

Species Status Assessment (SSA) Report
for the
Yellow Banded Bumble Bee
(*Bombus terricola*)
Version 1.1



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In order to produce the book's maps, graphs of seasonal activity patterns and lists of host plants, Leif Richardson gathered digital records of bumble bee specimens from museums, academic collections and private collectors. They also received some larger, aggregate datasets from individuals who had collected bee data for other purposes. The database included nearly 275,000 records of bumble bee specimens collected around North America from 1805 to the present. Leif Richardson continues to add to the database in order to use it for additional research and conservation projects, and as of August, 2018, it contains more than 500,000 specimen and observation records for North American bumble bees. This effort has benefited from contributions by more than 100 individuals and/ or institutions, and the data describe bee specimens held in more than 150 collections around the world. Below is a partial list of individuals and institutions that Leif Richardson gratefully acknowledge as direct contributors to this resource and to publications that are based on it. A full list of collections in which the specimens reside is available upon request.

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Executive Summary

This species status assessment reports the results of the comprehensive status review for the yellow banded bumble bee (*Bombus terricola*) and provides a thorough account of the species' overall viability. The yellow banded bumble bee (YBBB) is a bumble bee species native to the United States and Canada. The species occurs primarily in mixed woodland and wetland habitats.

To evaluate the biological status of the YBBB both currently and into the future, we assessed a range of conditions to allow us to consider the species' resiliency, redundancy, and representation (together, the 3Rs). The YBBB needs multiple resilient populations distributed widely across its range to maintain its persistence into the future and to avoid extinction. Factors that affect the viability of the species include the availability and connectivity of diverse and abundant floral resources, coupled with suitable nesting and overwintering habitat. As we consider the future viability of the species, more populations with high resiliency distributed across the known range of the species are associated with higher overall species viability.

The best available information indicates that historically the YBBB ranged across much of North America, including all or portions of 25 U.S. states and 12 Canadian provinces. However in recent decades the species' range has receded markedly in the southern and far western portions of its historical range and appears to have been extirpated from much of the Pacific Northwest, Southern Appalachians, and southeast plains. The YBBB's range also appears to have receded in some far northern Canadian provinces but this pattern is less certain because of generally low survey coverage in the region. The YBBB was once common and well represented within the core of its range (e.g., the upper Great Lakes region, New England, Ontario, Quebec, and the Canadian Maritime Provinces); however, it appears to have lost resiliency and likely to have lost representation in these areas. There are no current YBBB records in 11 of the U.S. states from which it is historically known: CT, GA, IL, IN, NC, NJ, OH, RI, TN, VA, and WA.

We assessed the status of the YBBB under the 3Rs by evaluating the occupancy and relative abundance of the species within and across Level II ecoregions. Historically, the YBBB's relative abundance percentages ranged from less than 1 to about 59 percent (mean 13 percent). Currently, the relative abundance percentages range from 0 to about 86 percent (mean 4 percent). Historical percent occupancy for the YBBB ranged from about 1 to 63 percent (mean 24 percent); the current percent occupancy ranges from 0 to 63 percent (mean 18 percent).

Our analysis of the past, current, and future influences on what the YBBB needs for long term viability revealed five primary influences affecting the future viability of the species. These influences are habitat loss and fragmentation, pesticide use, pathogens and parasites, the effects of small and isolated populations, and the effects of climate change. Throughout the range of the YBBB, habitat has been lost, degraded, and fragmented. Habitat loss by agriculture and urban and suburban development reduces the amount and connectivity of diverse and abundant floral resources throughout the colony life cycle; reduces nesting, mating, and overwintering habitat; and reduces connectivity of colonies and populations and ultimately the genetic health of the YBBB. Various insecticides, herbicides, and fungicides have the potential to affect the YBBB and its habitat directly and indirectly; some pesticides in use today are acutely toxic to bumble

bees. Additionally, the use of managed bees has increased, which has led to an increase of native and non-native pathogens and parasites in managed and native bee populations; pathogens and parasites can reduce fitness of the colony and cause mortality to individuals and colonies. Yellow banded bumblebee populations that are small and/or isolated are at risk of having reduced gene flow, reduced genetic diversity from inbreeding, and genetic drift, due to their smaller effective population size (number of reproductive bees) and haplodiploidy genetic structure (lower genetic diversity), and is exacerbated by environmental stressors. Evidence of ongoing climate change is apparent within the range of the species. Changes in climate can directly affect individual bees or disrupt the colony life cycle (i.e., timing of emergence and overwintering, the ability to survive hibernation, and the timing of mating and colony production). The ability of species to adapt to changes in climate is influenced by their genetic diversity, and whether non-climate related stressors, such as habitat fragmentation, small population size, and disease have already eroded it. Additionally, small and geographically isolated populations of bumble bees, and other eusocial insects, are at an increased risk of extinction due to their population and genetic structure, particularly when combined with environmental stressors. These populations are susceptible to greater genetic loss and low genetic variation, and have reduced evolutionary responses to changes in the environment, such as habitat fragmentation, pesticides, pathogens and parasites, and climate.

We developed four future plausible scenarios to assess the future viability of the YBBB in terms of the 3Rs. These scenarios are:

- Scenario A assumes plausible positive changes in stressor effects to YBBB, where relevant. Under this scenario, the effects of climate change are assumed to have either a negligible or positive effect on YBBB in some northern ecoregions.
- Scenario B is identical to Scenario A with the exception that the effects of climate change are not expected to produce any beneficial effects to the YBBB in the northern ecoregions.
- Under Scenario C, the negative effects of the various stressors are predicted to worsen or remain generally unchanged. The effects of climate change are expected to have a negative effect on the YBBB throughout the species' range with a concurrent decline in colony and population numbers.
- Scenario D models the future resiliency of the YBBB based on projected trends in the existing historical and current survey data. These data are assumed to reflect the effects of the various stressors to the species.

We limited our assessment of the future condition of the YBBB to approximately 20 years, or twenty YBBB colony life-cycles (*i.e.*, generations) because our modeled results become increasingly uncertain after two decades in the future, reducing our confidence in making longer-term projections beyond this timeframe. We also note that the existing historical and current survey data for the species demonstrates relatively rapid changes in the YBBB's occupancy and/or relative abundance in some ecoregions within the last several decades. This suggests that continued changes in the species' viability will also be apparent within the next several decades. Therefore, we consider 20 years to be a reasonable timeframe for assessing the future viability of the YBBB.

Under the generally positive assumptions of Scenario A, YBBB resiliency in three ecoregions is predicted to decline, two ecoregions are predicted to improve, and the rest are expected to remain relatively unchanged. Under Scenario B, which assumes climate change will have little effect on the YBBB in the northern ecoregions, the species' resiliency is expected to decline in three ecoregions and remain unchanged in the remainder. Under Scenario C, which assumes generally negative changes in the stressors, YBBB resiliency will remain relatively unchanged in 1 ecoregion and decline in the remaining 14. Notably, under Scenario C the YBBB is likely to be extirpated from six ecoregions where its resiliency is currently low. Under Scenario D, the extrapolation of trends in YBBB data, resiliency is projected to decline in five ecoregions (with the YBBB likely to be extirpated from one), to improve in two, and to remain relatively unchanged in the remaining eight ecoregions. We note, however, that the Scenario D model may underestimate the effects of some stressors.

Under no future scenario do we predict YBBB representation will improve, and given our assessment that some ecoregions will decline in resiliency even under the optimistic scenarios, representation and redundancy may actually decline too. Also, based on the general trends observed between the historical and current time periods, it is reasonable to conclude that the range of the species will not expand significantly under any future scenario and may in fact continue to contract, especially along the already receding southern boundary. The effect of this range contraction is likely a continued net loss of YBBB populations and overall redundancy.

Table of Contents

Executive Summary	5
List of Figures	10
List of Tables	13
Chapter 1: Introduction and Analytical Framework	15
Introduction	15
Analytical Framework	15
Chapter 2: Species Information	18
Taxonomy	18
Species Description	18
Range	19
Life History (Individual-level Ecology)	20
Habitat Needs	22
Nesting and Foraging Habitat Needs	24
Male and Gyne Foraging, Dispersal, and Mating Habitat Needs	25
Overwintering Habitat Needs	26
Population Needs	27
Chapter 3: Influences on Viability	32
Habitat Loss, Fragmentation, and Degradation	33
Pesticides	36
Pathogens and Parasites	42
Small and Isolated Populations	53
The Effects of Climate Change	56
Scientific Collection	59
Competition	62
Synergistic Effects	65
Beneficial Factors	67
Chapter 4: Past and Current Conditions	70
Chapter 5: Future Conditions	92
Literature Cited	104
Appendices	132
Appendix A: YBBB bee decadal occupancy and relative abundance by U.S. state or Canadian province	133

Appendix B: Trends in YBBB occupancy and relative abundance by level II ecoregion	135
Appendix C: Modeling Yellow Banded Bumble Bee (YBBB) Count Data	138

List of Figures

Figure 1. Species Status Assessment Framework	16
Figure 2. Comparison of adult YBBB forms. Left to right: queen, female worker, and male. (Illustration by Elaine Evans in Evans et al., 2008, p. 13)	19
Figure 3. Current range of the YBBB in blue; historical range shown in gray for comparison purposes. This range map was developed by drawing a 100 km (62 mi) buffer (per Colla et al. 2011) around 2010 to 2017 occurrence points and is generally inclusive of areas between positive occurrence locations. Minor smoothing of some edges was used where we judged appropriate to produce this reasonable approximation of the species' historical range.	20
Figure 4: Life history diagram of the yellow banded bumble bee (<i>Bombus terricola</i>).	20
Figure 5: YBBB foraging	Error! Bookmark not defined.
Figure 6: Influence diagram illustrating how environmental stressors influence YBBB habitat, demographics, growth, fitness, and resiliency	32
Figure 7. Trend in YBBB relative abundance and neonicotinoid application in eight states (MN, WI, MI, NY, MA, VT, NH, and ME) representing the core of the species' historical range in the U.S. (see Sheffield et al. 2016) Neonicotinoid amounts are the combined annual application, based on the 10-year average, of acetamiprid, clothianidin, imidacloprid, thiacloprid, and thiamethoxam (data from the USGS Pesticide National Synthesis Project, 2018).	40
Figure 8. Trend in YBBB relative abundance and glyphosate application in the U.S. The glyphosate figures represent the total active ingredient applied in the U.S. during the given year (from Benbrook 2016); the YBBB percentages are the decadal average for the given year.	41
Figure 9: Three mechanisms of pathogen and parasite infections between populations of managed and wild bees. Arrows indicated the direction of spread due to the mechanism. (From Graystock et al., 2016, p. 66).	43
Figure 10: The impact of multiple pressures (black text) on pollinator species across levels of biological organization (blue text). Black arrows span the levels at which each stressor has direct (solid) and indirect (dotted) effects. Vertical arrows show the most practical scale at which to study interactions between pressures. Green arrow = pesticide–pathogen–nutrition interactions at individual or colony scales; orange arrow = climate change–habitat interactions at population or species scale. (From: Vanbergen et al. 2013)	65
Figure 11. Historical range of the YBBB. This range map was developed by drawing a 100 km (62 mi) buffer (per Colla et al. 2011) around historical occurrence points and is generally inclusive of areas between positive occurrence locations. Minor smoothing of some edges was used where we judged appropriate to produce this reasonable approximation of the species' historical range.	73

- Figure 12. Level II ecoregions of North America (from <http://www.cec.org/sites/default/atlas/map/>, accessed May 7, 2018). 75
- Figure 13. Historical trends in YBBB occupancy and relative abundance across its range. Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of *Bombus* spp. specimens collected that were YBBB. 78
- Figure 14. Total hexagons that were surveyed for *Bombus* spp. and total individual *Bombus* specimens collected by time period. 79
- Figure 15. Current range of the YBBB in blue; historical range shown in gray for comparison purposes. This range map was developed by drawing a 100 km (62 mi) buffer (per Colla et al. 2011) around 2010 to 2017 occurrence points and is generally inclusive of areas between positive occurrence locations. Minor smoothing of some edges was used where we judged appropriate to produce this reasonable approximation of the species' historical range. 80
- Figure 16. Decadal trends in YBBB occupancy and relative abundance across all ecoregions. Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of *Bombus* spp. specimens collected that were YBBB. 83
- Figure 17. Decadal trends in YBBB occupancy and relative abundance within the core of the species' range (ecoregions 3.3, 5.1, 5.2, 5.3, and 5.4). 84
- Figure 18. Logarithmic relationship of current YBBB occupancy and land use across ecoregions. "Percent Developed" is the percentage of land within an ecoregion classified as agricultural and urban/built-up. 86
- Figure 19. Landcover within the current range of the YBBB in 2010. Brown shading indicates agricultural land, red indicates urban or built up land, and gray indicates generally undisturbed or natural land. (Data from: 2010 North American Landcover Dataset and Land Change Monitoring System at <https://landcover.usgs.gov/nalcms.php>). **Error! Bookmark not defined.**
- Figure 20. Methodology for describing YBBB resiliency at the ecoregion level. On the left is an excerpt from Table 7 (see Current Distribution and Status section above) showing the percentage values used to inform the resiliency matrix for ecoregion 5.1, shown at the right. Green shading indicates "high" resiliency, yellow indicates "moderate" resiliency, and red indicates "low" resiliency. In this example the current resiliency of ecoregion 5.1 is "high." 88
- Figure 21. Historical resiliency of the YBBB by ecoregion. Green shading indicates "high" resiliency, yellow indicates "moderate" resiliency, and red indicates "low" resiliency. 90
- Figure 22. Current resiliency of the YBBB by ecoregion. Green shading indicates "high" resiliency, yellow indicates "moderate" resiliency, and red indicates "low" resiliency. Gray

shading represents areas within the historical range of the YBBB where the species is presumed extirpated. 91

Figure 23. Predicted resiliency of the YBBB by ecoregion under Scenario A. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated. 96

Figure 24. Predicted resiliency of the YBBB by ecoregion under Scenario B. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated. 98

Figure 25. Predicted resiliency of the YBBB by ecoregion under Scenario C. Yellow shading indicates “moderate” resiliency, and red indicates “low” resiliency. Cross-hatching indicates ecoregions where the risk of YBBB extirpation is high and gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated. 100

Figure 26. Predicted resiliency of the YBBB by ecoregion under Scenario D. Cross-hatching indicates ecoregions where the risk of YBBB extirpation is high and gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated. 102

List of Tables

Table 1. The ecological requisites for survival and reproductive success of YBBBs (modified from USFWS 2016, p. 16)	27
Table 2. Bee Parasites for which there is evidence of anthropogenic spread to wild bee populations (From: Goulson and Hughes 2015, p. 13)	52
Table 3. U.S. states and Canadian provinces with YBBB records along with the years the species was first reported and last reported. Note that occurrence data for New Brunswick, Nova Scotia, and Prince Edward Island, together the Canadian Maritime Provinces, were combined and used collectively in our analyses.	72
Table 4. Level I and level II ecoregions, U.S. states, and Canadian provinces with YBBB records.	76
Table 5. Historical (1824 to 2009) YBBB occupancy and relative abundance by level II ecoregion. Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of <i>Bombus</i> spp. Specimens collected that were YBBB.	77
Table 6. Historical and current areal extent of YBBB range by level II ecoregion. “Percent of total” is the historical proportion of an ecoregion to the total YBBB historical range. “Percent change” is the change from the historical to the current areal extent.	82
Table 7. Current (2010 to 2017) YBBB occupancy and relative abundance by level II ecoregion. Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of the total number of <i>Bombus</i> spp. Specimens collected that were YBBB. Gray shading indicates ecoregions where the species is presumed extirpated.	82
Table 8. Percent agricultural and urban land cover within the ecoregions occupied by the YBBB (Data from: 2010 North American Landcover Dataset and Land Change Monitoring System at https://landcover.usgs.gov/nalcms.php).	85
Table 9. YBBB population estimates by time period and level II ecoregion. Estimates assume: 1) that a YBBB occurrence in a surveyed 100 sq km (38.6 sq mi) hexagon represents a population; 2) that surveyed hexagons are randomly distributed; and 3) land not classified as agricultural or urban is suitable habitat for the species. “Occupiable Hexagons” is the approximate number of hexagons potentially suitable for the species. For the historical period, this includes all land within an ecoregion, for the current period agricultural and urban land is excluded. “Estimated populations” is occupiable hexagons multiplied by YBBB occupancy. “Percent Change” is the change in estimated populations from the historical to the current time periods.	87

Table 10 Historical and current resiliency of YBBB populations by ecoregion. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency.	89
Table 11. Criteria for assessing potential YBBB stressors under Scenarios A, B, and C.	93
Table 12. Stressor assessment under Scenario A. Green shading indicates an assumption that conditions change favorably for the YBBB relative to the current condition; red shading indicates that conditions are predicted to worsen for the YBBB, and unshaded cells indicate no significant change in the stressor from the current condition. Cross hatching indicates that the YBBB is extirpated from that ecoregion.	94
Table 13. Predicted resiliency of YBBB by ecoregion under Scenario A with historical and current resiliencies provided for comparison. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency	95
Table 14. Stressor assessment under Scenario B. Green shading indicates an assumption that conditions change favorably for the YBBB relative to the current condition; red shading indicates that conditions are predicted to worsen for the YBBB, and unshaded cell indicate no significant change in the stressor from the current condition. Cross hatching indicates that the YBBB is extirpated from that ecoregion.	97
Table 15. Predicted resiliency of YBBB by ecoregion under Scenario B with historical and current resiliencies provided for comparison. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency.	97
Table 16. Stressor assessment under Scenario C. Red shading indicates that conditions are predicted to worsen for the YBBB and unshaded cells indicate no significant change in the stressor is expected. Cross hatching indicates that the YBBB is extirpated from that ecoregion.	99
Table 17. Predicted resiliency of YBBB by ecoregion under Scenario C with historical and current resiliencies provided for comparison. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Cross hatching indicates ecoregions where the risk of YBBB extirpation is high.	99
Table 18. Predicted resiliency of YBBB by ecoregion under Scenario D with historical and current resiliencies provided for comparison. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Cross hatching indicates ecoregions where the risk of YBBB extirpation is high.	101
Table 19. Comparison of YBBB resiliency by ecoregion. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Cross-hatching indicates ecoregions where the risk of YBBB extirpation is high.	103

Chapter 1: Introduction and Analytical Framework

Introduction

This report summarizes the results of a species status assessment (SSA) conducted for the yellow banded bumble bee (*Bombus terricola*), hereafter referred to as the YBBB. In 2015, we, the U.S. Fish and Wildlife Service (Service), received a petition to list the YBBB as an endangered or threatened species, and to designate critical habitat, under the Endangered Species Act of 1973, as amended (Act) (Defenders of Wildlife 2015, entire). In March of 2016, the Service found that the petition presented substantial scientific or commercial information indicating that the listing of the YBBB may be warranted (81 FR 14058). Thus, we conducted a SSA to compile the best scientific and commercial data available regarding the species' biology and factors that influence the species' viability.

Analytical Framework

The SSA report, the product of conducting a SSA, is intended to be a concise review of the species' biology and factors influencing the species, an evaluation of its biological status, and an assessment of the resources and conditions needed to maintain long-term viability. The intent is for the SSA report to be easily updated as new information becomes available, and to support all functions of the Endangered Species Program. As such, the SSA report will be a living document upon which other documents, such as listing rules, recovery plans, and 5-year reviews, would be based if the species warrants listing under the Act.

This SSA report for the YBBB is intended to provide the biological support for the decision on whether or not to propose to list the species as threatened or endangered and if so, whether or not to propose designating critical habitat. The process and this SSA report do not represent a decision by the Service whether or not to list a species under the Act. Instead, this SSA report provides a review of the best scientific and commercial information available strictly related to the biological status of the YBBB. The listing decision will be made by the Service after reviewing this document and all relevant laws, regulations, and policies, and a decision will be announced in the Federal Register.

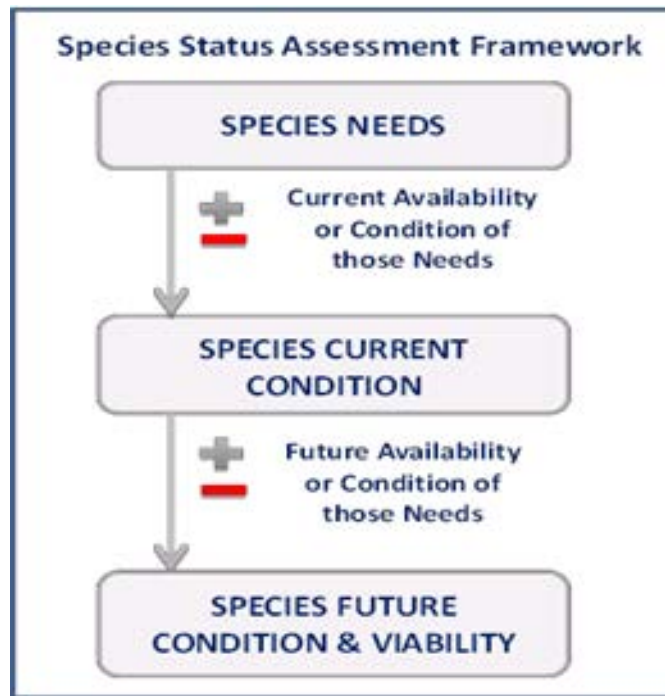


Figure 1. Species Status Assessment Framework

Using the SSA framework (figure 1), we consider what a species needs to maintain viability by characterizing the biological status of the species in terms of its resiliency, redundancy, and representation (together the “3Rs”) (Shaffer *et al.*, 2002, pp. 139–140; Wolf *et al.* 2015, entire; Smith *et al.* 2018, entire). For the purpose of this assessment, we generally define viability as the ability of the species to sustain populations in natural ecosystems within a biologically meaningful timeframe: in this case, twenty years. This represents twenty YBBB colony life cycles, and the available data suggest recent, significant changes in the species’ status within a similar timeframe. Twenty years is also a period that allows us to reasonably predict the potential effects of the various stressors within the range of the species. This is also consistent with the time scale for which we have previous data available on the species.

The 3 Rs are defined as follows:

Resiliency means having sufficiently large populations for the species to withstand stochastic events (arising from random factors). We can measure resiliency based on metrics of population health, such as population size, if that information exists. Resilient populations are better able to withstand demographic stochasticity (e.g., random fluctuations in birth rates), environmental stochasticity (e.g., annual variations in rainfall), natural disturbances, and the effects of human activities.

Redundancy means having a sufficient number of populations for the species to withstand catastrophic events (such as a rare destructive natural event or episode involving many populations). Redundancy is about spreading the risk and can be measured through the duplication and distribution of populations across the range of the species. Generally, the greater the number of populations a species has distributed over a larger landscape, the better it can withstand catastrophic events.

Representation means having the breadth of genetic makeup of the species to adapt to changing environmental conditions. Representation can be measured through the genetic diversity within and among populations and the ecological diversity (also called environmental variation or diversity) of populations across the species' range. The more representation, or diversity, a species has, the more it is capable of adapting to changes (natural or human caused) in its environment. In the absence of species-specific genetic and ecological diversity information, we evaluate representation based on the extent and variability of habitat characteristics within the geographical range.

The decision whether to list a species is based *not* on a prediction of the most likely future for the species, but rather on an assessment of the species' risk of extinction. Therefore, to inform this assessment of extinction risk, we describe the species' current biological status and assess how this status may change in the future under a range of scenarios to account for the uncertainty of the species' future. We evaluate the current biological status of the species by assessing the primary factors negatively and positively affecting the species to describe its current condition in terms of the 3Rs. We then evaluate the future biological status by describing a range of plausible future scenarios representing a range of conditions for the primary factors affecting the species and forecasting the most likely future condition for each scenario in terms of the 3Rs. As a matter of practicality, the full range of potential future scenarios and the range of potential future conditions for each potential scenario are too large to analyze and describe them individually. These scenarios do not include all possible futures, but rather include specific plausible scenarios that represent examples from the continuous spectrum of possible futures.

Chapter 2: Species Information

Taxonomy

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, Williams *et al.* 2008, entire; Williams *et al.*, 2014, entire; Koch and Strange 2009, p. 98). There are approximately 250 bumble bee species worldwide, 200 of which are considered to have highly social behavior and colony structure comprised of adults and juveniles in which the adults each have distinct characteristics, roles, and responsibilities. In the United States and Canada, there are approximately 46 native bumble bee species. Forty bumble bee species are grouped into 8 sub-genera. The remaining 6 bumble bee species are cuckoo bumble bees (subgenus *Psithyrus*) (Williams *et al.* 2008, pp. 49-50; Williams *et al.*, 2014, entire).

The YBBB (*Bombus terricola*, Kirby 1837) belongs to the distinct subgenus *Bombus* sensu stricto (*Bombus* s. str.), which in North America includes four additional bumble bee species: Franklin's bumble bee (*B. franklini*), the rusty patched bumble bee (*B. affinis*), the western bumble bee (WBB)(*B. occidentalis*), and the cryptic bumble bee (*B. cryptarum*) (Williams *et al.* 2008 p. 53; Williams *et al.* 2012, p. 10).

Yellow banded bumble bees and WBBs can be similarly colored in parts of their ranges where their ranges overlap; however, studies using mitochondrial COI-barcoding, morphological, and biogeographical analysis confirmed they are two separate species (Sheffield *et al.* 2016, p. 2-3, Williams *et al.* 2012, p. 19). Their geographic ranges meet at the Rocky Mountains (Bertsch *et al.* 2010, p. 232, Sheffield *et al.* 2016, p. 5). Yellow banded bumble bee and WBB have been found to overlap in the two northeastern-most counties of Montana, and they are distinguishable by different color patterns of yellow hairs on the second segment (Dolan, *et al.* 2017, pp. 137-38, 141; Dolan, 2016, p. 148). The two species (YBBB and WBB) do have different distributions and for the most part occupy different ecozones: YBBB has a Boreal distribution, and the WBB has a Cordilleran distribution. They share the same habitat area in the mountains and plateaus of eastern British Columbia and the plains of southern Alberta and Saskatchewan (Sheffield *et al.* 2016, entire). The WBB is found throughout British Columbia, and in the south spills east of the mountains at least as far as Regina, Saskatchewan (Sheffield *et al.* 2016, entire; S. Cannings, pers. comm., 2018; C. Sheffield, pers. comm., 2018).

The best available information indicates the YBBB is a unique species and valid taxon (Bertsch *et al.* 2010, pp. 239-240; Williams *et al.* 2012, pp. 9-19; Owen and Whidden 2013, pp. 335-342; Sheffield *et al.* 2016, pp. 3-9) and the species is listed as a valid taxon in the Integrated Taxonomic Information System (ITIS) database (ITIS, 2017).

Species Description

Adult YBBBs have black hairs on their heads. Queens, workers, and males have a similar hair color pattern. The front of the thorax, and second and third segments have all yellow hairs. Their wings are slightly brown, and the hair on the legs and base of the abdomen are black.

Queens and workers have a similar distinct fringe of brownish-yellow hair on the fifth segment of the abdomen. Males have long yellow hair on top of the head and on the face, as well as a fringe of black-yellow hair on the fifth segment of the abdomen (figure 2). Queens are the largest bees in a colony and range from 17-19 millimeters (mm) (0.67-0.75 inches (in)) in length with an abdomen width of 9-10 mm (0.35-0.39 in). Female workers are 9-14 mm (0.35-0.55 in) in length with an abdomen width of 5-7 mm (0.20-0.28 in). Males are 13-17 mm (0.51-0.67 in) in length with abdomens 6-8 mm (0.24-0.31 in) in width (Mitchell 1962, pp. 519-520; Evans *et al.* 2008, p. 13).



Figure 2. Comparison of adult YBBB forms. Left to right: queen, female worker, and male. (Illustration by Elaine Evans in Evans *et al.*, 2008, p. 13)

Range

Most bumble bees have large geographic ranges, including the YBBB. Bumble bees originated and diversified in cool temperate latitudes and are a cold adapted, largely alpine group. While bumble bees are most diverse in temperate and montane regions of the world, their distribution expanded millions of years ago to colonize areas throughout the Northern Hemisphere, including the Arctic tundra, deserts, and subtropical forests. In the southern hemisphere native bumble species are found in regions of South America (Hines 2008, entire; Williams *et al.* 2014, p. 9-12)

The range of YBBB includes Newfoundland and Labrador and the Eastern Temperate and Boreal forest regions, south along higher elevations in the Appalachian Mountains, west through North Dakota and the Canadian Great Plains, to the Tundra and Taiga of Canada and the Mountain West, and in British Columbia (figure 3; Williams *et al.* 2014, p. 113; Hatfield *et al.*, 2015, entire; IUCN Red List 2018). Please see Chapter 4 – *Past and Current Conditions* for a more thorough discussion of the YBBB's range.

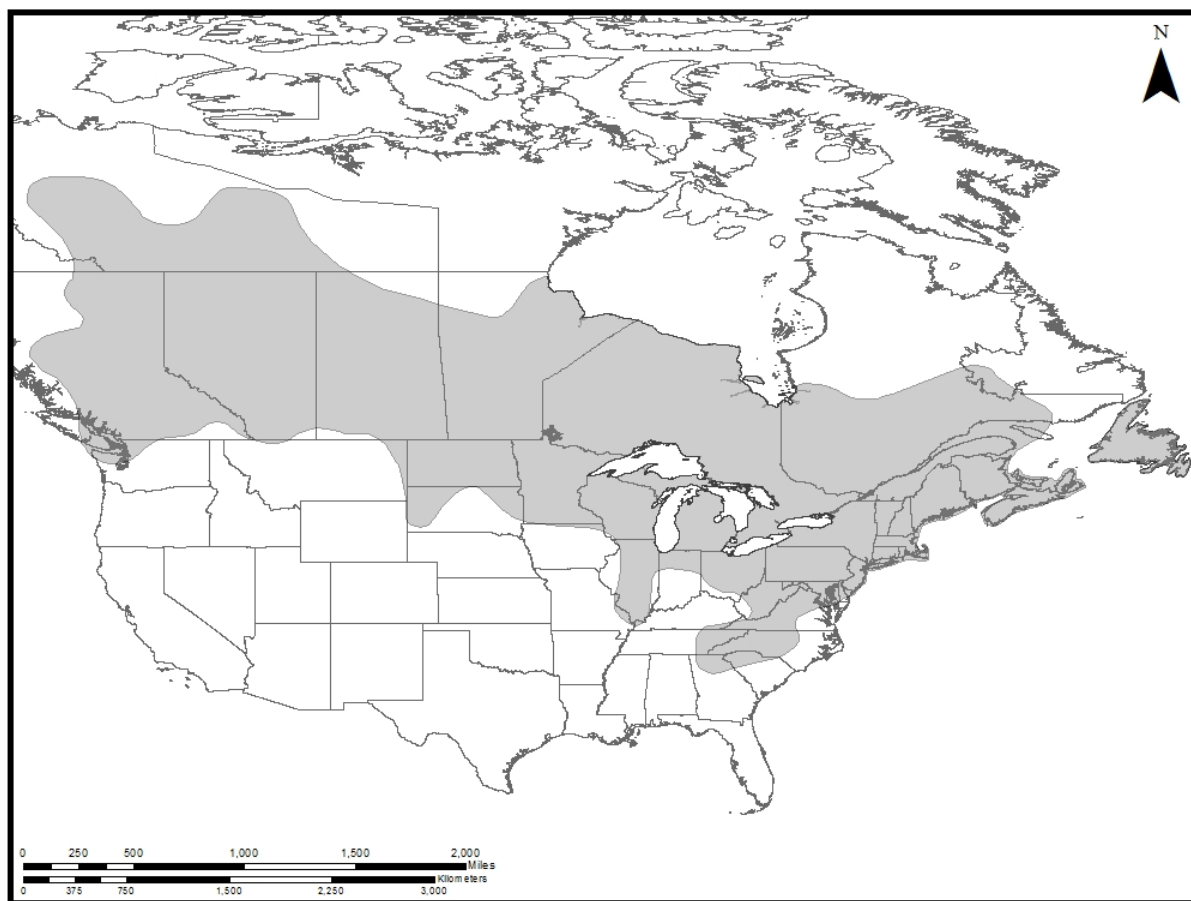


Figure 3. Historical range of the YBBB in gray. This range map was developed by drawing a 100 km (62 mi) buffer (per Colla *et al.* 2011) around historical occurrence points and is generally inclusive of areas between positive occurrence locations. Minor smoothing of some edges was used where we judged appropriate. Some areas with sparse survey data were included based on the professional judgement of bumble bee researchers with local knowledge.

Life History (Individual-level Ecology)

The YBBB is a highly social (eusocial) insect that lives in colonies that include a queen (or “foundress”), worker bees (sterile females), and reproductives (new queens or “gynes” and fertile male bees). Eusocial animals share three characteristics: not all individuals reproduce, there is cooperative care of brood and juveniles, and there is an overlap of generations that allows for older generations to help care for younger generations (Wilson 1971, pp. 4-5). Specifically, the queen is the mother and workers are daughters and there is reproductive division of labor, cooperative care, and one generation caring for the next, and the reproductives (new gynes and males) are siblings to the workers (J. Gibbs pers. comm 2018). The colonies are annual. and The founding queen, female workers, and males all die in the late summer or fall; only the mated gynes hibernate by overwintering in a state of dormancy (diapause) (figure 4). Yellow banded bumble bee colonies are medium-sized, producing an average of 150-430 workers, relatively few males, and 0 to 73 gynes in a single reproductive season (Evans 2018 pers comm., Koch and Strange 2009, p. 100; Macfarlane *et al.* 1994, p. 2).

The life cycle of the YBBB begins when the queen emerges from her underground overwintering site in the early spring. She forages at flowers to rebuild her body fat reserves and locates a suitable nest site, typically an underground hole such as an abandoned rodent nest. Some bumble bee queens (Vosnesensky bumble bee, *B. vosnesenskii*) have been found to exhibit nest fidelity, preferring to nest within 1 to 9 km (0.62-5.6 miles) of their natal colonies (Jha and Kremen 2013, pp. 2490-2493). The queen may re-arrange any existing rodent nesting materials (e.g., moss, grass, hair) to form a brood chamber. In the brood chamber the queen constructs a wax nectar pot in which she stores nectar to feed herself while incubating eggs and multiple wax cups to store pollen and nectar that will feed her future larvae. The queen lays multiple (up to twelve) fertilized eggs on top of a mass of pollen mixed with nectar and covers it with wax and then sits on the 'brood clump' to maintain the temperature at approximately 30 degrees Celsius (°C) (86 degrees Fahrenheit (°F)). In colder temperatures, bumble bees generate heat by shivering their flight muscles to keep the brood adequately warm. The eggs hatch after approximately four days and the larvae feed on the pollen mixed with nectar on which they were laid. As the larvae develop, the queen continues to guard the nest, care for the larvae, and forage for nectar for herself and for pollen and nectar to feed the growing larvae (Alford 1970, E. Evans, pers. comm., 2018; Goulson, 2010, pp. 5-6; Husband 1980 pp. 225-226; Williams *et al.* 2014, pp. 12-13, 113).

During the period that the larvae develop, they are reliant on an adult bee to feed them directly (Williams *et al.* 2014, p. 13). As the larvae grow, they spread out from the brood clump in the nest and spin silk cocoons in which they live until they pupate and emerge as adult female workers approximately 10 to 14 days later (Alford, 1970, Williams *et al.* 2014 pp. 12-13, Goulson 2010, p. 5-9). The life expectancy of newly hatched YBBB workers is estimated at 13.2 days (Rodd *et al.* 1980, p. 1719).

After the first batch of worker bees have emerged from the nest (approximately 3 to 4 weeks after the colony is founded), the queen remains in the nest to lay eggs and care for the young. The new workers divide the colony duties, with some bees predominantly foraging for pollen and nectar while others help the queen tend the developing brood and guard the nest.

