene, J. W. (1860). Review of the American Bombidae, together with a Description of several Species heretofe undescribed, being a Synopsis of the species of this family of Hymenopterous Insects thus far known to inhabit North America. *Annals of The Lyceum of Natural History of New York*, *7*(1), 168–176.

Grixti, J. C., Wong, L. T., Cameron, S. A., & Favret, C. (2009). Decline of bumble bees (Bombus) in the North American Midwest. *Biological Conservation*, *142*(1), 75–84. https://doi.org/10.1016/j.biocon.2008.09.027

Guzman, L. M., Johnson, S. A., Mooers, A. O., & M'Gonigle, L. K. (2021). Using historical data to estimate bumble bee occurrence: Variable trends across species provide little support for community-level declines. *Biological Conservation*, *257*, 109141. https://doi.org/10.1016/j.biocon.2021.109141

Hamblin, A. L., Youngsteadt, E., López-Uribe, M. M., & Frank, S. D. (2017). Physiological thermal limits predict differential responses of bees to urban heat-island effects. *Biology Letters*, *13*(6), 20170125. https://doi.org/10.1098/rsbl.2017.0125

Hannah, L., Flint, L., Syphard, A. D., Moritz, M. A., Buckley, L. B., & McCullough, I. M. (2014). Fine-grain modeling of species' response to climate change: Holdouts, stepping-stones, and microrefugia. *Trends in Ecology & Evolution*, *29*(7), 390–397. https://doi.org/10.1016/j.tree.2014.04.006

Hanson, R. L. (1991). *Evapotranspiration and Droughts.* (Water Supply Paper, pp. 99–104). U.S. Geological Survey Water-Supply Paper 2375. https://doi.org/10.3133/wsp2375

Hatfield, R., Colla, S., Jepsen, S., Richardson, L., Thorp, R., & Foltz Jordan, S. (2014). *IUCN Assessments for North American Bombus spp.* (p. 56). IUCN. https://xerces.org/sites/default/files/publications/14-065.pdf

Hatfield, R., Jepsen, S., Thorp, R., Richardson, L., & Colla, S. (2015). *Bombus suckleyi: The IUCN Red List of Threatened Species 2015* (p. 12). IUCN. https://doi.org/10.2305/IUCN.UK.2015-2.RLTS.T44937699A46440241.en

Hatfield, R., & LeBuhn, G. (2007). Patch and landscape factors shape community assemblage of bumble bees, Bombus spp. (Hymenoptera: Apidae), in montane meadows. *Biological Conservation*, *139*(1–2), 150–158. https://doi.org/10.1016/j.biocon.2007.06.019

Hayhoe, K., Wuebbles, D. J., Easterling, D. R., Fahey, D. W., Doherty, S., Kossin, J. P., Sweet, W. V., Vose, R. S., & Wehner, M. F. (2018). *Chapter 2: Our Changing Climate. Impacts, Risks, and Adaptation in the United States: The Fourth National Climate Assessment, Volume II*. U.S. Global Change Research Program. https://doi.org/10.7930/NCA4.2018.CH2

Heinrich, B. (1975). Thermoregulation in bumblebees: II. Energetics of warm-up and free flight. *Journal of Comparative Physiology*, *96*, 155–166.

Heinrich, B. (1979). *Bumblebee economics*. Harvard University Press.

Heinrich, B., & Vogt, F. D. (1993). Abdominal temperature regulation by arctic bumblebees. *Physiological Zoology*, *66*(2), 257–269.

Hendry, A. P., Kinnison, M. T., Heino, M., Day, T., Smith, T. B., Fitt, G., Bergstrom, C. T., Oakeshott, J., Jørgensen, P. S., Zalucki, M. P., Gilchrist, G., Southerton, S., Sih, A., Strauss, S., Denison, R. F., & Carroll, S. P. (2011). Evolutionary principles and their practical application. *Evolutionary Applications*, *4*(2), 159–183. https://doi.org/10.1111/j.1752-4571.2010.00165.x

Hicke, J. A., Lucatello, S., Mortsch, Li. A., Dawson, J., Aguilar, M. D., Enquist, C. A. F., Gimore, E. A., Gutzler, D. S., Harper, S., Holsman, K., Jewett, E. B., Kohler, T. A., & Miller, K. A. (2022). North America. In *Climate Change 2022: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 1929–2042). Cambridge University Press. https://doi.org/10.1017/9781009325844.016.

Higginson, A. D. (2017). Conflict over non-partitioned resources may explain between-species differences in declines: The anthropogenic competition hypothesis. *Behavioral Ecology and Sociobiology*, *71*(7). https://doi.org/10.1007/s00265-017-2327-z

Hines, H. M., & Hendrix, S. D. (2005). Bumble Bee (Hymenoptera: Apidae) Diversity and Abundance in Tallgrass Prairie Patches: Effects of Local and Landscape Floral Resources. *Environmental Entomology*, *34*(6), 1477–1484. https://doi.org/10.1603/0046-225x-34.6.1477

Hobbs, G. A. (1965). Ecology of Species of Bombus Latr. (Hymenoptera: Apidae) in Southern Alberta. II. Subgenus Bombias Robt. 1. *The Canadian Entomologist*, *97*(2), 120–128.

Hobbs, G. A. (1967). Ecology of Species of *Bombus* (Hymenoptera: Apidae) in Southern Alberta. VI. Subgenus Pyrobombus. *The Canadian Entomologist*, *99*(12), 1271–1292.

Hobbs, G. A. (1968). Ecology of species of *Bombus* (Hymenoptera: Apidae) in Southern Alberta: VII. Subgenus *Bombus*. *The Canadian Entomologist*, *100*(2), 156–164. https://doi.org/10.4039/Ent100156-2

Hodek, I., & Hodkova, M. (1988). Multiple role of temperature during insect diapause: A review. *Entomologia Experimentalis et Applicata*, *49*(1–2), 153–165. https://doi.org/doi.org/10.1111/j.1570-7458.1988.tb02486.x

Hoffmann, A. A., Chown, S. L., & Clusella-Trullas, S. (2013). Upper thermal limits in terrestrial ectotherms: How constrained are they? *Functional Ecology*, *27*(4), 934–949. https://doi.org/10.1111/j.1365-2435.2012.02036.x

Husband, R. W., Fischer, R. L., & Porter, W. (1980). Description and biology of bumblebees (Hymenoptera: Apidae) in Michigan. *The Great Lakes Entomologist*, *13*(4), 225–239.

Husband, R. W., & Sinha, R. N. (1970). A Revision of the Genus *Locustacarus* with a Key to Genera of the Family Podapolipidae (Acarina). *Annals of the Entomological Society of America*, *63*(4), 1152–1162. https://doi.org/10.1093/aesa/63.4.1152

Idaho Department of Fish and Game. (2017). *Idaho State Wildlife Action Plan*. Idaho Department of Fish and Game. http://fishandgame.idaho.gov/

Inouye, D. W. (2008). Effects of climate change on phenology, frost damage, and floral abundance of montane wildflowers. *Ecology*, *89*(2), 353–362. https://doi.org/10.1890/06-2128.1

IPCC. (2014). *Climate Change 2014: Synthesis Report Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovenmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)].* (p. 151). IPCC.

ISSSSP. (2021). *Region 6 Regional Forester and OR/WA State Director Special Status Species List*. The Interagency Special Status / Sensitive Species Program. https://www.fs.usda.gov/r6/issssp/policy/

Jackson, H. M., Johnson, S. A., Morandin, L. A., Richardson, L. L., Guzman, L. M., & M'Gonigle, L. K. (2022). Climate change winners and losers among North American bumblebees. *Biology Letters*, *18*(6), Article 6. https://doi.org/10.1098/rsbl.2021.0551

Janousek, W. M., Douglas, M. R., Cannings, S., Clément, M. A., Delphia, C. M., Everett, J. G., Hatfield, R. G., Keinath, D. A., Koch, J. B. U., McCabe, L. M., Mola, J. M., Ogilvie, J. E., Rangwala, I., Richardson, L. L., Rohde, A. T., Strange, J. P., Tronstad, L. M., & Graves, T. A. (2023). Recent and future declines of a historically widespread pollinator linked to climate, land cover, and pesticides. *Proceedings of the National Academy of Sciences*, *120*(5), Article 5. https://doi.org/10.1073/pnas.2211223120

Jha, S., & Kremen, C. (2013). Urban land use limits regional bumble bee gene flow. *Molecular Ecology*, *22*(9), 2483–2495. https://doi.org/10.1111/mec.12275

Johansen, C. (1977). Pesticides and Pollinators. *Annual Review of Entomology*, *22*, 177–192.

Johansen, C., & Mayer, D. F. (1990). *Pollinator protection: A bee & pesticide handbook*. Cheshire, Conn., USA : Wicwas Press, 1990.

Johnson, M. D., & Horn, C. M. (2008). Effects of Rotational Grazing on Rodents and Raptors in a Coastal Grassland. *Western North American Naturalist*, *68*(4), 444–452. https://doi.org/10.3398/1527-0904-68.4.444

Kammer, A. E., & Heinrich, B. (1974). Metabolic rates related to muscle activity in bumblebees. *The Journal of Experimental Biology*, *61*(1), 219–227.

Karger, D. N., Schmatz, D. R., Dettling, G., & Zimmermann, N. E. (2020). High-resolution monthly precipitation and temperature time series from 2006 to 2100. *Scientific Data*, *7*(248), Article 1. https://doi.org/10.1038/s41597-020-00587-y

Kearns, C. A., & Inouye, D. W. (1997). Pollinators, Flowering Plants, and Conservation Biology. *BioScience*, *47*(5), 297–307. https://doi.org/10.2307/1313191

Kearns, C. A., Inouye, D. W., & Waser, N. M. (1998). Endangered mutualisms: The Conservation of Plant-Pollinator Interactions. *Annual Review of Ecology and Systematics*, *29*(1), 83–112. https://doi.org/10.1146/annurev.ecolsys.29.1.83

Keaveny, E. C., & Dillon, M. E. (2022). Phat Queens Emerge Fashionably Late: Body Size and Condition Predict Timing of Spring Emergence for Queen Bumble Bees. *Insects*, *13*(10), 870. https://doi.org/10.3390/insects13100870

Kenna, D., Pawar, S., & Gill, R. J. (2021). Thermal flight performance reveals impact of warming on bumblebee foraging potential. *Functional Ecology*, 1365-2435.13887. https://doi.org/10.1111/1365-2435.13887

Kerr, J. T., Pindar, A., Galpern, P., Packer, L., Potts, S. G., Roberts, S. M., Rasmont, P., Schweiger, O., Colla, S. R., Richardson, L. L., Wagner, D. L., Gall, L. F., Sikes, D. S., & Pantoja, A. (2015). Climate change impacts on bumblebees converge across continents. *Science*, *349*(6244), 177–180. https://doi.org/10.1126/science.aaa7031

Kery, M., Royle, J. A., Schmid, H., Schaub, M., Volet, B., Häfliger, G., & Zbinden, N. (2010). Site-occupancy distribution modeling to correct population-trend estimates derived from opportunistic observations. *Conservation Biology*, *24*(5), 1388–1397.

Kimoto, C., DeBano, S. J., Thorp, R. W., Rao, S., & Stephen, W. P. (2012). Investigating Temporal Patterns of a Native Bee Community in a Remnant North American Bunchgrass Prairie using Blue Vane Traps. *Journal of Insect Science*, *12*(108), 1–23. https://doi.org/10.1673/031.012.10801

Kimoto, C., DeBano, S. J., Thorp, R. W., Taylor, R. V., Schmalz, H., DelCurto, T., Johnson, T., Kennedy, P. L., & Rao, S. (2012). Short-term responses of native bees to livestock and implications for managing ecosystem services in grasslands. *Ecosphere*, *3*(10), art88. https://doi.org/10.1890/es12-00118.1

Kissinger, C. N., Cameron, S. A., Thorp, R. W., White, B., & Solter, L. F. (2011). Survey of bumble bee (Bombus) pathogens and parasites in Illinois and selected areas of northern California and southern Oregon. *Journal of Invertebrate Pathology*, *107*(3), 220–224. https://doi.org/10.1016/j.jip.2011.04.008

Koch, J. B., McCabe, L. M., Love, B. G., & Cox-Foster, D. (2021). Genetic and Usurpation Data Support High Incidence of Bumble Bee Nest Invasion by Socially Parasitic Bumble Bee, Bombus insularis. *Journal of Insect Science*, *21*(5), Article 5. https://doi.org/10.1093/jisesa/ieab063

Koch, J. B., Strange, J., & Williams, P. (2012). *Bumble bees of the western United States*. USDA Forest Service and the Pollinator Partnership. 144 pp. https://www.xerces.org/sites/default/files/2018-05/12-053_01_Western_BB_guide.pdf

Kriesell, L., Hilpert, A., & Leonhardt, S. D. (2017). Different but the same: Bumblebee species collect pollen of different plant sources but similar amino acid profiles. *Apidologie*, *48*(1), 102–116. https://doi.org/10.1007/s13592-016-0454-6

Lämsä, J., Kuusela, E., Tuomi, J., Juntunen, S., & Watts, P. C. (2018). Low dose of neonicotinoid insecticide reduces foraging motivation of bumblebees. *Proceedings of the Royal Society B: Biological Sciences*, *285*(1883), 20180506. https://doi.org/10.1098/rspb.2018.0506

Lankau, R., Jørgensen, P. S., Harris, D. J., & Sih, A. (2011). Incorporating evolutionary principles into environmental management and policy. *Evolutionary Applications*, *4*(2), 315–325. https://doi.org/10.1111/j.1752-4571.2010.00171.x

Larrere, M., Lavenseau, L., Tasei, J. N., & Couillaud, F. (1993). Juvenile hormone biosynthesis and diapause termination in Bombus terrestris. *Invertebrate Reproduction & Development*, *23*(1), 7–14.