As mid to late summer arrives, the queen begins to lay unfertilized eggs that develop into fertile males and fertilized eggs that develop into fertile female larvae. These female larvae develop into gynes (potential reproductive females) as a result of receiving more food over a longer period of time and by receiving pheromonal cues from the queen (Goulson 2010, pp. 8, 24-25; Williams *et al.* 2014, p. 14). Once the gynes leave the nest, they forage heavily at flowers to build up significant fat reserves to survive overwintering. Based on studies of a closely related species, the buff tailed bumble bee, the gynes and males from individual colonies will disperse up to 10 kilometers (km) (6.2 miles (mi)) to mate (Lepais *et al.* 2010, pp. 826-829; Kraus *et al.* 2009, p. 249), and each queen mates only once with one male (Goulson 2010, p. 11, Owen and Whidden 2013, p. 524, 526). The newly-mated, potential new queens continue foraging until locating a suitable overwintering site. The founding queen, workers, and males from the original colony all die off in the fall and early winter.

The YBBB is one of the first bumble bee species to emerge each spring. They are known to be active and fly when air temperatures are near or below 0° C (32° F) (Heinrich 1972, entire). When temperatures increase and exceed 30° C (86° F), their activity declines and they may be susceptible to overheating, even in moderately warm conditions (Heinrich 1976a, pp. 882, 886; D. Goulson, pers.comm., 2018; Oyen *et al.* 2016, p. 55). Queens are observed April through September, workers mid-May to September, and males mid-June to September or October (Colla and Dumesh 2010, pp. 42, 53; Plath 1922, pp. 192-193). Historical observations have recorded queens in early to mid-April in Massachusetts and Michigan. The earliest workers have been observed in mid-May in Massachusetts and in mid-June in Michigan. In Michigan, the first males were observed in early July, but most of the gynes and males are presumed to be produced primarily July and August (Plath 1922, p.192; Husband 1980, pp. 226-227). In southwestern Ontario, one YBBB nest in a man-made wooden hive was reported starting on May 28 and brood rearing finished on July 29. The maximum number of worker bees were observed on August 19, with the worker and male bee numbers declining rapidly in late August. The colonies were terminated by September 9 (Macfarlane 1994, p. 5).

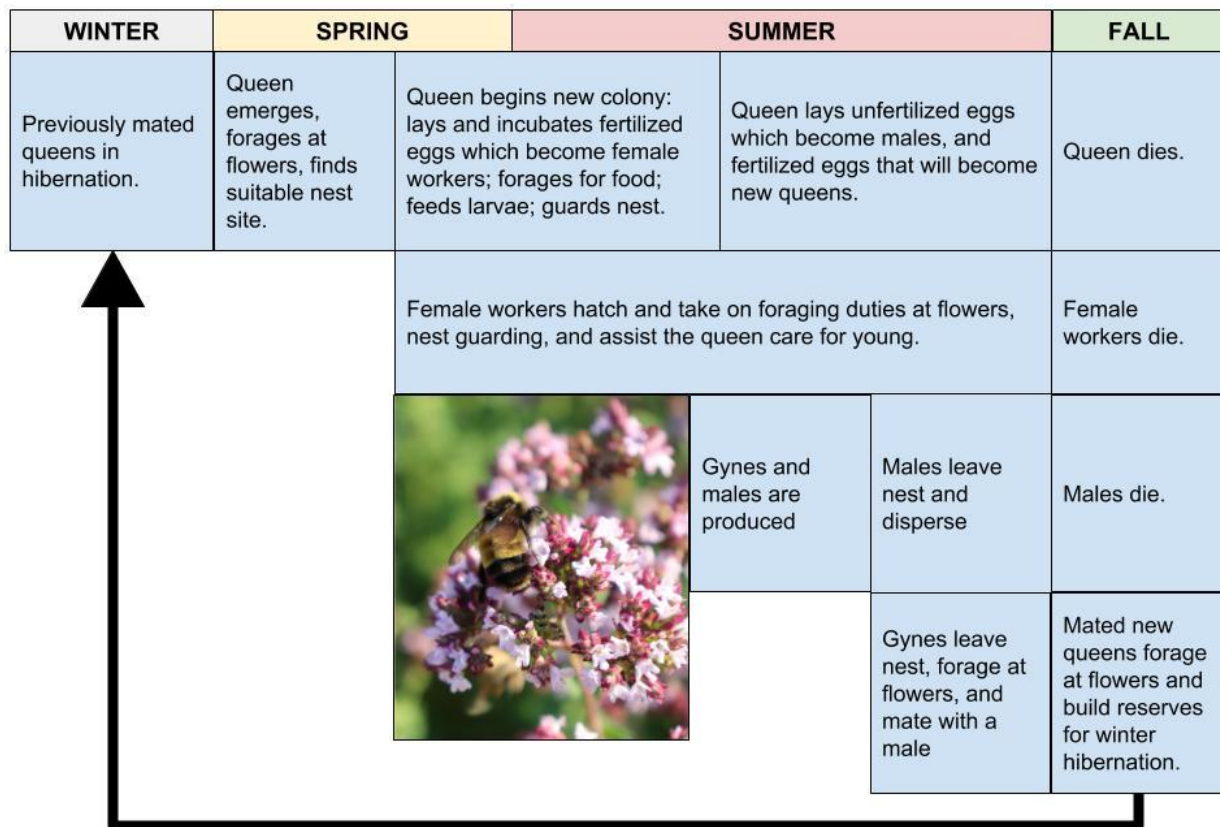


Figure 4: Life history diagram of the yellow banded bumble bee (*Bombus terricola*).
Photo credit: Molly Jacobson

Habitat Needs

The principal habitat of the sub-genus *Bombus* s. str. has been identified as alpine/arctic; montane/temperate forest; and prairie/steppe habitat (Williams *et al.* 2008, p. 55). Bumble bee species diversity is highest in cool temperate and montane situations (Williams *et al.* 2014, pp. 9-11). Early-spring emerging bumble bees, such as the YBBB, are associated with mixed and woodland habitats and wetlands, and likely have co-evolutionary relationships with early spring woodland ephemeral plants (Colla and Dumesh, 2010, pp. 42, 48; Colla 2016, pp. 413, 416, 421; Williams *et al.* 2014, p. 113). Examples of natural environments where yellow banded bumble bees have been observed are close to or within a variety of forest types, including riparian woodland, mature deciduous and conifer forests, and treeline conifer forest. They also utilize wetlands, undisturbed bogs, blueberry barrens alpine tundra, and prairies (Batra 1993, pp. 125-126; Heinrich 1976a, p. 874; Williams *et al.* 2014, p. 113, Koch and Strange 2009, p. 100).

The yellow banded bumble bee, like most *Bombus* species, can utilize a variety of floral resources in various habitats depending on availability and time of year. Their preferences based on their specific life history, nutritional requirements, and evolutionary history are largely unknown, however these specific preferences are documented to exist in other 'generalist' *Bombus* species that have been researched (see *Habitat Needs, Habitat Loss, and Effects of Climate Change* sections), and in all likelihood exist for YBBB but are yet to be researched and described. Williams *et al.* 2014 (pp. 111-122) describes YBBB habitat as primarily "close to or within wooded areas and wetlands", unlike the four other North American species that are in the same sub-genus. The RPBB habitat is described as close to or within woodland, urban parks, and gardens. The WBB habitat is primarily open grassy areas, urban parks, and gardens, chaparral and shrub areas, and mountain meadows. Franklin's bumble bee habitat was open grassy coastal prairies and the Cryptic bumble bee habitat is Tundra/Taiga (Williams *et al.* 2014, pp. 111-122).

Bumble bee tongue length and body-size play a role in governing food-plant preferences. The YBBB is a short-tongued species that forages from a variety of flowers with short, shallow corollas (figure 5). While they may not forage as easily at flowers with long corollas, they have been observed piercing the lower corolla to extract nectar (Medler, 1962, p. 217; Heinrich 1976a, pp. 875-882; Koch and Strange 2009, p.99).



Figure 5: Yellow banded bumble bee foraging
Photo credit: Leif Richardson

While bumble bees forage at a variety of flower species, research has demonstrated that in order to forage efficiently they must rely on individual experience and memory. Bumble bees gather

information from the environment, learn, and use memory to increase their foraging efficiency. Foragers return to the nest with the odor of a newly discovered food source and for several minutes run across the nest, fan their wings, and bump into nestmates releasing pheromones. The floral odor and pheromonal signal stimulates new recruit foraging activity for the floral resources carried by the successful forager. Because the bumble bee recruitment system does not include the precise location of the food, they must rely on their memory and searching ability to find the food source (Colla 2016, p. 413; Dornhaus and Chittka 1999, p. 38; Dornhaus and Chittka 2001, 570-575; Dornhaus and Chittka 2003, pp. 183, 185-188; Durisko *et al.* 2011, pp. 52-54; Townsend-Mehler and Dyer 2012, pp. 277, 282; Chittka and Thomson 1997, pp. 396-397).

Nesting and Foraging Habitat Needs

Nest sites are located underground in downward sloping tunnels and typically in a pre-existing burrow such as an abandoned rodent nest (Hobbs, 1968, p. 156-164; Williams *et al.* 2014, pp. 13, 21, 113). Nesting habitat may be a limiting factor for bumble bees due to long search times required to locate suitable sites, low levels of natural sites, niche overlap with other bee species, and high frequency of nest usurpation by other bees (Hines and Hendrix 2005, p. 1481; Richards 1978, p. 315). Additionally, bumble bee queens that exhibit nest fidelity, such as the Vosnesensky bumble bee, may be limited by the availability of suitable habitat (Jha and Kremen 2013, p. 2493). The nest needs to remain undisturbed until late summer until after the reproductive bees (gynes and males) leave the nest to forage, mate, and locate suitable overwintering habitat. 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 (Hines and Hendrix 2005, p. 1483) due to the presence of abandoned rodent nests and undisturbed habitat with diverse floral resources.

Foraging habitat with abundant and diverse floral resources across the landscape is required over the entire growing season, from early spring through early fall. While the YBBB forages on a variety of plant species, the availability of floral resources can fluctuate throughout the season as various plant species bloom and diminish. The YBBB requires nectar and pollen continuously throughout the season to support the colony and because there is limited storage capability in the nest. The food is collected, returned to the nest, and consumed by the colony (Colla and Dumesht 2010, p. 48; Heinrich 1976a, p. 887; Williams *et al.* 2014, pp. 15-16).

In the early spring, founding queens need ample pollen and nectar after emerging from hibernation to restore energy reserves to succeed in locating a nest site, founding a new colony, and foraging for the colony until the first worker bees emerge. As one of the earliest spring bumble bee species to emerge from winter hibernation, YBBBs utilize woodland ephemerals and early blooming bushes and trees (Colla and Dumesht 2010, pp. 42, 48). The queens appear on early blooming plants, such as *Salix* sp. (willow), which can bloom before the last snowfall of the year and before all of the winter snow has melted (Heinrich 1976a, p. 876; Husband *et al.*, 1980 pp. 225-226). They also forage on early spring blooming trees such as cherry (*Prunus*), maple (*Acer*), alder (*Alnus*), poplar (*Populus*), willow (*Salix*), horse chestnut (*Aesculus*), redbud (*Cercis*), and sassafras (*Sassafras*) (Vaughan and Black 2006, p. 3).

The YBBB may require floral resources that are close to the nest site; studies of other *Bombus* species indicate foraging distances are typically less than 1 km (0.62 miles) from nest sites (Knight *et al.* 2005, p. 1816; Wolf and Moritz 2008, p. 422; Dramstad 1996, pp. 170-182; Osborne *et al.* 1999, pp. 524-526; Rao and Strange 2012, pp. 909-913). This could be especially true in the spring while the queen alone is establishing a new colony and caring for the brood. In the late summer and fall, colonies produce the reproductive bees (gynes and males). Access to abundant and diverse floral resources is particularly important at this time because it influences the size and fitness of the last cohort of offspring in the colony, including fertile males and new queens needed to ensure the founding of new colonies in the following spring (Hatfield and LeBuhn 2007, pp. 156-157).

Colony growth can be significantly affected by pollen type (plant species) and the varying nutritional quality and quantity (Colla *et al.* 2016, pp. 413, 417). Greater pollen diversity and availability have been linked to increased larval growth. When there is constant food availability, the size of individual bees and colonies increases (Colla 2016, p. 413; Tasei and Aupinel 2008, pp. 401-406). Larger bees increase colony survival because they are more efficient at foraging and thermoregulation. Larval queens require more food to develop than workers and larger hibernating queens have been found to have higher winter survival rates (Sutcliffe and Plowright 1990, pp. 1120-1123). The egg laying rate of queens in well provisioned colonies is higher than in colonies with limited pollen resources (Burns 2004, p. 151). Pollen-deprived colonies experience nutritional stress that delays larval bee development and produces smaller bees over the course of an annual colony cycle. This can lead to a shortage of adult worker bees that cannot forage as efficiently as larger worker bees due to their smaller body size (Sutcliffe and Plowright 1988, p. 1056). Smaller bees have less body surface area on which to collect pollen and smaller pollen baskets on their hind legs, as well as lower energy reserves than larger bees. Nutritional stress can affect memory and learning and lead to reduced foraging efficiency, increased competition, and overall decline in colony fitness (Colla *et al.* 2016, p. 413) (see *Chapter 3 - Synergistic Effects* section regarding nutrition).

Male and Gyne Foraging, Dispersal, and Mating Habitat Needs

Mating habitat occurs within foraging habitat areas that are connected across the landscape. Gynes and males typically disperse from the natal area prior to mating to avoid sibling mating (Darvill *et al.* 2012, pp. 3988-3989). In healthy populations, two mechanisms in bumble bees that minimize inbreeding are the dispersal of males and potential new queens (reproductives) from the natal area prior to mating, and the forced dispersal of young males (4-5 days old) prior to gynes from the colony by the worker bees to avoid sibling mating (Darvill *et al.* 2012, pp. 3988-3989).

The gynes and male dispersal distance for YBBBs is estimated to be between 1 and 10 km (0.6 and 6.2 miles) based on closely related species in the same subgenus (buff tailed bumble bee, *Bombus terrestris* and rusty patched bumble bee, *B. affinis*) (Lepais *et al.* 2010, p. 829; Kraus *et al.* 2009, pp. 249-251; USFWS 2016, p. 11). They typically disperse in late summer-early fall and the gynes require areas rich with floral resources to build up sufficient fat reserves to survive the winter. When males disperse to forage for themselves and to search for mates, they scent-mark locations (with pheromones) along a flight circuit and then repeatedly patrol the circuit for

gynes (O'Neill *et al.* 1991, p. 604; Jennersten *et al.* 1991, pp. 322-323; Williams *et al.* 2014, pp. 14-15, 113). YBBBs are monoandrous; the queen mates with a single male instead of multiple males (Darvill *et al.* 2006, p. 602; Owen and Whidden 2013, p. 525).

Overwintering Habitat Needs

The YBBB's overwintering habitat is an underground burrow that the new queen digs to an adequate depth and may be located close to wooded areas due to their early spring emergence and reliance on woodland spring ephemeral flowers; however, overwintering preferences vary between bumble bee species (Alford 1969, p. 169; Goulson 2010, p. 11). The site needs to remain undisturbed from late fall through the spring while the queen is in hibernation. Queens of another ground hibernating bumble bee species (red tailed bumble bee, *Bombus lapidarius*) have been observed locating overwintering habitat by hovering over the ground and then landing in various places and scratching at the surface of the ground. They begin excavating a new burrow by using their mandibles and legs to dig and move loose soil out of the way. The amount of time it takes and depth to which they dig depends on the soil type. If the site is not suitable, and a rock or root is encountered, she will abandon that burrow and begin digging another until she has a suitable small oval cavity (Alford 1969, p. 158; Goulson 2010, p. 11).

For queens to survive overwintering, the sites need to be accessible to large quantities of pollen and nectar in the fall because substantial body fat reserves are needed for hibernation (Goulson 2010, pp. 10-11). Approximately 80 percent of fat stored in queens is used during the first half of the hibernation period (Alford 1969, pp. 158-159). Queens without suitable fat reserves will die before spring emergence and it is likely one of the most important requirements for survival. A closely related species in the same subgenus (buff tailed bumble bee) requires a critical wet weight of about 0.6 grams (g) (0.02 ounces) prior to hibernation (Beekman *et al.* 1998, pp. 207, 210).

Overwintering sites close to early spring floral resources are critical for newly emergent queens to restore body fat reserves rapidly and for adequate energy to locate a suitable nest site and establish a new colony (Alford 1969, pp. 158-159; Goulson 2010, pp. 10-11; Williams *et al.* 2014, p. 113). Table 1 summarizes the ecological requirements for the YBBB.

Table 1. The ecological requisites for survival and reproductive success of YBBBs (modified from USFWS 2016, p. 16)

Life Stage	Winter	Spring	Summer	Autumn
Queen		Diverse and abundant floral resources; suitable nesting habitat in the vicinity of previous overwintering habitat	Diverse and abundant floral resources; suitable nesting habitat	Diverse and abundant floral resources; suitable nesting habitat
Worker females		Diverse and abundant floral resources in proximity to nesting habitat	Diverse and abundant floral resources in proximity to nesting habitat	Diverse and abundant floral resources in proximity to nesting habitat
Males			Diverse and abundant floral resources; suitable mating habitat	Diverse abundant floral resources; suitable mating habitat
Gynes (new foundress queens)	Suitable overwintering habitat		Diverse and abundant floral resources; suitable mating habitat	Diverse and abundant floral resources; suitable mating habitat; suitable overwintering habitat

Population Needs

In the absence of population information specific to the YBBB, we relied on the best available information from the closely related rusty patched bumble bee (RPBB) as a surrogate. The following section is summarized from the RPBB SSA report (USFWS 2016, pp. 17-21).

To support population viability, the YBBB requires healthy demographics and sufficient habitat that is connected at the landscape level. The population structure of YBBB operates similarly to a metapopulation (i.e., an assemblage of connected populations). A population of YBBB is a collection of colonies connected by suitable habitat within a 1-10 km radius of a colony and overlap with other adjacent populations. A colony represents one reproductive unit, founded by a single queen. The number of successful colonies, not individuals, which occur within a given geographic area (USFWS 2016, p. 17), determines the population size.

The number of YBBB colonies required to ensure long-term persistence of the population is unknown and likely varies across spatial scales. Small populations are inherently more vulnerable to extirpation due to environmental and demographic stochasticity (Goulson and

Darvill 2008, pp. 197-198); generally, the larger the population the higher the likelihood of persistence over time (Hanski 1999, p. 36). The number of colonies in a population is determined by the number of queens, which is determined by the number of mated gynes. The number of mated gynes and their overwinter survival is influenced by habitat quality and quantity, specifically, the quality and density of floral resources and the proximity of these resources to nest sites and overwintering sites (USFWS 2016, p. 17).

As previously mentioned, YBBB colonies are considered medium-sized, producing an average of 390 workers and males and 32 gynes in a single reproductive season (Koch and Strange 2009, p. 100; Macfarlane *et al.* 1994, pp. 3-4). Populations are composed of many colonies; habitat quality and quantity must be sufficient to support many colonies in a given area to, for example, facilitate mate-finding.

In addition to habitat availability, the number of mated gynes, and hence the number of colonies, is also influenced by the number of fertile males and whether the landscape matrix is conducive to dispersal of reproductives (USFWS 2016, p. 17). YBBB founding queens typically forage up to approximately 1 km (0.62 miles) from nest sites, based on other *Bombus* species (Knight *et al.* 2005, p. 1816; Wolf and Moritz 2008, p. 422; Dramstad 1996, pp. 170-182; Osborne *et al.* 1999, pp. 524-526; Rao and Strange 2012, pp. 909-913). Based on studies of a closely related species, the buff tailed bumble bee, gynes and males can disperse up to 10 km (6.2 miles) in order to mate, therefore the landscape must be permeable, diverse and with high connectivity in order for unrelated gynes and males to successfully find suitable habitat and to mate with each other (Lepais *et al.* 2010, pp. 826-829; Kraus *et al.* 2009, p. 249). Local dispersal patterns of bee can be affected by topographic features and human-altered habitat (Jha and Kremen 2013, p. 2484). The hazards of human development: urban and suburban areas and large mono-cultural agricultural lands are that they can limit the dispersal of bees due to the lack of suitable habitat. Specifically, increased impervious surfaces reduce available ground nesting sites for queens, limit foraging habitat, and therefore may cause a reduced dispersal bees from colonies for mating (Jha and Kremen 2013, p. 2490-2492). Thus, connectivity among colonies is essential for successful recruitment of next year's queens, and therefore, is influential in determining population size.

Population size also affects population viability through genetic health (heterozygosity). Small populations have lower levels of genetic diversity (heterozygosity), which reduces the capacity of a population to respond to environmental change and may lead to reduced population fitness, such as longevity and fecundity, via inbreeding depression (Darvill *et al.* 2006, p. 608). Populations of monoandrous (colonies headed by a single queen who mates with a single male) social species, such as YBBB, are especially vulnerable to inbreeding depression, because the rate of genetic drift in a population is determined by the effective population size, which is much lower than the number of individuals occupying an area (Goulson and Darvill 2008, pp. 197-198; Darvill *et al.* 2006, p. 602; USFWS 2016, p. 17). A recent genetic study found signs of YBBB inbreeding and major reductions in the effective population size of YBBBs in eastern Canada, which increases their extinction risk due to the diploid male vortex (higher numbers of infertile males instead of workers and gynes) (Kent *et al.* 2018, entire) (see *Chapter 3 - Small and Isolated Populations*).

Climatic conditions affect the availability of requisite resources, and hence, bumble bee numbers. Pollen and nectar availability, especially in spring and fall when floral resources are scarcer, are influenced by environmental conditions (Holm 1966, pp. 156-157); in years with unfavorable weather, the supply of food is limited, leading to smaller and fewer colonies. Thus, population viability requires occupying areas with a diversity of environmental conditions (spatial heterogeneity) to ensure floral resources are available throughout the season and year-to-year despite variations in climatic variables, such as temperature and precipitation. Similarly, spatial heterogeneity increases the likelihood of asynchrony among colonies, a pre-requisite for metapopulation long-term persistence (Hanski 1999, p. 28). In spatially heterogeneous populations, it is unlikely that the entire population will contemporaneously experience the same environmental conditions, thus ensuring that not all colonies comprising a population will fail due to unfavorable conditions (USFWS 2016, p. 18).

In summary, the significant determinants of population-level viability for YBBB are a healthy demography and sufficient quality habitat to support this demography. The demography of YBBB populations is a function of their population size (the number of successful colonies) and the population growth rate over time. The population size required to support a viable population is likely variable across spatial scales and is unknown, but generally speaking, the larger the population, the more genetically healthy and thus the more robust to extirpation. A precise estimate of the area of habitat required to support a viable population is dependent on the density and quality of floral resources, but given the large amount of food needed to support successful colonies, it is reasonable to assume a large area is required. Another important aspect of population viability is connectivity among colonies to ensure mating of unrelated reproductives and connectivity among populations to maintain within population genetic diversity. Lastly, the degree of spatial heterogeneity across the population area reduces the chances of all colonies failing concurrently due to poor environmental conditions, and thus, is important for long-term persistence (USFWS 2016, p. 17).

The 3Rs

In this section, we describe the ecological requirements at the population and species levels in terms of the 3Rs.

Resiliency

Resiliency is the ability to sustain populations in the face of environmental variation and transient perturbations. YBBBs' resiliency is a function of the number of populations and the distributions of the populations relative to degree and spatial extent of environmental stochasticity. Generally, the greater the number of healthy populations and their spatial heterogeneity, the greater likelihood of sustaining populations through time. Healthy populations are better able to recover from stochastic events and withstand variation in the environment. Thus, the greater the number of healthy populations, the more resiliency the species possesses. Environmental stochasticity acts at local and regional scales, and thus, populations can fluctuate in synchrony over broad geographical areas (Hanski 1999, p. 372). Additionally, over longer periods, landscape and habitat changes can be synchronized over large areas, leading to correlated extinction risks among populations at a larger regional scale (Hanski

1999, pp. 381-382). For example, analyses of butterfly, moth, and aphid dynamics over wide areas in Britain indicate that populations can fluctuate in synchrony over areas of at least 100,000 sq km (39,000 sq miles) (Hanski 1999, pp. 381-382). Thus, having populations distributed across a diversity of environmental conditions helps guard against concurrent losses of populations at local and regional scales by inducing asynchronatic fluctuations among populations. The greater the degree of spatial heterogeneity (specifically, the diversity of ecoregions occupied by YBBB), the greater the resiliency the species will possess. Lastly, resiliency is also influenced by the degree of connectivity among populations. Movement among populations is essential for genetic health via gene flow and demographic rescue. Thus, connectivity among YBBB populations is also a requisite for resiliency (USFWS 2016, pp. 19-20).

Representation

Representation is the ability of a species to adapt to physical (*e.g.*, climate conditions, habitat conditions or structure across large areas) and biological (*e.g.*, novel diseases, pathogens, predators) changes in its environment both presently and into the future; it is the evolutionary capacity or flexibility of the species. Representation is the range of variation found in a species, and this variation, called adaptive diversity, is the source of the species' adaptive capabilities. YBBB's adaptive diversity 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 (USFWS 2016, p. 20).

Genetic diversity is the primary fuel for adapting to changing environmental conditions (Hendry *et al.* 2011, pp. 164-165); for adaptation to occur there must be variation upon which to act (Lankau *et al.* 2011, p. 320). The genetic diversity of the YBBB is determined by its allele diversity (size of its gene pool), which is influenced by the level of gene flow among populations and rates of genetic drift within populations. The degree of landscape connectivity influences gene flow (Lankau *et al.* 2011, p. 320; USFWS 2016, p. 20).

Many bumble bee species exhibit high levels of gene flow at multiple spatial scales (Woodard *et al.* 2015 and refs within, p. 2924; Lozier *et al.* 2011, pp. 4880-4882; Cameron *et al.* 2011b, pp. 664-665), and as such, show little genetic structure at local or regional scales. The genetic differentiation of the YBBB is unknown but where habitat and population connectivity (gene flow) are adequate, the species may not exhibit high genetic differentiation. Natural and man-made barriers (*e.g.*, large water bodies, elevation gradients, and extensive changes in land use patterns, etc.) can limit dispersal (Woodard *et al.* 2015, p. 2924) and lead to genetic structuring. Bee dispersal and therefore gene flow can be significantly reduced by human development as well as reduce the availability of inter-connected and overlapping nesting, foraging and overwintering habitats (Jha and Kremen 2013, pp. 2490-2492). A recent genetic study found signs of YBBB inbreeding and major reductions in the effective population size of YBBBs in eastern Canada, which increases their extinction risk due to the diploid male vortex (higher numbers of infertile males instead of workers and gynes) (Kent *et al.* 2018, entire) (see *Chapter 3 - Small and Isolated Populations*). To preserve the breadth of genetic diversity, it is important to maintain high levels of gene flow among populations. Genetic variation can be negatively affected by genetic drift, which is driven by population size; small populations experience

stronger drift than large ones (Zayed 2009, p. 246). Thus, preserving the genetic diversity of YBBB requires maintaining large populations and connectivity among the populations (USFWS 2016, p. 20).

Phenotypic diversity (the physiological, ecological, and behavioral variation expressed by YBBB) 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 YBBB 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. YBBB representation is, therefore, described as having healthy populations widely distributed across a breadth of ecoregions, spread across the United States and Canada (USFWS 2016, p. 21). Ecoregions are geographical units defined by habitat-relevant characteristics including climate, geology, soil, landform, vegetation, wildlife, and human land-use (Commission for Environmental Cooperation (CEC) 1997, pp. 4–6; Wiken *et al.* 2011, pp. 10–11).

Redundancy

Species-level redundancy reflects the ability of a species to withstand catastrophic events, and is best achieved by having multiple, widely distributed populations relative to the spatial occurrence of catastrophic events. Having multiple (redundant) populations would maintain species persistence, such as having populations widely distributed enough so that all are not subjected to the same stressor simultaneously. In addition to guarding against a single or series of catastrophic events extirpating all populations of YBBB, 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, and as such, representation supporting redundancy. Thus, YBBB redundancy is described as having multiple, healthy populations widely distributed across the breadth of adaptive diversity relative to the spatial occurrence of catastrophic events (USFWS 2016, p. 21).

Synopsis

Viability is the ability to sustain populations over time. To do this, YBBB needs a sufficient number and distribution of populations to withstand environmental stochasticity (resiliency), changes in the environment (representation), and catastrophes (redundancy) (Kent *et al.* 2018, entire; USFWS 2016, p. 21).

Chapter 3: Influences on Viability

Based on the YBBB's life history and habitat needs discussed previously we identified stressors (negative influences) and the contributing sources of those stressors that affect the species' current condition and viability (figure 6).

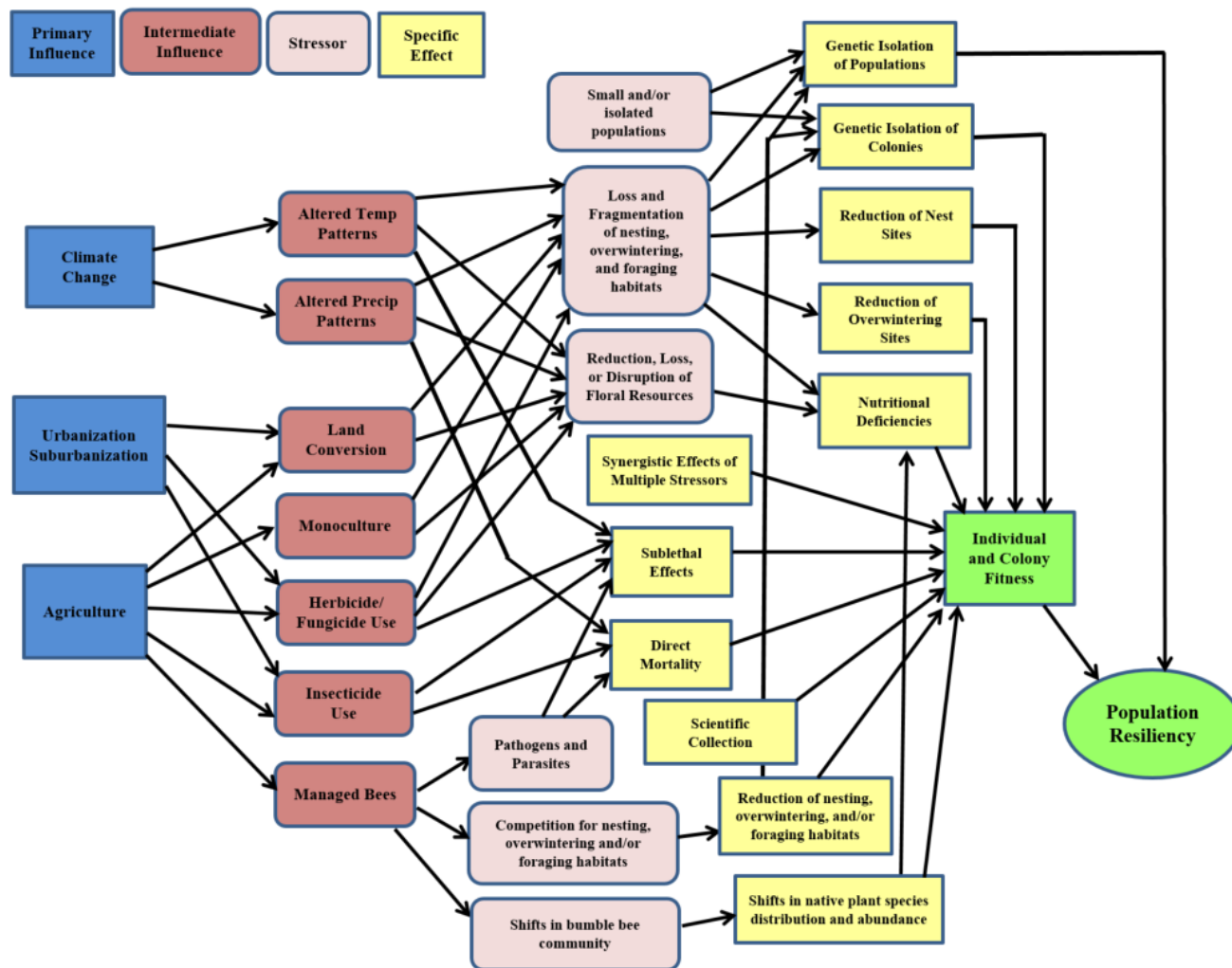


Figure 6: Influence diagram illustrating how environmental stressors influence YBBB habitat, demographics, growth, fitness, and resiliency

Habitat Loss, Fragmentation, and Degradation

Habitat loss, fragmentation, and degradation are recognized as having significant impacts to many *Bombus* species such as the YBBB (Colla 2016, p. 419; Hatfield and LeBuhn 2007 pp. 154-157; Hines and Hendrix 2005, pp. 1481-1483; Grixti *et al.*, 2009 pp. 79-81; Potts *et al.* 2010, p. 348). High bumble bee species richness and abundance have been associated with diverse floral resources, particularly when surrounded by a complexity of natural habitats across the landscape (Hatfield and LeBuhn 2007, pp. 154-157; Hines and Hendrix 2005, pp. 1481-1483). Due to their limited flight range, long colony cycle, and nesting and overwintering requirements, bumble bees are sensitive to the negative effects of habitat fragmentation (Grixti *et al.* 2009, p. 76 who cites Alford 1975 and Sakagami 1976; Richards 1978, entire). Habitat loss, fragmentation, and degradation reduces the amount or accessibility of suitable feeding, nesting, and overwintering habitat. It also reduces the connectivity required for healthy populations to expand in response to environmental or demographic changes and to maintain genetic diversity.

The YBBB is associated with mixed and woodland habitats, prairies, and wetlands (e.g., Colla 2016, p. 421; Colla and Dumesh 2010, p. 48; Williams 2014 *et al.*, p. 113). The dispersal distance of individual YBBBs is likely between 1 and 10 km (0.62 miles and 6.2 miles) (Lepais *et al.* 2010, pp. 819, 829; Kraus *et al.* 2009, pp. 249-251; USFWS 2016, p. 11). The YBBB requires nesting, overwintering, mating, and foraging habitat areas that connect, overlap, and are in relatively close proximity to each other (See *Chapter 2 - Habitat Needs* for detailed information).

Throughout its range, habitat historically occupied by YBBBs has been lost, degraded, or fragmented. The conversion of natural habitat to farmlands, urban, and suburban areas is the primary cause of bumble bee habitat loss (Goulson *et al.* 2015, p. 2; Jacobson *et al.* 2018, p. 437). Woodland habitat was cleared extensively for agriculture in the northeastern United States from the 1600s to the mid-1800s. Following this period, the westward shift of large agriculture led to substantial forest regrowth in abandoned agricultural lands over the next 150 years. However, since 1999 the loss of forested habitat is again on the rise in the Northeast United States, and approximately 24,000 acres of forestlands are converted annually for residential and commercial development (Foster *et al.* 2017, p. 11). Other factors contributing to the current 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 (Foster *et al.* 2017, p. 1-7). In the upper Midwest, forest, savanna, and prairie ecosystems were replaced with cropland from 1850 to 1935, and since that time there has been little recovery of these ecosystems (Rhemtulla *et al.* 2007, entire). The decline in native North American prairie land since the time of European settlement is estimated to be as high as 99.9 percent (Sampson and Knopf 1994, pp. 418-419; Goulson 2010, pp. 182-183). More recently, the growing worldwide demand for food has led to changes in farming practices and agricultural intensification within the YBBB's range. Remaining fragments of tallgrass prairies and wetlands in the upper Midwest continue to be lost due to the expansion of corn and soybean cultivation (Wright and Wimberly 2013, pp. 4134-4139). Consolidation of small diverse farms into industrial monocultures with mechanical disturbances over large areas has also led to the loss of natural areas around smaller fields of diverse crops (Goulson 2010, pp. 181-186).

Uncultivated field margins and patches of natural habitat can support a diversity of flowering plants that provide pollen and nectar through the entire season, as well as nesting and overwintering habitat for the YBBB. However, the patch size, location, and distribution, coupled with plant diversity determine how sufficient the features are for supporting healthy bee populations (Ockinger and Smith 2007, pp. 55-57). While large, single crop fields provide a mass of flowering crops when planted with bee preferred crops for a short period of time, the YBBB requires foraging habitat with a variety of floral resources throughout the season to support colony reproduction, growth, and fitness (DeBarros 2010, p. 6; Goulson and Darvil 2008, p. 193; Goulson *et al.* 2015, p. 4; Grixti *et al.*, pp. 79-82). Mass flowering plants can rapidly increase the growth of colonies, but reproductive success is dependent upon continuous floral resources throughout the season and at the landscape level beyond the area of single farms (Westphal *et al.* 2009, pp. 191-192). Changes in the distribution and abundance of diverse floral resources can play a critical role in governing bee populations (Ogilvie *et al.* 2017, pp 1507-1515).

Habitat loss and degradation leads to the reduction of abundant and diverse floral resources. 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, the production of smaller and fewer individuals, and an early shift to male production (Beekman *et al.* 1998, pp. 1535-1543; Sutcliffe *et al.* 1990, p. 1123; Sutcliffe *et al.* 1988, pp. 1056-1057). The quantity and quality of floral resources affects the growth and fitness of the first round of worker bumble bees, and this affects the long-term success of the entire colony (Hatfield and LeBuhn 2007, pp. 156-157). 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 *et al.* 2016, pp. 413 who cites Hobbs 1962; Heinrich 1976b; Durisko *et al.* 2011, entire; Townsend-Mehler and Dyer 2012, entire). 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 *et al.* 2016, pp. 416-417; Moerman *et al.* 2016, pp. 6-10; Tasei and Aupinel 2008, entire). Additionally, colonies produce reproductive bees (gynes and males) at the end of the season. Access to abundant and diverse floral resources is particularly important at this time because it influences the size and fitness of the last cohort of offspring in the colony, as well as the number of queens produced, which affects the propagation of the population in the following year (Hatfield and LeBuhn 2007, pp. 156-157).

Urban and suburban development also contribute to habitat loss, fragmentation, and degradation. In the United States, approximately one million hectares (2.47 million acres; 10,000 sq km; 3,900 sq miles) of farmland and natural habitat are converted to urban areas each year (Larson *et al.* 2013, p. 1). Urban and suburban gardens often provide the only food source in an otherwise barren landscape, attracting high densities of some bee species (Goulson and Darvil 2008, p. 201). However, these gardens often contain a high proportion of horticulturally modified variants of plants and exotic plant species that provide poor quality forage and nutrition, because they provide little or no food, or their food is inaccessible to insects. Some exotic plants have

been found to be less attractive to native bees than native plants (Bates *et al.* 2011, p. 5; Frankie *et al.* 2005, entire; Goulson and Darvil 2008, p. 201).

Residential, commercial, and institutional turf grasses (approximately 164,000 sq km (40.5 million acres; 63,000 sq miles) in the United States are often maintained to minimize “weedy” floral resources that are required by foraging bumble bees. Additionally, turf grasses are often treated with insecticides, which further degrade the habitat (Larson *et al.* 2013, p. 1) (see *Pesticides* section below). Therefore, turf grasses do not support the high levels of bumble bee diversity as are found in adjacent natural habitats and can hamper foraging (Evans *et al.* 2008, p.29; National Research Council 2007, p. 86).

Many urban environments provide little or no ground nesting or overwintering habitat, due to greater impervious surfaces and highly manipulated lawn and garden areas (Goulson and Darvil 2008, p. 194; Jha and Kremen 2013, pp. 2491-2492). Therefore, urbanization can negatively affect bumble bee diversity due to an increased proportion of impervious surfaces and a decreased proportion of forest and arable lands, and forest, field, and pasture boundaries. This results in a decrease in foraging areas, as well as nesting and overwintering sites (Ahrne *et al.* 2009, p. 4).

The widespread use of a variety of pesticides and herbicides in agricultural, urban, and suburban areas degrades bumble bee habitat. For example, herbicide use is implicated in the loss of milkweed habitat (approximately 58 percent in the Midwest) (Pleasants *et al.* 2013, p. 136, 139). Common milkweed (*Asclepias syriaca*) is an important food source for the YBBB because the high nectar content of milkweed and high caloric content of the nectar allows YBBBs to better regulate their body temperatures and forage more efficiently (Heinrich 1972, pp. 57, 59-61; Heinrich 1976a, p. 877). Drift of agrochemicals into adjacent natural habitat areas where foraging and nesting occur can negatively affect floral diversity and soils in which bee nest and hibernate (Potts *et al.* 2010, p. 350). In some agricultural settings, the use of managed bees (commercial bumble bees and honey bees) can increase disease and competition for nesting, foraging, and overwintering resources (Jacobson *et al.* 2018, pp. 441-442) (See *Pathogen*, *Pesticides*, and *Competition* sections below).

In North America, the rapid increase in urban and suburban areas since the 1950s and large amounts of impervious surfaces limit the density of ground nesting bees and may limit bee foraging (Jha and Kremen 2013, p. 2491). Urbanized areas are generally warmer than the surrounding landscape due to the presence of altered surfaces (urban heat island effect) (Gill *et al.*, 2007, pp.115-116). This may exacerbate the effects of warming temperatures on YBBB population in some areas. Infrastructure and impervious cover can contribute to bumble bee habitat fragmentation, acting as barriers to bee dispersal and restricting the gene flow of isolated populations across human altered habitats (Jha and Kremen 2013, pp. 2490-2494). Bee gene flow is dependent upon the availability of nesting habitat because the dispersal of queens involves two steps: the dispersal from the natal colony nest site to an overwintering site and the dispersal to a final nest site the following spring (Jha and Kremen 2013, pp. 2487). Habitat fragmentation can lead to small and isolated populations and the production of sterile males (diploids) instead of females, which reduces genetic diversity and increases the risk of population extinction (Grixti *et al.* 2009, p. 76) (see *Small and Isolated Populations* section below).

Reduced genetic diversity also increases the population's vulnerability to effects of infectious diseases, pesticides and nutritional stress (Goulson and Darvil 2008, pp. 196-197; Graystock *et al.*, 2016, p. 65).

Parts of the range where patches of protected habitat and unique ecological niches occur may provide pockets of habitat for YBBBs. A 2017 study in the White Mountain National Forest detected a relatively high abundance of YBBB at higher elevations (40 percent of collected *Bombus* records) (Tucker and Rehan 2017, p. 7-8). While these high elevation and boreal ecozone areas may provide refugial habitat for YBBBs, the restriction and isolation of populations to these areas could lead to evolutionary consequences detrimental to the species, including reduced gene flow among populations, reduced community fitness, and further decline of the species ((Hatfield *et al.* 2015, pp. 5-6; Tucker and Rehan 2017, p. 8) (See *Small and Isolated Populations* section below). Additionally, the YBBB may be the primary pollinator of some species of native alpine, forest, and bog dwelling plants that are uncommon and the fragmentation of habitat and subsequent isolation and loss of YBBB populations could negatively affect those habitat areas (Tucker and Rehan 2017; Jacobson *et al.* 2018, p. 443).

In summary, habitat loss by agriculture and urban and suburban development reduces diverse and abundant floral resources throughout the colony life cycle; reduces nesting, mating, and overwintering habitat; and reduces connectivity of colonies and populations and consequently the genetic health of the YBBB. Therefore, it is reasonable to conclude that habitat loss within the range of the YBBB is an ongoing stressor to the species.

Pesticides

Synthetic pesticides, which include a wide variety of insecticides, herbicides, and fungicides (among others), were introduced in the 1940s and are widely used in agriculture, urban/suburban settings, and in some natural environments (Aspelin 2003, Ch.2 pp. 8–9, Part 4, entire; Fernandez-Cornejo *et al.* 2014, entire). Insecticides are specifically designed to control “pest” insects; however, the potential effects to non-target insect species, including bees, have long been recognized (Butler *et al.* 1943, entire; Way and Synge 1948, entire; Marletto *et al.* 2003, entire). Since the 1990s, the introduction and widespread use of a new class of synthetic insecticides, neonicotinoids, has been implicated in the decline of native pollinators, including *Bombus* spp. (Goulson *et al.* 2008, pp. 9–10; Potts *et al.* 2010, pp. 346–347, 349–350). The use of herbicides, which are chemicals designed to control undesirable plants or “weeds,” has increased markedly since the introduction of herbicide-resistant crops in the mid-1990s (Dill *et al.* 2007, entire; Perry *et al.* 2016, entire; Benbrook 2016, entire). In intensively farmed areas, herbicide use can reduce native floral resources, likely reducing the foraging efficiency of resident bees. Additionally, some herbicides have been shown to cause direct toxic effects to individual bees (Herbert *et al.* 2014, pp. 3461–3462; Balbuena *et al.* 2015, pp. 2801–2804; Bohnenblust *et al.* 2015, pp. 149–150, Elston *et al.* 2013, entire; Robinson *et al.* 2017, entire). Fungicides, which are used to protect plants and seeds from fungal pathogens, can also cause direct toxic effects (alone or acting synergistically with other pesticides) to exposed bees and have been linked to declines in four declining *Bombus* spp. (Pilling and Jepson 1993, entire; Ladurner *et al.* 2005, pp. 454–456, Zhu *et al.* 2014, pp. 8–10; Park *et al.* 2015 pp. 5–7; Robinson *et al.* 2017, entire; McArt *et al.* 2017, p. 6).

The route of pesticide exposure may vary based on the characteristics of the specific pesticide mixture being applied, the method of application, the time of day of application, and local weather conditions. Bumble bees can be exposed through direct topical contact with the product, ingestion of pesticide residues in pollen, nectar, water, or guttation drops (xylem sap exuded at the edges of leaves of some vascular plants), and inhalation of volatile products (Sanchez-Bayo and Goka 2014, p. 12, Krupke *et al.* 2012, entire). Pesticide exposure(s) can be “acute,” meaning that exposure occurs once or over a short period of time, or “chronic,” meaning that exposure occurs over a longer period (Botias *et al.* 2015, entire; Botias *et al.* 2016, entire). The life-stage of an individual bumble bee may determine an exposure pathway. For example, free-flying queens and workers can be exposed via multiple routes, including contact with foliar sprays or contaminated seed dust, or ingestion of contaminated pollen, nectar, guttation fluid, and water, while larval bees may be exposed via maternal transfer or ingestion of contaminated pollen or nectar brought back to the nest. Additionally, because some pesticides are known to accumulate in the soil, overwintering queens may be exposed while hibernating, and all bumble bees may be exposed while in their subterranean nests.

Exposure to different pesticides, alone or in combination, can cause a wide variety of toxic effects in bumble bees. Impacts range from direct mortality to sublethal effects such as neurological and behavioral impairment, reduced foraging efficiency, suppression of immune function, reproductive, impaired memory and learning or other physiological effects (Thompson 2003, entire; Mommaerts and Smagghe 2011, entire; Desneux *et al.* 2007, entire; Laycock *et al.* 2012, entire; Laycock and Cresswell 2013, entire; Feltham *et al.* 2014, pp. 319–322; Gill and Raine 2014, pp. 5–11; Moffat *et al.* 2015, entire; Simon-Delso *et al.* 2015, entire; Arce *et al.* 2016, entire; Baron *et al.* 2017b, entire; Ellis *et al.* 2017, entire; Motta *et al.* 2018, entire). In agricultural areas especially, as well as in urban and suburban settings, bumble bees can be exposed to individual pesticides or, perhaps more likely, combinations of pesticides (Main *et al.* 2014, pp. 5–11; Hladik *et al.* 2016, pp. 472–476; Botias *et al.* 2017, pp. 76–80) and toxic responses may vary based on the life-stage of the individual bumble bee as well as other physiological stressors the bee may be subject to, such as dietary deficiencies, heat or cold stress, or pathogen or parasite infection (Piiroinen, *et al.* 2016, p. 11). The indirect effects of pesticides, specifically herbicides, may also include nutritional deficiencies or altered bee energy budgets as native floral resources are reduced.

Each YBBB colony is composed of a single queen supported by an average of 390 workers and males. Late in the season, the queens in successful colonies will produce new queens and males, which disperse from the nest and are the basis for the following year’s bumble bee population (see *Chapter 2 - Life History* section). Because of this particular life history, the colony itself can be considered the fundamental reproductive unit for the species. Therefore the effects of pesticide exposure on colony health (e.g. the number of workers produced, their foraging efficiency, their capacity to maintain and defend the nest, etc.) is more important than the effects to an individual bumble bee (excepting of course the queen and fall reproductives). Numerous laboratory and field studies have demonstrated that pesticide exposure (primarily neonicotinoids, discussed below) can reduce bumble bee colony health, reproductive output, and queen production (Gill *et al.* 2012, entire; Laycock *et al.* 2012, entire; Whitehorn *et al.* 2012, entire; Elston *et al.* 2013, entire; Laycock and Cresswell 2013, pp. 3–7; Fauser-Misslin *et al.* 2014, pp.

455–457; Scholer and Krischik 2014, pp. 5–8; Goulson 2015, pp. 4–7; Arce *et al.* 2016, pp. 6–7; Baron *et al.* 2017a, entire; Baron *et al.* 2017b, entire; Ellis *et al.* 2017, pp. 1729–1731). These colony-level effects may be a significant driver of the observed decline in the YBBB (Evans *et al.* 2008, 14–17; Cameron *et al.* 2011b, entire; Jacobson *et al.* 2017, p. 441).

Neonicotinoids

Neonicotinoids are a versatile class of insecticide introduced in the early-1990s that are now widely used in agricultural areas for crop protection and in other areas for the protection of lawns and ornamental plantings (Jeschke *et al.* 2010, pp. 2897, 2900–2902; Larson *et al.* 2013, entire; Simon-Delso *et al.* 2015, pp. 11–13). The neonicotinoids act primarily on the central nervous system of exposed insects; however, in bumble bees the ultimate effects can range from behavioral, immunological, and reproductive abnormalities to death, depending on the exposure (Jeschke *et al.* 2010, p. 2899; Mommaerts *et al.* 2010, pp. 210–212; Laycock *et al.* 2012, entire; Scholer and Krischik 2014, pp. 5–8; Moffat *et al.* 2015, entire; Stanley *et al.* 2015, pp. 5–6; Stanley and Raine 2016, pp. 1135–1138; Baron *et al.* 2017a, entire; Baron *et al.* 2017b, pp. 4–6; Simmons and Angelini 2017, pp. 5–6).

Two factors related to the application and mode of action of neonicotinoids make this class of insecticide particularly problematic for bumble bees. Traditionally, insecticides were most often applied as foliar sprays or soil treatments to address specific, identified insect pest threats (Jeschke *et al.* 2010, p. 2897; Tooker *et al.* 2017, p. 2) and over the years, a variety of management practices were identified to help minimize the risk of insecticide application to non-target pollinators (Hooven *et al.* 2013, pp. 8–14; Tooker *et al.* 2017, p. 2). For example, insecticide spraying can be done during periods when bees are less likely to be active in the area, and therefore less likely to be exposed. While neonicotinoids can also be applied as a spray or soil treatment for controlling specific pests, they are increasingly used as a prophylactic “seed dressing,” which means a crop seed is coated with the insecticide prior to planting (Jeschke *et al.* 2010, pp. 2900–2901). Because they are long-lasting “systemic” insecticides, neonicotinoids on treated seeds are incorporated into the plant’s tissues, including the pollen, nectar, and guttation fluids. Bumble bees are therefore chronically exposed to the insecticide from the time crops emerge (via guttation drops) throughout a crop’s bloom period (via pollen and nectar) (Girolami *et al.* 2009, pp. 1813–1814; Reetz *et al.* 2011, pp. 603–605). Additionally, neonicotinoids are known to contaminate perennial wild plants growing at the margins of seed-treated crop fields, as well as annual and perennial ornamental plants, thus providing another potential chronic exposure pathway for foraging bumble bees (Botias *et al.* 2015, pp. 12735–12738; Botias *et al.* 2016, pp. 273–276).

During the planting of neonicotinoid-coated seeds, fugitive dust from the mechanical seed planters can also expose bees directly to neonicotinoids at levels high enough to cause toxic effects (Forster 2009, entire; Girolami *et al.* 2011, pp. 6–8; Marzaro *et al.* 2011, pp. 124–125; Tapparo *et al.* 2012, entire; Sampson-Robert *et al.* 2015, entire; Xue *et al.* 2015, entire). This is especially concerning because crop planting usually occurs in the springtime, the period when recently emerged queen YBBBs are actively foraging and searching for suitable nesting locations. Neonicotinoid-treated seeds are now nearly universal for some crops. Within the YBBB’s range in the United States, it is estimated that nearly all land planted in corn and up to

75 percent of the land planted in soybeans (the top two crops by area) now use neonicotinoid-treated seeds (Douglas and Tooker 2015, p. 5092). Similar trends occur within portions of the YBBB's range in Canada. For example, in the prairie region of Canada, where 98 percent of that country's canola is grown, nearly all seed is coated with neonicotinoids (with 8.5 million hectares (21 million acres) planted in 2012) (Main *et al.* 2014, p. 2). Neonicotinoids are water soluble and relatively persistent in the environment, meaning they degrade slowly and can be transported into adjacent non-agricultural habitats (Jeschke *et al.* 2010, pp. 2898–2899). Chemical analysis of surface water samples from agricultural areas in the Midwest United States, the prairie region of Canada, and Ontario, Canada showed seasonal correlations between the use of neonicotinoid-treated seeds and neonicotinoid levels in nearby waters, demonstrating the ubiquity and mobility of neonicotinoids in these environments (Hladik *et al.* 2014, entire; Main *et al.* 2014, entire; Struger *et al.* 2017, entire).

The use of neonicotinoids has increased dramatically since their introduction in the mid-1990s, even as the use of other classes of insecticides has trended downward or remained steady. This is even more striking because the higher effectiveness of neonicotinoids makes their application rate markedly lower than that of traditional insecticides (Douglas and Tooker 2015, p. 5093). A temporal analysis of neonicotinoid use and YBBB survey data shows a precipitous decline in the species' relative abundance from the 1990s to the 2000s, coincident with the introduction and widespread use of these pesticides (figure 7). While this analysis does not necessarily demonstrate a causal relationship it does suggest that neonicotinoids may be playing a role in the apparent decline of the YBBB, especially given the known toxicity of neonicotinoids to bumble bees.

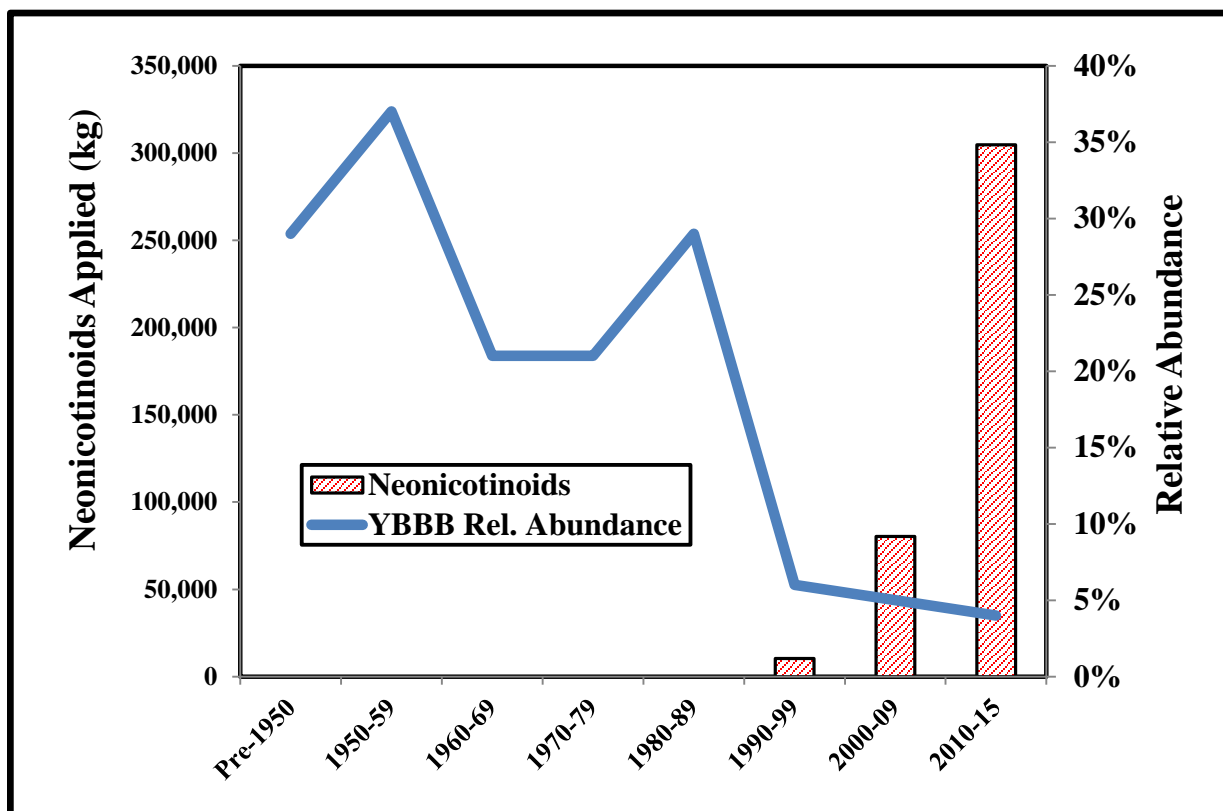


Figure 7. Trend in YBBB relative abundance and neonicotinoid application in eight states (MN, WI, MI, NY, MA, VT, NH, and ME) representing the core of the species' historical range in the U.S. (see Sheffield et al. 2016) Neonicotinoid amounts are the combined annual application, based on the 10-year average, of acetamiprid, clothianidin, imidacloprid, thiacloprid, and thiamethoxam (data from the USGS Pesticide National Synthesis Project, 2018).

Glyphosate

Glyphosate is a broad-spectrum, systemic herbicide first marketed in 1974. The product was used moderately for agricultural, commercial, and residential purposes up until the mid-1990s, when genetically engineered herbicide-tolerant (GEHT) crops were introduced. Following this innovation, the application of glyphosate in agricultural areas increased dramatically (Benbrook 2016, pp. 1–2, 5–9; Dill 2005, pp. 219–221). While generally considered to have low toxicity to terrestrial insects, some studies suggest glyphosate may cause sub-lethal effects to exposed bees (Balbuena *et al.* 2015, entire; Helmer *et al.* 2014, entire; Herbert *et al.* 2014, entire; Motta *et al.* 2018, entire). Perhaps a more significant effect of intense glyphosate use is a reduction in floral resources (i.e. flowering “weeds”, such as milkweed, aster, and goldenrod) in agricultural landscapes, thus affecting the quality and availability of pollen and nectar required by YBBBs throughout the colony life cycle.

Prior to the mid-1990s, both crops and weeds were susceptible to the effects of glyphosate, therefore its application was generally restricted to the periods before or after the crop growing season. However beginning in 1996, GEHT crops (initially soybeans, corn, and cotton, now also canola, alfalfa, wheat, sugar beets, and others) were introduced which allowed for (and encouraged) the application of glyphosate for weed control in farm fields throughout the crop growing season (Benbrook 2016, pp. 1–2). These intense weed control efforts can reduce the diversity and health of wild floral resources in and adjacent to treated fields during the entire period of YBBB colony growth and reproduction. This likely affects bumble bee foraging efficiency and may contribute to dietary deficiencies that reduce colony and reproductive success (Marshall et al. 2001, p. 71; Richards 2001, p. 168, 170; Boutin *et al.* 2014, pp. 299–304; Bohnenblust *et al.* 2016, pp. 149–150; Rollin *et al.* 2016, entire).

The widespread adaptation of GEHT crops led to a very rapid rise in the amount of glyphosate applied over the last two decades. In 2014, the USDA estimated that about 94 percent of U.S. soybean acres and 89 percent of U.S. corn acres were planted with GEHT seeds (USDA 2017, entire). The available data for Canada indicate a similar increase in GEHT crop adaptation (in Ontario primarily soybean and corn and in Manitoba primarily canola, soybean, and corn) with a commensurate increase in glyphosate usage (Wilson 2012, pp. 43–44; Farm and Food Care Ontario 2015, p. 17). Factors contributing to the continued increase in glyphosate use include the expected introduction of new herbicide-tolerant crops, the emergence of glyphosate-resistant weeds requiring more intense glyphosate applications, and the declining price of the product (Benbrook 2016, pp. 7–9). Because of the documented increase in the use of glyphosate over the last two decades, this SSA report specifically focuses on that chemical, however it is noteworthy that new GEHT crops are being developed that may lead to similar increases in the application of other herbicides (e.g. 2,4-D and dicamba) with similar resulting effects to YBBB (Benbrook 2016, p. 2).

A temporal analysis of glyphosate use and YBBB survey data shows a precipitous decline in YBBB relative abundance beginning in the 1990s, when glyphosate use began increasing rapidly (figure 8). This is similar to the pattern observed for neonicotinoid insecticides, discussed above. While this does not necessarily demonstrate a causal relationship, it does suggest that glyphosate use may also be playing a role in the apparent decline of the YBBB.

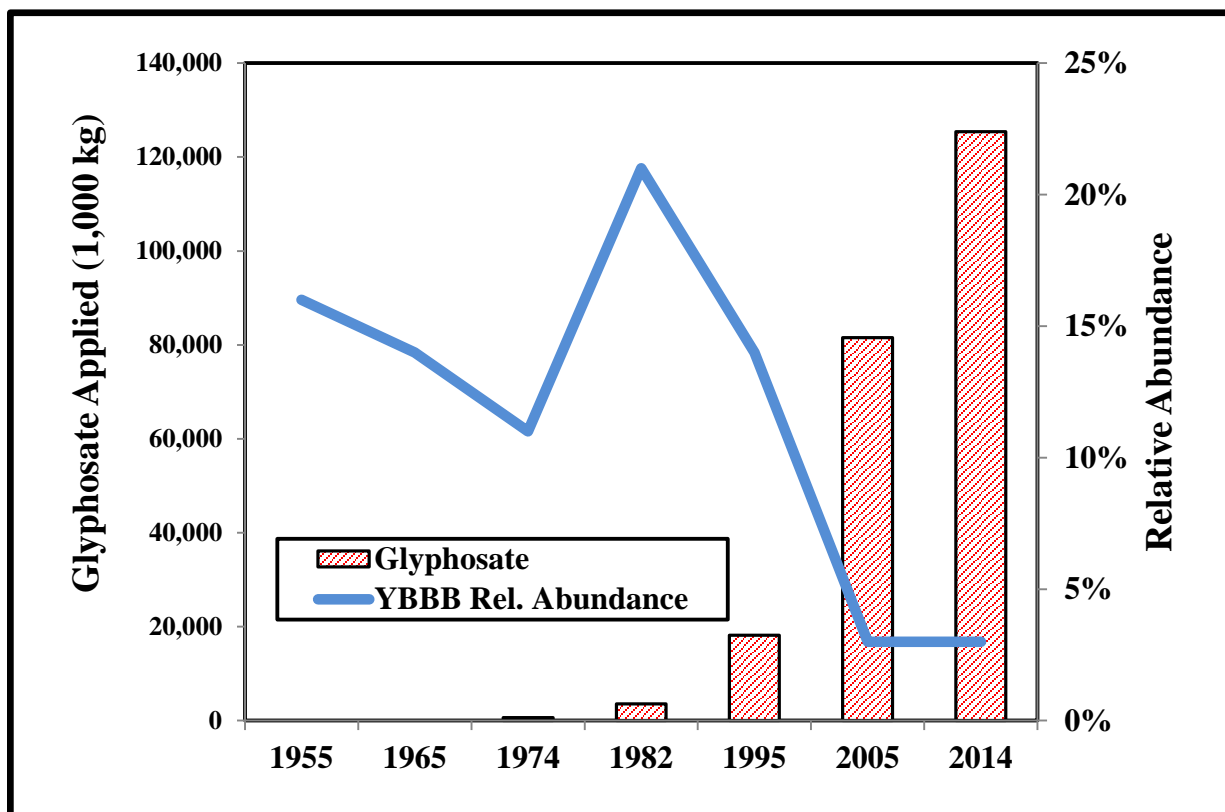


Figure 8. Trend in YBBB relative abundance and glyphosate application in the U.S. The glyphosate figures represent the total active ingredient applied in the U.S. during the given year (from Benbrook 2016); the YBBB percentages are the decadal average for the given year.

Summary

Various insecticides, herbicides, and fungicides have the potential to affect the YBBB and its habitat directly and indirectly. Multiple scientific studies indicate that many current-use pesticides, including neonicotinoid insecticides, are toxic to wild bumble bees. While the application of some traditional insecticides has decreased over the last several decades, the use of neonicotinoids has increased exponentially since their development in the early 1990s. Additionally, the widespread adoption of herbicide-resistant crops has greatly increased the use of glyphosate, likely decreasing the diversity of floral resources in agricultural landscapes. Exposure to fungicides has been found to be strongest predictor of the deadly pathogen *N. bombi* in four declining bumble bee species, including the YBBB (McArt *et al.* 2017, pp. 1-7). Therefore, it is reasonable to conclude that pesticide use within the range of the YBBB is an ongoing stressor to the species.

Pathogens and Parasites

Overview

Bee parasites and pathogens occur naturally in the environment, and North American bumble bees likely co-evolved with a variety of them, including mites, protozoa, viruses, bacteria, fungi, and parasitoid flies and wasps. Over the past 30 years, the production and worldwide movement of managed bees (honey bees and commercial bumble bees) have tripled due to the decline of native pollinators and the growing demand for food (Evans 2017, p. 3435; Velthuis and van Doorn 2006, p. 429; Goulson *et al.* 2015, p. 1). Concurrent with the increased deployment of managed bees, the prevalence of native and/or non-native pathogens and parasites in managed and native bees has also increased.

The majority of bee disease research has been conducted on honey bees and a few common bees in Europe, however important knowledge gaps remain including the prevalence of certain pathogens in natural systems, aspects of the life history of bee pathogens and parasites, the historical distribution and host species of bee parasites, and the virulence and natural population dynamics of many insect parasites (Colla 2016, p. 419; Goulson and Hughes 2015, pp. 11-13).

Despite data gaps in pathogen and parasite epidemiology and the susceptibility of different bumble bee species, pathogens and parasites in wild and managed bee populations are considered to be a significant threat to several bumble bee species (Arbetman 2013, pp. 489-490; Cameron 2011a, pp. 662, 665; Colla 2006, pp. 465-466; Evans 2017, entire; Furst *et al.* 2014, p. 366; Goulson *et al.* 2015, pp. 2-3; Goulson and Hughes 2015, p. 11; Graystock 2013b, pp. 1,7-8; Graystock 2016, pp. 70-71; Hatfield 2015, p. 5; Kissinger *et al.* 2011, p. 223; Otterstatter and Thomson 2008, pp. 1, 6; Sachman-Ruiz *et al.* 2015, pp. 8-9; Singh *et al.* 2010, pp. 1-12; Szabo *et al.* 2012, p. 235; Wilfert *et al.* 2016, p. 596).

There are three mechanisms of disease emergence within and between populations of managed and wild bees: competitive and nutritional stress, pathogen spillover, and pathogen spillback (Graystock *et al.* 2016, entire) (figure 9). Competitive and nutritional stress increases the susceptibility of wild bees to pathogens and parasites when high densities of managed bees are introduced (i.e., pathogen and parasite facilitation). Pathogen spillover occurs when managed bees transmit pathogens or parasites to wild bees, or if the pathogen or parasite is already present, artificially increases the pathogen or parasite level in the wild bee population. Pathogen spillback occurs when a natural pathogen or parasite in the wild bee population is transmitted to a managed bee population where it thrives in the high bee densities and spills back into the wild population at unnaturally high levels (Graystock *et al.* 2016, entire; Goulson and Hughes 2015, pp. 12-14).

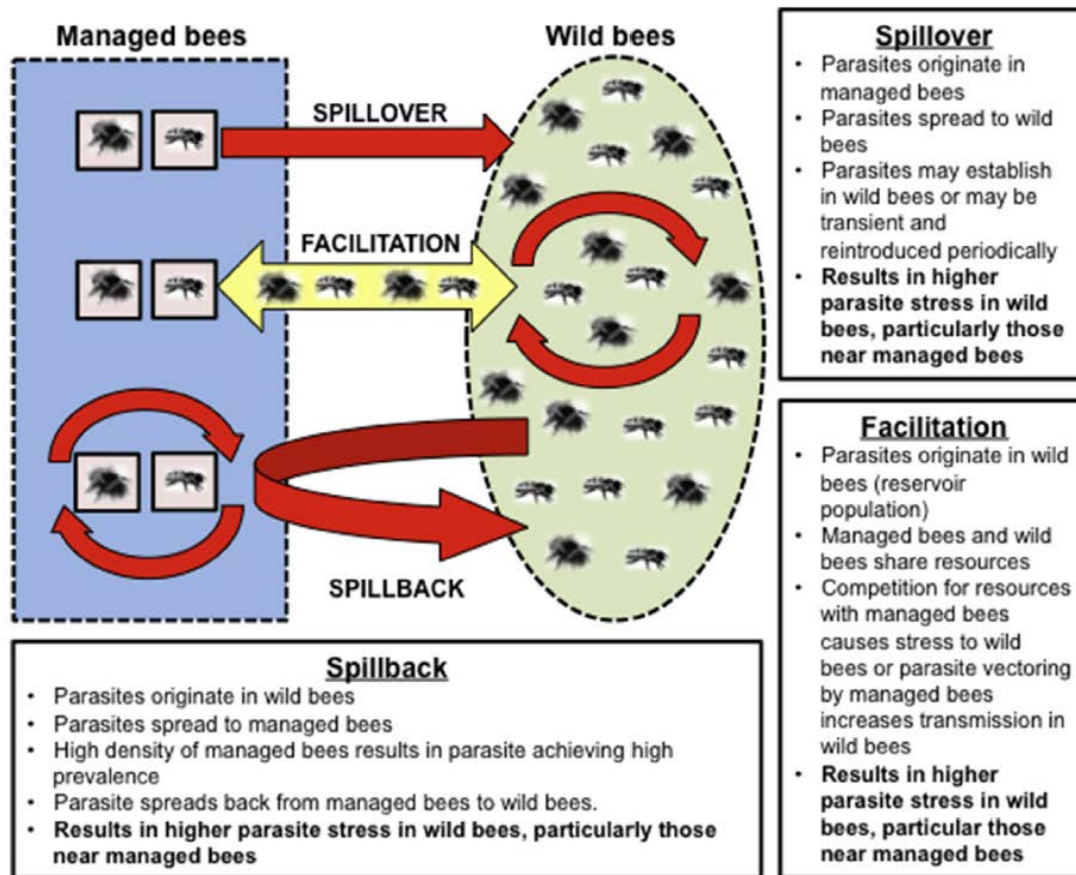


Figure 9: Three mechanisms of pathogen and parasite infections between populations of managed and wild bees. Arrows indicated the direction of spread due to the mechanism. (From Graystock *et al.*, 2016, p. 66).

Pathogen spillover from domesticated populations to wild populations is a main source of emerging infectious diseases (EIDs) (Morse 1995, pp.7-9; Meeus *et al.* 2011, p. 663; Daszak *et al.* 2000, pp. 445-448). EIDs are diseases that have recently appeared in a population or the incidence or geographic range of an infection is rapidly increasing or threatens to increase in the near future. They are ranked as one of the top five causes of species extinction worldwide (Graystock *et al.* 2016, p.65). Small or declining populations are at an increased risk when exposed to EIDs because the source host acts as a continuous reservoir, causing repeated spillover events and disease outbreaks in the already vulnerable populations (Furst 2014, p. 364). Several pathogens that infect honey bees, commercial bumble bees, and wild bumble bees have recently been cataloged as EIDs (*Nosema ceranae*, *Apicystis bombi*, and Deformed Wing Virus (DWV) (Brown 2017, pp. 747-760; Sachman-Ruiz 2015 p. 2; Wilfert *et al.* 2016, p. 596). For these reasons, in 2017 the introduction of non-native bumble bee species and their spread of novel infectious diseases were listed as one of the top emerging issues for global conservation and biodiversity (Sutherland *et al.* 2017, p. 34).