Laycock, I., & Cresswell, J. E. (2013). Repression and Recuperation of Brood Production in Bombus terrestris Bumble Bees Exposed to a Pulse of the Neonicotinoid Pesticide Imidacloprid. *PLoS ONE*, *8*(11), e79872. https://doi.org/10.1371/journal.pone.0079872

Laycock, I., Lenthall, K. M., Barratt, A. T., & Cresswell, J. E. (2012). Effects of imidacloprid, a neonicotinoid pesticide, on reproduction in worker bumble bees (Bombus terrestris). *Ecotoxicology*, *21*(7), 1937–1945. https://doi.org/10.1007/s10646-012-0927-y

Lepais, O., Darvill, B., O'Connor, S., Osborne, J. L., Sanderson, R. A., Cussans, J., Goffe, L., & Goulson, D. (2010). Estimation of bumblebee queen dispersal distances using sibship reconstruction method: ESTIMATION OF BUMBLEBEE QUEEN DISPERSAL. *Molecular Ecology*, *19*(4), 819–831. https://doi.org/10.1111/j.1365-294X.2009.04500.x

Lhomme, P., & Hines, H. M. (2019). Ecology and Evolution of Cuckoo Bumble Bees. *Annals of the Entomological Society of America*, *112*(3), Article 3. https://doi.org/10.1093/aesa/say031

Liczner, A., & Colla, S. R. (2019). A systematic review of the nesting and overwintering habitat of bumble bees globally. *Journal of Insect Conservation*, *23*(5–6), 787–801.

Louvrier, J., Chambert, T., Marboutin, E., & Gimenez, O. (2018). Accounting for misidentification and heterogeneity in occupancy studies using hidden Markov models. *Ecological Modelling*, *387*, 61–69. https://doi.org/10.1016/j.ecolmodel.2018.09.002

Lozier, J. D., Strange, J. P., & Heraghty, S. D. (2023). Whole genome demographic models indicate divergent effective population size histories shape contemporary genetic diversity gradients in a montane bumble bee. *Ecology and Evolution*, 1–13. https://doi.org/10.1002/ece3.9778

Lozier, J. D., Strange, J. P., Stewart, I. J., & Cameron, S. A. (2011). Patterns of range-wide genetic variation in six North American bumble bee (Apidae: Bombus) species: POPULATION GENETICS OF NORTH AMERICAN BOMBUS. *Molecular Ecology*, *20*(23), 4870–4888. https://doi.org/10.1111/j.1365-294X.2011.05314.x

Lundberg, H. (1980). Effects of weather on foraging-flights of bumblebees (Hymenoptera, Apidae) in a subalpine/alpine area. *Ecography*, *3*(2), 104–110. https://doi.org/10.1111/j.1600-0587.1980.tb00715.x

Lundin, O., Rundlöf, M., Smith, H. G., Fries, I., & Bommarco, R. (2015). Neonicotinoid Insecticides and Their Impacts on Bees: A Systematic Review of Research Approaches and Identification of Knowledge Gaps. *PLOS ONE*, *10*(8), e0136928. https://doi.org/10.1371/journal.pone.0136928

Macfarlane, R. P., Lipa, J. J., & Liu, H. J. (1995). Bumble Bee Pathogens and Internal Enemies. *Bee World*, *76*(3), 130–148. https://doi.org/10.1080/0005772x.1995.11099259

Macfarlane, R. P., Patten, K. D., Royce, L. A., Wyatt, B. K. W., & Mayer, D. F. (1994). Management potential of sixteen North American bumble bee species. *Melanderia*, *50*(1), 1–12.

Maebe, K., Hart, A. F., Marshall, L., Vandamme, P., Vereecken, N. J., Michez, D., & Smagghe, G. (2021). Bumblebee resilience to climate change, through plastic and adaptive responses. *Global Change Biology*, *27*(18), 4223–4237. https://doi.org/10.1111/gcb.15751

Mahony, C. R., Wang, T., Hamann, A., & Cannon, A. J. (2022). A global climate model ensemble for downscaled monthly climate normals over North America. *International Journal of Climatology*, *42*(11), 5871–5891.

Malfi, R. L. (2023, July 27). *States Make Way For Pesticide Reforms*. The Xerces Society for Invertebrate Conservation. https://www.xerces.org/blog/states-make-way-for-pesticide-reforms

Malfi, R. L., & Roulston, T. H. (2014). Patterns of parasite infection in bumble bees (*Bombus* spp.) of Northern Virginia: Parasitism in bumble bees of Virginia. *Ecological Entomology*, *39*(1), 17–29. https://doi.org/10.1111/een.12069

Manley, R., Boots, M., & Wilfert, L. (2015). REVIEW: Emerging viral disease risk to pollinating insects: ecological, evolutionary and anthropogenic factors. *Journal of Applied Ecology*, *52*(2), 331–340. https://doi.org/10.1111/1365-2664.12385

Martin, M., Hatfield, R., May, E., Richardson, L., & Jepsen, S. (2023). *Strategy to Protect State and Federally Recognized Bumble Bee Species of Conservation Concern: Washington State*. The Xerces Society for Invertebrate Conservation; with the Interagency Special Status and Sensitive Species Program USDA Forest Service, REgion 6 and USDI Oregon/Washington Bureau of Land Management.

Martinet, B., Dellicour, S., Ghisbain, G., Przybyla, K., Zambra, E., Lecocq, T., Boustani, M., Baghirov, R., Michez, D., & Rasmont, P. (2021). Global effects of extreme temperatures on wild bumblebees. *Conservation Biology*, *35*(5), 1507–1518. https://doi.org/10.1111/cobi.13685

Martinet, B., Lecocq, T., Smet, J., & Rasmont, P. (2015). A Protocol to Assess Insect Resistance to Heat Waves, Applied to Bumblebees (Bombus Latreille, 1802). *PLOS ONE*, *10*(3), e0118591. https://doi.org/10.1371/journal.pone.0118591

Martinet, B., Zambra, E., Przybyla, K., Lecocq, T., Anselmo, A., Nonclercq, D., Rasmont, P., Michez, D., & Hennebert, E. (2020). Mating under climate change: Impact of simulated heatwaves on the reproduction of model pollinators. *Functional Ecology*, *35*(3), 739–752. https://doi.org/10.1111/1365-2435.13738

Maxfield-Taylor, S., Skyrm, K. M., & Rao, S. (2011). First record of Sphaerularia bombi (Nematoda: Tylenchida: Sphaerularidae), a parasite of bumble bee queens, in the Pacific Northwest. *The Pan-Pacific Entomologist*, *87*(2), 134–137. https://doi.org/10.3956/2011-01.1

McCorquodale, D. B., Beresford, R. G., Francis, J. M., Thomson, C. E., & Bartlett, C. M. (1998). PREVALENCE OF *SPHAERULARIA BOMBI* (NEMATODA: TYLENCHIDA: SPHAERULARIIDAE) IN BUMBLE BEE QUEENS (HYMENOPTERA: APIDAE) ON CAPE BRETON ISLAND, NOVA SCOTIA, CANADA. *The Canadian Entomologist*, *130*(6), 877–882. https://doi.org/10.4039/Ent130877-6

McGarigal, K., Wan, H. Y., Zeller, K. A., Timm, B. C., & Cushman, S. A. (2016). Multi-scale habitat selection modeling: A review and outlook. *Landscape Ecology*, *31*(6), 1161–1175. https://doi.org/10.1007/s10980-016-0374-x

McMahon, D. P., Fürst, M. A., Caspar, J., Theodorou, P., Brown, M. J. F., & Paxton, R. J. (2015). A sting in the spit: Widespread cross-infection of multiple RNA viruses across wild and managed bees. *Journal of Animal Ecology*, *84*(3), 615–624. https://doi.org/10.1111/1365-2656.12345

McNeil, D. J., McCormick, E., Heimann, A. C., Kammerer, M., Douglas, M. R., Goslee, S. C., Grozinger, C. M., & Hines, H. M. (2020). Bumble bees in landscapes with abundant floral resources have lower pathogen loads. *Scientific Reports*, *10*(1). https://doi.org/10.1038/s41598-020-78119-2

Meeus, I., Brown, M. J. F., De Graaf, D. C., & Smagghe, G. (2011). Effects of Invasive Parasites on Bumble Bee Declines: Parasites and Bumble Bee Declines. *Conservation Biology*, *25*(4), 662–671. https://doi.org/10.1111/j.1523-1739.2011.01707.x

Miller-Struttmann, N. E., Geib, J. C., Franklin, J. D., Kevan, P. G., Holdo, R. M., Ebert-May, D., Lynn, A. M., Kettenbach, J. A., Hedrick, E., & Galen, C. (2015). Functional mismatch in a bumble bee pollination mutualism under climate change. *Science*, *349*(6255), 1541–1544. https://doi.org/10.1126/science.aab0868

Minnameyer, A., Strobl, V., Bruckner, S., Camenzid, D., Oystaeyen, A. V., Wackers, F., Williams, G. R., Yañez, O., Neumann, P., & Straub, L. (2021). Eusocial insect declines: Insecticide impairs sperm and feeding glands in bumblebees. *Science of The Total Environment*, *785*(146955), 1–12. https://doi.org/10.1016/j.scitotenv.2021.146955

Mola, J. M., Miller, M. R., O'Rourke, S. M., & Williams, N. M. (2020). Wildfire reveals transient changes to individual traits and population responses of a native bumble bee Bombus vosnesenskii. *Journal of Animal Ecology*, *89*(8), 1799–1810. https://doi.org/10.1111/1365-2656.13244

Mola, J. M., Richardson, L. L., Spyreas, G., Zaya, D. N., & Pearse, I. S. (2021). Long-term surveys support declines in early season forest plants used by bumblebees. *Journal of Applied Ecology*, *58*(7), 1431–1441. https://doi.org/10.1111/1365-2664.13886

Mola, J. M., & Williams, N. M. (2018). Fire-induced change in floral abundance, density, and phenology benefits bumble bee foragers. *Ecosphere*, *9*(1). https://doi.org/10.1002/ecs2.2056

Mommaerts, V., Hagenaars, A., Meyer, J., De Coen, W., Swevers, L., Mosallanejad, H., & Smagghe, G. (2011). Impact of a perfluorinated organic compound PFOS on the terrestrial pollinator

Bombusterrestris (Insecta, Hymenoptera). *Ecotoxicology*, *20*(2), 447–456. https://doi.org/10.1007/s10646-011-0596-2

Mommaerts, V., Sterk, G., & Smagghe, G. (2006). Hazards and uptake of chitin synthesis inhibitors in bumblebees Bombus terrestris. *Pest Management Science*, *62*, 752–758. https://doi.org/10.1002/ps.1238

Montana Natural Heritage Program. (2023, July 3). *Suckley Cuckoo Bumble Bee—Montana Field Guide*. Natural Heritage Program and Montana Fish, Wildlife & Parks. https://fieldguide.mt.gov/speciesDetail.aspx?elcode=IIHYM24350

Morandin, L. A., Winston, M. L., Franklin, M. T., & Abbott, V. (2005). Lethal and sub-lethal effects of spinosad on bumble bees (Bombus impatiens Cresson). *Pest Management Science*, *61*, 619–626. https://doi.org/10.1002

Mullin, C. A., Chen, J., Fine, J. D., Frazier, M., & Frazier, J. (2015). The formulation makes the honey bee poison. *Pesticide Biochemistry and Physiology*, 1–27. http://dx.doi.org/doi: 10.1016/j.pestbp.2014.12.026

Murray, T. E., Coffey, M. F., Kehoe, E., & Horgan, F. G. (2013). Pathogen prevalence in commercially reared bumble bees and evidence of spillover in conspecific populations. *Biological Conservation*, *159*, 269–276. https://doi.org/10.1016/j.biocon.2012.10.021

Newsholme, E. A., Crabtree, B., Higgins, S. J., Thornton, S. D., & Start, C. (1972). The activities of fructose diphosphatase in flight muscles from the bumble-bee and role of this enzyme in heat generation. *Biochemical Journal*, *128*(1), 89–97. https://doi.org/10.1042/bj1280089

Öckinger, E., & Smith, H. G. (2006). Semi-natural grasslands as population sources for pollinating insects in agricultural landscapes: Population sources for pollinators. *Journal of Applied Ecology*, *44*(1), 50–59. https://doi.org/10.1111/j.1365-2664.2006.01250.x

Ogilvie, J. E., & CaraDonna, P. J. (2022). The shifting importance of abiotic and biotic factors across the life cycle of wild pollinators. *Journal of Animal Ecology*, *91*(12), 2412–2423.