The European honey bee (*Apis mellifera*) has been domesticated and heavily managed for thousands of years. Due to their close social interactions in colonies of up to 60,000 individuals in a single hive, honey bees are highly susceptible to infectious diseases. Approximately 24 viruses have been identified in different honey bee life stages and several have been identified as

a threat to other pollinator species, including bumble bees (Manley *et al.* 2015, p. 2; Singh *et al.* 2010, pp. 1-2).

Similarly, conditions in commercial bumble bee facilities that propagate, sell, and transport bumble bees to areas for pollination often increase the abundance, and possibly virulence, of pathogens and parasites. The high density of bees, as well as a higher survival rate of infected bees as a result of an abundant food supply, can lead to higher parasite loads and longer disease transmission periods than are found in wild bee populations (Colla 2006, p. 465; Evans 2017, p. 36; Meeus *et al.*, 2011, pp. 663-667; Murray *et al.* 2013, pp. 273-274; Otterstatter and Thomson 2008, p. 5). Additionally, commercially reared bumble bees are fed pollen collected and processed by honey bees, providing an exposure route for many bee pathogens, including *Nosema bombi*, *N. cernae*, *Apicystis bombi*, *Crithidia bombi*, DWV, Black Queen Cell Virus (BQCV), Sac Brood Virus (SBV), chalk brood, and American foulbrood (Goulson *et al.* 2015, p. 3; Graystock 2016, p. 71) and the spread of disease likely.

Declines of the YBBB and the American bumble bee (*B. pensylvanicus*) have been found to be associated with high densities of commercial bumble bees (Szabo *et al.* 2012, pp. 235-237). In North America, two native *Bombus* species were initially used for commercial bumble bee colony production. From 1992 through 1994, the common eastern bumble bee, *Bombus impatiens* (native species to the eastern U.S. and Canada) and the WBB (*Bombus occidentalis*) native to western United States and Canada, were reared in European production factories alongside Europe's native buff tailed bumble bee (*Bombus terrestris*, also a member of sub-genus *Bombus* sensu stricto). The bees were then imported back to the U.S. for open field and greenhouse pollination. During this time, commercial colony production also began in eastern Canada using wild common eastern bumble bees and in California using the WBB. By 1997, the production and use of the WBB ceased as a result of a major infestation of *Nosema bombi* and collapse of the WBB commercial rearing operations (Cameron 2016, p. 4386; Velthuis and van Doorn 2006, pp. 427, 432). The subsequent decline of four North American species belonging to *Bombus* sensu stricto, including two eastern species (YBBB and RPBB) and two western species (WBB and Franklin bumble bee, *B. franklini*), have been associated with significantly elevated levels of *N. bombi*; however the precise role of *N. bombi* in these species' declines remains difficult to determine (Cameron *et al.* 2011a, p. 665; Cameron *et al.* 2016, pp. 4386-4387; Evans 2017, p. 37; Graystock *et al.* 2016, p. 69; Szabo 2012, pp. 235-237). New research from Canada has found that YBBB and other species enter the "quads" housing commercial bumble bees in agricultural fields, providing additional opportunities for disease transfer (Hicks *et al.* 2018, pp. 1-9). Two additional *Bombus* species, *B. huntii* and *B. vosnesenskii*, will soon be available from commercial suppliers for pollination in the western US; this may increase the likelihood of pathogen spread and exacerbate this particular stressor in western ecoregions where YBBB still occurs (L. Richardson pers. comm. 2018).

Exposure to pesticides may increase bumble bee susceptibility to pathogens. An increased susceptibility to pathogens, increased prevalence of infection; and increased pathogen-induced mortality has been observed in honey bees (Goulson *et al.* 2015, p. 5). Fungicides are typically the most abundant pesticide detected in bumble bees (Botias *et al.* 2017, pp. 73-81). A recent study by McArt *et al.* (2017) found that exposure to the fungicide Chlorothalonil® was the

strongest predictor of *N. bombi* prevalence in the four declining bumble bee species, including the YBBB (McArt *et al.* 2017, pp. 1-7).

Low genetic diversity may also increase bumble bee susceptibility to pathogens and parasites. Wild bee communities may lack an evolved resistance to novel pathogens (Manley *et al.* 2015, p. 1), and some bee populations with reduced genetic diversity have a high prevalence of parasites (Whitehorn *et al.* 2014, p. 670). In a Massachusetts study, *Nosema* infection levels were significantly higher in rare bumble bee species (American bumble bee and yellow bumble bee, *B. fervidus*) than in common species (Gillespie 2010, pp. 742, 744). A national study found that declining populations of two species (American bumble bee and the WBB, *B. occidentalis*) also had significantly higher *N. bombi* infection levels as well as lower genetic diversity compared to bumble bee species with stable populations (there were an insufficient number of samples of YBBB and RPBB available for analysis) (Cameron *et al.* 2011a, p. 664-666). A genetic study found that YBBB have several immune related genes with signatures indicating that pathogen spillover has played a role in their decline, as well as low effective population size and inbreeding in eastern Canada (Kent *et al.* 2018, entire).

Due to the reliance on managed bees in agriculture, numerous options for reducing the risks associated with the anthropogenic spread of bee pathogens and parasites have been identified. Some safeguards proposed to minimize negative effects to wild bees include: monitoring pathogens and parasites in commercial stock; preventing the escape of commercial bumble bees; improving regulations on the transport of managed bees across state and international boundaries; creating nesting and foraging habitat for wild bumble bee populations; and rearing locally native species for commercial use (see Evans 2017; pp. 38-40 for details). For additional strategies see: Goulson *et al.*, 2015, pp. 5-7; Goulson and Hughes 2015, pp. 14-17; Graystock *et al.* 2016, pp. 71-72; Meeus *et al.* 2011., p. 668; Murray *et al.* 2013, p. 275; Wilfert *et al.* 2016, p. 596.

Pathogens and parasites that may affect the YBBB:

Nosema bombi

Nosema bombi is a microsporidian intestinal parasite (related to fungi) that infects commercially produced and wild bumble bees. Bumble bee workers, larvae, queens and reproductives spread the parasite within the colony. *Nosema bombi* spores are spread through fecal oral contact both within the nest and can transfer to bees from other nests at shared flowers.

N. bombi significantly reduces bumble bee colony growth and fitness; reduces reproductive performance of males and gynes; lowers survival rate of workers; and reduces lifespan. Smaller colonies that are infected have a lower chance of reaching the stage of sexual reproduction (Colla *et al.* 2006, p. 465; Graystock 2013b, p. 6; Meeus *et al.* 2011, p. 666; Otti and Schmidt-Hempel 2007, pp. 118-123; Rutrecht and Brown 2008, pp. 505-511).

It is suspected that the export and import of commercial bumble bees during the 1990s and the presence of high densities of commercial bumble bee colonies can increase parasite loads and stress on native bumble bees. In the United States, *Nosema bombi* was historically present and

widely dispersed, but infection rates were low and then increased significantly in the mid-1990s, just before the first reported decline of several *Bombus* species (Cameron *et al.* 2016, pp. 4388-4389). *Bombus* species that have undergone range reductions have been found to have a significantly higher prevalence of *N. bombi*, while the prevalence of *N. bombi* in stable bumble bee species has remained relatively low (Bushmann *et al.* 2012, pp. 5-6; Cameron *et al.* 2016, pp. 4388-4390; Gillespie 2010, p. 742; Malfi and Roulston 2014 pp. 26-27). While there is evidence that *N. bombi* spillover has contributed to the decline of some North American bumble bee species, it is undetermined whether it is a direct cause of the declines (Cameron 2016, pp. 4388-4390; Colla *et al.* 2016, p. 419; Evans 2017, pp. 36-37; Graystock *et al.* 2016, p. 69; Manley *et al.* 2015, p. 8; Meeus *et al.* 2011, pp. 665-666; Otterstatter and Thomson 2008, p. 6).

Low genetic diversity may increase bumble bee susceptibility to pathogens and parasites. In a Massachusetts study, *Nosema* infection levels were significantly higher in rare bumble bee species than in common species (Gillespie 2010, pp. 742, 744). A national study found that declining populations of two species in the sub-genus *Bombus* s.s. also had significantly higher *N. bombi* infection levels as well as lower genetic diversity compared to bumble bee species with stable populations (Cameron *et al.* 2011, pp. 664-666). And, a recent genetic study of YBBB in eastern Canada found that YBBB is showing signs of inbreeding, a decline in their effective population size, and that pathogen spillover may be playing a role in the current range reduction of the YBBB (Kent *et al.*, 2018, entire).

Apicystis bombi

Apicystis bombi is a protozoan parasite found in bumble bees and honey bees in North America, South America, Europe and Asia (Graystock *et al.* 2015c, p. 1). It is found in bee feces and on flowers and therefore is assumed to be transmitted orally, but is possibly transmitted from infected queens to the eggs as well (Graystock *et al.* 2015, p. 1; Goulson and Hughes 2015, p. 13).

Apicystis bombi degrades bee body fat, has neurological effects, and causes mortality in adult bees (Goulson and Hughes 2015, p. 13). In Canada, 2.0 percent and 2.5 percent of early emerging YBBB and RPBB queens, respectively, were infected and they died on average within 14 days (Macfarlane 1995, p. 134). A study of infected buff tailed bumble bee found that the stored body fat in infected bees was significantly reduced and 22 percent of the bees died after 15 days. Reduced body fat reserves in hibernating queens would lower overwintering survival and lower their success in founding new colonies in the spring. The study also found that infected bees had an increased demand for carbohydrates. For infected workers, this would likely result in reduced pollen collected for the developing colony (foraging efficiency) as the worker bees would instead need to forage for nectar to compensate for reduced body fat reserves (Graystock *et al.* 2015, p. 6).

In 2009, the parasite was introduced to South America with the importation of non-native commercial bumble bees (buff-tailed bumble bee). The buff tailed bumble bee is rapidly expanding its range in South America and there is evidence that virulent *Apicystis bombi* can infect native *Bombus* species and is likely contributing to the decline of native bumble bee

species (Arbetman *et al.* 2013, pp. 492-492; Goulson and Hughes 2015, pp. 13-14; Schmid-Hempel *et al.* 2014, pp. 833-834).

In central Mexico, *Apicystis bombi* was the most frequently detected pathogen in the imported commercial bumble bee species, the common eastern bumble bee, and the pathogen is currently cataloged as an EID (Sachman-Ruiz *et al.* 2015, p. 6). In Ontario, *A. bombi* was detected in 1.8 percent of bees at all study locations, but not detected in commercial bumble bees, possibly due to the restricted diet fed to greenhouse bees and reduced fat tissue available for infection (Colla 2006, p. 463; Sachman-Ruiz *et al.* 2015, p. 6).

Crithidia bombi

Crithidia bombi is a bumble bee intestinal trypanosomatid/protozoan parasite that can severely reduce colony founding success by queens, reduce fitness of colonies, slow colony growth, and reduce survival and foraging efficiency of workers. Infected bees have been observed with reduced pollen loads and a reduction in flower visits per minute. It is highly transmissible as it is ingested from bumble bee feces within the nest and is known to spread when bees share flowers (Cordes *et al.* 2012, p. 210 ; Kissinger and Cameron *et al.*, 2011, p. 223; Malfi and Roulston 2014, p. 24; Otterstatter and Thomson 2008, p. 2; Shykoff and Schmid-Hempel 1991, pp.120-122).

Food-stressed worker bees infected with *Crithidia bombi* were found to have a 50 percent increased mortality rate, which can stress colonies and contribute to the declines of susceptible species. Emergent queens infected with the parasite were found to delay colony initiation in the spring (Brown *et al.*, 2003, p. 995-1000; Kissinger and Cameron *et al.*, 2011, p. 223). Smaller bees may be more likely to be parasitized and can delay worker production in infected queens (Malfi and Roulston, 2014, p. 25). *Crithidia bombi* has been found to become virulent in bumble bees with a restricted diet (Goulson *et al.* 2015, p. 5). Research indicates that *C. bombi* was not an introduced species to North America and may be naturally occurring (Cordes *et al.* 2012, pp. 212-214; Otterstatter and Thomson 2008, p. 2).

Crithidia bombi can be more abundant in commercial bee colonies than wild bumble bees, and the prevalence of *C. bombi* detected in wild bumble bees has been found to be higher near sites using commercially produced bumble bees than in wild bees captured 2 km (1.2 miles) or further from such sites. These studies found that the prevalence and intensity of *C. bombi* infections declined with increasing distance from the greenhouses (Colla *et al.* 2006, pp. 462-466; Graystock, *et al.* 2016, pp. 69; Otterstatter and Thomson 2008, pp. 2-7). However, in Massachusetts and Virginia, *C. bombi* infections in common eastern bumble bees were found in relatively isolated areas. Gillespie *et al.* (2010, pp. 737-747) reported that it was unknown whether the individuals sampled were wild or from nearby managed colonies in Massachusetts, while Malfi and Roulston (2014, pp. 23-27) reported that the bees were not collected near commercial greenhouses in Virginia. *C. bombi* has been found to occur frequently in common bumble bee species (Cordes *et al.* 2012, p. 212; Gillespie *et al.* 2010, p. 744; Malfi and Roulston 2014, p. 25). In a broad nationwide survey of 36 species, *C. bombi* commonly occurred in about half of the bumble bee species and at relatively low levels in species with apparently stable populations. The survey did not find evidence that *C. bombi* was involved in the decline of

bumble bees in the United States and whether spillover is occurring or has occurred could not be determined (Cordes *et al.* 2012, p. 214). More research is needed to determine the potential threat of *C. bombi* to the YBBB.

Locustacarus buchneri

The tracheal mite *Locustacarus buchneri* is a widespread bumble bee parasite that feeds on the blood-like fluid (haemolymph) of its hosts, damaging the trachea and respiratory system, piercing the tracheal wall, reducing fitness, and shortening the lifespan of individual bees. The mites overwinter in the trachea of young queens, reproduce inside the queen in the spring and the larvae from female mites spread to other bees. Large numbers of *L. buchneri* can accumulate in worker bees, causing them to become lethargic and to cease foraging (Goulson *et al.* 2010, p.72; Otterstatter and Whidden 2004, pp. 352-356).

Locustacarus buchneri has been found to be relatively host specific with the highest prevalence in bumble bee species belonging to the subgenus *Bombus s.s.*, including the YBBB, however it is unknown why the subgenera *Bombus s.s.* species are more frequently parasitized. The species share similar nesting requirements, overwintering ecology, floral preferences and phenology and may increase their susceptibility to infection (Otterstatter and Whidden 2004, p.355). The parasite has been reported in approximately 25 bumble bee species and while the YBBB comprised only 18 percent of all the bees analyzed, they accounted for approximately 83 percent of all bee parasitized by *L. buchneri* (Otterstatter and Whidden 2004, p. 353).

Locustacarus buchneri can be prevalent in both wild and commercial bumble bee colonies (Colla *et al.*, 2006, pp. 462-466), and there is concern about the potential effects resulting from impacts from high-density managed bees in proximity to wild bees that may be stressed by other factors. Infected bee colonies may be a threat to uninfected bee colonies when infected workers enter colonies other than their own ('drifting workers') or if *L. buchneri* is spread by workers at shared food sources (Colla *et al.* 2006, p. 465; Otterstatter and Whidden 2004, pp. 351,355). It is suspected that the worldwide trade in bumble bees has led to a global redistribution of various strains of *L. buchneri* (Goulson *et al.* 2015, p. 12) and spillover has been observed in Japan, and commercial bumble bee colonies delivered to Mexico were infected with *L. buchneri* from Europe and the United States (Graystock *et al.*, 2016, p. 69; Sachman-Ruiz *et al.* 2015, pp. 5-9).

Viral Pathogens

Honey bees have been linked to the spread of diseases and parasites among honey bees and other pollinators, including bumble bees. The naturally occurring frequency of viral infections in bumble bee populations that are not associated with honey bee apiaries is unknown (Meeus *et al.* 2011, p. 666), however approximately 24 viruses have been detected in various life stages of the honey bee (*Apis mellifera*) and many honey bee viral pathogens and parasites pose a threat to wild bumble bees (Colla 2016, p. 419; Evans 2017, p. 39; Manley *et al.* 2015, p. 2; Singh *et al.*, 2010, p. 2).

In the United States, five common honey bee viruses were found to also infect bumble bees: Deformed Wing Virus (DWV), Black Queen Cell Virus (BQCV), Sac Brood Virus (SBV),

Kashmir Brood Virus (KBV), and Israeli Acute Paralysis Virus (IAPV). These were detected in bumble bees in the vicinity of honey bee apiaries (Singh *et al.* 2010, pp. 4-8). Less common honey bee viruses also detected in bumble bees are Acute Bee Paralysis Virus (ABPV) and Chronic Bee Paralysis Virus (CBPV).

Various viral infections can lead to colony losses; symptoms of these infections include deformed wings, discoloration, hair loss, bloated abdomens, trembling, paralysis, brood and adult mortality (Singh *et al.* 2010, p. 2). The major route of transmission is primarily through shared floral resources while foraging for pollen, as well as in stored pollen and fecal matter within the nest (Singh *et al.* 2010, pp. 6-12).

Deformed wing virus (DWV), and its vector the Varroa mite (*Varroa destructor*), is considered a major threat to the world's honey bees and has been identified as a major contributor to global honeybee mortalities. Its prevalence in honey bees has been linked to its prevalence in wild bumble bees and is identified as an EID with impacts on bumble bee survival (Furst *et al.* 2014, pp. 364-366; Manley *et al.* 2015, p. 2; Wilfert *et al.* 2016, pp. 594-596).

Deformed wing virus was found to be more virulent in bumble bees because oral transmission of the virus caused wing deformities, while in honey bees the direct injection of the virus via the *Varroa* mite bite has to occur for symptoms to manifest. In addition, while asymptomatic honey bees test positive for DWV, bumble bees that tested positive for DWV were symptomatic, suggesting a difference in virulence (Genersch *et al.* 2006, p. 63).

Nosema ceranae

Nosema ceranae (sometimes misidentified as *Nosema apis*), is a honey bee microsporidian intestinal parasite with a broad range of hosts. *Nosema ceranae* was recently cataloged as an EID within bumble bee populations.

Nosema ceranae is a natural parasite to the Asian honey bee (*Apis cerana*) that jumped hosts and is infective to the European honey bee, the honey bee that occurs in North America. Graystock *et al.* (2013a) found that bumble bees can be infected by ingesting spores from flowers visited by infected bees and that it is highly virulent, causing rapid mortality. Symptoms include suppressed immune response, impaired learning and flower handling, increased hunger, and increased mortality contributing to colony losses (Graystock 2016, p. 68; Graystock *et al.* 2013a, pp. 116-117).

As of 2016, *N. ceranae* is found everywhere that the European honey bee occurs, including North America, and analysis has shown that *N. ceranae* introduction, spread, and prevalence have increased over time (Colla 2016, p. 419; Furst *et al.* 2014, pp. 364-366; Goulson *et al.* 2015, p. 2; Graystock *et al.* 2016, p. 68). *Nosema ceranae* has been found to be infective to bumble bee species in Argentina, the United Kingdom, and China where there is evidence of pathogen spillover (Graystock *et al.* 2013a, pp. 116-117), and has been observed in wild bumble bees in North America as well. New research has found *Nosema ceranae* in both a wild and commercial bumble bee specimen in Canada and therefore YBBB may be significantly at risk for this pathogen (Hicks *et al.* 2018, pp. 1-9).

Aethina tumida

The Small hive beetle (*Aethina tumida*) is a native parasite to the Africanized honey bee (*Apis mellifera scutellata*) that has spread to the European honey bee and was introduced to the United States in the late 1990s. While honey bees were thought to be their primary host, the Small hive beetle is attracted to stored pollen and wax and has become an invasive species in commercial bumble bee colonies as well (Evans *et al.* 2008, pp. 31-32; Goulson *et al.* 2015, p.2). When introduced to common eastern bumble bee colonies the Small hive beetle destroyed the colonies by consuming wax, pollen, and nectar as well as completing their life cycle in the colony while destroying bumble bee brood cells (Colla 2016, p. 419). Small hive beetles were found to readily invade commercial colonies of common eastern bumble bees in North America and there is a risk that it has the potential to impact wild bumble bees nests as well (Spiewok and Neumann 2006, pp. 627-632). The Small hive beetle occurs in many states where the YBBB occurs.

Ascosphaera spp.

Ascosphaera spp. (chalkbrood) is a lethal intestinal fungal parasite and brood disease most commonly found in honey bee pollen stores, honey, larval feces, and nesting material. There are 28 species of the parasite, some of which are pathogenic and are found in a variety of other bee species, including the larvae of the buff tailed bumble bee in Europe (Evison *et al.* 2012, p. 1; Goulson and Hughes 2015, p. 13; Maxfield-Taylor *et al.* 2015, p. 2).

Previously only recorded in honey bees, bumble bees have also been found to carry the spores (Evison *et al.* 2012, p. 3). In 2015, adult bumble bees in the U.S. were observed with the fungus and larval chalkbrood disease was detected (Maxfield-Taylor *et al.* 2015, entire). The fungus was detected on two western U.S. bee species: the Nevada bumble bee, *B. nevadensis* and Yellow faced bumble bee, *B. vosnesenskii*; and one species that is found throughout the United States, the brown belted bumble bee, *B. griseocollis*. The study highlights the potential risks to native bees, including the YBBB, via infected honey bees, honey bee pollen commonly fed to commercially reared bumble bees, and potential pathogen spillover from infected bees (Maxfield-Taylor *et al.* 2015, pp.1-9). However, we have no information to conclude that *Ascosphaera spp.* may be, or if it is, to what extent it is, affecting the YBBB.

Conopid fly parasitism

Conopid flies infect bumble bees by laying eggs inside the bee's abdomen. They can affect colony fitness, reproduction, and worker foraging behavior. The prevalence of conopids has been found to increase over the course of the spring and summer in Canada (Gillespie *et al.* 2010, p. 738). They occur within the range of the YBBB.

High levels of parasitism by conopid flies coupled with *Crithidia bombi* and *Nosema bombi* infection are likely to negatively affect bumble bee populations in Massachusetts (Gillespie *et al.* 2010, pp. 743-745). In western Massachusetts, the infection rate of conopid flies was 20 to 60 percent, which was higher than previous studies conducted in Canada (10 percent), possibly due to differences in study methodologies or study area climatic differences. The parasitism rates of rare bees and common bumble bees were found to be the same and parasitized bees had a shorter

lifespan. Conopids were found to infect more females than males and intermediate-sized bees more than large or small bees.

In Virginia, conopid fly parasitism was found to be higher in more common bee species (common eastern bumble bee; brown belted bumble bee, and two-spotted bumble bee (*B. bimaculatus*). Larger bee body size and the bee species population cycles that closely align with the seasonal patterns of conopid fly attacks may be contributing factors; however additional research is needed to understand the relative risk of conopid fly parasitism among the various species, including the YBBB (Malfi and Roulston 2014, pp. 25-27).

Several pathogens and parasites are present in bumble bee communities in which YBBB are present, sometimes at elevated levels due to pathogen spillover, spillback, and facilitation. The risk of parasite and pathogen outbreaks could have a significant effect on bumble bee species such as the YBBB that may lack an evolved resistance to these and novel infectious diseases. Pathogen spillover may be playing a role in the decline of some YBBB populations, particularly those with reduced effective population size and inbreeding. Several additional stressors are likely acting in combination with pathogens and parasites to cause recent declines in some bumble bee species, including agrochemicals, small population size, competition, habitat loss, nutritional deficiencies, and changing climate (Cameron 2016, p. 4390; Goulson *et al.* 2015, p. 5; Kent *et al.* 2018, entire; Kerr *et al.* 2015, pp. 177-180; Manley *et al.* 2015, p. 2; Potts *et al.* 2010, pp. 348-351).

The YBBB co-evolved with and is host to a cuckoo bumble bee species in the subgenus *Psithyrus*. The main hosts for the Ashton cuckoo bumble bee (*Bombus ashtoni*) are YBBB and RPBB, and the YBBB is a potential host for four additional cuckoo bumble bee species (*B. citrinus*, *B. insularis*, *B. suckleyi*, and *B. variabilis*) (Lhomme and Hines 2018, p. 6-7). These species are obligate social parasites that invade the YBBB nest and remove the queen. They reproduce in the colony and the YBBB workers rear their brood (Lavery and Harder 1988, p. 966; Husband 1980, p. 227). We have no evidence that cuckoo bumble bees have a significant negative effect on YBBB populations, and in fact, the Ashton Cuckoo Bumble Bee, as well as *B. suckleyi* and *B. variabilis*, are declining precipitously and may be linked to the decline of their hosts ((Lhomme and Hines 2018, p. 9).

Table 2. Bee Parasites for which there is evidence of anthropogenic spread to wild bee populations (From: Goulson and Hughes 2015, p. 13)

Parasite	Parasite taxa	Host	Spread to	Pathology	Refs
<i>Apicystis bombi</i>	Neogregarine	Bumblebees	Honeybees?	Parasitizes adult bumblebees. Faecal-oral transmission. Degrades fat body, has neurological affects and can cause mortality. Spillover from commercial bees to wild bumblebees; implicated in bumblebee declines in Argentina. Can infect honeybees.	Liu et al. 1974; Plischuk & Lange 2009; Plischuk et al. 2011; Arbetman et al. 2013; Graystock et al. 2013b; Maharramov et al. 2013; Graystock et al. 2014
<i>Crithidia bombi</i>	Trypanosome	Bumblebees	Bumblebees	Parasitizes adult bees. Faecal-oral transmission and context-dependent virulence. Appears unable to infect honeybees. Spillover from commercial bumblebees to wild bumblebees; implicated in bumblebee declines in Argentina.	Schmid-Hempel 2001; Brown et al. 2003; Graystock et al. 2014; Schmid-Hempel et al. 2014
<i>Nosema bombi</i>	Microsporidian	Bumblebees	Bumblebees	Parasitizes adult bees. Faecal-oral transmission, reducing worker survival and colony fitness. Spillover from commercial bumblebees implicated in bumblebee declines in North America.	Colla et al. 2006; Otti & Schmid-Hempel 2007; Cameron et al. 2011
<i>Nosema ceranae</i>	Microsporidian	Honeybees	Bumblebees	Emerging disease of adult bees. Natural parasite of <i>Apis ceranae</i> , jumped host to <i>Apis mellifera</i> , and then to bumblebees. Spillover from honeybees and commercial bumblebees to wild bumblebees. Can lead to mortality.	Plischuk et al. 2009; Fries 2010; Graystock et al. 2013a; Fürst et al. 2014; Graystock et al. 2014;
<i>Locustacarus buchneri</i>	Mite	Bumblebees	Bumblebees	Tracheal mite, feeding on haemolymph of adult bumblebees. May cause lethargy, altered foraging behaviour and reduced lifespan. Spillover has taken place from commercial bumblebees to wild bumblebees in Japan.	Goka et al. 2000; Otterstatter & Whidden 2004; Otterstatter et al. 2005; Goka et al. 2006;
Deformed wing virus (DWV)	Iflaviridae	Honeybees	Bumblebees Solitary bees?	Parasite of brood and adults. In honeybees, infected brood may develop into adults with deformed wings; infection in adults has neurological affects and can cause mortality. Has been detected in wild and commercially produced bumblebees. In bumblebees, infections of brood can also lead to adults with deformed wings, and infections of adults can cause mortality. Has been detected in solitary bees, but infectivity or pathology unknown	Genersch et al. 2006; Chen & Siede 2007; de Miranda & Genersch 2010; Evison et al. 2012; Fürst et al. 2014; Manley et al. 2015; McMahon et al. 2015
Slow bee paralysis virus (SBPV)	Iflaviridae	Honeybees	Bumblebees?	Causes paralysis of adult honeybees. Has been detected in wild bumblebees, but infectivity or pathology unknown.	Chen & Siede 2007; McMahon et al. 2015
Israeli acute paralysis virus (IAPV)	Dicistroviridae	Honeybees	Bumblebees	Causes paralysis and mortality in adult honeybees. Has been detected in wild bumblebees. Reduces brood production in bumblebees.	Chen & Siede 2007; Singh et al. 2010; Meeus et al. 2014
Acute bee paralysis virus (ABPV)	Dicistroviridae	Honeybees	Bumblebees	Causes paralysis and mortality in adult honeybees. Has been detected in wild bumblebees. Also causes paralysis in bumblebees	Bailey & Gibbs 1964; Meeus et al. 2010; McMahon et al. 2015

Parasite	Parasite taxa	Host	Spread to	Pathology	Refs
Kashmir bee virus (KBV)	Dicistroviridae	Honeybees	Bumblebees	Causes mortality in adult honeybees. Has been detected in wild bumblebees. Delays oviposition and reduces brood production in bumblebees.	Chen & Siede 2007; Meeus et al. 2014
Black queen cell virus (BQCV)	Dicistroviridae	Honeybees	Bumblebees? Solitary bees?	Causes mortality of queen larvae. Has been detected in wild bumblebees and solitary bees, but infectivity or pathology unknown.	Chen & Siede 2007; McMahon et al. 2015; Manley et al. 2015
Sacbrood virus (SBV)	Dicistroviridae	Honeybees	Bumblebees? Solitary bees?	Lethal disease of honeybee larvae. Has been detected in wild bumblebees and solitary bees, but infectivity or pathology unknown.	Chen & Siede 2007; Manley et al. 2015; McMahon et al. 2015
<i>Ascosphaera</i> spp.	Fungus	Honeybees	Bumblebees	Lethal, specialist brood disease (chalkbrood). Also infects solitary bees. Detected in wild bumblebees. Infections reported from adult bumblebees.	Aronstein & Murray 2010; Evison et al. 2012; Maxfield-Taylor et al. 2015;
<i>Aethina tumida</i>	Small hive beetle	Honeybees	Bumblebees	Emerging parasite, spreading from Africa to America, Australasia and Asia over last decade. Larvae feed on honey and pollen, and can destroy colonies. Can parasitize bumblebees	Spiewok & Neumann 2006; Hoffmann et al. 2008

Small and Isolated Populations

Small and geographically isolated populations of bumble bees, and other eusocial insects, are at an increased risk of extinction due to their population and genetic structure, particularly when combined with environmental stressors. These populations are susceptible to greater genetic loss and low genetic variation, and have reduced evolutionary responses to changes in the environment, such as habitat fragmentation, pesticides, pathogens and parasites, and climate (Chapman *et al.* 2001, entire; Tucker and Rehan 2017, p. 8; Whitehorn *et al.* 2011, p. 1195; Whitehorn *et al.* 2014, entire; Zayed and Packer 2005, entire; Zayed 2009, pp. 241-246, entire).

The YBBB colony is founded by a single queen and consists primarily of non-reproductive female workers and relatively few reproductive gynes and males (Macfarlane *et al.*, 1994, pp. 3-4) (see *Chapter 2 - Species Information*). Because only the queens produce offspring, and each colony supports only a single queen, the colony is considered a single reproductive unit. Populations are comprised of numerous colonies within a given area (multicolonial). The effective population size (number of reproductive bees) is determined by the queens and unrelated reproductive males, and is inherently smaller than the census size (number of individual bees) because the majority of the colony are worker bees (Chapman *et al.* 2001, p. 652). Healthy YBBB populations that are composed of numerous colonies maintain genetic diversity when males and queens disperse to mate with unrelated individuals in other populations. When YBBB populations are small or geographically isolated, they are more susceptible to a reduction in fitness due to inbreeding (Zayed 2009, p. 244).

In healthy populations, two mechanisms in bumble bees that minimize inbreeding are the dispersal of males and potential new queens (reproductives) from the natal area prior to mating and the forced dispersal of young males (4-5 days old) prior to gynes from the colony by workers to avoid sibling mating (Darvill *et al.* 2012, p. 3988-3989; Plowright and Pallett 1979, p. 289).

However, the range contraction of several North American bumble bee species, including the YBBB, has led to isolation of populations in areas of their range as dispersing individuals cannot reach neighboring populations. Yellow banded bumble bee dispersal and potentially nest fidelity (queens sometimes preferring to nest close to their natal colonies per Jha and Kremen (2013, p. 2493)) are influenced by changes in landscape composition, topographic features, and climatic conditions. Bees from temperate regions, such as the YBBB, appear to be dependent on habitat quality at the landscape level for reduced inbreeding, lineage survival (queens that survive overwintering and spring emergence), and adequate nesting density (Lopez-Urbe *et al.* 2017, p. 502-504; Carvell *et al.* 2017, p. 547-549; Jha and Kremen 2013, p. 2487-2493). Genetic analysis has found that in eastern Canada the YBBB has reduced genetic diversity due to lower effective population size and is showing signs of inbreeding (Kent *et al.* 2018, entire).

Scientific evidence from New Hampshire provides an example of progressive YBBB population isolation. In that state Jacobson *et al.* (2018, p. 439) report that the relative abundance of the YBBB in the native bee community has declined over the past 150 years from 23.7 percent to 6.8 percent. Historically found throughout the state, the species' range contracted to include only the northern portion of the state through the mid-1900s. By 2016, YBBB occurrence and distribution were greatly reduced from previous levels and restricted to the White Mountain region of the state (Jacobson *et al.* 2018, p. 441; Tucker and Rehan, 2017, p. 8). The range contraction has also included a significant shift in elevation from 61 meters (200 feet) historically to an average of 527 meters (1,729 feet) in 2016 (Jacobson *et al.* 2018, p. 441; Tucker and Rehan, 2017, entire). Restriction of the species to refugia areas at higher elevations could be detrimental to the species' long-term persistence due to reduced gene flow between populations, reduced community fitness, and reduced species viability in these small populations (Tucker and Rehan 2017, p. 8).

A confounding factor for the YBBB's effective population size is its haplodiploidy nature. This genetic sex-determining mechanism occurs in bumble bees and other insects because males are produced from unfertilized eggs and females are produced from fertilized eggs. Female haplodiploid insects have two complete sets of chromosomes, one from each parent (diploid) and males have a single set of unpaired chromosomes (haploid). Consequently, the haploid males have half the number of chromosomes as the diploid females. Because there are fewer genes compared to diploid organisms, haplodiploids have a smaller effective population size and reduced genetic variation (heterozygosity) (Packer and Owen 2001, p. 8; Zayed 2009, p. 239).

A significant genetic risk to small and/or isolated populations of YBBB (i.e., populations with a limited number of connected colonies) is the production of diploid males, and the "diploid male vortex." Diploid males are produced in populations with reduced genetic variation due to sibling mating, inbreeding, and genetic drift (Zayed 2009 p. 241; Whitehorn *et al.* 2009, p. 2). It occurs when a diploid female mates with a haploid male that is a sibling (match-mated), which produces diploid males instead of females (Zayed 2009, p. 239). Diploid males are sterile or have very

low reproductive viability and if they mate, they typically produce sterile offspring. The reduced number of female workers and reproductive bees (new queens and males), slows colony growth and size, and reduces the health of the colony. Fewer reproductive bees and diminished colony size leads to reduced genetic diversity within and among colonies (Chapman *et al.*, 2001, p. 653; Colla 2016, p.413; Whitehorn *et al.* 2009, pp. 2, 5-7).

The diploid male vortex is the rapid deterioration of population growth and size that occurs after the initial production of diploid males. The combination of small population dynamics, environmental stochasticity, and genetic drift continues to reduce genetic variation, leading to an even higher production of diploid males and fewer females in the population. This has the domino vortex effect of a rapid reduction in colony growth rate, fitness, and survival. Negative population growth rates then rapidly occur and ultimately lead to extinction (Zayed 2009, p. 241-243; Zayed and Packer 2005, p. 10743-10745). Diploid males are found in many declining social bee species and are a sensitive indicator of the loss of genetic diversity and increased extinction risk (Zayed *et al.*, 2009, p. 241-243; Darvill *et al.* 2006, p. 608). Inbreeding that has been detected in eastern Canadian populations of YBBB significantly increases their risk of extinction due to the high level of diploid male production and the diploid male vortex estimated to occur in these populations (Kent *et al.* 2018, entire).

Additionally, YBBBs are monoandrous; the queen mates with a single male instead of multiple males, which reduces the amount of genetic variation in each colony and increases susceptibility to inbreeding (Darvill *et al.* 2006, p. 602; Owen and Whidden 2013, p. 524-526). Therefore, small and isolated populations of YBBB are particularly vulnerable to the negative genetic effects of low effective population size, haplodiploidy, and monandry (Chapman *et al.* 2001, entire; Zayed *et al.* 2009, entire), which can be exacerbated by multiple environmental stressors such as habitat fragmentation, increasing temperatures, pesticides, and pathogens and parasites.

Bee populations and communities commonly experience large annual fluctuations in population size (multiple sources in Murray *et al.* 2009, pp. 211-212), which exposes small and/or isolated populations to an increased risk of extirpation due to natural variation in environmental conditions (USFWS 2016, p.51). However, even when suitable habitat is present and conditions are favorable, populations may become extirpated or extinct because of stochastic events and natural catastrophes, such as drought, floods, and fires (USFWS 2016, p.51). Small and isolated populations are more susceptible to these events than larger populations with higher levels of connectivity. The events can exacerbate each other and be more likely to cause extirpation or extinction in small populations (Shaffer 1981, p. 131).

Populations of the YBBB that are small and/or isolated populations are at risk of having reduced gene flow, reduced genetic diversity from inbreeding, and genetic drift. This is due to their smaller effective population size (number of reproductive bees) and haplodiploidy genetic structure (lower genetic diversity), and is exacerbated by multiple environmental stressors. Reduced genetic diversity also leads to populations being less adaptable to ongoing environmental pressures such as habitat fragmentation, increasing temperatures, pathogens and parasites, and pesticides and therefore at a higher risk of extinction.

The Effects of Climate Change

Changes in climate that may affect the YBBB include changes in temperature and precipitation patterns (e.g., early snow melt, late frost events, increasing drought, and highly variable or extreme temperatures). These changes can directly affect individual bees or disrupt the colony life cycle (i.e., timing of emergence and overwintering, the ability to survive hibernation, and the timing of mating and colony production). The effects of climate change may also affect the habitat conditions required for nesting and overwintering (e.g. soil moisture, humidity, and temperature). They can disrupt community interactions and bumble bee food resources such as the timing, quality, and quantity of flowering resources, plant species composition, and competition for habitat (Hatfield *et al.*, 2015, p. 5; Inouye, 2008, pp. 358-361; see citations in Cameron *et al.*, 2011b, pp. 35-40; Ogilvie *et al.* 2017, entire; Witmer *et al.*, 2012, p. 208).

Bumble bees originated and diversified in cool temperate latitudes and are a cold adapted, largely alpine group. This evolutionary history may limit the YBBBs' ability to adapt to warming temperatures (Hines, 2008, p. 58-59; Kerr *et al.* 2015, p. 179; Oyen *et al.*, 2016, p. 55). Individual YBBBs are well-insulated, with longer hairs, and are physiologically adapted to pollinate in cooler conditions. They are known to be active and fly when air temperatures are near or below 0 °C (32 °F) (Heinrich 1972, p. 49). When temperatures increase and exceed 30°C (86 °F), their activity declines and they may be susceptible to overheating, even in moderately warm conditions (Heinrich 1976a, pp. 882, 886; D. Goulson, pers.comm., 2018; Oyen *et al.* 2016, p. 55).

The body temperature of the YBBB is dependent on external air temperature; however, they do generate some internal heat when active. During extreme warming events, bumble bees make behavioral adjustments to thermoregulate, such as seeking out shade, burrows, or wet cool places. However, the physiological stress and energy requirements of these adjustments are costly and can reduce individual fitness and survival (Kerr *et al.* 2015, p. 179; Sunday *et al.* 2014, pp. 5613-5614). Larger bees generate the most heat, so queens and large workers are especially susceptible to overheating, particularly when they are in flight and foraging (Goulson 2010, p.14-17; Oyen *et al.* 2016, p. 53). Bumble bee nests also require thermoregulation to maintain the narrow temperature range required for brood growth and development (28 to 32 °C / 82 to 89 °F). During extreme conditions, worker bees use their wings to fan air in the nest, but this behavioral response may limit individual workers from meeting other colony demands such as foraging or brood care, which can slow or cease. Additionally, individual workers may experience physiological stress as they attempt to cool the nest. Therefore, even with nest thermoregulation, colony fitness may be reduced during periods of high temperatures (O'Donnell and Foster 2001, pp. 397-398).

Evidence of ongoing climate change is apparent within the range of the YBBB. Accelerated warming of the atmosphere in the United States has resulted in the warmest decades on record since 1979 and over the past 20 years the number of high temperature records have far exceeded the number of low temperature records (Reidmiller *et al.* 2018, pp. 64-66). Since the early 1900s, the length of the frost-free season, the time between the last freeze in the spring to the first freeze in the fall, has also increased (Reidmiller *et al.* 2018, p. 64). These trends are predicted to continue in the future regardless of the climate change projection model (Reidmiller *et al.* 2018,

pp. 64-66). Average annual temperatures are expected to continue increasing, and record-setting hot years will become more common (Reidmiller *et al.* 2018, pp. 64-66). Extreme high temperatures are projected to increase even more than average temperatures, with heat waves becoming more intense (Reidmiller *et al.* 2018, pp. 64-66). These changes are likely to have significant consequences for the cold adapted YBBB. As the southern portion of the species' range contracts, and in some areas are now relegated to higher elevation habitats due to increasing temperatures, the species will become more reliant on the availability of suitable habitat in the northern portion of their range. However, temperature projections indicate a proportionally greater increase in temperature in the northern part of the YBBB's range as compared to the southern part (Reidmiller *et al.* 2018, pp. 64-66, 91), which may affect the viability of the species.

Between 1901 and 2010, bumble bee species in North America and Europe demonstrated range losses of approximately 300 km (186 miles) along the southern limits of the species' ranges (Colla 2016, p. 421; Kerr *et al.*, 2015, pp.177-180). This contraction of the historical ranges of bumble bees is attributed to the effects of a warming climate and their ancestral origins in cool temperate conditions. However, as temperatures have increased many bumble bees have not expanded the northern limits of their ranges (Colla 2016, p. 421; Hines 2008, entire; Kerr *et al.*, 2015, pp.177-180). The capacity of bumble bees to colonize previously unoccupied areas and to maintain new populations in previously unoccupied areas may be insufficient, possibly due in part to floral resource gaps (Kerr *et al.*, 2015, p. 179; Ogilvie *et al.* 2017, p. 2-7) as well as limited nesting and overwintering sites, and niche availability in the native bee community.

The YBBB has declined steeply in the southern portions of its range. While the species persists in the northern areas of its historical range, it has not expanded its range northward as temperatures increase (Bartomeus *et al.* 2013, pp. 4656-4657; Colla 2016, p. 420; Kerr *et al.*, 2015, pp. 177-180). In New Hampshire, the range of YBBB has contracted from a statewide distribution to the higher elevations of the White Mountains regions (Jacobson *et al.* 2018, pp. 441-442; Tucker and Rehan, 2016, entire). This is consistent with additional research that documents a shift of some bumble bee species ranges to higher elevations when southern portions of the range have been lost, rather than expanding their ranges toward northern latitudes (Jacobson *et al.* 2018, p. 442; Ploquin *et al.* 2013, entire; Pyke *et al.* 2016, entire). A reduction in the synchrony between the bumble bee life cycle and the peak flowering of plants at these elevations has also been observed to reduce bumble bee production and abundance (Pyke *et al.* 2016, pp.15-16). Range contraction, limited suitable bumble bee habitat on mountaintops, reduced land surface area and floral resources with increasing altitude, and competition with high altitude bumble bee species could have negative effects on the YBBB including isolation or loss of populations, reduced genetic diversity, and inbreeding. (See *Small and Isolated Populations* section above for details).

Changes in temperature and precipitation alter the bloom time, abundance, and diversity of floral resources that are required at critical times of the YBBB's life cycle, such as after spring emergence and before winter hibernation (See *Chapter 2 - Habitat Needs* and *Chapter 3 - Habitat Loss* section above for details). Warmer springs can result in plants blooming earlier, before the queens have emerged from hibernation. Late spring frost events often damage or kill spring flowers that have bloomed early. This reduces the availability of nectar and pollen for

queens during the critical period when they have emerged and are establishing nests. There is an increased risk of decreased health or starvation of the queen, particularly when temperatures are higher and body functions and energy expenditure are higher. Plants that are not pollinated due to a decrease in bumble bee abundance and out of sync phenology will not produce seed and the recruitment of important floral resources in subsequent years gradually declines (Abu-Asab *et al.* 2001, pp. 598-611; Aldridge *et al.* 2011, entire; Inouye 2008, entire; Memmott *et al.* 2007, pp. 713-714; Vesterlund and Sorvari 2014, pp. 217-219). The increase in warm periods during mid-winter in temperate areas, followed by more seasonal cool temperatures can be potentially harmful to overwintering females because warm temperatures can increase their metabolism and increase the risk of starvation, particularly if the winter lasts a normal long period (J. Gibbs pers. comm. 2018). Warm temperatures that extend later into the fall season may delay hibernation of new queens. As they continue to forage, fewer floral resources are available because the late fall blooming plant species have bloomed and died (e.g., goldenrod and asters). These plant species are critical for the winter survival of new queens; however, they are declining in abundance in parts of their range as a result of warming temperatures (Willis *et al.* 2008, pp. 17030-17031).

Changes in climate and phenological shifts that decrease floral diversity and abundance reduce the diet breadth necessary for healthy bumble bee colonies. While YBBB has been observed foraging on a variety of plant species, bumble bees have different nutritional requirements and dietary preferences (Forrest *et al.*, pp. 438-439; Memmott *et al.* 2007, pp. 712-714). Pollen from some plants is more nutritionally valuable than others and negative health effects can result from a lack of diverse foraging options and diets throughout the bumble bee's long active season (Colla 2016, pp. 413-417; Genissel *et al.* 2002, pp. 333-335; Tasei and Aupinel 2008, pp. 397-398, 405; Sutcliffe and Plowright 1988, pp. 1053-1057; Vaudo *et al.* 2015, pp. 134-137). Bumble bee colony development, growth, and fitness have been found to vary by species when fed the same diet under the same conditions. Pollen type (plant species) and the varying nutritional quality and quantity of various plant pollens can significantly affect colony growth (Moerman *et al.* 2016, pp. 6-10). The pollen type that is collected by bumble bees can also vary by species depending on population trends and life history traits, and declining bumble bee species may be especially susceptible to decreases in preferred plant foods (Williams and Osborne 2009, p. 376; Kleijn and Raemakers 2008, pp. 1818-1821).

In the northern Canadian ecoregions and northeastern U.S. forests of the YBBB's range, temperatures are higher, trees are blooming earlier, treeline is increasing in elevation, and drought is stressing the native forests (Williamson *et al.*, 2009, pp. v-20; Rustad *et al.* 2012, pp. 15-21). If plant and wildlife species move northward in latitude, the potential establishment of new niches and occupation by those species are expected to lag significantly, occurring at a slower rate than the rate of future climate change. This is because the rate of dispersal and establishment of vegetation and wildlife are not based solely on temperature and precipitation. Soil conditions, day length, the rate at which plant species migrate, competition with native species that persist in the habitat, and because some species require other species' presence to function all influence the migration rate and successful establishment of southern plant species in more northerly latitudes (Williamson *et al.*, 2009, pp. v-20; Rustad *et al.* 2012, pp. 15-21). Given the YBBB's lack of ability, to date, to shift northward, these trends in plant migration rates will further compound the asynchronies in bee/plant phenology.

In addition to temperature change, the frequency and intensity of heavy precipitation has increased in parts of the YBBB range in the United States and spring precipitation is projected to increase 10 to 15 percent between the 1990s and 2050 (Reidmiller *et al.* 2018, pp. 68-71). However, soil surface moisture is expected to decrease due to warmer temperatures, increased evaporation rates, and fewer soaking rain events where steady precipitation soaks into the ground instead of running off. Longer and more frequent droughts, and more severe heavy precipitation events, are becoming more common across the species' range and these trends are projected to continue into the future (Reidmiller *et al.* 2018, pp. 68-71, 972; National Oceanic and Atmospheric Administration, 2017). Drought has been shown to result in fewer flowers and fewer flowers containing nectar. Drought can also reduce flower size, number of flowers, and reduced production of pollen and nectar (Phillips *et al.* 2017, pp. 3326-3327 and 3230-3233). Both heavy rain events and drought conditions are stressors that could negatively affect YBBB survival because the queens overwinter and colonies nest underground, both of which can be affected by flood and drought conditions, as has been seen in rodent burrows systems that become flooded and can develop high humidity microclimates (Witmer *et al.* 2012, p. 8).

The ability of a species to adapt to changes in climate is influenced by its genetic diversity, which may already be reduced or at risk of being reduced by non-climate related stressors, such as habitat fragmentation, small population size, and disease. While genetic diversity and adaptability vary by species, environmental change often occurs too rapidly for many species to adapt (Reidmiller *et al.* 2018, pp. 261-262, 270; Zayed 2009 p. 246). Due to their population and genetic structure, bumble bees are at a higher risk of losing genetic diversity and several North American bumble bee species that are in decline have significantly lower genetic diversity when compared to bumble bee species that are stable (non-declining). Additionally, small bee populations are at an even higher risk of reduced genetic diversity, inbreeding depression, and the diploid male vortex and once these have occurred are believed to have limited potential evolutionary responses to changes in their environment, such as the effects of climate change (Cameron *et al.* 2011a, pp. 662, 664-665; Zayed 2009, entire). In summary, the effects of climate change in combination with multiple stressors such as habitat fragmentation, small population size, pesticides, pathogens and parasites is likely impacting the YBBB and will continue to add to increased stress in the future (Goulson *et al.*, 2015, pp. 4-5; Potts *et al.* 2010, p. 351; Vanbergen *et al.* 2013, pp. 251-258; Williams and Osborn 2009, p. 371).

Scientific Collection

Scientific assessments and inventories of bumble bees traditionally use lethal collection protocols that could affect small or isolated populations of declining or otherwise at-risk species, such as the YBBB. Of particular concern is the lethal collection of spring queens and fall reproductives (males and gynes), which has a higher likelihood of affecting local bumble bee populations than the collection of individual worker bees (Droege *et al.*, 2017, pp. 2-3; Gezon *et al.* 2015, p. 1050; Gibbs *et al.* 2017, pp. 2, 7-9; Shepherd *et al.* 2003, p.48; Minter *et al.* 2014a, pp. 260-261, Minter *et al.* 2014b, p. 816). Spring queens emerge and forage heavily to produce offspring for the new colony. The gynes and males disperse from the nest in late summer and early fall to mate. The new fall queens forage heavily until they hibernate and are the basis for the following year's bumble bee population. Therefore, the effects of collection on colony health (e.g. the collection of spring queens and fall reproductives) are more detrimental to a colony than

the collection of an individual worker bumble bee (e.g. the number of workers collected). Healthy bumble bee populations are composed of numerous colonies within a given area (multicolonial) (Chapman *et al.* 2001, p. 652), therefore populations that are isolated due to habitat fragmentation and range contraction may also be vulnerable to the effects of collection. However, non-lethal protocols for conducting bumble bee surveys and for the collection of samples for genetic testing have been developed and their use is encouraged in areas where at-risk bumble bees may be present. Balancing the benefits of improved scientific understanding against the effects of sample collection, especially to vulnerable wildlife populations, continues to be a complex issue (Minteer *et al.* 2014a, pp. 260-261; Rocha *et al.* 2014, pp. 814-815; Krell and Wheeler 2014, pp. 815-816; Minteer *et al.* 2014b, p. 816).

Environmental monitoring can include the collection and killing of biological specimens to gather information on the presence, absence, abundance, and distribution of species; identify new species; document biodiversity; assess the evolution of species; conduct genetic assessments; and monitor changes in species' health. Modern scientific collection can require adherence to permitting regulations, and follow ethics guidance (e.g., Kent *et al.* 2018, p. 10). Consideration of impacts to the species and limiting the number of collected specimens to substantially below what could affect a species' population are often part of a monitoring plan. In general, collections are more often made to monitor a species' health, evolution, phenotype, and/or morphological diversity, rather than solely for species identification.

The purpose of many bee monitoring programs is to inventory all bee species in a given area and to assess long-term population trends of the bee community. However, some bee research is focused on hypotheses where the results have a broader application, such as evaluating the impact of an anthropogenic stressor on bee populations in a representative location, the response of bee populations to individual restoration actions for future management application, and the survey of bee communities in representative habitat areas and applied to comparable habitat areas (Tepedino *et al.* 2015, pp. 281-282). These assessments and inventories traditionally use survey methods that are lethal to the captured bees so they can be taken back to a laboratory for positive identification and/or further processing. This is necessary for bee species that are difficult to identify without the aid of a microscope (Tepedino & Stanton 1981, entire; Gezon *et al.* 2015, p. 1044). However, bumble bees are relatively large and are more easily identified without a microscope based on their size, pile coloration and markings, and other physical features (Droege *et al.*, p. 3; Gezon *et al.* 2015, p. 1046). Stored specimens can be used for future reference or for genetic analysis. For studies of bumble bee genetics, however, non-lethal genetic sampling techniques have been developed. The collection of a small piece of appendage, excreta (feces), or fluid (hemolymph) have little to no effect on the performance of wild bumble bees and are equally effective for conducting genetic analysis (Holehouse *et al.* 2003, pp. 277-284; Scriven *et al.* 2013, p. 228; Droege *et al.* 2017, p. 76).

A variety of lethal trapping and netting methods are used to collect bees, such as vane traps, malaise traps, pan traps set at ground-level, and elevated pan traps. These methods allow for the collection of large numbers of all bee species during each sampling event and at the same site repeatedly throughout the bee flying season and sometimes for multiple years (Gezon *et al.*, 2015, p. 1045). The predominant bee species that are captured by traps can vary depending on the trap type and the methods used (Geroff *et al.* 2014, pp. 951-952). For example, bumble bees

were captured at much higher rates in elevated pan traps than in ground-level pan traps, possibly due to a difference in foraging height (Geroff *et al.* 2014, pp. 961-962). Vane traps, such as Japanese beetle traps, are popular due to their ease of use and effectiveness in capturing large numbers of insects (Ptasznik *et al.*, 2015, p.6). They attract all bees and are used to inventory bee communities. However, bumble bees are highly attracted to vane traps and are highly susceptible to capture (Geroff *et al.*, 2014, pp. 961-962; Gibbs *et al.*, 2017, pp. 2-3; Kimoto *et al.* 2012, pp. 7-12, 20, 23; Ptaszniak 2015, pp. 27-28). Vane traps have been found to capture more *Bombus* spp. (bumble bees) than all of the other trap types combined (Geroff *et al.*, pp. 952-963). Bumble bees made up 62.1 percent of all bees genera captured in vane traps (Stephen and Rao 2005, p. 375). Fourteen species of bumble bees, primarily queens and workers, dominated vane trap captures; and the number of bumble bee species captured declined in the second year of study (Kimoto *et al.* 2012, pp. 7-12, 20, 23).

While lethal bee sampling conducted once every two weeks using ground-level pan traps and netting found that bee community structure was not affected, the effects of more frequent sampling (weekly or daily), and the declines in the abundance of individual bee species over an entire sampling season, have not been assessed (Gezon *et al.* 2015, pp. 1044, 1049-1052). Likewise, the effects of lethal collection on the YBBB has not been assessed, however the loss of queens foraging in the spring before establishing new colonies and in the fall before mating and hibernating may be particularly important to small or isolated populations. Particularly in parts of their range where population genetic analysis of YBBB has found major reductions in their effective population size and indications that genetic inbreeding is occurring (Kent *et al.* 2018, entire).

With rising awareness of bumble bee conservation, many citizen science bumble bee monitoring programs use non-lethal protocols (e.g. net collections) and photography to collect survey data. Images of bumble bees and other related data are submitted electronically for expert identification or verification (Bug Guide at bugguide.net 2018; Bumble Bee Watch 2018; Donovall and vanEngelsdorp 2008, entire; iNaturalist 2018; Ohio State University 2018; University of Illinois 2018; University of Minnesota 2018; Vermont Center for Ecostudies 2018; and Wisconsin Bumble Bee Brigade. <http://wiatri.net/inventory/bbb/>). Techniques and detailed instructions have been developed for capturing, preparing, and photographing the bees so that photography will capture subtle distinctive features and for reliable species identification. The Service developed a national protocol framework for the inventory and monitoring of bees on National Wildlife Refuges (NWRs) which may also be used by other organizations and individuals to monitor bees in any given habitat or location (Droege *et al.* 2017, pp. 70-79). The protocol framework provides all of the most common lethal sampling methods, as well as a non-lethal sampling method, Standard Operating Procedure 7 (SOP 7). Standard Operating Procedure 7 is provided for those NWRs that have bees that are listed as threatened or endangered. The framework suggests that the non-lethal protocol may also be warranted if there is the possibility of impacting populations of rare species or vulnerable life stages, such as newly emergent spring queens and late summer-fall males and queens foraging before winter hibernation (Droege *et al.* 2017, pp. 2, 70-79). The non-lethal protocol is one of many options that are available for use and it is unknown if NWRs within the range of the YBBB will choose to use the non-lethal method. However, in 2018, four NWRs in the Service's Northeast region are conducting inventories of bumble bee species to measure species richness and relative

abundance, and investigate bumble bee relationships with habitat variables using a non-lethal survey method adapted from the USFWS National Protocol Framework for the Inventory and Monitoring of Bees (Droege *et al.*, 2017, pp. 70-77) and the USFWS Rusty-Patched Bumble Bee Survey Protocol (USFWS 2018, entire). Hand-netting will be conducted three times during the growing season (June - August) and photographic vouchers of bumble bee species for identification will be taken in the field (R. Longenecker, pers. comm., 2018).

Competition

While the agriculture industry relies heavily on the use of managed bees (commercial honey bees and bumble bees) for crop pollination, the increased use of managed bees over the last 30 years comes with negative ecological effects to wild bumble bees (Goulson 2003, p. 2; Evans 2017, entire; Sutherland *et al.* 2017, p. 34). Most commercially reared bumble bees are produced in Canada and Europe and traded internationally to service markets all over the world (Evans 2017, p. 35). The consequences of the booming international commercial bumble bee trade for agricultural pollination and the global trade of bumble bees are among the top 15 emerging environmental issues likely to affect global diversity due to the lack of coordinated, international measures to prevent species invasions (Aizen *et al.* 2018, pp. 1-5)

The common eastern bumble bee and the European honey bee are regularly used in the U.S. for the pollination of crops such as tomatoes, sweet peppers, blueberries, and strawberries (Velthuis and van Doorn 2006, pg. 431; Geldmann and González-Varo 2018, p. 392). The common Eastern bumble bee, which is native to eastern North America, is reared in eastern Canadian and north central U.S. production facilities (Cameron *et al.* 2016, pp. 4386, 4389). The European honey bee, which is not native to North America, was imported to the U.S. by the early colonists and the beekeeping industry has grown to approximately 2.78 million colonies in 2016. They are bred and distributed extensively throughout the country, however new strains of the European honey bee are periodically imported to the U.S. from Europe (USDA 2016). Managed bees unavoidably escape from hives (Morandin *et al.* 2001, entire; Goulson *et al.* 2002a, pp. 270-271; Goulson *et al.* 2008, p. 195; Ings *et al.* 2006, p. 941). These bees compete with wild bee species for floral resources and escaped managed bumble bees also compete for nesting and overwintering habitat (Goulson 2003, p. 2; Dafni *et al.* 2010, pp. 103-104; Bushmann *et al.* 2012, p. 9; Evans 2017, p. 37). Native bumble bees may be vulnerable to competitive displacement (Ings *et al.* 2006, entire; Morales *et al.* 2013, p. 529) because wild *Bombus* species have a similar ecological role to managed bees and a high level of overlap in plant resources has been documented (Matsumura *et al.* 2004, Thomson 2006). These negative impacts have led managed bee invasions and disease to be listed as a top emerging concern for the conservation of biodiversity (Aizen *et al.* 2018, p. 1-5; Evans 2017 p. 35; Sutherland *et al.* 2017, p. 34). For the purposes of this section we focus on the impacts of managed bumble bees on wild bumble bees and, where applicable, YBBBs.

The extent of competition with managed bumble bees and honey bees, and resulting impacts on YBBBs and other wild bee populations is likely to vary with the density of the managed bees (Cane and Tepedino 2017, p. 208). The competitive effects have been shown to be strongest closer to managed colonies (typically < 800 m (2,625 feet)), with reduced or no effects as distances increased to 1200 m (3,937 feet) (Thomson 2004, entire; Thomson 2006, entire;

Elbgami *et al.* 2014; entire). This suggests that the impacts of managed bees on wild bumble bees could be relatively localized (<1 km (0.62 miles) from sources of managed bees). While it is clear that managed honey and bumble bees compete with wild *Bombus* species for resources, managed bees and wild bumble bees may be able to coexist in areas where resources are plentiful (Mallinger *et al.* 2017, p. 25). However, in agricultural landscapes, as well as in urban and suburban settings, both floral resources (especially outside of the monoculture bloom period) and nesting and overwintering resources could be limited (Hines and Hendrix 2005, p. 1481). Additionally, honey bee colonies are increasingly being placed into natural areas where there is better forage and less pesticide use, which may be a particular risk to wild bees (S. Colla, pers comm., 2018). In an area where wild *Bombus* species co-occur with managed honey bees in the United Kingdom, wild *Bombus* foragers tended to be smaller as a result of competition with honey bees (Goulson and Sparrow 2009, p. 180). Furthermore, the WBB foraging rates were shown to be lower when in close proximity to managed honey bee hives (Thomson 2004, pp. 467-468).

Managed bumble bees are captively reared year round and colonies forage at any time of the year if food is available or provided by producers. If they are produced for an earlier seasonal emergence, they may deplete early spring resources before wild *Bombus* species emerge and establish nests. Managed bumble bees with larger bodies are also able to thermoregulate at lower temperature and therefore utilize floral resources for longer foraging periods than native bumble bees (Goulson 2003, p. 9; Dafni *et al.* 2010, p. 103). This can create an uneven distribution of resources in favor of managed bumble bee species (Goulson 2003, p. 9). While the YBBB is typically the earliest emerging *Bombus* species within its range, if managed common eastern bumble bees are emerging earlier than their naturally occurring conspecifics, this could increase competition for resources at a critical time in the YBBB life cycle. Furthermore, competition with commercial bumble bees for pollen and nectar may cause changes in wild bumble bee floral use and niche breadth (range of resources utilized by a species), resulting in changes in wild bumble bee fitness (Mallinger *et al.* 2017, p. 2).

In addition to competition over floral resources, when managed bumble bees escape, become established, and reproduce in the wild, they compete with wild bumble bees for nesting locations and overwintering habitat (Matsumura *et al.* 2004, pp. 59-60; Dafni *et al.* 2010, entire; Bushmann *et al.* 2012, p. 9; Mallinger *et al.* 2017, p. 2). Both the YBBB and the common eastern bumble bee typically nest in underground cavities. Nesting habitat may be a limiting factor for bumble bees due to long search times required to locate suitable sites, low levels of natural sites, niche overlap with other bee species, and high frequency of nest usurpation by other bee species (Hines and Hendrix 2005, p. 1481; Richards 1978, p. 315). Active nest invasions and usurpations by introduced *Bombus* species have been noted (Matsumura *et al.* 2004, pp. 59-60; Inouye *et al.* 2008, p. 142; Dafni *et al.* 2010, p. 104).

Once established, the high reproductive success of managed bumble bees may permit them to become established, spread, and outcompete native bumble bees (Ings *et al.* 2006, p. 940, 946), with one study showing that the buff tailed bumble bee had four times the reproductive output of native bumble bee species. Managed bumble bee colonies produce more gynes and males than native colonies under similar rearing and field placement conditions, which enables them to reproduce rapidly in the environment (Ings *et al.* 2006, p. 946). Artificially shorter hibernation

periods that are selected for in commercial populations may result in higher gyne production (Beekman *et al.* 1998, entire; Ings *et al.* 2006, p. 946). In addition, managed bees can negatively impact wild bumble bee reproduction; proximity to higher densities of honey bees was shown to reduce WBB reproductive success (measured as gyne number, gyne ratio, mean gyne size, and male sightings (Thomson 2004, pp. 464-467). Managed bees can have further negative impacts on native bee populations due to mating disruption (Evans 2017, p. 37), where native queens will mate with introduced males resulting in offspring that are not viable (Goka 2010, p. 3; Tsuchida *et al.* 2010, entire). Similarly, there is the risk of hybridization and introgression that will potentially modify wild bumble bee communities and their natural genetic diversity, leading to the displacement of the wild species by hybrids or the introduced species (Ings *et al.* 2006, p. 941; Tsuchida *et al.* 2010, entire).

Given the expanding range of the common eastern bumble bee resulting from its commercial use and the current and historical distribution for the YBBB (Colla and Packer 2008, entire; Ratti and Colla 2010, entire; Cameron *et al.* 2011a, p. 663), the YBBB is competing with the managed common eastern bumble bee through a considerable portion of its current range. These areas include New England, the Great Lakes Region, Ontario, and British Columbia (Colla and Packer 2008, entire; Ratti and Colla 2010, entire; Cameron *et al.* 2011a, p. 663), although the negative effects of competition on the YBBB can be expected wherever their range overlaps with agriculture and the use of managed bumble bees. Artificially elevated and expanded populations of the naturally occurring common eastern bumble bee are likely to exacerbate the effects of competition on an already declining YBBB population. However, while we are confident that competition with managed bumble bees can have negative impacts on native *Bombus* species, it is unclear to what extent this is occurring with YBBB and the common eastern bumble bee in the portions of their range where they naturally co-occur.

Synergistic Effects

The YBBB is exposed to a variety of stressors that can interact to affect the species synergistically, meaning that the effects of two or more stressors are more harmful than the sum of the effects of each stressor acting alone. Synergistic effects of multiple stressors have been observed to be more harmful than one stressor alone in many wildlife species, including bumble bees (Coors and DeMeester 2008, pp. 1822-1826; Gill *et al.* 2012, entire; Goulson *et al.* 2015, p. 5; Sih *et al.* 2004; Potts *et al.* 2010, p. 349) (figure 10). Research indicates that several significant interacting stressors are acting in combination to cause recent declines in some bumble bee species, including agrochemicals, small population size, pathogens, habitat loss, diet, and changing climate (Cameron 2016, p. 4390; Colla and Packer 2008, p. 1388; Goulson *et al.* 2015, p. 5; Kerr *et al.* 2015, pp. 177-180; Manley *et al.* 2015, p. 2; Potts *et al.* 2010, pp. 348-351).

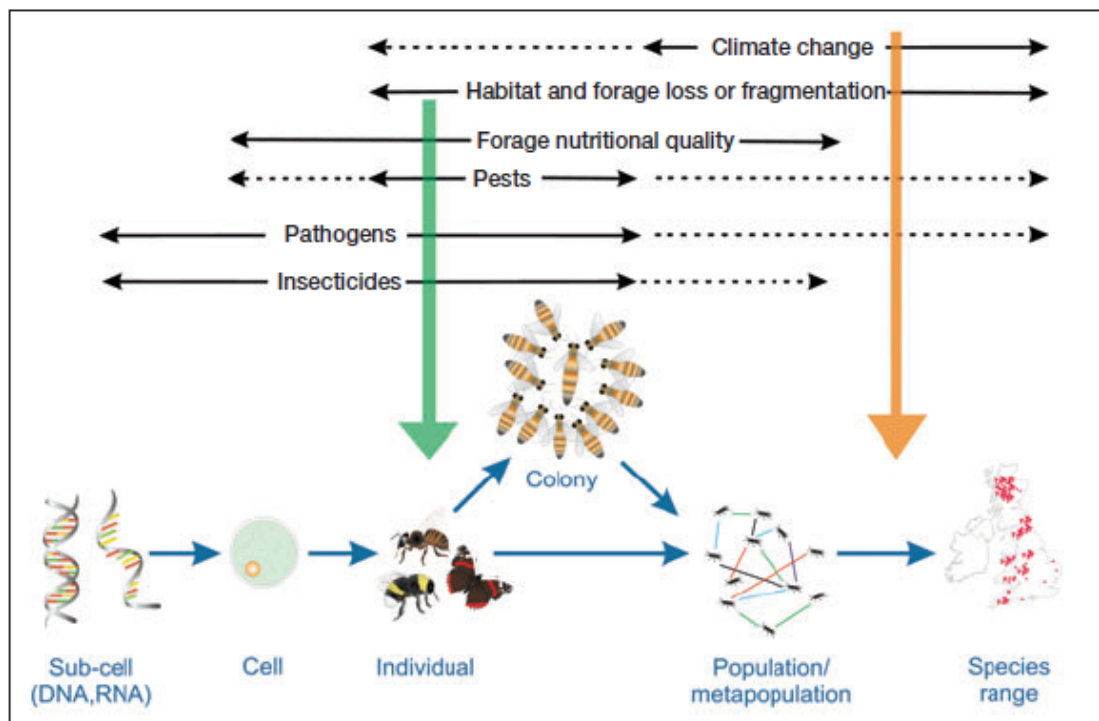


Figure 10: The impact of multiple stressors (black text) on pollinator species across levels of biological organization (blue text). Black arrows span the levels at which each stressor has direct (solid) and indirect (dotted) effects. Vertical arrows show the most practical scale at which to study interactions between pressures. Green arrow = pesticide–pathogen–nutrition interactions at individual or colony scales; orange arrow = climate change–habitat interactions at population or species scale. (From: Vanbergen *et al.* 2013)

Habitat loss combined with the effects of climate change increases nutritional stress and bumble bee susceptibility to pathogens and parasites. This is due to reduced quality and quantity of floral resources at a landscape level as well as changes in the timing of availability of floral resources (Vanbergen *et al.* 2013, pp. 254-256). Nutritional stress in bumble bees has been shown to dramatically reduce their survival of parasitic infections and result in elevated mortality rates (Brown *et al.* 2000, pp. 425-426). Bumble bees experience increased physiological stress

due to environmental degradation, such as loss of floral and nesting resources, presence of pesticides, and the effects of climate change, which likely increases the effects of pathogens (Cameron *et al.* 2016, p. 4390).

The synergistic effects of nutritional stress, pathogens and parasites, and pesticides have been shown to impact learning and memory in bumble bees which can lead to reduced foraging efficiency, increased competition, and decline in colony fitness (Colla *et al.* 2016, p. 413). Habitat loss and fragmentation reduce floral resource availability and magnify nutritional stress on bees. The YBBB forages at a variety of flower species that differ in morphology, scent, color, and quantity and quality of pollen and nectar rewards. They forage efficiently by learning and remembering the floral resources that are available across the landscape over the course of the season. Bumble bees deprived of adequate amounts of protein (pollen) can experience an immune-induced reduction in memory and reduced ability to learn when their immune system is stimulated (Riddell and Mallon 2005, pp. 136-139). Pathogens and parasites can also trigger an immune response in bumble bees that impairs their ability to learn. The immune-induced impairment reduces memory and the ability to learn the color of pollen and nectar rich flowers, which can lead to reduced foraging efficiency and nutritional stress on the colony (Alghamdi *et al.* 2006, pp. 479-481). Activation of the bumble bee immune response also increases food consumption, which can further increase exposure to pesticides (Goulson *et al.* 2015, p. 5). Chronic exposure to pesticides can result in memory impairment, reduced learning speed, significantly impaired short-term memory, and deficits in navigation, foraging, and colony growth (Goulson *et al.* 2015, p. 5; Stanley *et al.* 2015, pp. 4-6; Moffat *et al.* 2015, pp. 5-7).