Ogilvie, J. E., Griffin, S. R., Gezon, Z. J., Inouye, B. D., Underwood, N., Inouye, D. W., & Irwin, R. E. (2017). Interannual bumble bee abundance is driven by indirect climate effects on floral resource phenology. *Ecology Letters*, *20*(12), 1507–1515. https://doi.org/10.1111/ele.12854

Oregon Department of Agriculture. (2017). *Oregon Approved Invertebrate List*. https://www.oregon.gov/oda/shared/Documents/Publications/IPPM/OregonApprovedInvertebrateList.pdf

Oregon Department of Agriculture, Oregon State University Extension Service, & Oregon Department of Forestry. (2022). *The Oregon Bee Project Strategic Plan 2: A strategic plan for keeping Oregon's bee pollinators healthy (2022-2027)*. https://www.oregonbeeproject.org/strategic-plan

Otterstatter, M. C., Gegear, R. J., Colla, S. R., & Thomson, J. D. (2005). Effects of parasitic mites and protozoa on the flower constancy and foraging rate of bumble bees. *Behavioral Ecology and Sociobiology*, *58*(4), 383–389. https://doi.org/10.1007/s00265-005-0945-3

Otterstatter, M. C., & Thomson, J. D. (2008). Does Pathogen Spillover from Commercially Reared Bumble Bees Threaten Wild Pollinators? *PLoS ONE*, *3*(7), e2771. https://doi.org/10.1371/journal.pone.0002771

Otterstatter, M. C., & Whidden, T. L. (2004). Patterns of parasitism by tracheal mites (Locustacarus buchneri) in natural bumble bee populations. *Apidologie*, *35*(4), 351–357. https://doi.org/10.1051/apido:2004024

Otti, O., & Schmid-Hempel, P. (2007). Nosema bombi: A pollinator parasite with detrimental fitness effects. *Journal of Invertebrate Pathology*, *96*(2), 118–124. https://doi.org/10.1016/j.jip.2007.03.016

Oyen, K. J., & Dillon, M. E. (2018). Critical thermal limits of bumble bees (*Bombus impatiens*) are marked by stereotypical behaviors and are unchanged by acclimation, age, or feeding status. *Journal of Experimental Biology*, jeb.165589. https://doi.org/10.1242/jeb.165589

Oyen, K. J., Giri, S., & Dillon, M. E. (2016). Altitudinal variation in bumble bee (Bombus) critical thermal limits. *Journal of Thermal Biology*, *59*, 52–57. https://doi.org/10.1016/j.jtherbio.2016.04.015

Palmier, K. M., & Sheffield, C. S. (2019). First records of the Common Eastern Bumble Bee Bombus impatiens Cresson (Hymenoptera: Apidae, Apinae, Bombini) from the Prairies Ecosystem in Canada. *Biodiversity Data Journal*, *7*, 1–18. https://doi.org/10.3897/BDJ.7.e30953

Palmier, K. M., Tripodi, A. D., Cameron, A. D. S., Strange, J. P., & Sheffield, C. S. (2020). First record of *Crithidia expoeki* (Trypanosomatida: Trypanosomatidae) from native Canadian bumble bees (Hymenoptera: Apidae: *Bombus*). *The Canadian Field-Naturalist*, *134*(1), 16–20. https://doi.org/10.22621/cfn.v134i1.2299

Parks, S. A., Holsinger, L. M., Miller, C., & Parisien, M.-A. (2018). Analog-based fire regime and vegetation shifts in mountainous regions of the western US. *Ecography*, *41*, 910–921. https://doi.org/10.1111/ecog.03378

Parsche, S., & Lattorff, H. M. G. (2018). The relative contributions of host density and genetic diversity on prevalence of a multi-host parasite in bumblebees. *Biological Journal of the Linnean Society*, *125*(4), 900–910.

Pearman, P. B., Guisan, A., Broennimann, O., & Randin, C. F. (2008). Niche dynamics in space and time. *Trends in Ecology & Evolution*, *23*(3), 149–158. https://doi.org/10.1016/j.tree.2007.11.005

Peng, W., Li, J., Boncristiani, H., Strange, J. P., Hamilton, M., & Chen, Y. (2011). Host range expansion of honey bee Black Queen Cell Virus in the bumble bee, Bombus huntii. *Apidologie*, *42*(5), 650–658. https://doi.org/10.1007/s13592-011-0061-5

Phillips, B. B., Shaw, R. F., Holland, M. J., Fry, E. L., Bardgett, R. D., Bullock, J. M., & Osborne, J. L. (2018). Drought reduces floral resources for pollinators. *Global Change Biology*, *24*(7), 3226–3235. https://doi.org/10.1111/gcb.14130

Pimsler, M. L., Oyen, K. J., Herndon, J. D., Jackson, J. M., Strange, J. P., Dillon, M. E., & Lozier, J. D. (2020). Biogeographic parallels in thermal tolerance and gene expression variation under temperature stress in a widespread bumble bee. *Scientific Reports*, *10*(1). https://doi.org/10.1038/s41598-020-73391-8

Pisa, L. W., Amaral-Rogers, V., Belzunces, L. P., Bonmatin, J.-M., Downs, C. A., Goulson, D., Kreutzweiser, D. P., Krupke, C., Liess, M., McField, M., Morrissey, C. A., Noome, D., Settele, J., Simon-Delso, N., Stark, J. D., Van der Sluijs, J. P., Van Dyck, H., & Wiemers, M. (2015). Effects of neonicotinoids and fipronil on non-target invertebrates. *Environmental Science and Pollution Research*, *22*, 68–102. https://doi.org/10.1007/s11356-014-3471-x

Plath, O. E. (1927). Notes on the Nesting Habits of Some of the Less Common New England Bumble-bees. *Psyche: A Journal of Entomology*, *34*(2), 122–128. https://doi.org/10.1155/1927/43469

Plowright, R. C., Pendrel, B., & McLaren, I. (1978). THE IMPACT OF AERIAL FENITROTHION SPRAYING UPON THE POPULATION BIOLOGY OF BUMBLE BEES (BOMBUS LATR.: HYM.) IN SOUTH-WESTERN NEW BRUNSWICK. *The Canadian Entomologist*, *110*, 1145–1156.

Poinar, G. O. (1974). The presence of Sphaerularia bombi (Tylenchida: Nematoda) a nematode parasite of Bombus queens (Apidae: Hymenoptera), in California. *The Pan-Pacific Entomologist*, *50*(3), 304–305.

Poinar, G. O., & Van Der Laan, P. A. (1972). Morphology and Life History of Sphaerularia Bombi. *Nematologica*, *18*(2), 239–252. https://doi.org/10.1163/187529272X00476

Potter, K. A., Arthur Woods, H., & Pincebourde, S. (2013). Microclimatic challenges in global change biology. *Global Change Biology*, *19*(10), 2932–2939.

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: Trends, impacts and drivers. *Trends in Ecology & Evolution*, *25*(6), 345–353. https://doi.org/10.1016/j.tree.2010.01.007

Pradervand, J.-N., Pellissier, L., Randin, C. F., & Guisan, A. (2014). Functional homogenization of bumblebee communities in alpine landscapes under projected climate change. *Climate Change Responses*, *1*(1). https://doi.org/10.1186/s40665-014-0001-5

Pyke, G. H., Thomson, J. D., Inouye, D. W., & Miller, T. J. (2016). Effects of climate change on phenologies and distributions of bumble bees and the plants they visit. *Ecosphere*, *7*(3). https://doi.org/10.1002/ecs2.1267

Raine, N. (2018). An alternative to controversial pesticides still harms bumblebees. *Nature*, *561*, 40–41. https://doi.org/10.1038/d41586-018-05917-0

Rasmont, P., & Iserbyt, S. (2012). The Bumblebees Scarcity Syndrome: Are heat waves leading to local extinctions of bumblebees (Hymenoptera: Apidae:Bombus)? *Annales de La Société Entomologique de France (N.S.)*, *48*(3–4), 275–280. https://doi.org/10.1080/00379271.2012.10697776

Ratti, C. M., & Colla, S. R. (2010). Discussion of the presence of an eastern bumble bee species (Bombus impatiens Cresson) in western Canada. *The Pan-Pacific Entomologist*, *86*(2), 29–31. https://doi.org/10.3956/2009-19.1

Ratti, C. M., Higo, H. A., Griswold, T. L., & Winston, M. L. (2008). Bumble bees influence berry size in commercial Vaccinium spp. Cultivation in British Columbia. *The Canadian Entomologist*, *140*(3), 348–363. https://doi.org/10.4039/n07-065

Richards, K. W. (1978). Nest site selection by bumble bees (Hymenoptera: Apidae) in southern Alberta. *The Canadian Entomologist*, *110*(3), 301–318. https://doi.org/10.4039/ent110301-3

Robson-Hyska, C. (2017). *Detection of pathogen spillover between managed honey bees (Apis mellifera L.) and native pollinators (Bombus spp.) through quantification of RNA viruses*. http://hdl.handle.net/1993/32736

Rotheray, E. L., Osborne, J. L., & Goulson, D. (2017). Quantifying the food requirements and effects of food stress on bumble bee colony development. *Journal of Apicultural Research*, *56*(3), 288–299. https://doi.org/10.1080/00218839.2017.1307712

Roulston, T. H., & Goodell, K. (2011). The role of resources and risks in regulating wild bee populations. *Annual Review of Entomology*, *56*, 293–312. https://doi.org/10.1146/annurev-ento-120709-144802

Rutrecht, S. T., & Brown, M. J. F. (2008). Within colony dynamics of *Nosema bombi* infections: Disease establishment, epidemiology and potential vertical transmission. *Apidologie*, *39*(5), 504–514. https://doi.org/10.1051/apido:2008031

Rutrecht, S. T., Klee, J., & Brown, M. J. F. (2007). Horizontal transmission success of *Nosema bombi* to its adult bumble bee hosts: Effects of dosage, spore source and host age. *Parasitology*, *134*(12), 1719–1726. https://doi.org/10.1017/S0031182007003162

Sachman-Ruiz, B., Narváez-Padilla, V., & Reynaud, E. (2015). Commercial Bombus impatiens as reservoirs of emerging infectious diseases in central México. *Biological Invasions*, *17*(7), 2043–2053. https://doi.org/10.1007/s10530-015-0859-6

Sanderson, B. M., Knutti, R., & Caldwell, P. (2015). A representative democracy to reduce interdependency in a multimodel ensemble. *Journal of Climate*, *28*(13), 5171–5194.

Schmid-Hempel, P. (2001). On the evolutionary ecology of host-parasite interactions: Addressing the question with regard to bumblebees and their parasites. *Naturwissenschaften*, *88*(4), 147–158. https://doi.org/10.1007/s001140100222

Schmid-Hempel, R., Eckhardt, M., Goulson, D., Heinzmann, D., Lange, C., Plischuk, S., Escudero, L. R., Salathé, R., Scriven, J. J., & Schmid-Hempel, P. (2014). The invasion of southern South America by imported bumblebees and associated parasites. *Journal of Animal Ecology*, *83*(4), 823–837. https://doi.org/10.1111/1365-2656.12185

Schmid-Hempel, R., & Tognazzo, M. (2010). Molecular Divergence Defines Two Distinct Lineages of Crithidia bombi (Trypanosomatidae), Parasites of Bumblebees: CRITHIDIA BOMBI LINEAGES. *Journal of Eukaryotic Microbiology*, *57*(4), 337–345. https://doi.org/10.1111/j.1550-7408.2010.00480.x

Scholer, J., & Krischik, V. (2014). Chronic Exposure of Imidacloprid and Clothianidin Reduce Queen Survival, Foraging, and Nectar Storing in Colonies of Bombus impatiens. *PLoS ONE*, *9*(3), e91573. https://doi.org/10.1371/journal.pone.0091573

Schweitzer, D. F., Capuano, N. A., Young, B. E., & Colla, S. R. (2012). *Conservation and Management of North American Bumble Bees*. NatureServe, USDA Forest Service.

Scohier, A., Ouin, A., Farruggia, A., & Dumont, B. (2012). Is there a benefit of excluding sheep from pastures at flowering peak on flower-visiting insect diversity? *Journal of Insect Conservation*, *17*(2), 287–294. https://doi.org/10.1007/s10841-012-9509-9

Shaffer, M. L., & Stein, B. A. (2000). Safeguarding Our Precious Heritage. In J. S. Adams, B. A. Stein, & L. S. Kutner, *Precious Heritage: The Status of Biodiversity in the United States*. Oxford University Press. https://doi.org/10.1093/oso/9780195125191.003.0017

Sheffield, C. S., Pindar, A., Packer, L., & Kevan, P. G. (2013). The potential of cleptoparasitic bees as indicator taxa for assessing bee communities. *Apidologie*, *44*, 501–510.