The YBBB is exposed to several classes of pesticides while foraging. The combination of pesticides, such as neonicotinoids and pyrethroids, causes higher bumble bee mortality and colony failure than exposure to a single pesticide (Gill *et al.* 2012, entire). Fungicides are typically the most abundant pesticide detected in bumble bees. Fungicides can act synergistically with insecticides (neonicotinoids and pyrethroids) and increase the toxicity of the insecticides (Botias *et al.* 2017, pp. 73-81; Goulson *et al.* 2015, p.5).

The interaction of pesticides and pathogens can lead to an increase in bumble bee susceptibility to disease and an increase in the effects of disease and parasites on bumble bee survival. The interactive effects of pesticides and parasites can cause a reduction in bumble bee queen survival and worker lifespan, leading to reduced colony fitness and success. Fungicide and insecticide exposure can result in a higher prevalence of pathogens, such as *Nosema sp.*, as well as impaired immune function and increased pathogen-induced mortality (Fauser-Missilin *et al.*, pp. 455-457; Goulson *et al.* 2015, p. 5; Pettis *et al.* 2013, pp. 4-7; Pettis *et al.* 2012 pp. 155-157; Aufavre *et al.*, 2012, pp. 3-5). Exposure to the fungicide Chlorothalonil was the strongest predictor of the prevalence of the pathogen *Nosema bombi* in four declining bumble bee species, including the YBBB (McArt *et al.* 2017, pp. 1-7).

Reduced genetic diversity increases the vulnerability of bumble bee populations to infectious diseases, pesticides, and nutritional stress (Goulson and Darvil 2008, pp. 196-197; Graystock *et al.*, 2016, p. 65). Compared to bumble bee species with stable populations, two closely related species (the American bumble bee and the WBB) with declining populations and lower genetic diversity were found to have significantly higher infection levels of the pathogen *N. bombi*

(Cameron *et al.* 2011a, pp. 664-666). Inbreeding and reduced genetic diversity in small and isolated populations caused by habitat fragmentation reduce bumble bee resistance to parasites and infection, which in turn decreases colony fitness and increases mortality (Colla 2016, p. 421; Whitehorn *et al.* 2011, pp. 1199-1200; Whitehorn *et al.* 2014, pp. 670-672). Specifically, they have been found to be more susceptible to parasitic tracheal mites and *Nosema sp.* infections (Gillespie 2010, pp. 744-745; Whitehorn *et al.* 2014, p. 670). Additionally, Yellow banded bumble bee populations in eastern Canada have genetic signatures that indicate that pathogen spillover has played a role in their decline in combination with reduced effective population size and inbreeding (Kent *et al.* 2018, entire).

Changes in habitat, landscape features and composition, topographic features, and climatic conditions influence patterns of YBBB dispersal and potentially nest fidelity. Isolated or small populations of bees in habitat that lacks adequate floral resources, nesting, and overwintering habitat are particularly susceptible to reduced genetic diversity and rapid reduction in colony growth rate, fitness, and survival, due to the diploid male vortex (see *Small and Isolated Population* section above). Similarly, various insecticides, herbicides, and fungicides have the potential to directly and indirectly affect YBBB habitat and populations. The combined effects of habitat fragmentation and changes in climate where isolated populations occur can be magnified because the ability of species to migrate in response to those pressures may be limited due to habitat loss/fragmentation, and vice versa (Vanbergen *et al.* 2013, p. 255). Genetic diversity in isolated populations can become further degraded, and the synergistic effects with habitat fragmentation and changes in climate may be compounded.

Beneficial Factors

We are aware of relatively few conservation measures that specifically target bumble bees in general or the YBBB specifically.

In Canada, the YBBB is not listed as threatened or endangered under the Species at Risk Act (SARA). In 2015, it was listed as ‘special concern’ under SARA and by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) because it has declined by at least 34 percent in areas of southern Canada (COSEWIC 2015, p. vi-viii). This status means that a management plan must be drafted that will help to ensure that the species does not become threatened or endangered. However, special concern status does not provide regulatory protections for the YBBB. The YBBB is not listed as threatened or endangered by any Canadian provinces. It is designated as a species of ‘special concern’ by the province of Ontario and designated as ‘vulnerable’ by the province of Nova Scotia. In Quebec, the YBBB is included on a list of species likely to be designated threatened or vulnerable. However, neither status provides any regulatory protection. The YBBB is listed as a species of special concern in Wisconsin, Michigan, Minnesota, and Maine, but the special concern status does not convey any regulatory protection. The YBBB are listed as threatened in Connecticut and Vermont. The threatened status in Vermont designates that the bee has a high possibility of becoming endangered in the near future. It is protected by Vermont’s Endangered Species law (10 V.S.A. Chap. 123); the state listing confers protection against take of individuals and protection for critical habitat, if that has been designated, unless permitted to do so (see section 5403: <https://legislature.vermont.gov/statutes/fullchapter/10/123>). In Connecticut, the bee is

protected under Connecticut Endangered Species Act Chapter 495, which identifies the bee as a threatened species (i.e., likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range in the state). The state listing confers protection against take of individuals, unless permitted per Sec. 26-311 of Ch. 495 (see Chapter 495 at https://www.cga.ct.gov/current/pub/chap_495.htm#sec_26-305). The YBBB is currently on the International Union for Conservation of Nature (IUCN) Red List of threatened species.

Some organizations, universities, and state agencies have initiated volunteer bee monitoring programs or surveys to collect baseline data on bumble bees. A 2018 Canadian study has found that recent controls that have been established to improve the reliability of citizen science data and the standardization of data gathering efforts provide substantial amounts of data and new information about changing species distributions and phenologies. And when combined with professionally collected survey data, these datasets can provide large, reliable datasets that change our understanding of species' across large areas (Soroye *et al.* 2018, pp. 1-9). Some programs use non-lethal bumble bee survey protocols using photography and cell phone applications or online portals to collect and submit bumble bee and habitat data for verification by experts (Bumble Bee Watch 2018; Donovall and vanEngelsdorp 2008, entire; iNaturalist 2018; Ohio State University 2018; University of Illinois 2018; University of Minnesota 2018; Vermont Center for Ecostudies 2018, and Wisconsin Bumble Bee Brigade. <http://wiatri.net/inventory/bbb/>). However, lethal sampling protocols are more commonly used (see *Scientific Collection* section above).

Within the range of YBBB, some State Wildlife Action Plans list pollinators, bumble bees, or YBBB and recommend public outreach and encourage integrated pest management practices, however species-specific conservation actions for YBBB are not assigned. Federal agency programs such as the U.S. Department of Agriculture (USDA) 2014 Farm Bill programs (e.g. Conservation Stewardship Program) can provide support to private landowners to conduct pollinator conservation (USDA 2015, entire; NSAC 2016, pp. 11, 16, 21, 23, 25, 26-27, 39-40). In North Dakota, a partnership was established between USDA- Natural Resources Conservation Service (NRCS), North Dakota State University, the Service, and private landowners throughout the state to conduct a 3 year survey of native pollinating insects, including the YBBB and other declining pollinators to better understand their status, distribution, and habitat use. The information will help landowners to manage and maintain existing important habitat, and to identify management actions providing long-term benefits to the species (NDSU 2018a, pp. 1-4; NDSU 2018b, pp. 1-10; NRCS-USFWS 2018, pp. 1-2). A Working Lands for Wildlife partnership in New England between NRCS and the Service is under development to improve the ability of landowners in these states to implement voluntary, cost-effective conservation practices that benefit native pollinators, including the YBBB (NRCS 2018, pp. 1-15). If implemented, these voluntary conservation practices could reduce the effects of stressors to native pollinators such as exposure to pesticides and habitat loss and degradation. Some agricultural producers also utilize integrated pest and pollinator management plans.

Outreach by a variety of organizations and agencies has led to increased awareness and interest in pollinators and the planting of pollinator gardens in urban and suburban areas and in schoolyards. These gardens can provide additional floral resources for pollinators but the direct benefits to YBBB populations may be limited due to the species' need for unfragmented areas of

abundant and diverse floral resources throughout the season, as well as access to contiguous nesting and overwintering habitat (see *Habitat Loss* section for more details) (Goulson *et al.*, 2015, p. 6; Hatfield and LeBuhn 2007, entire). Planting guides to benefit wild pollinators in general have been developed by the Xerces Society-Pollinator Conservation Resource Center and the Pollinator Partnership (Xerces Society 2018; Pollinator Partnership 2018).

The potential to establish conservation breeding colonies and captive breeding programs for native bumble bee species have been considered for the purpose of re-introducing bees into now vacant areas of their historical range or to introduce them into northern areas outside of their historical range (Wildlife Preservation Canada 2018), for the purposes of assisted migration against the effects of climate change. However, concerns have been raised that bumble bee relocation presents multiple risks to wild bumble bee populations. The risks include complex and irreversible consequences from the spread of pathogens and parasites, genetic influences on taxonomy and population structure, and competition with wild bumble bees. Habitat management and other activities to reduce the impacts of stressors have been suggested as an alternative approach to the experimental relocation of bumble bee colonies (Lozier *et al.*, 2018, p. 286; Xerces Society 2015, p. 1-4).

To date, we do not have information on whether these activities have attempted to address any of the identified stressors to the YBBB. While some general conservation measures have been identified for bumble bees, very few of them have been acted upon, such as improved management of the movement of managed bee colonies to reduce the threat of pathogens and parasites, improvements in pesticide use, addressing genetic threats, reducing competition with managed bees, improving and restoring bumble bee habitat at the landscape level, and reducing the risk of climate change (Cameron *et al.* 2011, entire; Colla 2016, pp. 418-421; Goulson *et al.* 2015, pp. 5-7; Hatfield *et al.* 2015, p. 6; IUCN 2018).

Yellow banded bumble bees are subject to numerous stressors range-wide: habitat loss, fragmentation and degradation, pesticide use, pathogens and parasites; and the effects of climate change, all compounded by the effects of small and isolated populations. Independently, all of these stressors reduce viability; however, the additive and synergistic effects of numerous stressors reduce population size and increase population isolation. Small and isolated populations (as discussed in *Small and Isolated Populations Section* above) can have deleterious effects on haplodiploid species such as the YBBB. It is difficult to tease out the effects of one stressor when they appear to interact. It is likely that several of these stressors are acting synergistically on the species, and the combination of multiple stressors is likely more harmful than a single stressor acting alone.

Chapter 4: Past and Current Conditions

Historical Condition

The historical status of the YBBB is the baseline or reference condition that provides the context for our analysis of the species' current and future conditions. For the purposes of this SSA report, all YBBB records prior to 2010 are considered "historical" occurrences (e.g., occurrences dating from 1824 to 2009).

Uncertainty - Uncertainty regarding the historical range and condition of the YBBB results from inconsistencies in the data, including highly variable survey effort (geographically and temporally) and poor or nonexistent documentation of survey methodologies. For example, bumble bee surveys in the northern portions of most Canadian provinces are sparse, while some areas around population centers or universities have been relatively well surveyed over time (although consistent, long-term surveys of specific sites are rare). These factors make it difficult or impossible for us to assess directly historical YBBB abundances or other demographic trends or to analyze potential co-variables that may influence YBBB populations across its range; however, we are able to use the best available information to infer some of these parameters.

It is also important to recognize that our assessment of the YBBB's historical condition does not necessarily reflect the species' "natural" condition (*i.e.*, prior to Euro-American settlement). This is because nearly all occurrence data for the species were collected after widespread land disturbing activities such as commercial logging, agricultural conversion, and urban development had already modified large swaths of habitat within the YBBB's range (Clawson 1979, pp. 1168–1169; Ramankutty and Foley 1999, pp. 385–394; Rhemtulla *et al.* 2007, pp. 64–65; Schulte *et al.* 2007, pp. 1090, 1098–1101). Therefore, it is possible, and perhaps likely, that the YBBB had already undergone declines in range, abundance, and/or population connectedness prior to any efforts to document its presence or condition.

In summary, while a paucity of survey data introduces significant uncertainty into our assessment of the YBBB in some far northern portions of its range, in much of the U.S. and southern Canadian areas, the data appear sufficient to assess the status of the species with a higher degree of confidence.

Historical Range - Based on the best available information, the YBBB was historically known from 25 U.S. states and 12 Canadian provinces and territories (table 3). The first documented YBBB specimen was collected at Isle Royale, Michigan (MI) in 1824, with no further records reported until 1884. From 1884 to 1949, specimens were reported from 17 additional states and provinces. Therefore by the mid-20th century, the overall continental-scale distribution of the species was known. From 1950 to 2010, the species was determined in six more states, generally infilling areas within its known range (figure 11).

Our determination of the YBBB's historical range in the far western United States is complicated by potential confusion in the identification of YBBB and WBB (*B. occidentalis*) specimens (Koch *et al.* 2012, pp. 102–108). While the taxonomic status of the YBBB and WBB as separate species is no longer in question (see Chapter 2 – Taxonomy), prior to 1913, and in some cases

more recently (see Milliron 1971, pp. 51–67), the WBB was considered a sub-species of the YBBB. Therefore, many early WBB specimens from areas where the ranges of the two species are now understood to be adjacent or overlap, were recorded as “*B. terricola*” or “*B. terricola-occidentalis*.” While most museum specimens have since been examined to confirm and update their classification as either YBBB or WBB, it is probable that some misidentified historical specimens remain in the *Bombus* database (Strange 2018, pers. comm.; Richardson 2017, pers. comm.). This issue is pertinent in our determination of the YBBB’s historical presence in (or absence from) the states of Idaho (ID), Utah (UT), Washington (WA), Montana (MT), Wyoming (WY), Alaska (AK), and Iowa (IA).

The *Bombus* database contains YBBB records from ID, UT, WY, AK, and IA (specimens collected in 1968, 1918, 1964, 1962, and 1930, respectively). However, the best available information suggests these reports are either erroneous or suspect. Inspection of the ID specimen determined that a label transcription error had resulted in the collection location being recorded as Geneva, ID instead of the correct Geneva, New York (NY) (Idaho Department of Fish & Game 2018, entire). The same occurred for the IA specimen, which was collected in Washington County, Maine, and a specimen reported from AK, which was determined to be erroneous and based on a misidentification or a digitization error. Therefore, we have no data to indicate the historical range of the YBBB included ID, IA, or AK.

Three YBBB specimens collected in WY in 1964 and held at the Yale University Peabody Museum of Natural History were recently reclassified as WBB. A single specimen from UT is reported to be held in the Smithsonian National Museum of Natural History; however, we were unable to locate this record in the museum’s database to confirm its validity. We note that the reported location of this specimen is approximately 770 km (480 mi) south of the nearest other confirmed YBBB record in South Dakota (SD). Additionally, the scientific literature is consistent that WY, ID, and UT are not within the range of the YBBB (Evans *et al.* 2008, pp. 14–16; Koch and Strange 2009, entire; (Koch *et al.* 2012, pp. 106–108; Sheffield *et al.* 2016, entire) nor are these areas reported to maintain suitable habitat for the species (Strange 2018, pers. comm.). Therefore, we do not include the states of WY, ID, or UT as being within the historical range of the YBBB.

There are several historical YBBB records from central and southwestern MT, however these specimens were recently determined to be WBB (Dolan *et al.* 2017, pp. 137–138). In 2011 and 2015, YBBB specimens were collected from northeastern MT; therefore the species has only been recorded in MT since 2011. Based on the relatively close proximity of these collections with historical records of the species from similar habitat areas of Saskatchewan (SK), Canada, it is likely the YBBB was historically present in MT. We emphasize that while these state record determinations help define the range of the species, they are not generally significant to our data analysis because we do not rely on political boundaries to define our primary analytical units for assessing the status of the YBBB.

While the scientific literature does not include WA within the range of the YBBB (Evans *et al.* 2008, pp. 14–16; Koch and Strange 2009, entire; (Koch *et al.* 2012, pp. 106–108; Sheffield *et al.* 2016, entire), we reviewed the best available information and determined that the northwest corner of the state was likely within the historical range of the species. There is a single 1906

record of the YBBB from San Juan Island, WA. While we could not locate any additional information with which to confirm the taxonomy of this record, we note that there are several credible YBBB records (dating from 1929 to 1988) from areas of British Columbia (BC) within about 80 km (50 mi) of the reported WA state location. Additionally, there are multiple current records of the YBBB from an area of BC approximately 290 km (180 mi) northwest of the WA record. Therefore we conclude that WA was likely within the historical range of the YBBB.

The YBBB was not reported in the Yukon Territory (YT), Canada until 2014, but again, it is likely that the species did occur there historically. We base this conclusion on the historical presence of the species in adjacent areas of BC and the Northwest Territories (NT), which have suitable habitat similar to the areas of YT where the YBBB was recently confirmed. Again, this conclusion does not significantly affect our assessment because we do not rely on political boundaries to define our primary analytical units.

Table 3. U.S. states and Canadian provinces with valid YBBB records along with the years the species was first confirmed and last confirmed. Note that occurrence data for New Brunswick, Nova Scotia, and Prince Edward Island, together the Canadian Maritime Provinces, were combined and used collectively in our analyses.

U.S. State and Abbreviation		YBBB Record Year		Canadian Province and Abbreviation		YBBB Record Year	
		Earliest	Latest			Earliest	Latest
Connecticut	CT	1904	2009	Alberta	AB	1884	2016
Georgia	GA	1943	1943	British Columbia	BC	1911	2015
Illinois	IL	1894	1965	Manitoba	MB	1909	2017
Indiana	IN	1950	1950	New Brunswick	NB	1898	2015
Maine	ME	1898	2016	Newfoundland	NL	1905	2015
Maryland	MD	1976	2012	Northwest Territories	NT	1922	2016
Massachusetts	MA	1891	2017	Nova Scotia	NS	1905	2015
Michigan	MI	1824	2014	Ontario	ON	1886	2016
Minnesota	MN	1906	2016	Prince Edward Island	PE	1940	2015
Montana	MT	2011	2015	Quebec	QC	1906	2016
New Hampshire	NH	1889	2015	Saskatchewan	SK	1888	2016
New Jersey	NJ	1912	1978	Yukon Territory	YT	2014	2014
New York	NY	1885	2015				
North Carolina	NC	1912	2009				
North Dakota	ND	1910	2012				
Ohio	OH	1964	1981				
Pennsylvania	PA	1902	2017				
Rhode Island	RI	1974	1974				
South Dakota	SD	1929	2010				
Tennessee	TN	1934	2004				
Vermont	VT	1891	2016				
Virginia	VA	1998	1998				
Washington	WA	1906	1906				
West Virginia	WV	1967	2018				
Wisconsin	WI	1905	2017				

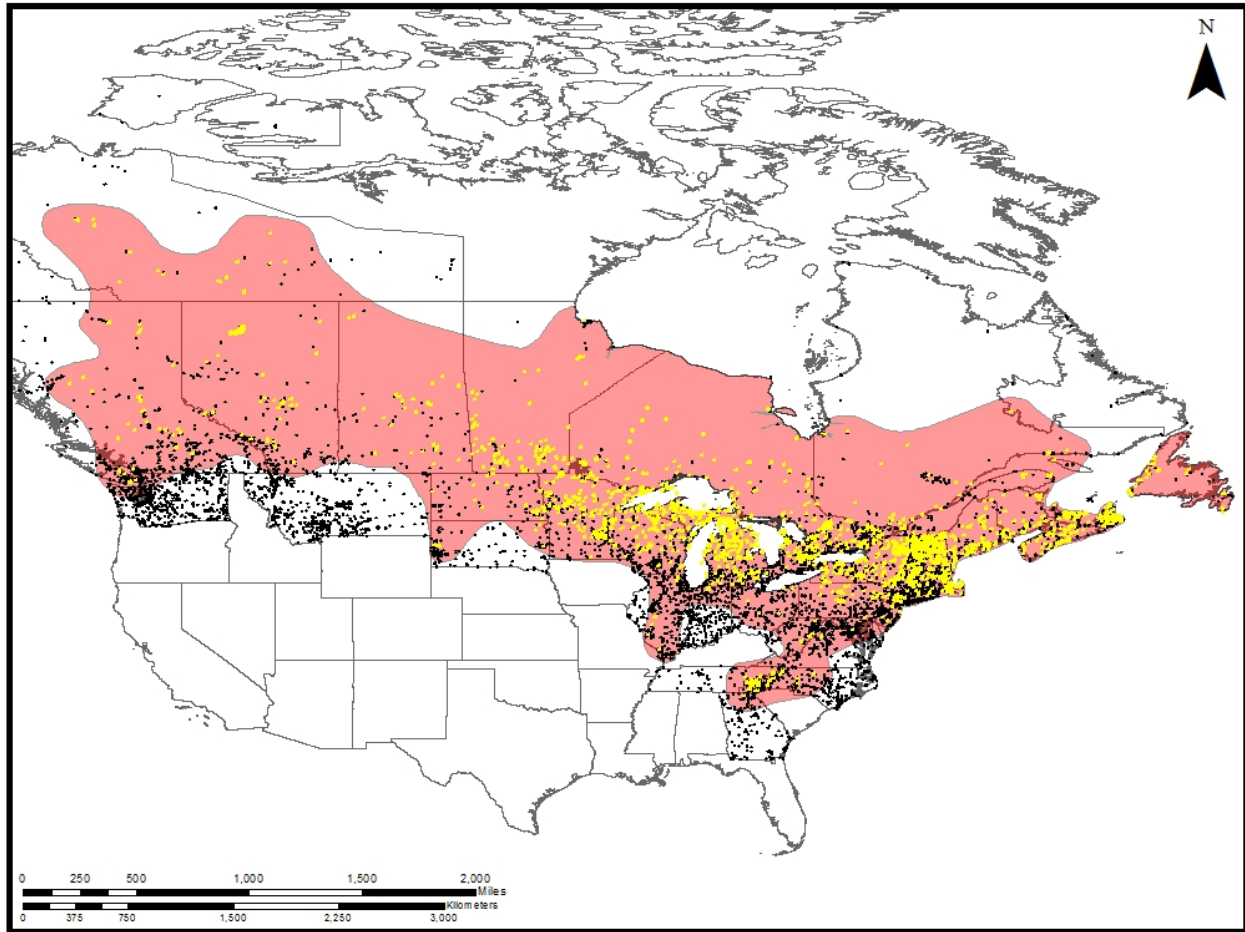


Figure 11. Historical range of the YBBB in red. Yellow dots represent positive YBBB occurrences and black dots represent surveyed locations that were negative for YBBB. This range map was developed by drawing a 100 km (62 mi) buffer (per Colla *et al.* 2011) around historical (1824 to 2009) occurrences and is generally inclusive of areas between positive occurrence locations. Minor smoothing of some edges was used where we judged appropriate. Some areas with sparse survey data were included based on the professional judgement of bumble bee researchers with local knowledge.

Analytical Units – North American political boundaries are not based on any environmental or habitat characteristics known to be relevant to the natural distribution of the YBBB. Nor are the variably-sized and shaped spatial units produced by these political boundaries (e.g. counties, states, provinces, etc.) particularly well-suited for analyzing the YBBB distribution or condition across its range. Therefore we selected two other geographical analytical units, hexagonal grids and ecoregions, to help assess the historical, current, and future condition of the species. The use of these analytical units is intended to help standardize the available survey data and assess the species using habitat-based delineations. For convenience, we do use state and province names when describing the geographic range of the species and have also included summaries of the survey data based on state/province boundaries (Appendix A).

Because of inconsistencies in the survey data, specifically, intense survey effort and specimen records in some localized areas and relatively few surveys and specimen records across larger areas, our fundamental analytical units are based on a 100 square kilometer (sq km) (38.6 square mile (sq mi)) hexagonal grid overlaid on the map of North America. A hexagonal grid system

(as opposed to a rectangular grid system) is preferred here where YBBB population connectedness and potential movement paths are a concern and where a geographic information system (GIS) is used to analyze the data (Birch *et al.* 2007, entire). As described in the status assessment report for the rusty patched bumble bee (USFWS 2016, p. 11), the 100 sq km (38.6 sq mi) grid size is based on the reported dispersal distances for the closely related buff-tailed bumble bee (*B. terrestris*) (Kraus *et al.* 2009, p. 249; Lepais *et al.* 2010, pp. 826–827). Because of the colonial life history of that species, whereby only the queens produce offspring and each colony supports a single queen, the colony is considered a single reproductive unit. Therefore it is reasonable to assume that a buff-tailed bumble bee occurrence within a grid likely represents a colony or colonies, or for our purposes, a population. Because the life history of the YBBB is very similar to that of the buff-tailed bumble bee (and the RPBB), we use the same construct and assume that individual YBBBs detected within a 100 sq km (38.6 sq mi) grid represent a YBBB population. Analyzing the data based on population units, as opposed to individual specimen records, helps standardize YBBB occurrence data across the species' range and provides a more accurate representation of the functional status of the species.

At a larger scale, we consider YBBB survey data by ecological region or “ecoregion.” Ecoregions are geographical units defined by habitat-relevant characteristics including climate, geology, soil, landform, vegetation, wildlife, and human land-use (Commission for Environmental Cooperation (CEC) 1997, pp. 4–6; Wiken *et al.* 2011, pp. 10–11; Omernek and Griffith 2014, entire). While there are limited data on how some of these characteristics specifically affect the YBBB, it is reasonable to conclude that, either alone or in combination, they are relevant in determining the natural distribution and status of the species.

The CEC (1997, entire) and Wiken *et al.* (2011, entire) define 15 broad level I terrestrial ecoregions in North America. These major ecological settings are most useful for analysis at the global or intercontinental scale. Within the level I ecoregions, 52 level II ecoregions are defined (figure 12). These areas are delineated to allow for analysis at the national or regional scale. Finally, about 200 level III ecoregions are nested within the level II regions. These units incorporate more localized characteristics and allow for analysis at the regional scale. Because of the YBBB's large historical range and inconsistent survey data (both geographically and temporally), we determined that the level II ecoregion is the most appropriate scale for assessing the species' status.



Historical Distribution – The YBBB historically occurred in 18 of 52 level II ecoregions (within 8 of 15 level I ecoregions) primarily distributed across the northern United States and most Canadian provinces (table 4).

Table 4. Level I and level II ecoregions, U.S. states, and Canadian provinces with YBBB records.

Level I Ecoregion		Level II Ecoregion	U.S. States	Canadian Provinces
Taiga	3.3	Taiga Plains		AB, BC, NT, YT
	3.4	Taiga Shield		NT, QC
Hudson Plains	4.1	Hudson Plains		MB, ON, QC
Northern Forests	5.1	Softwood Shield		MB, ON, NL, QC, SK
	5.2	Mixed Wood Shield	MI, MN, WI	MB, ON, QC
	5.3	Atlantic Highlands	CT, MA, ME, NH, NJ, NY, VT	NB, NS, QC
	5.4	Boreal Plain		AB, BC, MB, SK
Northwestern Forested Mountains	6.1	Boreal Cordillera		BC, YT
	6.2	Western Cordillera	SD	AB, BC
Marine West Coast Forest	7.1	Marine West Coast Forest	WA	BC
Eastern Temperate Forests	8.1	Mixed Wood Plains	CT, IN, MA, RI, ME, MI, MN, NH, NY, OH, PA, VT, WI	NB, NS, PE, ON, QC
	8.2	Central USA Plains	IL, OH, MI, WI	
	8.3	Southeastern USA Plains	IL, MD, NC, NJ, PA	
	8.4	Ozark, Ouachita-Appalachian Forests	GA, MD, NC, OH, NJ, PA, TN, VA, WV	
	8.5	Southeast USA Coastal Plains	MA, NJ, NY	
Great Plains	9.2	Temperate Prairies	MN, ND, SD	AB, MB, SK
	9.3	West Central Semi-Arid Prairies	MT, ND, SD	AB, SK
North American Deserts	10.1	Cold Deserts		BC

The historical distribution of the YBBB varied widely across ecoregions, with percent occupancy (based on YBBB occurrences within 100 sq-km (38.6 sq-mi) grids with *Bombus* spp. survey data) ranging from about 1 to 63 percent (mean 20 percent), and relative abundance ranging from less than 1 to about 59 percent (mean 11 percent) (table 5). These data suggest that the species was very common, based on occurrence percentages of 40 or greater, in the Taiga Plains, Softwood Shield, Mixed Wood Shield, Atlantic Highlands, and Boreal Plain (ecoregions 3.3, 5.1, 5.2, 5.3, and 5.4); therefore, we consider these to represent the core of the YBBB's historical range where it was apparently a dominant *Bombus* spp. The species also appeared to be relatively common, based on occurrence percentages between about 12 and 40 percent, in the Hudson Plains, Mixed Wood Plains, Ozark, Ouachita-Appalachian Forests, and Temperate Prairies (ecoregions 4.1, 8.1, 8.4, and 9.2). The data suggest that the YBBB was historically rare in the Taiga Shield, Boreal Cordillera, Western Cordillera, Marine West Coast Forest, Central USA Plains, Southeastern USA Plains, Southeast USA Coastal Plains, West Central Semi-Arid Prairies, and Cold Deserts (ecoregions 3.4, 6.1, 6.2, 7.1, 8.2, 8.3, 8.5, 9.3, and 10.1). The survey data in the Taiga Plains, Taiga Shield, and Hudson Plains (ecoregions 3.3, 3.4, and 4.1) are particularly sparse; therefore, we are less certain of the YBBB's historical distribution in those ecoregions. However, because these landscapes are generally homogeneous and largely undisturbed by human activity, the available survey data may sufficiently characterize the species' historical condition in those ecoregions.

In summary, the YBBB was historically well distributed and likely a dominant bumble bee species across much of its range (e.g. the core ecoregions), but relatively rare and likely patchily distributed at the periphery and perhaps in areas that underwent intense timber clearing and agricultural and/or urban conversion. The observations of bumble bee researchers generally support this conclusion (Evans, *et al.* 2008, pp. 14–17; Williams and Osborne 2009, pp. 370–371).

Table 5. Historical (1824 to 2009) YBBB occupancy and relative abundance by level II ecoregion. Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of *Bombus* spp. specimens collected that were YBBB.

Ecoregion		Occupancy			Relative Abundance		
		YBBB+	Total	%YBBB	YBBB	Total	%YBBB
3.3	Taiga Plains	19	41	46%	53	413	13%
3.4	Taiga Shield	3	28	11%	9	257	4%
4.1	Hudson Plains	6	31	19%	21	495	4%
5.1	Softwood Shield	68	170	40%	219	980	22%
5.2	Mixed Wood Shield	583	921	63%	6,208	10,611	59%
5.3	Atlantic Highlands	552	1,269	43%	3,247	12,535	26%
5.4	Boreal Plain	73	148	49%	567	2,670	21%
6.1	Boreal Cordillera	2	130	2%	2	4,964	0.04%
6.2	Western Cordillera	26	360	7%	48	2,638	1.8%
7.1	Marine West Coast Forest	5	438	1%	9	8,924	0.1%
8.1	Mixed Wood Plains	553	2,696	21%	2,645	32,415	8%
8.2	Central USA Plains	33	504	7%	68	9,489	1%
8.3	Southeastern USA Plains	18	695	3%	25	4,989	1%
8.4	Ozark, Ouachita-Appalachian Forests	84	682	12%	195	5,350	4%
8.5	Southeast USA Coastal Plains	22	259	8%	4	245	2%
9.2	Temperate Prairies	117	357	33%	698	5,268	13%
9.3	West Central Semi-Arid Prairies	14	299	5%	18	2,512	1%
10.1	Cold Deserts	10	135	7%	8	927	0.9%
Totals		2,188	9,163	24%	14,044	105,682	13%

Historical Trends – To better understand the historical condition of the YBBB, we analyzed potential trends in the species’ condition between 1824 and 2009. As discussed previously, the survey data vary widely in geographical and temporal consistency; therefore, we binned the survey data to ensure sufficient survey data were available to compare ecoregions throughout the historical period. The majority of the data were combined by decade, however because early records were particularly sporadic, all data from 1824 to 1949 were combined.

Results of our trend analysis of historical YBBB occurrence and relative abundance data combined across all ecoregions indicate that up until the 2000s, the species’ overall condition appeared to be relatively stable, with percent occupancy ranging from about 23 to 27 percent (mean 26 percent) and relative abundance ranging from about 11 to 24 percent (mean 18 percent). However, data from the 2000s, the last decade in our defined historical period, indicate a probable departure from this trend, with overall percent occupancy declining to 15 percent and relative abundance declining to 3 percent (figure 13).

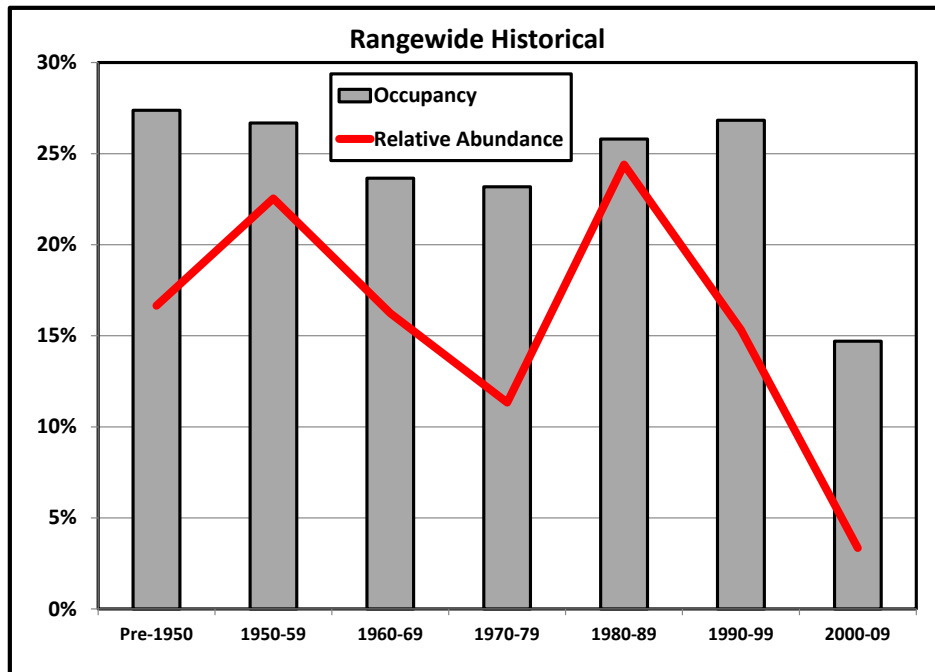


Figure 13. Historical trends in YBBB occupancy and relative abundance across its range. Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of *Bombus* spp. specimens collected that were YBBB.

When the historical data are parsed by individual ecoregion, the temporal patterns of YBBB occupation and relative abundance vary, but a general downward trend in one or both parameters is apparent in 7 of the 18 level II ecoregions (see Appendix B). We note that the majority of ecoregions without distinct downward trends in either parameter (including three ecoregions with apparent upward trends), are either at the periphery of the species' range where the data suggest YBBB has always been rare and/or in ecoregions where survey data are sparse, which increases our uncertainty.

Current Condition

We considered occurrence data from 2010 to 2017 to assess the current condition of the YBBB. This period of time was selected primarily because the scientific evidence indicates many bumble bee species are undergoing rapid changes in range or abundance and because increased survey effort has resulted in significantly more survey records for this period than for any previous survey period (figure 14). However, as with the historical dataset, the consistency of current survey data varies widely and is especially sparse in the northern portions of most Canadian provinces (e.g. Taiga Plains and Hudson Plains, level II ecoregions 3.3 and 4.1, respectively). While the year 2010 is an artificial break point, we conclude that the 2010 to 2017 survey data are appropriate for assessing the YBBB's current condition with respect to its long-term historical condition.

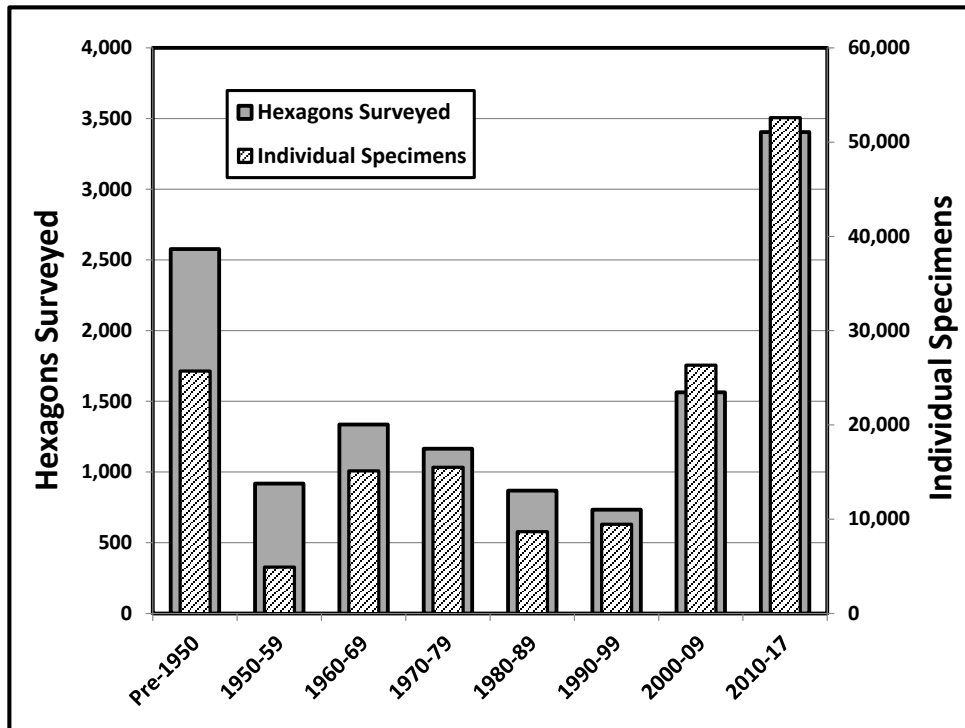


Figure 14. *Bombus* spp. survey effort by time period. Total number of hexagons surveyed for bumble bees and total individual *Bombus* specimens collected.

Current Range – Based on the best available information, the current range of the YBBB is reduced from its historical range (figures 15 and 16). The net areal extent of this loss is approximately 3,936,426 sq km (1,519,854 sq mi), or about 23 percent, with the extent of loss varying by ecoregion and ranging from about 6 to 100 percent (table 6). During the current period, surveys did not confirm the species in 11 of the 25 U.S. states (44 percent) from which it had historically been reported. Losses are mostly concentrated in the mid-Atlantic and Great Lakes regions. The historical states with no current records of the YBBB include: CT, GA, IL, IN, NC, NJ, OH, RI, TN, VA, and WA. The species remains extant in the Canadian provinces from which it was historically known, though its range may have contracted at the northern and western periphery. It is important to note however that very low survey effort in the northern Canadian provinces make us less certain about losses from this area. The range contraction described here is generally consistent with the conclusions of other bumble bee researchers (Hatfield *et al.* 2015 entire; Kerr *et al.* 2015, entire).

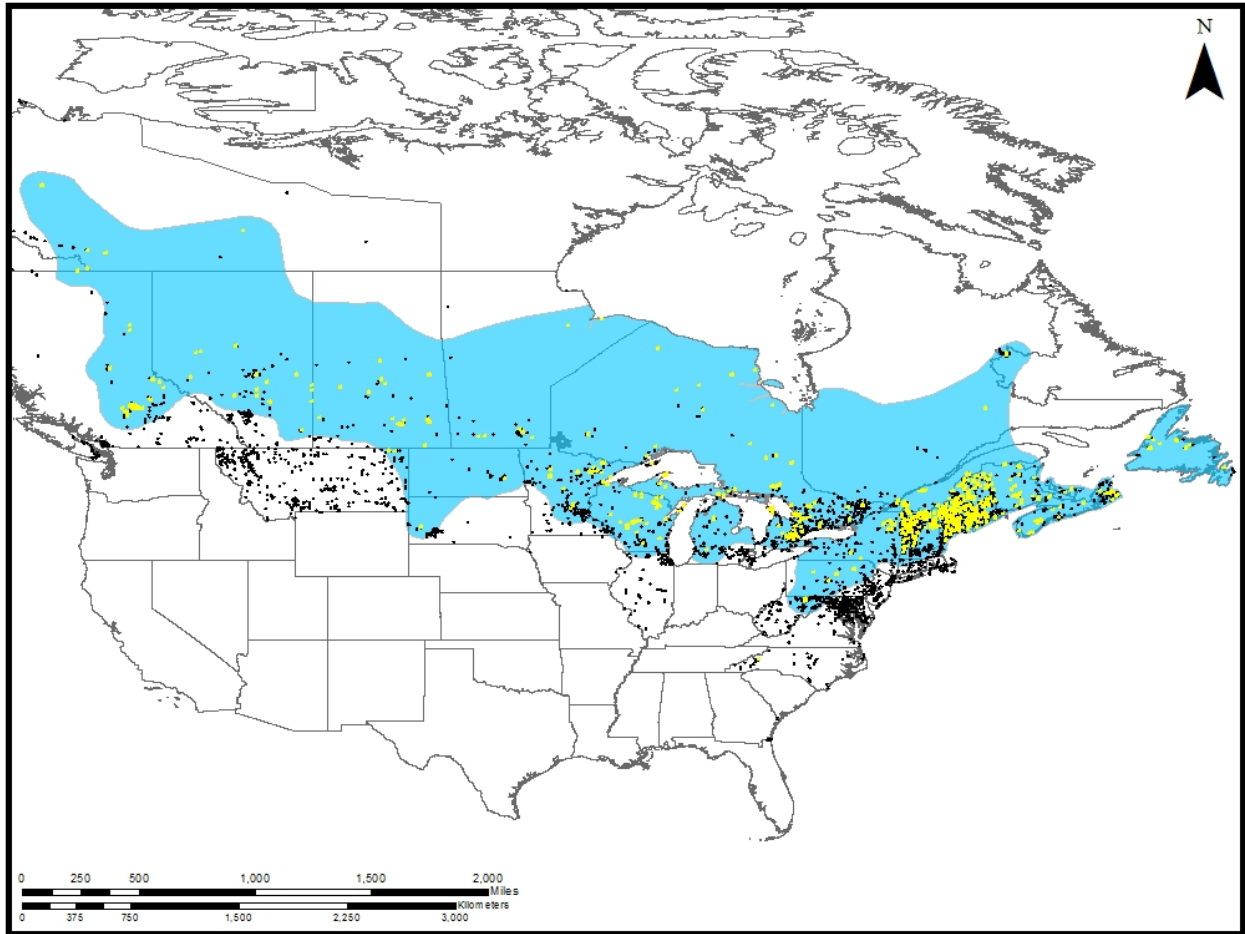


Figure 15. Current range of the YBBB in blue. Yellow dots represent positive YBBB occurrences and black dots represent surveyed locations that were negative for YBBB. This range map was developed by drawing a 100 km (62 mi) buffer (per Colla et al. 2011) around 2010 to 2017 occurrence points and is generally inclusive of areas between positive occurrence locations. Minor smoothing of some edges was used where we judged appropriate. Some areas with sparse survey data were included based on the professional judgement of bumble bee researchers with local knowledge.

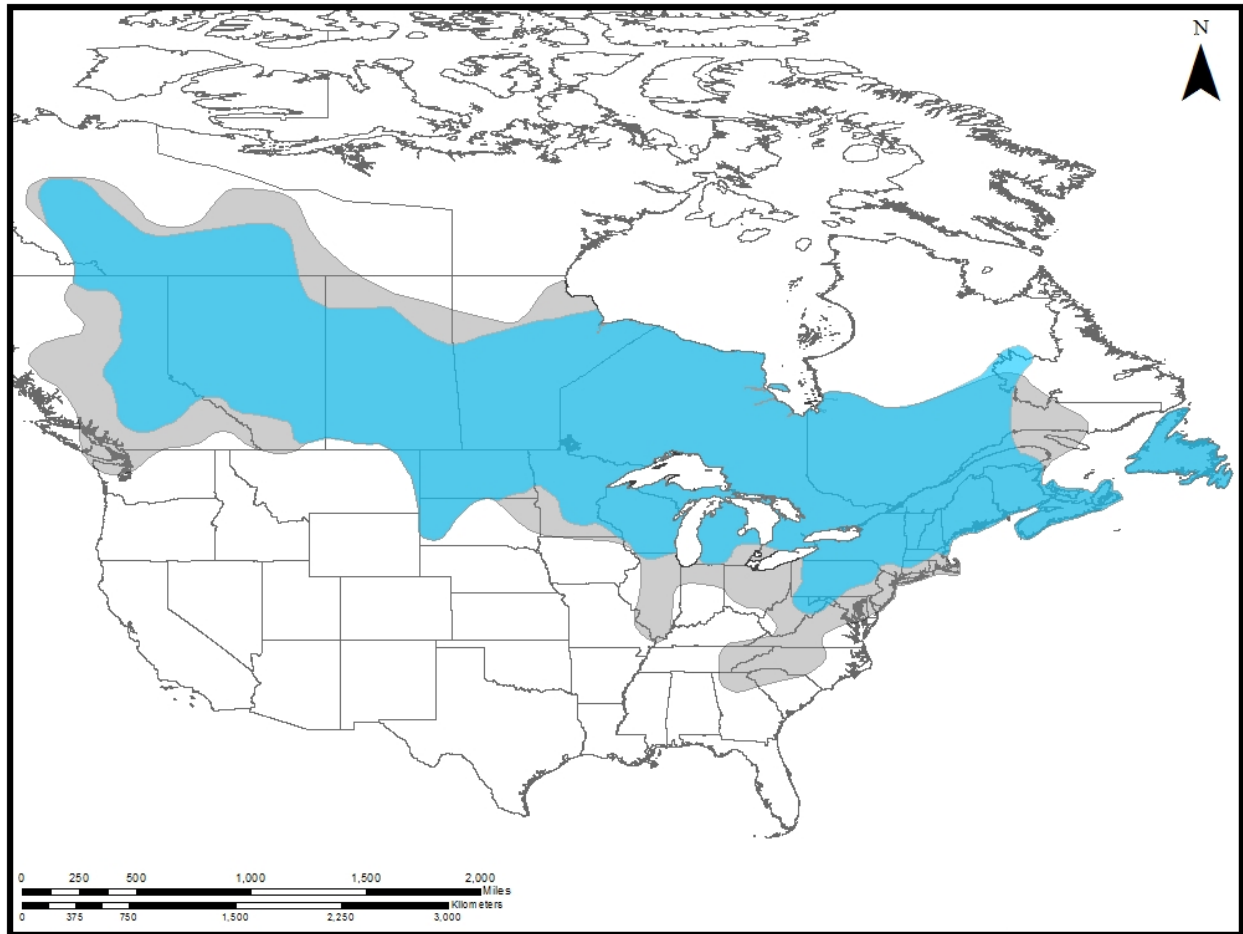


Figure 16. YBBB range contraction. Gray shading represents the historical range of the species; blue shading represents the current range.

Table 6. Historical and current areal extent of YBBB range by level II ecoregion. “Percent of total” is the historical proportion of an ecoregion to the total YBBB historical range. “Percent change” is the change from the historical to the current areal extent. Gray shading indicates ecoregions where the species is presumed extirpated.

Level II Ecoregion		Historical Range		Percent of Total	Current Range		Percent Change
		Sq Km	Sq Mi		Sq Km	Sq Mi	
3.3	Taiga Plains	1,850,243	714,379	11%	1,575,438	608,277	-15%
3.4	Taiga Shield	963,930	372,173	6%	337,349	130,250	-65%
4.1	Hudson Plains	1,064,663	411,066	6%	1,008,094	389,225	-5%
5.1	Softwood Shield	3,707,544	1,431,483	21%	3,141,848	1,213,068	-15%
5.2	Mixed Wood Shield	1,312,534	506,769	8%	1,307,462	504,811	0%
5.3	Atlantic Highlands	526,455	203,264	3%	493,989	190,729	-6%
5.4	Boreal Plain	2,084,121	804,679	12%	2,078,404	802,472	0%
6.1	Boreal Cordillera	202,692	78,259	1%	63,779	24,625	-69%
6.2	Western Cordillera	1,105,538	426,848	6%	494,162	190,796	-55%
7.1	Marine West Coast Forest	237,818	91,822	1%	137	53	-100%
8.1	Mixed Wood Plains	1,151,466	444,581	7%	986,899	381,042	-14%
8.2	Central USA Plains	325,939	125,845	2%	86,953	33,572	-73%
8.3	Southeastern USA Plains	306,949	118,513	2%	71	27	-100%
8.4	Ozark, Ouachita-Appalachian Forests	425,364	164,233	2%	130,775	50,492	-69%
8.5	Southeast USA Coastal Plains	46,313	17,882	0.3%	0	0	-100%
9.2	Temperate Prairies	1,010,707	390,234	6%	816,430	315,224	-19%
9.3	West Central Semi-Arid Prairies	885,073	341,727	5%	815,975	315,048	-8%
10.1	Cold Deserts	127,087	49,068	1%	60,245	23,261	-53%
Totals:		17,334,436	6,692,826	100%	13,398,010	5,172,972	-23%

Current Distribution - Along with the observed contraction in range, the YBBB appears to have been extirpated from 3 of the 18 ecoregions from which it was historically known (table 7).

Table 7. Current (2010 to 2017) YBBB occupancy and relative abundance by level II ecoregion. Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of *Bombus* spp. specimens collected that were YBBB. Gray shading indicates ecoregions where the species is presumed extirpated.

Ecoregion		Occupancy			Relative Abundance		
		YBBB+	Total	%YBBB	YBBB	Total	%YBBB
3.3	Taiga Plains	5	8	63%	11	39	28%
3.4	Taiga Shield	4	12	33%	13	144	9%
4.1	Hudson Plains	3	5	60%	19	22	86%
5.1	Softwood Shield	18	42	43%	103	2,301	4%
5.2	Mixed Wood Shield	67	269	25%	104	1,498	7%
5.3	Atlantic Highlands	227	646	35%	619	14623	4%
5.4	Boreal Plain	13	41	32%	39	86	45%
6.1	Boreal Cordillera	2	123	2%	2	1,192	0.2%
6.2	Western Cordillera	21	82	26%	194	348	56%
7.1	Marine West Coast Forest	0	143	0%	0	1,013	0%
8.1	Mixed Wood Plains	231	1,111	21%	693	20,933	3%
8.2	Central USA Plains	3	123	2%	3	508	1%
8.3	Southeastern USA Plains	0	188	0%	0	3946	0%
8.4	Ozark, Ouachita-Appalachian Forests	4	200	2%	5	1603	0.3%
8.5	Southeast USA Coastal Plains	0	79	0%	0	88	0%
9.2	Temperate Prairies	22	107	21%	28	393	7%
9.3	West Central Semi-Arid Prairies	4	194	2%	24	3,768	1%
10.1	Cold Deserts	4	32	13%	2	85	2%
Totals		628	3,405	18%	1,859	52,590	4%

The current data confirm the observed decline in overall YBBB occurrence and relative abundance percentages that were noted in the last decade of the historical period (figure 17). During the current period, the percentage of hexagons where YBBBs were collected varied widely by ecoregion, ranging from 0 to more than 63 percent (mean 17 percent). Importantly,

the data indicate the species may be extirpated from the Marine West Coast Forest, Southeastern USA Plains, and Southeast USA Coastal Plains (ecoregions 7.1, 8.3, and 8.5, respectively). The current relative abundance percentages ranged from 0 to about 86 percent (mean 4 percent). Averaged across all ecoregions, the current percent occupancy is 18 percent and the relative abundance is 4 percent.

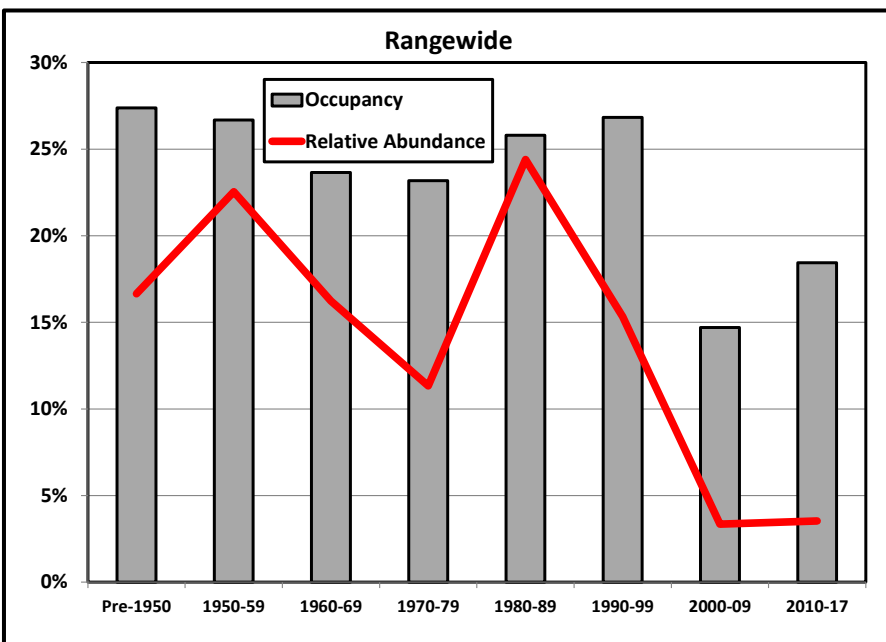


Figure 17. Decadal trends in YBBB occupancy and relative abundance across all ecoregions. Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of *Bombus* spp. specimens collected that were YBBB.

This trend is notably apparent in the core of the YBBB's historical range, the Taiga Plains, Softwood Shield, Mixed Wood Shield, Atlantic Highlands, and Boreal Plain (ecoregions 3.3, 5.1, 5.2, 5.3, and 5.4) where the species' historical occurrence percentage ranged from 35 to 65 percent (mean 51 percent) and its relative abundance ranged from 8 to 32 percent (mean 26 percent). During the current period these parameters have declined to 33 and 4 percent, respectively (figure 18).

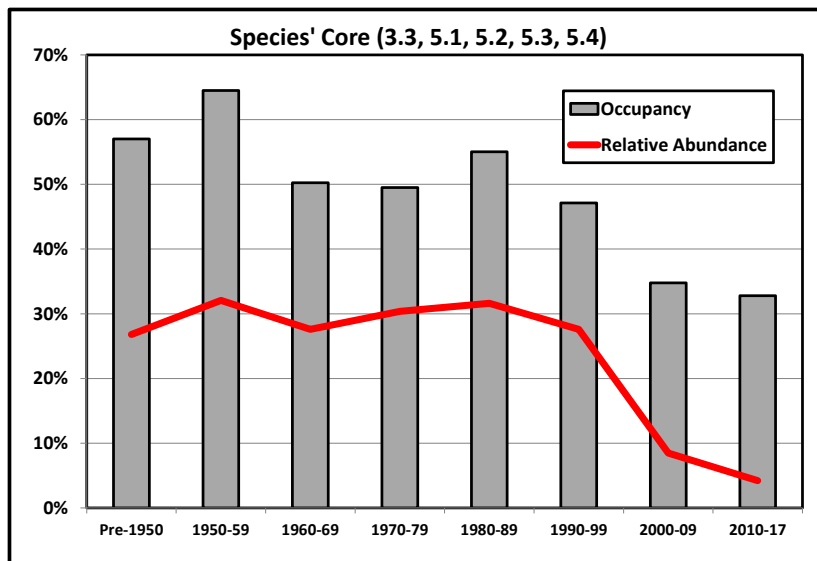


Figure 18. Decadal trends in YBBB occupancy and relative abundance within the core of the species' range (ecoregions 3.3, 5.1, 5.2, 5.3, and 5.4).

3Rs Analysis

Redundancy - Redundancy describes the ability of a species to withstand catastrophic events and can be understood as the spreading of risk among multiple populations to minimize the potential loss of the species from catastrophic events. Redundancy is characterized by having multiple, resilient populations distributed within the species' ecological settings and across its range. For the YBBB, we base our analysis of redundancy on the number of occupied ecoregions and the estimated number of populations within each ecoregion.

There is no genetic or demographic information available with which to identify the number or distribution of YBBB populations; therefore we estimated potential historical and current YBBB populations using the available *Bombus* spp. survey data, the area of each ecoregion within the range of the species, and the percentage of land cover within each ecoregion that is potentially suitable habitat for the species. These population estimates assume that a YBBB occurrence in a surveyed 100 sq km (38.6 sq mi) hexagon represents a discrete population and that the surveyed hexagons are randomly distributed within the ecoregions. Neither of these assumptions are necessarily supported by the data or scientific literature; however we consider the resulting estimates of YBBB population numbers useful for assessing the relative condition of the species within and across ecoregions.

Our YBBB population estimates are derived by multiplying the historical or current area (in sq km/sq mi) of each ecoregion by the ecoregion's current or historical YBBB percent occupancy (see tables 5 and 7). We then divided the result by 100 sq km (38.6 sq mi) to give an estimate of occupied hexagons or "populations." Because there are no sufficiently detailed land cover data available for the historical period, we assumed that all land within the historical range of the species was potentially suitable habitat. For the current period we provide two estimates. In the first current population estimate, all land within the current range of the species is assumed to be suitable habitat. This is identical to the method used for the historical period. For the alternative

current population estimate, we used the available land cover data to exclude agricultural and urban (e.g., developed) land as suitable habitat for the YBBB (table 8 and figure 19).

Table 8. Percent agricultural and urban land cover within the ecoregions occupied by the YBBB (Data from: 2010 North American Landcover Dataset and Land Change Monitoring System at <https://landcover.usgs.gov/nalcms.php>).

	Ecoregion	Agricultural	Urban	Total
3.3	Taiga Plains	0%	0%	0%
3.4	Taiga Shield	0%	0%	0%
4.1	Hudson Plains	0%	0%	0%
5.1	Softwood Shield	1%	0%	1%
5.2	Mixed Wood Shield	4%	0%	4%
5.3	Atlantic Highlands	3%	2%	5%
5.4	Boreal Plain	15%	0%	15%
6.1	Boreal Cordillera	0%	0%	0%
6.2	Western Cordillera	2%	0%	2%
7.1	Marine West Coast Forest	2%	2%	4%
8.1	Mixed Wood Plains	38%	6%	44%
8.2	Central USA Plains	82%	10%	92%
8.3	Southeastern USA Plains	52%	9%	61%
8.4	Ozark, Ouachita-Appalachian Forests	18%	4%	22%
8.5	Southeast USA Coastal Plains	13%	36%	49%
9.2	Temperate Prairies	83%	1%	83%
9.3	West Central Semi-Arid Prairies	38%	0%	38%
10.1	Cold Deserts	15%	1%	15%

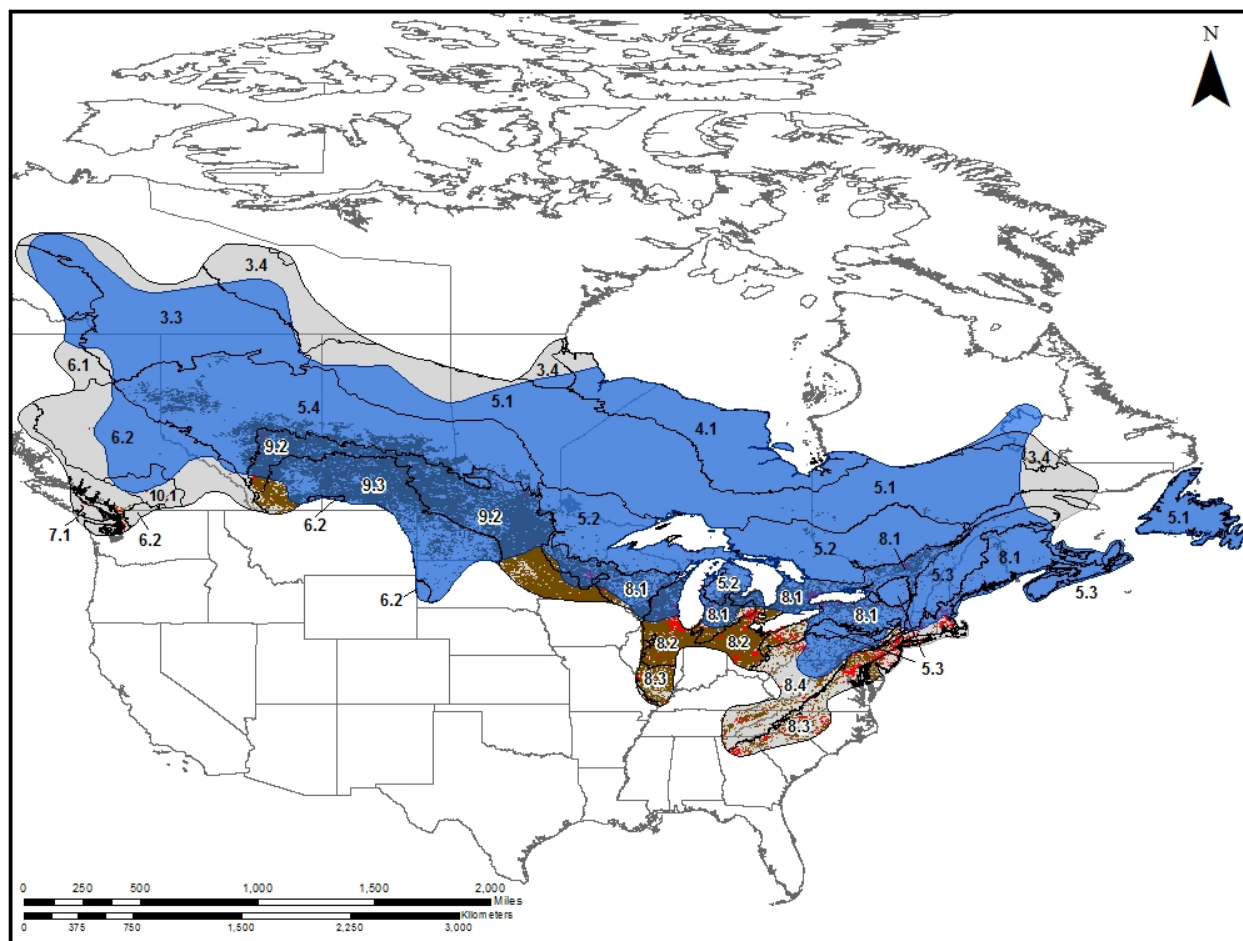


Figure 19. Landcover within the range of the YBBB. Blue and gray shading indicates the current and historical range of the species, respectively. Brown shading indicates agricultural land and red indicates urban or built up land (areas not shaded brown or red within the range of the YBBB are generally undisturbed or natural land). Numbers indicate level II ecoregions. (Data from: 2010 North American Landcover Dataset and Land Change Monitoring System at <https://landcover.usgs.gov/nalcms.php>).

Our assumption that undeveloped land cover within an occupied ecoregion is potentially suitable habitat for the YBBB is based on observations indicating the species utilizes a wide range of habitat types within its historical range. This construct is supported by a comparison of current YBBB occupancy and land use across ecoregions, where an inverse relationship exists between percent occupancy and developed (e.g., agricultural and urban) land (figure 20).

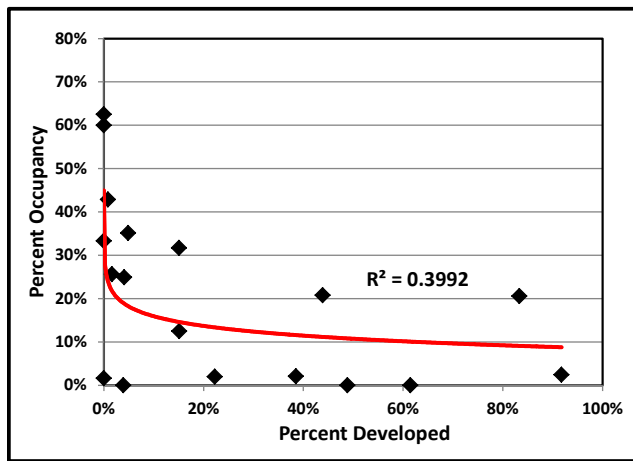


Figure 20. Logarithmic relationship of current YBBB occupancy and land use across ecoregions. “Percent Developed” is the percentage of land within an ecoregion classified as agricultural and urban/built-up.

Because there are no detailed land cover data for the historical time period, our estimates of historical YBBB population numbers may be high in some ecoregions where agriculture and urban development had already altered the landscape. However we expect these discrepancies to be relatively minor with respect to the overall pattern of YBBB population changes. We base this on evidence that recent increases in “agricultural intensity” (*i.e.*, the increasing use of monoculture crops, consolidation of fields with a resultant loss of natural edge habitat, increased reliance on GEHT crops and novel pesticides, etc.) is more closely tied to pollinator decline than simply the amount of land cover classified as agricultural. In other words, agricultural practices during much of the historical time period were generally more compatible with healthy pollinator communities than current practices.

Based on the methodology described above, we estimate there were historically about 55,272 YBBB populations spread across the species’ range (table 9). The number of populations in each ecoregion varied substantially, ranging from 27 to 14,830. We estimate that currently there are a total 43,829 or 47,364 YBBB populations, depending on our assumptions regarding the suitability of currently developed land as habitat for the species. Our analysis indicates that YBBB populations have declined in 14 ecoregions (by about 9 to 100 percent) and increased in four ecoregions (by about 9 to 194 percent). Rangewide, the current number of YBBB populations appears to be 14 to 21 percent lower than the historical numbers, depending on the assumptions regarding the suitability of currently developed land cover. And as noted

previously, the data suggest that three historical ecoregions currently support no YBBB populations.

Table 9. Estimated YBBB populations (e.g. occupied hexagons) by time period and level II ecoregion. “Historical” estimates assume that the land area within the historical range of the YBBB, by ecoregion, was potentially suitable habitat for the species. “Current Entire” estimates assume that the land area within the current range of the species, by ecoregion, is potentially suitable habitat for the YBBB while “Excluding Developed” assumes that land classified as agricultural or urban is not suitable habitat for the YBBB. Gray shading indicates ecoregions where the YBBB is presumed to be extirpated.

Level II Ecoregion		Estimated YBBB Populations			Percent Change (from Historical)	
		Historical	Current		Entire	Excluding Developed
			Entire	Excluding Developed		
3.3	Taiga Plains	8,574	9,846	9,846	15%	15%
3.4	Taiga Shield	1,033	1,124	1,124	9%	9%
4.1	Hudson Plains	2,061	6,049	6,049	194%	194%
5.1	Softwood Shield	14,830	13,465	13,465	-9%	-9%
5.2	Mixed Wood Shield	8,308	3,257	3,128	-61%	-62%
5.3	Atlantic Highlands	2,290	1,736	1,671	-24%	-27%
5.4	Boreal Plain	10,280	6,590	5,594	-36%	-46%
6.1	Boreal Cordillera	31	10	10	-67%	-67%
6.2	Western Cordillera	798	1,266	1,244	59%	56%
7.1	Marine West Coast Forest	27	0	0	-100%	-100%
8.1	Mixed Wood Plains	2,362	2,052	1,228	-13%	-48%
8.2	Central USA Plains	213	21	3	-90%	-98%
8.3	Southeastern USA Plains	79	0	0	-100%	-100%
8.4	Ozark, Ouachita-Appalachian Forests	524	26	19	-95%	-96%
8.5	Southeast USA Coastal Plains	39	0	0	-100%	-100%
9.2	Temperate Prairies	3,312	1,679	292	-49%	-91%
9.3	West Central Semi-Arid Prairies	414	168	78	-59%	-81%
10.1	Cold Deserts	94	75	75	-20%	-20%
		55,272	47,364	43,829	-14%	-21%

We emphasize that this methodology produces an estimate of population numbers only and does not consider potential population connectivity or fragmentation effects, YBBB genetics, demography, or population dynamics, or other potential factors that may influence the condition of the YBBB within ecoregions. It is also probable that patches of unsuitable habitat exist in some undeveloped areas and that patches of suitable habitat exist within some agricultural or urban areas. However, at the scale of our analyses we do not expect these deviations to significantly affect the results of our overall assessment.

Representation - Representation describes the ability of a species to adapt to changing environmental conditions over time. It is characterized by the breadth of genetic and environmental diversity within and among populations. Because there are insufficient data to describe the genetic, phenotypic, or behavioral diversity of the YBBB across its range, we consider its condition in various ecological settings, represented by the 18 level II ecoregions, as an indicator of the species’ representation. The ecoregions are characterized by differences in climate, vegetation, geology, and other habitat variables to which we assume the resident YBBB has adapted.

As mentioned previously, the YBBB is historically known from 18 different level II ecoregions, indicating a historically high degree of heterogeneity. The best available information suggests the species has been extirpated from three of these ecoregions, possibly reducing overall YBBB representation. Additionally, it is possible that the 23 percent reduction in range described above

has resulted in the loss of genetically or phenotypically adapted YBBBs from certain habitats within an ecoregion. For example, research shows that in the Atlantic Highlands ecoregion (ecoregion 5.3), the YBBB is found in some higher elevation sites but is no longer found in lower elevation habitats (Tucker and Rehan 2017, pp. 7–8) and the data suggests this upslope shift may be occurring rangewide (Richardson 2018, pers. comm.). This loss of low-elevation YBBBs may represent a decline in the species’ adaptive capacity (heterogeneity).

Resiliency - Resiliency describes the ability of a species to withstand natural variation in environmental conditions or population demographics and is positively related to population size, growth rate, and connectivity among populations. In general, populations need abundant colonies within habitat patches of adequate size and quality to persist in the face of natural demographic or environmental perturbations.

As previously mentioned, we have little direct information with which to characterize the resiliency of individual YBBB populations. Therefore we developed an indirect method for describing and comparing the species’ resiliency across ecoregions based on the best available data, the occupancy and relative abundance data derived from *Bombus* spp. surveys. Assessing the condition of the YBBB using relative abundance data is complicated by the variability in survey effort (as previously discussed) and by potential changes in the abundance of other co-occurring *Bombus* spp. For example, if all co-occurring *Bombus* spp. (including YBBB) decline equally, their relative abundances will not change, thus absolute declines in YBBB (or other *Bombus* spp.) may not be apparent. Here we also apply YBBB occupancy data to help determine if YBBB population numbers (and by extension, individual YBBBs) have changed over time.

Our methodology involves the use of matrices to plot each metric (percent occupancy and relative abundance) along an axis (figure 21). The numerical bins for each axis are based on quartiles for the relevant data in the historical dataset, our reference condition for the species. The results are used to describe YBBB resiliency as “high,” “moderate,” or “low” in each ecoregion. It is important to recognize that these scores describe the species’ resiliency at a particular point in time (or within a particular time frame) and do not take into account any potential trends in the data. However, separate scoring of the historical and current data along with modeled projections for these parameters is used later to help determine trends and assess the species’ future viability.

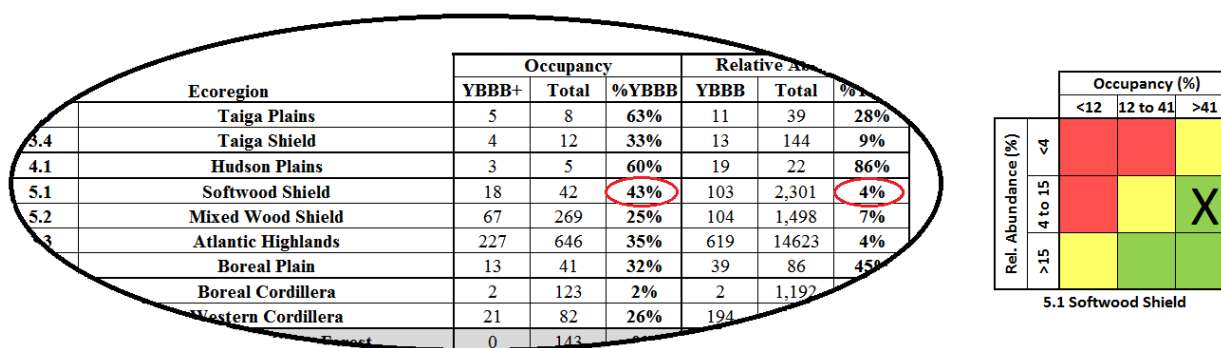


Figure 21. Methodology for describing YBBB resiliency at the ecoregion level. On the left is an excerpt from Table 7 (see Current Distribution section above) showing the percentage values used to inform the resiliency matrix

for ecoregion 5.1, shown at the right. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. In this example the current resiliency of ecoregion 5.1 is “high.”