Sheffield, C. S., Richardson, L., Cannings, S., Ngo, H., Heron, J., & Williams, P. H. (2016). Biogeography and designatable units of Bombus occidentalis Greene and B. terricola Kirby (Hymenoptera: Apidae) with implications for conservation status assessments. *Journal of Insect Conservation*, *20*(2), 189–199. https://doi.org/10.1007/s10841-016-9853-2

Shykoff, J. A., & Schmid-Hempel, P. (1991). Parasites delay worker reproduction in bumblebees: Consequences for eusociality. *Behavioral Ecology*, *2*(3), 242–248. https://doi.org/10.1093/beheco/2.3.242

Singh, R., Levitt, A. L., Rajotte, E. G., Holmes, E. C., Ostiguy, N., vanEngelsdorp, D., Lipkin, W. I., dePamphilis, C. W., Toth, A. L., & Cox-Foster, D. L. (2010). RNA Viruses in Hymenopteran Pollinators: Evidence of Inter-Taxa Virus Transmission via Pollen and Potential Impact on Non-Apis Hymenopteran Species. *PLOS ONE*, *5*(12), e14357. https://doi.org/10.1371/journal.pone.0014357

Sirois-Delisle, C., & Kerr, J. T. (2018). Climate change-driven range losses among bumblebee species are poised to accelerate. *Scientific Reports*, *8*(1), 14464. https://doi.org/10.1038/s41598-018-32665-y

Siviter, H., Koricheva, J., Brown, M. J. F., & Leadbeater, E. (2018). Quantifying the impact of pesticides on learning and memory in bees. *Journal of Applied Ecology*, *55*(6), 2812–2821. https://doi.org/10.1111/1365-2664.13193

Sjödin, N. E. (2007). Pollinator behavioural responses to grazing intensity. *Biodiversity and Conservation*, *16*(7), 2103–2121. https://doi.org/10.1007/s10531-006-9103-0

Sjödin, N. E., Bengtsson, J., & Ekbom, B. (2008). The influence of grazing intensity and landscape composition on the diversity and abundance of flower-visiting insects. *Journal of Applied Ecology*, *45*(3), 763–772. https://doi.org/10.1111/j.1365-2664.2007.01443.x

Sladen, F. W. L. (1912). *The humble-bee its life-history and how to domesticate it, with descriptions of all the British species of Bombus and Psithyrus*. MacMillan and Co., Limited.

Smallidge, P. J., & Leopold, D. J. (1997). Vegetation management for the maintenance and conservation of butterfly habitats in temperate human-dominated landscapes. *Landscape and Urban Planning*, *38*(3–4), 259–280. https://doi.org/10.1016/s0169-2046(97)00038-8

Smith, D. R., Allan, N. L., McGowan, C. P., Szymanski, J. A., Oetker, S. R., & Bell, H. M. (2018). Development of a Species Status Assessment Process for Decisions under the U.S. Endangered Species Act. *Journal of Fish and Wildlife Management*, *9*(1), 302–320. https://doi.org/10.3996/052017-jfwm-041

Somme, L., Vanderplanck, M., Michez, D., Lombaerde, I., Moerman, R., Wathelet, B., Wattiez, R., Lognay, G., & Jacquemart, A.-L. (2015). Pollen and nectar quality drive the major and minor floral choices of bumble bees. *Apidologie*, *46*(1), 92–106. https://doi.org/10.1007/s13592-014-0307-0

Soroye, P., Newbold, T., & Kerr, J. (2020). Climate change contributes to widespread declines among bumble bees across continents. *Science*, *367*(6478), 685–688. https://doi.org/10.1126/science.aax8591

Sponsler, D. B., Requier, F., Kallnik, K., Classen, A., Maihoff, F., Sieger, J., & Steffan-Dewenter, I. (2022). Contrasting patterns of richness, abundance, and turnover in mountain bumble bees and their floral hosts. *Ecology*, *103*(7), e3712. https://doi.org/10.1002/ecy.3712

Sramkova, A., & Ayasse, M. (2009). Chemical ecology involved in invasion success of the cuckoo bumblebee Psithyrus vestalis and in survival of workers of its host Bombus terrestris. *Chemoecology*, *19*(1), Article 1.

Stark, J. M., & Norton, J. M. (2015). The invasive annual cheatgrass increases nitrogen availability in 24-year-old replicated field plots. *Oecologia*, *177*(3), 799–809. https://doi.org/10.1007/s00442-014-3093-5

Staten, P. W., Lu, J., Grise, K. M., Davis, S. M., & Birner, T. (2018). Re-examining tropical expansion. *Nature Climate Change*, *8*(9), 768–775. https://doi.org/10.1038/s41558-018-0246-2

Strange, J. P., Baur, A., & Koch, J. B. (2014). A scientific note on Bombus (Psithyrus) insularis invasions of bumblebee nests and honeybee hives in the western USA. *Apidologie*, *45*(5), 554–556. https://doi.org/10.1007/s13592-014-0271-8

Suhonen, J., Rannikko, J., & Sorvari, J. (2015). The Rarity of Host Species Affects the Co-Extinction Risk in Socially Parasitic BumblebeeBombus(Psithyrus) Species. *Annales Zoologici Fennici*, *52*(4), 236–242. https://doi.org/10.5735/086.052.0402

Suhonen, J., Rannikko, J., & Sorvari, J. (2016). Species richness of cuckoo bumblebees is determined by the geographical range area of the host bumblebee. *Insect Conservation and Diversity*, *9*(6), 529–535.

Sunday, J. M., Bates, A. E., & Dulvy, N. K. (2012). Thermal tolerance and the global redistribution of animals. *Nature Climate Change*, *2*(9), 686–690. https://doi.org/10.1038/nclimate1539

Sutcliffe, G. H., & Plowright, R. C. (1990). The effects of pollen availability on development time in the bumble bee Bombus terricola K. (Hymenoptera: Apidae). *Canadian Journal of Zoology*, *68*(6), 1120–1123. https://doi.org/10.1139/z90-166

Swain, S., & Hayhoe, K. (2015). CMIP5 projected changes in spring and summer drought and wet conditions over North America. *Climate Dynamics*, *44*, 2737–2750. https://doi.org/doi:10.1007/s00382-014-2255-9

Tasei, J.-N., & Aupinel, P. (2008). Nutritive value of 15 single pollens and pollen mixes tested on larvae produced by bumblebee workers (Bombus terrestris, Hymenoptera: Apidae). *Apidologie*, *39*(4), 397–409. https://doi.org/10.1051/apido:2008017

Tehel, A., Streicher, T., Tragust, S., & Paxton, R. J. (2022). Experimental cross species transmission of a major viral pathogen in bees is predominantly from honeybees to bumblebees. *Proceedings of the Royal Society B: Biological Sciences*, *289*(1969), 20212255. https://doi.org/10.1098/rspb.2021.2255

The Trustees of the Natural History Museum. (n.d.). *Bombus*. Bumble Bees. Retrieved April 10, 2024, from https://www.nhm.ac.uk/research-curation/research/projects/bombus/ps.html

The Xerces Society, & Thorp, R. W. (2010). *Petition to List Franklin's Bumble Bee (Bombus franklini) as an Endangered Species under the U.S. Endangered Species Act*. https://www.xerces.org/publications/policy-statements/petition-to-list-franklins-bumble-bee-under-esa-2010

Thompson, H. M., & Hunt, L. V. (1999). Extrapolating from Honeybees to Bumblebees in Pesticide Risk Assessment. *Ecotoxicology*, *8*(3), 147–166.

Thomson, D. M. (2004). Competitive interactions between the invasive european honey bee and native bumble bees. *Ecology*, *85*(2), 458–470. https://doi.org/10.1890/02-0626

Thomson, D. M. (2016). Local bumble bee decline linked to recovery of honey bees, drought effects on floral resources. *Ecology Letters*, *19*(10), 1247–1255. https://doi.org/10.1111/ele.12659

Thomson, J. D. (2010). Flowering phenology, fruiting success and progressive deterioration of pollination in an early-flowering geophyte. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *365*(1555), 3187–3199. https://doi.org/10.1098/rstb.2010.0115

Thorp, R., Horning, D. S., & Dunning, L. (1983). Bumble Bees and Cuckoo Bumble Bees of California (Hymenoptera: Apidae). *Bulletin of the California Insect Survey*, *23*, 1–79.

Tokarev, Y. S., Huang, W.-F., Solter, L. F., Malysh, J. M., Becnel, J. J., & Vossbrinck, C. R. (2020). A formal redefinition of the genera *Nosema* and *Vairimorpha* (Microsporidia: Nosematidae) and reassignment of species based on molecular phylogenetics. *Journal of Invertebrate Pathology*, *169*, 107279. https://doi.org/10.1016/j.jip.2019.107279

Townsend-Mehler, J. M., & Dyer, F. C. (2011). An integrated look at decision-making in bees as they abandon a depleted food source. *Behavioral Ecology and Sociobiology*, *66*(2), 275–286. https://doi.org/10.1007/s00265-011-1275-2

Tripodi, A. D., Szalanski, A. L., & Strange, J. P. (2018). Novel multiplex PCR reveals multiple trypanosomatid species infecting North American bumble bees (Hymenoptera: Apidae: Bombus). *Journal of Invertebrate Pathology*, *153*, 147–155. https://doi.org/10.1016/j.jip.2018.03.009

U.S. Environmental Protection Agency. (2021a). *Climate Change Indicators in the United States. Heat Waves.* U.S. Environmental Protection Agency. https://www.epa.gov/climate-indicators/climate-change-indicators-heat-waves

U.S. Environmental Protection Agency. (2021b). *Climate Change Indicators in the United States. U.S. and Global Temperatures.* https://www.epa.gov/climate-indicators/climate-change-indicators-us-and-global-temperature

U.S. Fish and Wildlife Service. (2018). *Species Status Assessment for the Yellow Banded Bumble Bee (Bombus terricola), Version 1.0.* Hadley, Massachusetts. 146 pp.

U.S. Forest Service. (2019). *Colville National Forest Plan*. United States Department of Agriculture, Forest Service. 250 pp. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd673192.pdf

Vanderplanck, M., Martinet, B., Carvalheiro, L. G., Rasmont, P., Barraud, A., Renaudeau, C., & Michez, D. (2019). Ensuring access to high-quality resources reduces the impacts of heat stress on bees. *Scientific Reports*, *9*(1). https://doi.org/10.1038/s41598-019-49025-z

Vaudo, A. D., Tooker, J. F., Grozinger, C. M., & Patch, H. M. (2015). Bee nutrition and floral resource restoration. *Current Opinion in Insect Science*, *10*, 133–141. https://doi.org/10.1016/j.cois.2015.05.008

Vázquez, D. P., & Simberloff, D. (2003). Changes in interaction biodiversity induced by an introduced ungulate: Introduced species and interaction biodiversity. *Ecology Letters*, *6*(12), 1077–1083. https://doi.org/10.1046/j.1461-0248.2003.00534.x

Vesterlund, S.-R., Lilley, T. M., van Ooik, T., & Sorvari, J. (2014). The effect of overwintering temperature on the body energy reserves and phenoloxidase activity of bumblebee Bombus lucorum queens. *Insectes Sociaux*, *61*(3), 265–272. https://doi.org/10.1007/s00040-014-0351-9

Vogt, F. D. (1986a). Thermoregulation in Bumblebee Colonies. I. Thermoregulatory versus Brood-Maintenance Behaviors during Acute Changes in Ambient Temperature. *Physiological Zoology*, *59*(1), 55–59. https://doi.org/10.1086/physzool.59.1.30156090

Vogt, F. D. (1986b). Thermoregulation in Bumblebee Colonies. II. Behavioral and Demographic Variation throughout the Colony Cycle. *Physiological Zoology*, *59*(1), 60–68. https://doi.org/10.1086/physzool.59.1.30156091

Vogt, F. D., Heinrich, B., Dabolt, T. O., & McBath, H. L. (1994). Ovary development and colony founding in subarctic and temperate-zone bumblebee queens. *Canadian Journal of Zoology*, *72*(9), 1551–1556. https://doi.org/10.1139/z94-206

Walther-Hellwig, K., Fokul, G., Frankl, R., Büchler, R., Ekschmitt, K., & Wolters, V. (2006). Increased density of honeybee colonies affects foraging bumblebees. *Apidologie*, *37*(5), 517–532. https://doi.org/10.1051/apido:2006035

Waser, N. M., & Price, M. V. (2016). Drought, pollen and nectar availability, and pollination success. *Ecology*, *97*(6), 1400–1409. https://doi.org/10.1890/15-1423.1

Washington Department of Fish and Wildlife. (2015). *Washington's State Wildlife Action Plan: 2015 Update*. Washington Department of Fish and Wildlife.

Washington Department of Fish and Wildlife, Idaho Department of Fish and Game, Oregon Department of Fish and Wildlife, & The Xerces Society for Invertebrate Conservation. (2023). *Pacific Northwest Bumble Bee Atlas*. PNW Bumble Bee Atlas. https://www.pnwbumblebeeatlas.org/

Washington State Department of Agriculture. (2018). *Managed Pollinator Protection Plan*. 11 pp. https://agr.wa.gov/getmedia/819f8d22-37b1-484d-a522-31f60875f9c9/101-681managedpollinatorprotectionplan.pdf

Weaving, H., Terblanche, J. S., Pottier, P., & English, S. (2022). Meta-analysis reveals weak but pervasive plasticity in insect thermal limits. *Nature Communications*, *13*(1), Article 1. https://doi.org/10.1038/s41467-022-32953-2

Weidenmüller, A., Kleineidam, C., & Tautz, J. (2002). Collective control of nest climate parameters in bumblebee colonies. *Animal Behaviour*, *63*(6), 1065–1071. https://doi.org/10.1006/anbe.2002.3020

Westphal, C., Steffan-Dewenter, I., & Tscharntke, T. (2006). Foraging trip duration of bumblebees in relation to landscape-wide resource availability. *Ecological Entomology*, *31*, 389–394.

White, S. A., & Dillon, M. E. (2023). Climate warming and bumble bee declines: The need to consider sub-lethal heat, carry-over effects, and colony compensation. *Frontiers in Physiology*, *14*. https://www.frontiersin.org/articles/10.3389/fphys.2023.1251235

Whitehorn, P. R., O'Connor, S., Wackers, F. L., & Goulson, D. (2012). Neonicotinoid Pesticide Reduces Bumble Bee Colony Growth and Queen Production. *Science*, *336*(6079), 351–352. https://doi.org/10.1126/science.1215025

Wilfert, L., Long, G., Leggett, H. C., Schmid-Hempel, P., Butlin, R., Martin, S. J. M., & Boots, M. (2016). Deformed wing virus is a recent global epidemic in honeybees driven by *Varroa* mites. *Science*, *351*(6273), 594–597. https://doi.org/10.1126/science.aac9976

Williams, N. M., & Hemberger, J. (2023). Climate, pesticides, and landcover drive declines of the western bumble bee. *Proceedings of the National Academy of Sciences*, *120*(7), Article 7. https://doi.org/10.1073/pnas.2221692120

Williams, N. M., Mola, J. M., Stuligross, C., Harrison, T., Page, M. L., Brennan, R. M., Rosenberger, N. M., & Rundlöf, M. (2019). Fantastic bees and where to find them: Locating the cryptic overwintering queens of a western bumble bee. *Ecosphere*, *10*(11). https://doi.org/10.1002/ecs2.2949

Williams, P., Colla, S., & Xie, Z. (2009). Bumblebee Vulnerability: Common Correlates of Winners and Losers across Three Continents. *Conservation Biology*, *23*(4), 931–940. https://doi.org/10.1111/j.1523-1739.2009.01176.x

Williams, P. H. (2021). Not just cryptic, but a barcode bush: PTP re-analysis of global data for the bumblebee subgenus *Bombus s. str* . supports additional species (Apidae, genus *Bombus*). *Journal of Natural History*, *55*(5–6), 271–282. https://doi.org/10.1080/00222933.2021.1900444

Williams, P. H., Cameron, S. A., Hines, H. M., Cederberg, B., & Rasmont, P. (2008). A simplified subgeneric classification of the bumblebees (genus Bombus). *Apidologie*, *39*(1), 46–74. https://doi.org/10.1051/apido:2007052

Williams, P. H., Thorp, R. W., Richardson, L., & Colla, S. (2014). *Bumble bees of North America: An identification guide*. Princeton University Press.

Willmer, P. (2012). Ecology: Pollinator–Plant Synchrony Tested by Climate Change. *Current Biology*, *22*(4), R131–R132. https://doi.org/10.1016/j.cub.2012.01.009

Wilson, E. E., Holway, D., & Nieh, J. C. (2006). Cold anaesthesia decreases foraging recruitment in the New World bumblebee,Bombus occidentalis. *Journal of Apicultural Research*, *45*(4), 169–172. https://doi.org/10.1080/00218839.2006.11101343

Winter, K., Adams, L., Thorp, R., Inouye, D. W., Day, L., Ascher, J., & Buchmann, S. (2006). *Importation of Non-Native Bumble Bees into North America: Potential Consequences of Using Bombus terrestris and Other Non-Native Bumble Bees for Greenhouse Crop Pollination in Canada, Mexico, and the United States. A white paper of the North American Pollinator Protection Campaign (NAPPC)* (p. 35) [White paper].

Witmer, G., W., Moulton, R., S., & Swartz, J., L. (2012). Rodent Burrow Systems in North America: Problems Posed and Potential Solutions*. *Proceedings of the Vertebrate Pest Conference*, *25*. https://doi.org/10.5070/v425110499

Wolf, S., Hartl, B., Carroll, C., Neel, M. C., & Greenwald, D. N. (2015). Beyond PVA: Why Recovery under the Endangered Species Act Is More than Population Viability. *BioScience*, *65*(2), 200–207. https://doi.org/10.1093/biosci/biu218

Woodard, S. H., Lozier, J. D., Goulson, D., Williams, P. H., Strange, J. P., & Jha, S. (2015). Molecular tools and bumble bees: Revealing hidden details of ecology and evolution in a model system. *Molecular Ecology*, *24*(12), 2916–2936. https://doi.org/10.1111/mec.13198

Wu-Smart, J., & Spivak, M. (2018). Effects of neonicotinoid imidacloprid exposure on bumble bee (Hymenoptera: Apidae) queen survival and nest initiation. *Environmental Entomology*, *47*(1), 55–62. https://doi.org/10.1093/ee/nvx175

Xie, Z., Williams, P. H., & Tang, Y. (2008). The effect of grazing on bumblebees in the high rangelands of the eastern Tibetan Plateau of Sichuan. *Journal of Insect Conservation*, *12*(6), 695–703. https://doi.org/10.1007/s10841-008-9180-3

Zattara, E. E., & Aizen, M. A. (2021). Worldwide occurrence records suggest a global decline in bee species richness. *One Earth*, *4*(1), 114–123.

Zayed, A. (2009). Bee genetics and conservation. *Apidologie*, *40*(3), 237–262. https://doi.org/10.1051/apido/2009026

Zayed, A., & Packer, L. (2005). Complementary sex determination substantially increases extinction proneness of haplodiploid populations. *Proceedings of the National Academy of Sciences*, *102*(30), 10742–10746. https://doi.org/10.1073/pnas.0502271102

Ziehn, T., Chamberlain, M. A., Law, R. M., Lenton, A., Bodman, R. W., Dix, M., Stevens, L., Wang, Y.-P., & Srbinovsky, J. (2020). The Australian Earth System Model: ACCESS-ESM1.5. *Journal of Southern Hemisphere Earth Systems Science*, *70*(1), 193–214.

11 Appendices

11.1 Suggestions For Future Work

Suckley's cuckoo bumble bee is a very data poor species. Throughout the development of this assessment we, our expert group, and our modelers continuously encountered questions about *Psithyrus* life history characteristics, behavior, and ecology for which there is a lack of even basic information. We have compiled the following list of suggestions for future work, with the hope that they may inform future revisions to this living document as well as other *Psithyrus* status assessments and conservation efforts.

Psithyrus fecundity – very little information exists on fecundity in cuckoo bumble bees; for example, how quickly a Suckley's cuckoo bumble bee queen begins to lay eggs in the nest of a host colony after killing or displacing the foundress queen; how many eggs may be deposited and at what rate, and whether this is correlated with host colony size and age; and information on how male vs. female eggs are deposited. Limited information can be found in Hobbs 1965, Fisher 1984, Fisher 1987, Goulson 2010, and Koch 2021. In 1965, Hobbs reported a *B. suckleyi* queen laid 17 and 8 eggs in two cells respectively on top and near the center of a *B. nevadensis* nest, but does not indicate how long this took or when the process started after the *B. suckleyi* queen invaded the nest (Hobbs 1965*a*:127). In Fisher 1987, there is no specific data on fecundity but fascinating study of queen/host interaction in the nest, focused on *B. affinis* and *B. ashtoni*. On page 1032, there is a note that 1,872 male and 1,062 female *Psithyrus* were produced in 39 nests (cohorts A and B) with a resulting sex ratio of 0:64 (Fisher 1987:1032). Extrapolating from this, 1,872 + 1,062 = 2,934/2 = 75.2 *Psithyrus* per nest, but this result was not a specific goal of the study.

Host dynamics – at the start of this assessment and as described in the petition (Center for Biological Diversity 2020:4), *B. suckleyi* was thought to have only been confirmed as reproducing in *B. occidentalis* nests, but suspected in several other likely hosts based on historic observations and nest excavations (see SSA Section 3.1.4). However, in the development of this assessment we found confirmation that Suckley's cuckoo bumble bees reproduced in a *B. nevadensis* nest (Hobbs 1965*b*:120). Extensive research could be done on multiple *Psithyrus* and the dynamics with their hosts, including host preference, confirmation of successful reproduction in suspected hosts, rates of infestation, environmental stressors (specifically heat and resource availability) contributing to rates of success, and *Psithyrus* success across multiple hosts. See Antonovics and Edwards, 2011 for more information (Antonovics and Edwards 2011: entire).

*Genetics and specimen verification –*investigations of curated *B. suckleyi* specimens to verify their identity took place throughout the development of this assessment. This includes work done by our expert panel on specimens from across the range and in multiple collections, including the Canadian National Collection, the Oregon State Arthropod Collection, the American Museum of Natural History, and others. Currently, a verified DNA barcode for *B. suckleyi* is not publicly available, against which samples and specimens could be tested to confirm their identity. Additional verification work for specimens held in private collections may yield further insight into species abundance, distribution, and population structure. A verified DNA barcode for the species, along with other genomic work, could provide more information on how the effects of small population structure may have impacted the species, and form the foundation for additional field survey efforts such as eDNA sampling. Comparing genetics from the eastern and western portions of the range is of particular interest.

Expanded and targets survey effort – Our understanding of bumble bee distribution and their interaction with their environment – particularly their response to threats – would be improved with an expanded survey effort. Some evidence suggests that current strongholds for multiple species are at higher elevations, somewhat spatially removed from concentrations of particular threats. Targeted surveys in higher elevation locations, which are traditionally poorly-sampled due to remoteness, would improve our understanding of bumble bee distribution and provide insights into conservation.

11.2 Floral Associations

The table below summarizes known floral associations of Suckley's cuckoo bumble bee. These associations should not be considered a complete list of the floral resources used as forage by the species.

***Table A1**. Known floral associations for Suckley's, obtained from occurrence data and published sources (COSEWIC, 2019, p. 27; Koch et al., 2012, p. 134; P. H. Williams et al., 2014, p. 163).*

Family	Genus	Species	Common Name
Asteraceae	*Balsamorhiza*		balsamroot
	Centaurea		knapweeds
	Chrysothamnus	*viscidiflorus*	yellow or green rabbitbrush
	Chrysothamnus		rabbitbrush
	Cirsium		thistle
	Gaillardia		blanket flower
	Grindelia	*squarrosa*	curlycup gumweed
	Grindelia		gumplant
	Helianthella		little sunflower
	Helianthus		sunflower
	Helianthus	*multiflora*	manyflower sunflower
	Helichrysum		everlasting flower
	Heliomeris	*multiflora*	showy goldeneye
	Hymenoxys	*hoopesii*	owl's claws or orange sneezeweed
	Pyrrocoma	*crocea*	goldenweed
	Rudbeckia		coneflower
	Senecio		ragwort
	Solidago	*canadensis*	Canada goldenrod
	Solidago		goldenrod
	Symphyotrichum		American aster
	Taraxacum	*campylodes*	dandelion
	Taraxacum		dandelion
Boraginaceae	*Mertensia*	*ciliata*	mountain bluebell
	Mertensia	*lanceolata*	prairie bluebell
Ericaceae	*Vaccinium*		blueberry, cranberry, etc.
Fabaceae	*Astragalus*		milkvetch
	Melilotus		sweet clover
	Trifolium	*pratense*	red clover
	Trifolium		clover
Gentianaceae	*Frasera*		green gentian
Geraniaceae	*Geranium*		geranium or crane's bill
Lamiaceae	*Monarda*	*fistulosa*	wild bergamot
Onagraceae	*Chaemaenerion*		fireweed
Plantaginaceae	*Penstemon*		beardtongue
Rosaceae	*Cotoneaster*		cotoneaster
	Prunus	*cha*	plum
	Rubus		bramble, raspberry, blackberry, etc.

11.3 Stressors Excluded from the Analysis

Table A2. Stressors that were identified by the petition, in the literature, or by experts that were concluded to be insubstantial to the overall population resiliencies of Suckley's cuckoo bumble bee.

Stressor	Summary of affects to bees	Rationale for exclusion
Early spring frost effect on thermal suitability and survival	Spring frosts after the start of spring may create temperatures below the lower thermal thresholds of new queens and workers, potentially causing direct mortality.	It is expected that newly emerged queens can enter torpor (diapause) to withstand abnormal cold frosts. Additionally, *Psithyrus* tend to emerge later than other bumble bees which would lessen the potential effects. This effect is understudied and could use further research consideration.
Invasive plants effect on floral resources	Invasive plants create a monoculture, therefore reducing floral diversity (quality) and temporal availability.	This effect is mostly localized as well as complex; in some areas invasive plants are favored by bumble bees and they may provide an important source of nectar and pollen when other blooming species are scarce. In other cases, invasives create monocultures that offer little to no value to pollinators.
Mowing effect on floral resources	Mowing directly removes floral resources and may create short-term resource gaps for colonies.	This effect is localized and complex, as mowing could also stimulate floral growth. The influence likely depends on the seasonality and landscape context.
Mowing effects on nest sites and diapause sites	Mowing may compact soil and reduce nest sites and diapause sites when heavy machinery is used, especially on moist soils.	This effect is localized and complex and depends on soil moisture and intensity/frequency of mowing.
Phenological decoupling effect on floral resources	With climate change, floral resources are expected to bloom earlier as the start of spring advances. If the cues that plants use to emerge in the spring are different than those used by diapausing bees, a decoupling may occur between floral and bee emergence, which can create resource gaps.	Although this risk may increase in the future, currently, studies have shown that bumble bees appear to be keeping up with the advancing start of spring.
Predation effect on survival	Predation by vertebrates (birds, reptiles, and mammals) and invertebrates (cuckoo bees, and crab spiders) contribute to direct mortality of adults and the brood.	There is no evidence that predators have been increasing recently; as such, the natural rate of predation is unlikely to be a substantial threat to the resiliency of bumble bees.
Rodenticide effect on nest availability	Rodenticide application reduces rodent populations, and it may	Application is localized, and the relationship between rodenticide,

	influence nest site availability (lack of burrows).	rodent abundance, and nest sites is understudied in the literature.
Silviculture and other forestry activities effect on floral resources	Activities in timber stands may reduce floral resources by creating monocultures, introducing invasive plants, and compacting soil with heavy equipment.	Timber practices vary widely and have complex effects. Clear-cutting or timber thinning practices may increase early-succession habitats. The effect of timber activities on native pollinators is understudied.
Timber activities effect on survival	Timber activities may cause direct mortality if heavy equipment crushes active nest and diapause sites.	Timber activities and their effects on nest site and diapause site availability are understudied.
Wildfire effect on survival	Wildfires may cause direct mortality of individuals, especially if the fire has high-burn intensity or if the nest burrow is shallow.	This effect is short-term and localized. Underground nest sites are somewhat buffered from burns.
Wildfire effect on diapause	Wildfires could cause direct mortality to overwintering gynes or reduce the loose duff and litter used as diapause sites. It is also possible that fires may create additional diapause sites by adding woody debris.	Wildfires do not typically occur during the diapause period of bumble bees and diapause sites are understudied. The exact relationship of this influence is uncertain, and it is likely localized.

11.4 Additional Details on Occupancy Modelling Approach

The following summarizes information from Jackson et al. (2022), and additional analyses completed, in support of this SSA by Teagan Baiotto, Hanna Jackson, and Melissa Guzman.

Introduction

To aid in the assessment of *Bombus suckleyi*, we modeled bumble bee species occupancy across North America using the largest currently available dataset of bumble bee presence records (Richardson 2023) in conjunction with temperature, precipitation, and floral resource data. We quantified 1) the temporal trends of bumble bees in North America, 2) the effect of temperature, precipitation, and floral resource quality on bumble bee occupancy, and 3) projected our estimates into the future using climate projections to estimate trends in species' occupancy in future time periods.

Species distribution models are commonly used to determine species' range extents, habitat associations, and occupancy trends, and to predict future ranges under different climate change scenarios. While many studies have successfully used species distribution modeling with bee species, this approach does account for detection bias. Detection bias can be high for cryptic or rare species and in large-scale historical datasets in which survey methods, and detection, is likely to have changed over time. Moreover, species distribution models can be challenging for data-poor species.

This study uses a multi-species, Bayesian occupancy model to aid in the assessment of *B. suckleyi*, a species that is relatively cryptic and data poor. Occupancy models are an extension of species distribution modeling that explicitly account for potential biases in detection probability that, if ignored, can lead to spurious inferences. Occupancy models are apt for analyzing long-term, large-scale aggregate data, in which detection probability likely varies across surveys, like the Bumble bee occurrences of North America. The multi-species modeling approach is also apt, as it enables data-poor species to 'borrow strength' from more data-rich species. In this manner, a multi-species model can provide more statistical power for data-poor species than an analysis of the data-poor species alone. This approach allows us to model multiple bumble bee species in North America simultaneously but extract species-specific estimates for *B. suckleyi.* Finally, this multi-species approach is also beneficial, as it provides occupancy estimates for the hosts of *B. suckleyi*.

Methods

The analyses for the SSA generally followed methods outlined in Jackson et al. (2022, entire), with some modifications, outlined below.

Occurrence records. After data cleaning, the study used 796,985 occurrence records from 46 bumble bee species, dating 1900–2023. Researchers updated the models described in Jackson et al. (2022, entire) to include new records of *B. suckleyi*, and to remove some previously mis-identified records. In occupancy models, the occupancy probability of each species is estimated for each site in each time period of interest. Sites were generated by overlaying a 50x50-km grid onto North America. The records were split into 12 10-year eras, each of which was sub-divided into 10 1-year time intervals. We estimated occupancy in each era using 10 1-year 'visits' that represent replicate observations of the occupancy state of that species at that site in that era. For each species, non-detection was assigned at each site in each visit, if at least one other species of bumble bee was observed but that focal species was not (Kery et al. 2010, p. 1388).

Habitat and climate data. The models used data on temperature, precipitation, and floral resources. Models also included time (era) as a covariate to 1) assess overall temporal trends in occupancy, and 2) to account for any factors that are influencing the species but are not accounted for in the model due to lack of data (e.g., introduced species, pathogens, pesticides, wildfire, and livestock grazing, etc.). For the SSA, we allowed time to have a quadratic component (instead of simply linear as in Jackson et al. 2022).

We calculated floral resource data in an identical way to Jackson et al. (2022, p. 3) and based it on land use classification data for the Holocene, agricultural crop data, and previously established floral resource scores for bees, which were based on expert-opinion. Floral resource scores are held at 2020 levels in future projections of bumble bee occupancy, as there are not any future projections of this variable.

Historical and future climate data for this study were from the ACCESS1-3 model that was statistically downscaled by Karger et al. (2020, entire). These climate data are from global circulation models (GCMs) from the Coupled Model Intercomparison Project phase 5 (CMIP5), that are downscaled using the CHELSA algorithm, which provides improved estimates of temperature and precipitation in highly complex terrain (Karger et al. 2020, p. 2). For each of our sites from 1900-2006, we calculated: 1) the decadal mean of the average maximum temperature of the warmest month in each year and 2) the decadal monthly mean precipitation. We used climate data under RCP4.5 for the 2006-2020 time period.

Occupancy models.

The model used for prediction of occupancy [termed the **predictive model**] followed the equation:

$$\begin{gathered}\mathrm{logit}\left(\psi_{ijk}\right) = \psi_0 + \\ \psi\ \mathrm{species}_i + \\ \left(\beta\ \mathrm{area}\ x\ \mathrm{area}_j\right) + \\ (\ K\ x\ \beta\ \mathrm{era}_i) + (K^2\ \mathrm{x}\ \beta\ \mathrm{era2}\ _i) + \\ (\beta\ \mathrm{temp}_i\ x\ \ temp_{jk}) + (\beta\ \mathrm{temp2}\ x\ \ temp_{jk}^2) + \\ (\beta\ \mathrm{precip}_i\ x\ \ precip_{jk}) + \\ (\beta\ \mathrm{floral}\ _i\ x\ \ floral_{jk})\end{gathered}$$

Here, the i subscript indexes each specific species, j indexes each specific site, and k each specific era (e.g., time). Thus:

… ψ_{ijk} denotes occupancy of species i at site j, in time k while, ψ_0 denotes mean occupancy of all species in year one, and $\psi\ \mathrm{species}_i$ denotes a species-specific random effect for occupancy.

… $\beta\ \mathrm{area}$ denotes a fixed effect of site area, to account for the fact that some sites are smaller due to water bodies, while area_j represents the area of site j.

…$\beta\ \mathrm{era}_i$ denotes a species-specific effect of era (e.g., time), and *K* represents era (e.g.., time). Note, there is also a species-specific linear and quadradic term for era modeled, which is denoted by $\beta\ \mathrm{era2}\ _i$

…$\beta\ \mathrm{temp}_i$ denotes a species-specific effect of temperature, and $temp_{jk}$ represents the temperature at site j at time k. Note there is also a quadratic effect of temperature modeled, that is not species-specific (e.g., averaged across species), and denoted by $\beta\ \mathrm{temp2}$

…$\beta\ \mathrm{precip}_i$ denotes a species-specific effect of precipitation, and $precip_{jk}$ represents the precipitation at site j at time k.

…$\beta\ \mathrm{floral}_i$ denotes a species-specific effect of floral resources, and $floral_{jk}$ represents the precipitation at site j at time k.

The researchers also developed an additional model [termed the **regional model**] that specifically tested whether the temporal trend differed between ecoregions. This model was not used for prediction, rather solely to assess whether it was reasonable to assume that the overall, average, rangewide trends estimated in the predictive model were similar across different portions of the range. Here we only ran the model for regions where there was sufficient data to estimate a regional trend. The model used for this analysis followed the equation:

$$\begin{gathered}\mathrm{logit}\left(\psi_{ijk}\right) = \ \psi_0 + \\ \psi\ \mathrm{species}_i + \\ \left(\beta\ \ \mathrm{area}\ x\ \ \mathrm{area}_j\right) + \\ (\ K\ x\ \beta\ \ \mathrm{era}_{ir}) +\ (K^2\ \mathrm{x}\ \beta\ \mathrm{era2}\ _{ir})\end{gathered}$$

Here, the I subscript again indexes each specific species, j indexes each specific site, and k each specific era (e.g., time). In this case, however, there is also an index r, for each region. Thus, all model parameters are as defined above, with the exception that $\beta\ \mathrm{era}_{ir}$ and $\beta\ \mathrm{era2}_{ir}$ denote the linear, and quadratic components of a species-specific temporal trend in each region r, respectively.

Occupancy models contain an ecological process component that describes how ecological factors are influencing species' distribution across the landscape, and a detection process, that describes factors that influence probability of detection. The equations above capture the ecological process component. The detection process was subsequently modeled as:

$$\begin{gathered}\mathrm{logit}\left(\mathrm{p}_{ijk}\right) = \ \mathrm{p}_0 + \\ \mathrm{p\ species}_i + \\ \mathrm{p\ site.\,era}_{jk}\end{gathered}$$

Here p_{ijk} denotes the probability of detection for species i, at site j, in time period k, p_o represents the mean detection probability across sites, species, and time periods, and $\mathrm{p\ site.\,era}_{jk}$ a site-specific and era-specific random effect that allows detection to vary relatively independently across sites and between eras.

These models all assume that species-specific (as well as species-region specific, and site-era specific) slopes are normally distributed about some mean.

Future projections. Using the predictive model, we spatially project probability of occupancy for Suckley's cuckoo bumble bee across its range, currently, and into the future at multiple time steps

(2020-2040; 2040-2060; 2060-2080; 2080-2100). “Current” projections reflect estimated occupancy for the 2000-2020 time period. Future occupancy projections were made under two different scenarios. Both scenarios hold floral resources constant at their 2020 levels, as data are not available to project this variable into the future. Both scenarios use future projections of average maximum temperature of the warmest month in each year, and decadal monthly mean precipitation, from the same data set as was used to build the models.

Scenario 1 uses climate projections under RCP4.5, which represents a moderate climate change warming scenario, while scenario 2 uses projections under RCP8.5, which represents a high warming scenario. Scenario 2 also assumes that average, observed, historical rates of decline due to other, non-climatic factors continue into the future (i.e., trend momentum), while Scenario 1 does not project these historical trends into the future (i.e., no trend momentum). Thus, under scenario 1, any changes in future occupancy are based solely on changes in climate (assuming RCP4.5), while under scenario 2, any changes in future occupancy are based on both climate change (assuming RCP8.5), as well as historical rates of change due to non-climatic factors.

In these scenarios, RCP4.5 specifically projects a net change in radiative forcing of 4.5 W/m2 (~650 ppm CO2 eq) by 2100; this equates to an estimated average global mean surface temperature change of 1.8°C (likely range of 1.1 to 2.6°C) (3.2°F [2.0-4.7°F]) by 2100 (Thomson et al. 2011, entire; Collins et al. 2013, entire). In contrast, under RCP8.5, the net change in radiative forcing is 8.5 W/m2 by 2100 (~1370 ppm CO2 eq); this equates to an estimated average global mean surface temperature change of 3.7°C (likely range of 2.6 to 4.8°C) (6.7°F [4.7-8.6°F]) by 2100 (Riahi et al. 2011, entire; Collins et al. 2013, entire). These RCP scenarios were chosen, as they are the recommended upper and lower bounds scenarios for FWS planning and decision making (US Fish and Wildlife Service 2023, p. 10).
We used the same occupancy modeling framework and scenarios to spatially project future probability of occupancy for the confirmed and potential hosts of Suckley’s cuckoo bumble bee.

Table A3. Scenarios for analyzing the future condition of Suckley’s cuckoo bumble bee.

Factor	Scenario 1	Scenario 2
Climate change	RCP4.5	RCP8.5
Floral resources	Held constant at 2020 levels	Held constant at 2020 levels
Trend Momentum. Temporal declines observed after accounting for changes in climate and floral resources (e.g., temporal declines driven by other factors not explicitly included in models, like pesticides, etc.).	No trend momentum: Historical rate of temporal decline (driven by non-climatic factors) not projected into the future.	Continued trend momentum: Historical rate of temporal decline (driven by non-climatic factors) projected into the future.

Literature Cited

Collins, M., R. Knutti, J. Arblaster, J.L. Dufresne, T. Fichefet, P. Friedlingstein, X. Gao, W.J. Gutowski, T. Johns, G. Krinner, M. Shongwe, C. Tebaldi, A.J. Weaver, and M. Wehner. 2013. Long-term Climate Change: Projections, Commitments and Irreversibility. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V.Bex and P.M. Midgley (eds.)]. Cambridge U. Press, Cambridge, UK and New York, NY, USA. https://www.ipcc.ch/site/assets/uploads/2018/02/WG1AR5_Chapter12_FINAL.pdf

Jackson, H. M., Johnson, S. A., Morandin, L. A., Richardson, L. L., Guzman, L. M. and M'Gonigle, L. K. 2022. Climate change winners and losers among north 124American Bumblebees. Biology letters, 18(6), 20210 551. https://royalsocietypublishing.org/doi/pdf/10.1098/rsbl.2021.0551

Karger, D. N., D. R. Schmatz, G. Dettling, and N. E. Zimmermann. 2020. High-resolution monthly precipitation and temperature time series from 2006 to 2100. Scientific Data 7(1):248. https://www.nature.com/articles/s41597-020-00587-y

Kery M., Royle J.A., Schmid H., Schaub M., Volet B., Haefliger G., Zbinden N. 2010 Site-occupancy distribution modeling to correct population-trend estimates derived from opportunistic observations. Conservation Biology. 24: 1388–1397. doi:10.1111/j.1523-1739.2010.01479.x

Riahi, K., S, Rao, V. Krey, C. Cho, V. Chirkov, G. Fischer, G. Kindermann, N. Nakicenovic, and P. Rafaj. 2011. RCP 8.5—A scenario of comparatively high greenhouse gas emissions. Climatic Change 109: 33-57. https://doi.org/10.1007/s10584-011-0149-y

Richardson, L. 2023. Bumble bees of North America occurrence records database.

Thomson, A.M., K.V. Calvin, S.J. Smith, G.P. Kyle, A. Vilke, P. Patel, S Delgado-Arias, B. Bond-Lamberty, M.A. Wise, L.E. Clarke, and J.A. Edmonds. 2011. RCP4.5: a pathway for stabilization of radiative forcing by 2100. Climate Change 109:77-94. https://doi.org/10.1007/s10584-011-0151-4

U.S. Fish and Wildlife Service. 2023. Frequently Asked Questions: Use of Climate Scenarios and Projections in Conservation Planning and Decision-making.

11.5 Supplementary Tables and Figures

***Table A4.**. The confirmed hosts, likely preferred hosts (e.g., subgenus Bombus, including Bombus occidentalis, Bombus cryptarum, Bombus mckayi, Bombus terricola, and Bombus affinis – in light blue) and potential, but unconfirmed hosts (Bombus appositus, Bombus fervidus, and Bombus rufocinctus) that have been detected at least once in each Suckley's cuckoo bumble bee analytical unit, since 2000.*

	Confirmed hosts		**Potential hosts**						
Analytical unit	*Bombus nevadensis*	*Bombus occidentalis*	*Bombus affinis*	*Bombus cryptarum*	*Bombus mckayi*	*Bombus terricola*	*Bombus appositus*	*Bombus fervidus*	*Bombus rufocinctus*
Atlantic Highlands	0	0	0	0	0	1	0	1	1
Boreal Cordillera	1	0	0	1	1	1	0	1	1
Boreal Plains	1	1	0	1	0	1	0	1	1
Brooks Range Tundra	0	0	0	1	0	0	0	0	0
Cold Deserts	1	1	0	0	0	1	1	1	1
Hudson Plains	0	0	0	0	0	1	0	0	0
Marine West Coast Forests	1	1	0	0	0	0	1	1	1
Mixed Wood Plains	0	0	1	0	0	1	0	1	1
Mixed Wood Shield	1	0	1	0	0	1	0	1	1
Softwood Shield	0	0	0	0	0	1	0	0	1
South Central Semi-Arid Prairies	1	1	0	0	0	0	1	1	1
Taiga Plains	0	0	0	1	1	1	0	0	0
Temperate Prairies	1	1	1	1	0	1	0	1	1
West Central Semi-Arid Prairies	1	1	0	1	0	1	1	1	1
Western Cordillera	1	1	0	1	0	1	1	1	1
Totals	**9**	**7**	**3**	**7**	**2**	**12**	**5**	**11**	**12**

***Table A5**. Current condition summary information for each analytical unit, including the year of last detection for Suckley's cuckoo bumble bee, percent decline of Suckley's cuckoo bumble bee occupancy from historical to current, current estimated median occupancy for Suckley's cuckoo bumble bee and its two confirmed host species, B. occidentalis, and B. nevadensis, the number of likely preferred hosts from subgenus Bombus (not including B. nevadensis) detected at least once in each analytical unit since 2000, and a tally of the number of different potential, preferred, and confirmed host species detected since 2000.*

Analytical unit	**Last detection**	**Percent decline**	**Occupancy Suckley's**	**Occupancy *B. occidentalis***	**Occupancy *B. nevadensis***	**Preferred hosts**	**All potential hosts**
Atlantic Highlands	1924	-79	0.149			1	3
Boreal Cordillera	2019	-79	0.147		0.399	3	6
Boreal Plains	2022	-78	0.145	0.473	0.646	3	6
Brooks Range Tundra	2019	-79	0.147			1	1
Cold Deserts	2011	-87	0.061	0.358	0.727	2	6
Hudson Plains	1949	-80	0.133			1	1
Marine West Coast Forests	1982	-84	0.095	0.588	0.098	1	5
Mixed Wood Plains	1971	-83	0.106			2	4
Mixed Wood Shield	1995	-83	0.098		0.56	2	5
Softwood Shield	2010	-79	0.142			1	2
South Central Semi-Arid Prairies	2014	-89	0.013	0.312	0.651	1	5
Taiga Plains	1969	-77	0.161			3	3
Temperate Prairies	2018	-89	0.028	0.167	0.474	4	7
West Central Semi-Arid Prairies	2022	-83	0.1	0.452	0.761	3	7
Western Cordillera	2018	-80	0.14	0.481	0.619	3	7

Table A6*. Median occupancy of Suckley's cuckoo bumble bee in each analytical unit for the historical, current, and future time periods out to end of century, under* ***Scenario 1****. The interquartile range (IQR) which represents the 25th and 75th quantile of values, is also shown to represent the spread of values around the median in each analytical unit.*

	Historical		**Current**		**2030**		**2050**		**2070**		**2090**	
Analytical unit	median	IQR	median	IQR	median	IQR	median	IQR	median	IQR	median	IQR
Atlantic Highlands	0.712	*0.695 - 0.736*	0.149	*0.14 - 0.168*	0.096	*0.086 - 0.11*	0.087	*0.075 - 0.101*	0.079	*0.066 - 0.094*	0.075	*0.063 - 0.088*
Boreal Cordillera	0.706	*0.706 - 0.711*	0.147	*0.147 - 0.15*	0.108	*0.108 - 0.11*	0.112	*0.112 - 0.113*	0.111	*0.111 - 0.112*	0.110	*0.11 - 0.111*
Boreal Plains	0.661	*0.632 - 0.694*	0.145	*0.135 - 0.154*	0.084	*0.075 - 0.095*	0.067	*0.063 - 0.075*	0.048	*0.036 - 0.062*	0.033	*0.023 - 0.042*
Brooks Range Tundra	0.705	*0.705 - 0.705*	0.147	*0.147 - 0.147*	0.109	*0.109 - 0.109*	0.124	*0.124 - 0.124*	0.125	*0.125 - 0.125*	0.122	*0.122 - 0.122*
Cold Deserts	0.485	*0.295 - 0.650*	0.061	*0.028 - 0.112*	0.019	*0.008 - 0.042*	0.016	*0.006 - 0.032*	0.010	*0.003 - 0.024*	0.009	*0.003 - 0.018*
Hudson Plains	0.682	*0.663 - 0.692*	0.133	*0.123 - 0.139*	0.079	*0.071 - 0.085*	0.078	*0.07 - 0.086*	0.064	*0.056 - 0.074*	0.056	*0.048 - 0.066*
Marine West Coast Forests	0.580	*0.521 - 0.649*	0.095	*0.076 - 0.119*	0.060	*0.047 - 0.071*	0.052	*0.037 - 0.065*	0.052	*0.037 - 0.065*	0.041	*0.026 - 0.056*
Mixed Wood Plains	0.640	*0.392 - 0.690*	0.106	*0.042 - 0.132*	0.060	*0.021 - 0.079*	0.054	*0.019 - 0.069*	0.041	*0.013 - 0.057*	0.034	*0.012 - 0.055*
Mixed Wood Shield	0.590	*0.562 - 0.617*	0.098	*0.082 - 0.109*	0.051	*0.042 - 0.056*	0.056	*0.035 - 0.062*	0.033	*0.022 - 0.038*	0.011	*0.009 - 0.015*
Softwood Shield	0.685	*0.668 - 0.699*	0.142	*0.133 - 0.152*	0.096	*0.088 - 0.105*	0.093	*0.083 - 0.106*	0.094	*0.080 - 0.107*	0.09	*0.073 - 0.100*
South Central Semi-Arid Prairies	0.118	*0.110 - 0.232*	0.013	*0.012 - 0.03*	0.003	*0.003 - 0.008*	0.003	*0.002 - 0.006*	0.002	*0.002 - 0.004*	0.001	*0.001 - 0.003*
Taiga Plains	0.708	*0.690 - 0.730*	0.161	*0.153 - 0.168*	0.106	*0.097 - 0.116*	0.092	*0.08 - 0.111*	0.084	*0.071 - 0.103*	0.090	*0.071 - 0.110*
Temperate Prairies	0.263	*0.125 - 0.655*	0.028	*0.011 - 0.137*	0.014	*0.004 - 0.079*	0.01	*0.003 - 0.057*	0.006	*0.002 - 0.035*	0.004	*0.001 - 0.018*
West Central Semi-Arid Prairies	0.577	*0.462 - 0.649*	0.100	*0.073 - 0.123*	0.046	*0.025 - 0.064*	0.021	*0.013 - 0.035*	0.016	*0.009 - 0.025*	0.009	*0.005 - 0.014*
Western Cordillera	0.692	*0.62 - 0.725*	0.140	*0.098 - 0.16*	0.078	*0.046 - 0.104*	0.052	*0.028 - 0.086*	0.046	*0.025 - 0.081*	0.030	*0.014 - 0.063*

Table A7. *Median occupancy of Suckley's cuckoo bumble bee in each analytical unit for the historical, current, and future time periods out to end of century, under* ***Scenario 2***. *The interquartile range (IQR) which represents the 25th and 75th quantile of values, is also shown to represent the spread of values around the median in each analytical unit.*

	Historical		**Current**		**2030**		**2050**		**2070**		**2090**	
Analytical unit	median	IQR	median	IQR	median	IQR	median	IQR	median	IQR	median	IQR
Atlantic Highlands	0.712	*0.695 - 0.736*	0.149	*0.14 - 0.168*	0.031	*0.028 - 0.036*	0.004	*0.003 - 0.005*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Boreal Cordillera	0.706	*0.706 - 0.711*	0.147	*0.147 - 0.15*	0.037	*0.037 - 0.038*	0.005	*0.005 - 0.005*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Boreal Plains	0.661	*0.632 - 0.694*	0.145	*0.135 - 0.154*	0.031	*0.027 - 0.033*	0.003	*0.002 - 0.003*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Brooks Range Tundra	0.705	*0.705 - 0.705*	0.147	*0.147 - 0.147*	0.040	*0.040 - 0.040*	0.006	*0.006 - 0.006*	0.001	*0.001 - 0.001*	0.000	*0.000 – 0.000*
Cold Deserts	0.485	*0.295 - 0.650*	0.061	*0.028 - 0.112*	0.007	*0.003 - 0.014*	0.000	*0.000 - 0.001*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Hudson Plains	0.682	*0.663 - 0.692*	0.133	*0.123 - 0.139*	0.025	*0.022 - 0.028*	0.003	*0.003 - 0.003*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Marine West Coast Forests	0.580	*0.521 - 0.649*	0.095	*0.076 - 0.119*	0.020	*0.016 - 0.024*	0.002	*0.001 - 0.003*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Mixed Wood Plains	0.640	*0.392 - 0.690*	0.106	*0.042 - 0.132*	0.019	*0.007 - 0.025*	0.002	*0.001 - 0.003*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Mixed Wood Shield	0.590	*0.562 - 0.617*	0.098	*0.082 - 0.109*	0.014	*0.012 - 0.017*	0.001	*0.001 - 0.001*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Softwood Shield	0.685	*0.668 - 0.699*	0.142	*0.133 - 0.152*	0.032	*0.029 - 0.036*	0.004	*0.004 - 0.005*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
South Central Semi-Arid Prairies	0.118	*0.110 - 0.232*	0.013	*0.012 - 0.03*	0.001	*0.001 - 0.002*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Taiga Plains	0.708	*0.690 - 0.730*	0.161	*0.153 - 0.168*	0.035	*0.031 - 0.039*	0.004	*0.004 - 0.005*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Temperate Prairies	0.263	*0.125 - 0.655*	0.028	*0.011 - 0.137*	0.004	*0.001 - 0.026*	0.000	*0.000 - 0.002*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
West Central Semi-Arid Prairies	0.577	*0.462 - 0.649*	0.100	*0.073 - 0.123*	0.013	*0.007 - 0.021*	0.001	*0.000 - 0.001*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*
Western Cordillera	0.692	*0.62 - 0.725*	0.140	*0.098 - 0.16*	0.024	*0.013 - 0.035*	0.002	*0.001 - 0.004*	0.000	*0.000 – 0.000*	0.000	*0.000 – 0.000*

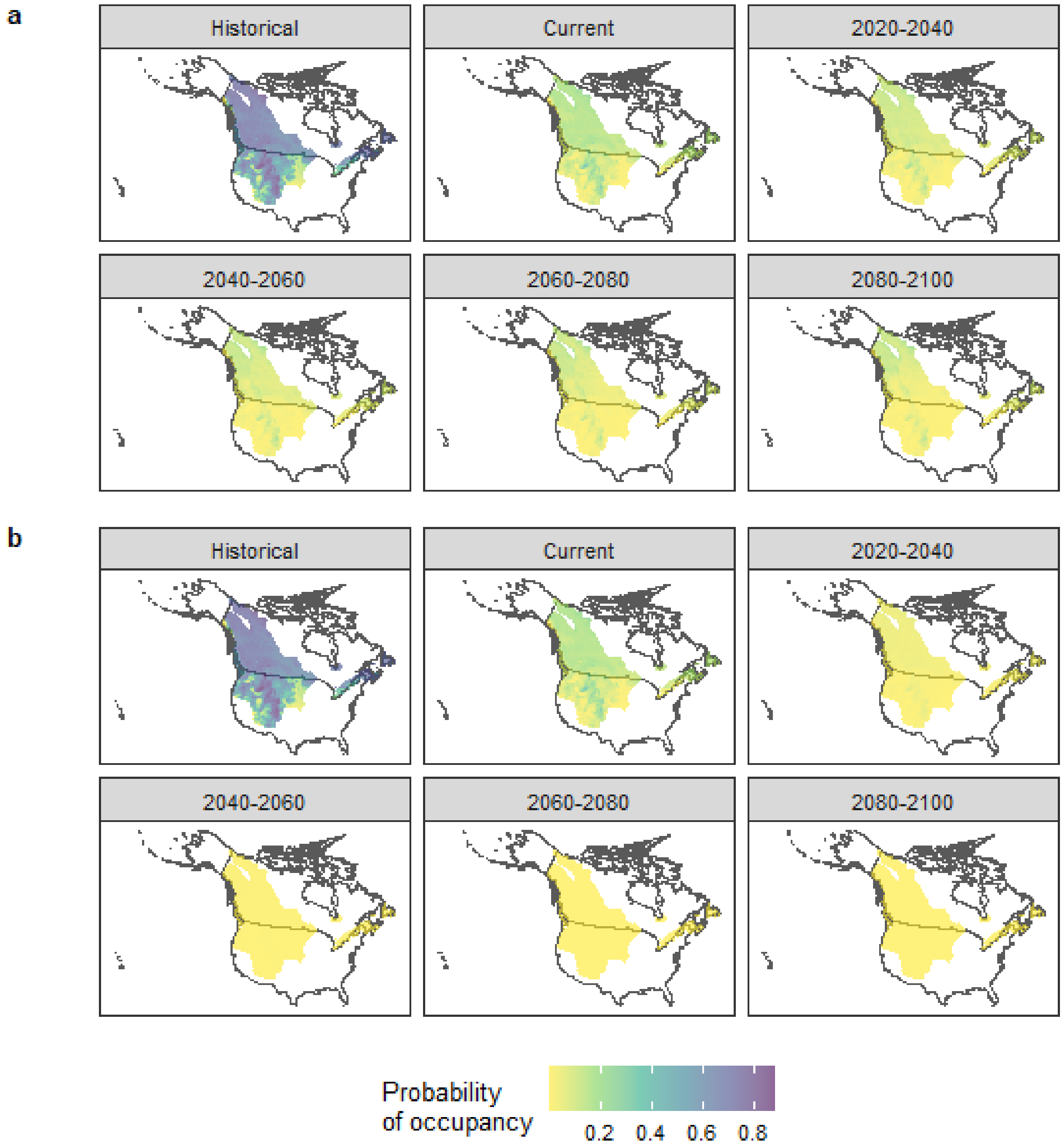


Figure A1*. Raw predicted probability of occupancy values for Suckley's cuckoo bumble bee, across the full range, at each time step, under Scenario 1 (a), and Scenario 2 (b).*

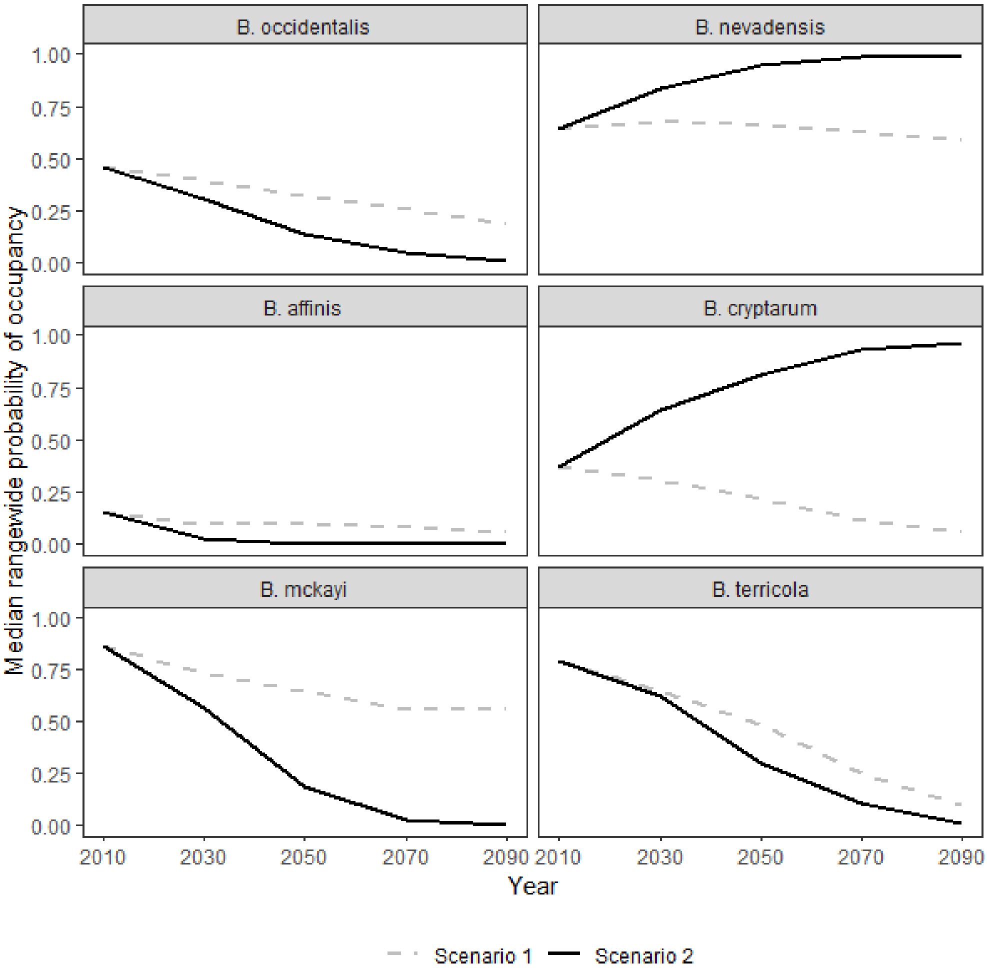


Figure A2. *Median estimated range wide probability of occupancy for two confirmed Suckley's cuckoo bumble bee host species (*B. occidentalis *and* B. nevadensis*), and four other likely preferred hosts in the Bombus sub-genus over time, under two different scenarios. The data for 2010 represents the midpoint of the current period (2000-2020), and the future projections represent the midpoints of the future time periods (2020-2040; 2040-2060; 2060-2080; 2080-2100), respectively. Scenario 1 assumes RCP4.5 and projects future occupancy based solely on climate change (e.g., historical rates of declines due to other factors are not projected), while Scenario 2 assumes RCP8.5, and projects future occupancy based on both climate change, and historical rates of decline due to non-climactic factors.*

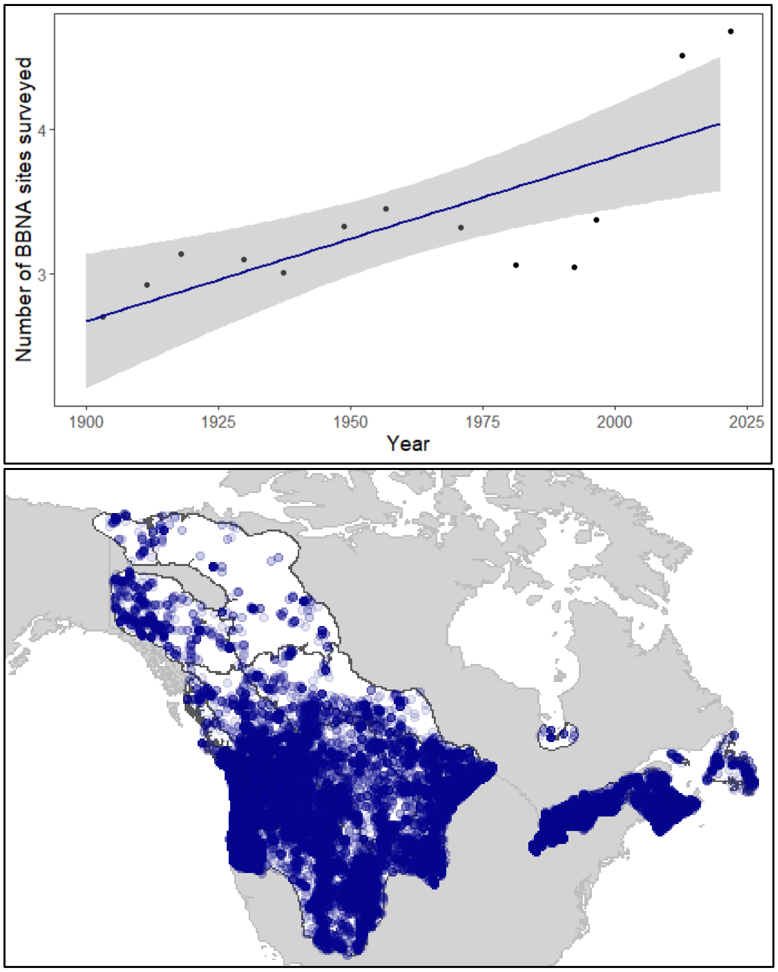


Figure A3. Top: *Number of unique Bumble Bees of North America locations surved by decade from 1900 to present, within the occurrence polygon of Suckley's cuckoo bumble bee. Values on the y axis are log-transformed, so the large uptick in surveys in the last two decades does not visually obscure trends from 1900 to 2000.* ***Bottom:*** *spatial depiction of the BBNA survey locations from 1900 to present, within the occurrence polygon of Suckley's cuckoo bumble bee. Data are from the Bumble Bees of North America (BBNA) survey database.*