Applying this method across ecoregions indicates that the YBBB currently has high resiliency in five ecoregions (28 percent), moderate resiliency in four ecoregions (22 percent), low resiliency in six ecoregions (33 percent), and no resiliency in the three ecoregions (17 percent) from which it is presumed extirpated. Considering the historical condition, resiliency appears to have increased in three ecoregions (17 percent), remained the same in eight ecoregions (44 percent), and declined in seven ecoregions (39 percent) (table 10 and figures 22 and 23).

Table 10 Historical and current resiliency of YBBB populations by ecoregion. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency.

	Ecoregion	Historical	Current
3.3	Taiga Plains		
3.4	Taiga Shield		
4.1	Hudson Plains		
5.1	Softwood Shield		
5.2	Mixed Wood Shield		
5.3	Atlantic Highlands		
5.4	Boreal Plain		
6.1	Boreal Cordillera		
6.2	Western Cordillera		
7.1	Marine West Coast Forest		Extirpated
8.1	Mixed Wood Plains		
8.2	Central USA Plains		
8.3	Southeastern USA Plains		Extirpated
8.4	Ozark, Ouachita-Appalachian Forests		
8.5	Southeast USA Coastal Plains		Extirpated
9.2	Temperate Prairies		
9.3	West Central Semi-Arid Prairies		
10.1	Cold Deserts		

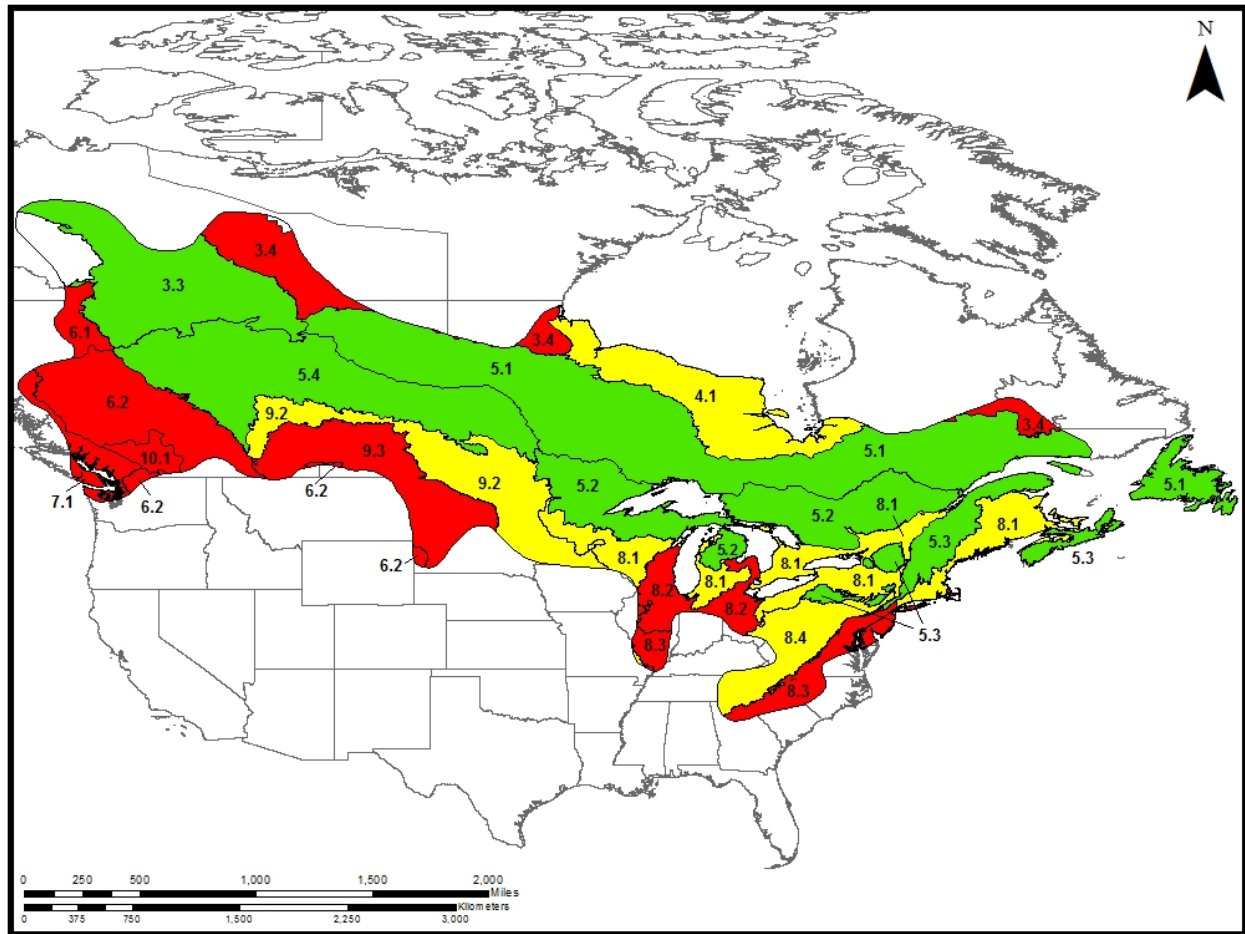


Figure 22. Historical resiliency of the YBBB by ecoregion. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency.

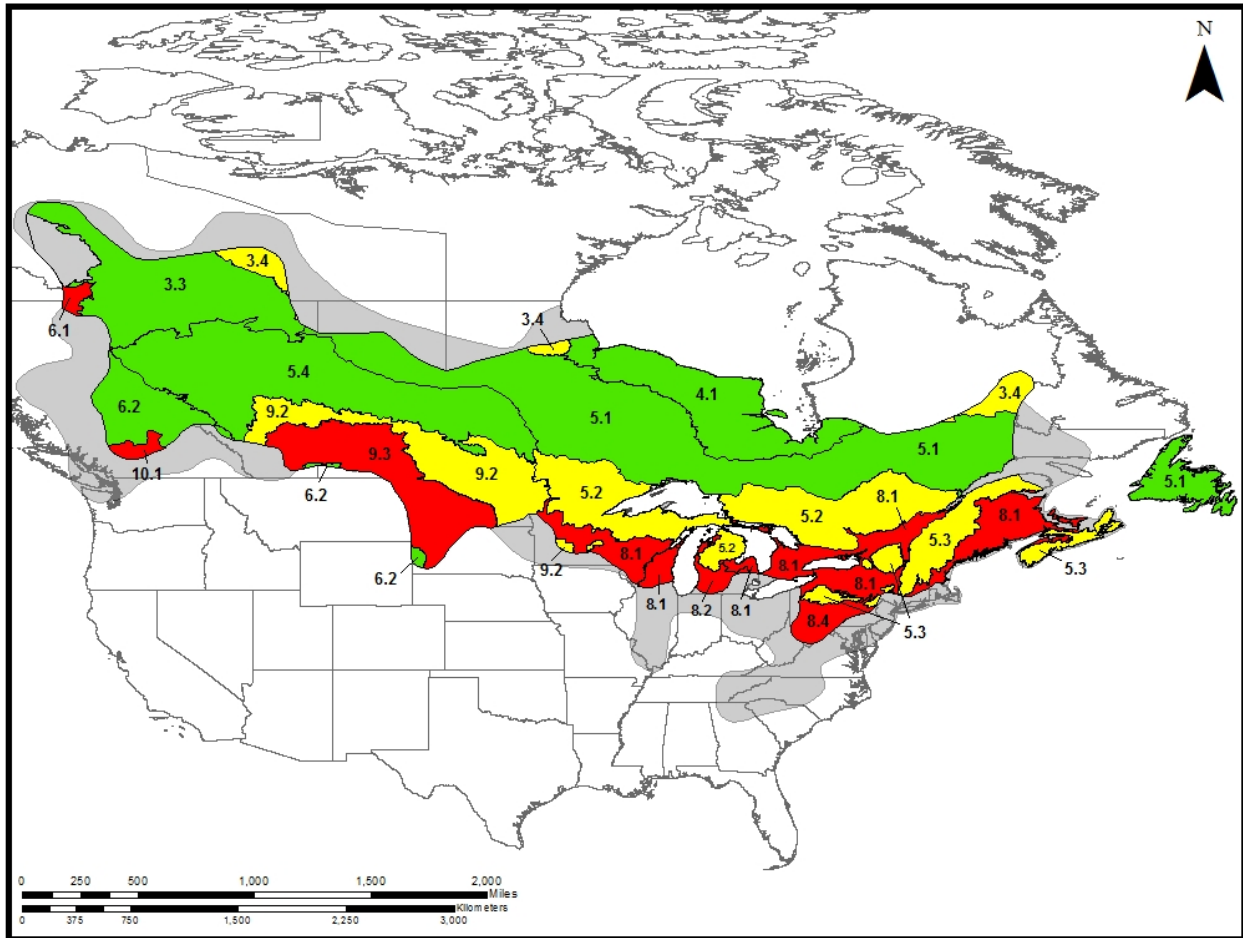


Figure 23. Current resiliency of the YBBB by ecoregion. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated.

Summary

The best available information indicates that historically the YBBB ranged across much of North America, including all or portions of 25 U.S. states and 12 Canadian provinces. However in recent decades the species’ range has receded markedly in the southern and far western portions of its historical range and appears to have been extirpated from much of the Pacific Northwest, Southern Appalachians, and southeast plains. The YBBB’s range also appears to have receded in some far northern Canadian provinces but this pattern is less certain because of generally low survey coverage in the region. The YBBB was once common and well represented within the core of its range (e.g., the upper Great Lakes region, New England, Ontario, Quebec, and the Canadian Maritime Provinces); however it appears to have lost resiliency and likely to have lost representation in these areas. There are no current YBBB records in 11 of the U.S. states from which it is historically known (figure 16).

Chapter 5: Future Conditions

Predicting the future condition of the YBBB is complicated by the wide geographic range of the species, the variety of stressors acting on the species (alone or in concert), and uncertainty regarding potential changes in the stressors and/or the degree to which they affect the species. Under the 3Rs analysis, the resiliency of the YBBB within the ecoregions informs our assessment of the species' overall redundancy and representation. Therefore, to assess the future viability of the YBBB, we first assessed the potential future resiliency of the species within each ecoregion under four different plausible future scenarios. We then used this information to help predict potential changes in the YBBB's redundancy and representation.

We limited our assessment of the future condition of the YBBB to approximately twenty years, or twenty YBBB colony life-cycles (*i.e.*, generations) because our modeled results become increasingly uncertain after two decades in the future, reducing our confidence in making longer-term projections beyond this timeframe. We also note that the existing survey data for the species demonstrates relatively rapid changes in the YBBB's occupancy and/or relative abundance in some ecoregions within the last several decades. This suggests that continued changes in the species' viability will also be apparent within the next several decades. Therefore, we consider twenty years to be a reasonable timeframe for assessing the future viability of the YBBB.

Methodology - We developed four plausible scenarios to help assess the future resiliency of the YBBB. In three of these scenarios, Scenarios A, B, and C, we used the best available scientific and commercial information, including our best professional judgement to predict plausible changes or trends in the primary stressors (habitat loss and fragmentation, pathogens and parasites, pesticides, small and/or isolated populations, and the effects of climate change; (see Chapter 3 *Influences on Viability Stressor* for additional information) and predict the effects these changes may have on the future viability of the YBBB. These factors were assessed individually and then considered collectively to predict future YBBB resiliency in each ecoregion. Because there is uncertainty regarding the future changes in these stressors and their potential to influence the YBBB, we assessed them under positive assumptions (Scenarios A and B) and negative assumptions (Scenario C). The fourth scenario, Scenario D, uses the historical and current data on YBBB occupancy and relative abundance to model these parameters into the future and does not explicitly incorporate the stressor information; however, we assume that the effects of the past stressors are implicitly incorporated into the data. Under Scenario D the modeled predictions are used to describe the YBBB's future resiliency using the method described above in the 3Rs Analysis section.

Under Scenarios A, B, and C there are uncertainties regarding the possible trends, interactions, and ultimate effects of the various stressors across the range of the YBBB that reduce our confidence in making projections beyond about twenty years in the future. Similarly, under Scenario D the modeled results become increasingly uncertain after two decades in the future.

Scenarios A, B, and C - These future scenarios are based on the best available published literature related to the effects of five main stressors on the YBBB, plausible trends or predictions for these stressors, and YBBB survey data. Scenarios A and B assume the stressors

change in a way that produces a generally positive effect on the YBBB. While both Scenarios A and B are positive, because of greater uncertainty surrounding the potential effects of climate change, they differ from each other in their assumptions regarding the way the effects of climate change may affect the YBBB in some ecoregions. We also note that under these scenarios it is implausible that some stressors will change, or change in a way that has a positive effect on the YBBB. For example, if introduced pathogens and parasites are already established in a wild bee population, it is not likely the negative effects will cease even if new pathogen introductions are eliminated. Likewise, the negative effects of climate change already observed in some ecoregions are unlikely to reverse even under the most optimistic plausible climate model.

Scenario C assumes the stressors will change in ways that result in generally negative effects to the YBBB. But again, even under generally negative assumptions, some stressors may not change, or change in a way that will necessarily have a negative effect on the YBBB.

The criteria and rationale used in our assessment of the five stressors within each ecoregion are summarized in table 11. We emphasize that the numerical criteria used to describe the status of each stressor and their potential effect on YBBB should not be considered absolute measures and should only be used to compare ecoregions. For example, there are no data available specifically linking a greater than 5-percent agricultural land use with bee pathogen spread. However, increased agricultural development can reasonably be expected to also increase the use of commercial bees for agricultural purposes, thus increasing the likelihood that bee pathogens and parasites will also be spread to wild bee populations. While not tied specifically to commercial bee usage (or any other particular stressor), the relationship between YBBB occupancy and land cover (figure 20) suggests that greater than about five percent development is generally associated with lower YBBB occupancies. Therefore, we selected 5 percent agricultural development to define the status of the pathogen and parasite and the pesticide stressors.

Table 11. Criteria for assessing potential YBBB stressors under Scenarios A, B, and C.

Stressor	Status	Criteria w/Reasonable Assumptions
Habitat Fragmentation/Loss	Minimal Development	0 to 5 percent developed; natural habitat largely intact.
	Light Development	6 to 25 percent developed; minor habitat fragmentation or loss.
	Moderate Development	26 to 75 percent developed; significant habitat fragmentation or loss.
	Heavy Development	>75 percent developed; severe habitat fragmentation or loss.
Pathogens and Parasites	Stable Prevalence	0 to 5 percent agricultural land use; minimal risk of introduced pathogens; natural infection/infestation rates.
	Increased Prevalence	>5 percent agricultural land use; use of commercial bees likely to increase pathogen introduction and spread.
	Decreased Prevalence	Changes in agriculture and/or husbandry of commercial bees limits introduction and spread of pathogens.
Pesticides	Little or no use	0 to 5 percent agricultural land use; minimal risk of pesticide exposure.
	Increased Use	>5 percent agricultural land use; pesticide applications likely to affect bees.
	Reduced Use	Changes in agricultural practices reduce pesticide effects to bees.
Small and/or Isolated Populations	Stable Populations	YBBB population dynamics and species distribution stable.
	Expanding Populations	YBBB populations increasing in number and/or distribution.
	Declining Populations	YBBB populations declining in number and/or becoming more isolated.
Effects of Climate Change	Stable Habitat Suitability	Habitat characteristics remain tolerable to YBBB.
	Increased Habitat Suitability	Changes in floral resources, temperatures, etc. provide a net benefit to YBBB
	Decreased Habitat Suitability	Changes in floral resources, temperatures, etc. are detrimental to YBBB

Under Scenarios A, B, and C, each stressor is assessed by ecoregion in relation to the current condition. The three possible results of this exercise are that changes in the stressor will have a positive effect on the YBBB in relation to the current condition, changes in the stressor will have

a negative effect on the YBBB in relation to the current condition, or the stressor or effects of the stressor will not change significantly from the current condition. Within an ecoregion, if there is a net change (either positive or negative) in two or more stressors, the resiliency of the ecoregion will likewise either increase or decrease from the current resiliency score (see table 10).

Scenario A assumes plausible positive changes in stressor effects to YBBB, where relevant. Under this scenario, the effects of climate change are assumed to have either a negligible or positive effect on YBBB in some northern ecoregions (table 12). This assumes that even with warming temperatures, these ecoregions are expected to maintain current conditions or perhaps improve for the YBBB, leading to an increase in colonies and/or populations. We note that the available scientific research does not necessarily support this hypothesis, however we include it here because the available YBBB survey data indicate the species' resiliency may have increased in some northern ecoregions. In the more southern ecoregions the effects of climate change are expected to continue exerting negative pressure on the YBBB under even the most optimistic plausible climate model. Under this scenario we also assume that the negative effects of pesticides decline in those ecoregions where agriculture is prevalent. This could be the result of changes in product formulations and/or significant reductions in pesticide use. The negative effects of pathogens and parasites are expected to continue in ecoregions where commercial bees are used (*i.e.*, ecoregions with greater than minimal development). We do not expect development (agricultural or urban) to change significantly under this scenario, so those ecoregions where development is minimal are not expected to see a change in either pesticides or pathogens and parasites. Table 12 provides a summary of the potential changes in YBBB stressors under the optimistic assumptions of Scenario A, table 13 shows the resultant changes in resiliency from the current condition (with historical resiliencies provided for context), and figure 24 is a map of the predicted resiliency of each ecoregion under Scenario A.

Table 12. Stressor assessment under Scenario A. Green shading indicates an assumption that conditions change favorably for the YBBB relative to the current condition; red shading indicates that conditions are predicted to worsen for the YBBB, and unshaded cells indicate no significant change in the stressor from the current condition. Cross hatching indicates that the YBBB is extirpated from that ecoregion.

	Ecoregion	Habitat Fragmentation/Loss	Pathogens and Parasites	Pesticides	Small and/or Isolated Populations	Effects of Climate Change
3.3	Taiga Plains	Minimal development	Stable prevalence	Little or no use	Expanding populations	Increased habitat suitability
3.4	Taiga Shield	Minimal development	Stable prevalence	Little or no use	Expanding populations	Increased habitat suitability
4.1	Hudson Plains	Minimal development	Stable prevalence	Little or no use	Expanding populations	Increased habitat suitability
5.1	Softwood Shield	Minimal development	Stable prevalence	Little or no use	Expanding populations	Increased habitat suitability
5.2	Mixed Wood Shield	Minimal development	Stable prevalence	Little or no use	Stable populations	Stable habitat suitability
5.3	Atlantic Highlands	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
5.4	Boreal Plain	Light development	Increased prevalence	Reduced use	Stable populations	Stable habitat suitability
6.1	Boreal Cordillera	Minimal development	Stable prevalence	Little or no use	Expanding populations	Increased habitat suitability
6.2	Western Cordillera	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
7.1	Marine West Coast Forest	Minimal development	Stable prevalence	Little or no use	Extirpated	Decreased habitat suitability
8.1	Mixed Wood Plains	Moderate development	Increased prevalence	Reduced use	Declining populations	Decreased habitat suitability
8.2	Central USA Plains	Heavy development	Increased prevalence	Reduced use	Declining populations	Decreased habitat suitability
8.3	Southeastern USA Plains	Moderate development	Increased prevalence	Reduced use	Extirpated	Decreased habitat suitability
8.4	Ozark, Ouachita-Appalachian Forests	Light development	Increased prevalence	Reduced use	Declining populations	Decreased habitat suitability
8.5	Southeast USA Coastal Plains	Light development	Increased prevalence	Reduced use	Extirpated	Decreased habitat suitability
9.2	Temperate Prairies	Heavy development	Increased prevalence	Reduced use	Declining populations	Decreased habitat suitability
9.3	West Central Semi-Arid Prairies	Moderate development	Increased prevalence	Reduced use	Declining populations	Decreased habitat suitability
10.1	Cold Deserts	Light development	Increased prevalence	Reduced use	Declining populations	Decreased habitat suitability

Table 13. Predicted resiliency of YBBB by ecoregion under Scenario A with historical and current resiliencies provided for comparison. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency.

Ecoregion		Historical	Current	A
3.3	Taiga Plains			
3.4	Taiga Shield			
4.1	Hudson Plains			
5.1	Softwood Shield			
5.2	Mixed Wood Shield			
5.3	Atlantic Highlands			
5.4	Boreal Plain			
6.1	Boreal Cordillera			
6.2	Western Cordillera			
7.1	Marine West Coast Forest		Extirpated	Extirpated
8.1	Mixed Wood Plains			
8.2	Central USA Plains			
8.3	Southeastern USA Plains		Extirpated	Extirpated
8.4	Ozark, Ouachita-Appalachian Forests			
8.5	Southeast USA Coastal Plains		Extirpated	Extirpated
9.2	Temperate Prairies			
9.3	West Central Semi-Arid Prairies			
10.1	Cold Deserts			

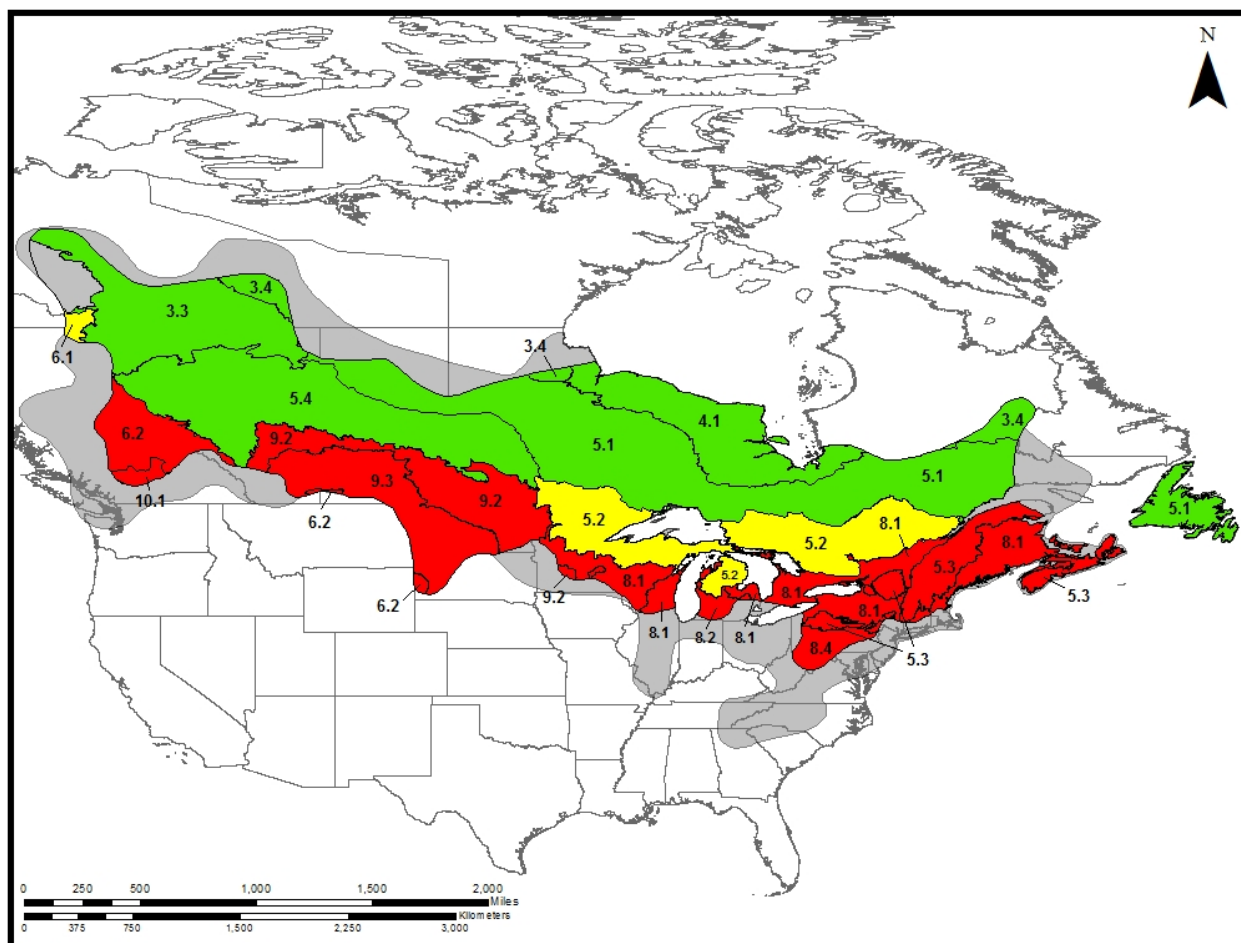


Figure 24. Predicted resiliency of the YBBB by ecoregion under Scenario A. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated.

Scenario B is identical to Scenario A with the exception that the effects of climate change are not expected to produce any beneficial effects to the YBBB in the northern ecoregions (tables 14 and 15, figure 25).

Table 14. Stressor assessment under Scenario B. Green shading indicates an assumption that conditions change favorably for the YBBB relative to the current condition; red shading indicates that conditions are predicted to worsen for the YBBB, and unshaded cell indicate no significant change in the stressor from the current condition. Cross hatching indicates that the YBBB is extirpated from that ecoregion.

Ecoregion		Habitat Fragmentation/Loss	Pathogens and Parasites	Pesticides	Small and/or Isolated Populations	Effects of Climate Change
3.3	Taiga Plains	Minimal development	Stable prevalence	Little or no use	Stable populations	Stable habitat suitability
3.4	Taiga Shield	Minimal development	Stable prevalence	Little or no use	Stable populations	Stable habitat suitability
4.1	Hudson Plains	Minimal development	Stable prevalence	Little or no use	Stable populations	Stable habitat suitability
5.1	Softwood Shield	Minimal development	Stable prevalence	Little or no use	Stable populations	Stable habitat suitability
5.2	Mixed Wood Shield	Minimal development	Stable prevalence	Little or no use	Stable populations	Stable habitat suitability
5.3	Atlantic Highlands	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
5.4	Boreal Plain	Light development	Increased prevalence	Reduced Use	Stable populations	Stable habitat suitability
6.1	Boreal Cordillera	Minimal development	Stable prevalence	Little or no use	Stable populations	Stable habitat suitability
6.2	Western Cordillera	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
7.1	Marine West Coast Forest	Minimal development	Stable prevalence	Little or no use	Extirpated	Decreased habitat suitability
8.1	Mixed Wood Plains	Moderate development	Increased prevalence	Reduced Use	Declining populations	Decreased habitat suitability
8.2	Central USA Plains	Heavy development	Increased prevalence	Reduced Use	Declining populations	Decreased habitat suitability
8.3	Southeastern USA Plains	Moderate development	Increased prevalence	Reduced Use	Extirpated	Decreased habitat suitability
8.4	Ozark, Ouachita-Appalachian Forests	Light development	Increased prevalence	Reduced Use	Declining populations	Decreased habitat suitability
8.5	Southeast USA Coastal Plains	Light development	Increased prevalence	Reduced Use	Extirpated	Decreased habitat suitability
9.2	Temperate Prairies	Heavy development	Increased prevalence	Reduced Use	Declining populations	Decreased habitat suitability
9.3	West Central Semi-Arid Prairies	Moderate development	Increased prevalence	Reduced Use	Declining populations	Decreased habitat suitability
10.1	Cold Deserts	Light development	Increased prevalence	Reduced Use	Declining populations	Decreased habitat suitability

Table 15. Predicted resiliency of YBBB by ecoregion under Scenario B with historical and current resiliencies provided for comparison. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency.

Ecoregion		Historical	Current	B
3.3	Taiga Plains	Green	Green	Green
3.4	Taiga Shield	Red	Yellow	Yellow
4.1	Hudson Plains	Yellow	Green	Green
5.1	Softwood Shield	Green	Green	Green
5.2	Mixed Wood Shield	Green	Yellow	Yellow
5.3	Atlantic Highlands	Green	Yellow	Red
5.4	Boreal Plain	Green	Green	Green
6.1	Boreal Cordillera	Red	Red	Red
6.2	Western Cordillera	Red	Green	Red
7.1	Marine West Coast Forest	Red	Extirpated	Extirpated
8.1	Mixed Wood Plains	Yellow	Red	Red
8.2	Central USA Plains	Red	Red	Red
8.3	Southeastern USA Plains	Red	Extirpated	Extirpated
8.4	Ozark, Ouachita-Appalachian Forests	Yellow	Red	Red
8.5	Southeast USA Coastal Plains	Red	Extirpated	Extirpated
9.2	Temperate Prairies	Yellow	Yellow	Red
9.3	West Central Semi-Arid Prairies	Red	Red	Red
10.1	Cold Deserts	Red	Red	Red

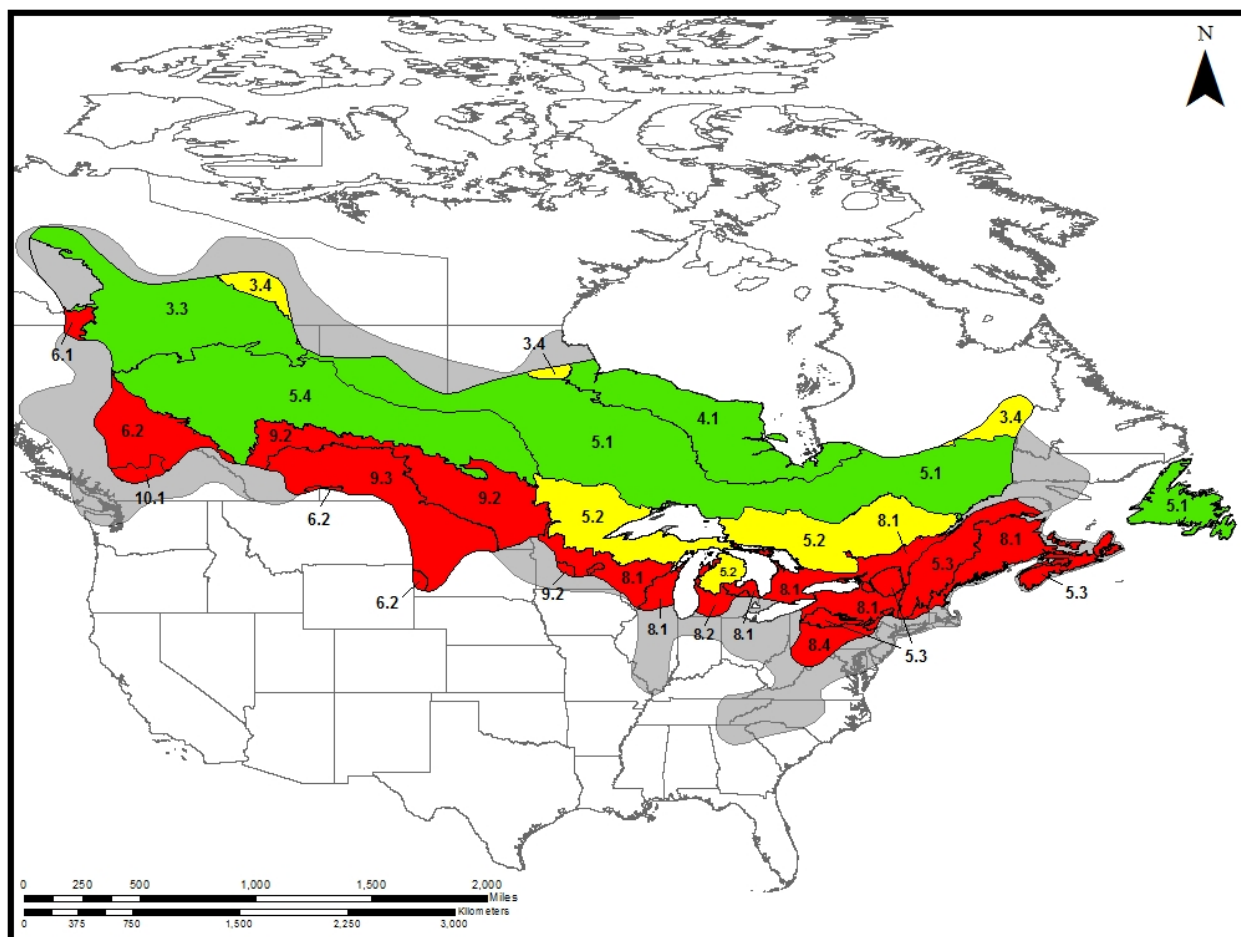


Figure 25. Predicted resiliency of the YBBB by ecoregion under Scenario B. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated.

Under Scenario C, the negative effects of the various stressors are predicted to worsen or remain generally unchanged (tables 16 and 17, figure 26). The effects of climate change are expected to have a negative effect on the YBBB throughout the species’ range with a concurrent decline in colony and population numbers. Additionally, under this scenario agriculture or other development increases in some ecoregions enough to move the ecoregion into the next classification unit (e.g., from minimally developed to light development). Agricultural intensity is projected to increase in all but the northern ecoregions, where the soil is generally not amenable to farming. In the ecoregions where development increases, we predict a concurrent increase in negative pesticide effects and in the introduction and transmission of bee pathogens and parasites.

The effects of the additional negative stressors are predicted to increase the likelihood of YBBB extirpation in the six ecoregions where its resiliency is currently low. This would obviously result in additional contraction of the species’ southern and western range and concurrent loss in representation. Based on our population estimates described earlier, the extirpation of YBBB from these ecoregions would represent a loss of about 1,500 populations, or about three percent of the roughly 45,000 current populations.

Table 16. Stressor assessment under Scenario C. Red shading indicates that conditions are predicted to worsen for the YBBB and unshaded cells indicate no significant change in the stressor is expected. Cross hatching indicates that the YBBB is extirpated from that ecoregion.

	Ecoregion	Habitat Fragmentation/Loss	Pathogens and Parasites	Pesticides	Small and/or Isolated Populations	Effects of Climate Change
3.3	Taiga Plains	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
3.4	Taiga Shield	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
4.1	Hudson Plains	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
5.1	Softwood Shield	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
5.2	Mixed Wood Shield	Light development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability
5.3	Atlantic Highlands	Light development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability
5.4	Boreal Plain	Light development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability
6.1	Boreal Cordillera	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
6.2	Western Cordillera	Minimal development	Stable prevalence	Little or no use	Declining populations	Decreased habitat suitability
7.1	Marine West Coast Forest	Light development	Stable prevalence	Little or no use	Extirpated	Decreased habitat suitability
8.1	Mixed Wood Plains	Moderate development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability
8.2	Central USA Plains	Heavy development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability
8.3	Southeastern USA Plains	Moderate development	Increased prevalence	Increased use	Extirpated	Decreased habitat suitability
8.4	Ozark, Ouachita-Appalachian Forests	light development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability
8.5	Southeast USA Coastal Plains	light development	Increased prevalence	Increased use	Extirpated	Decreased habitat suitability
9.2	Temperate Prairies	Heavy development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability
9.3	West Central Semi-Arid Prairies	Moderate development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability
10.1	Cold Deserts	Light development	Increased prevalence	Increased use	Declining populations	Decreased habitat suitability

Table 17. Predicted resiliency of YBBB by ecoregion under Scenario C with historical and current resiliencies provided for comparison. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Cross hatching indicates ecoregions where the risk of YBBB extirpation is high.

	Ecoregion	Historical	Current	C
3.3	Taiga Plains	Green	Green	Yellow
3.4	Taiga Shield	Red	Yellow	Red
4.1	Hudson Plains	Yellow	Green	Yellow
5.1	Softwood Shield	Green	Green	Yellow
5.2	Mixed Wood Shield	Green	Yellow	Red
5.3	Atlantic Highlands	Green	Yellow	Red
5.4	Boreal Plain	Green	Green	Yellow
6.1	Boreal Cordillera	Red	Red	Cross-hatched
6.2	Western Cordillera	Red	Green	Cross-hatched
7.1	Marine West Coast Forest	Red	Extirpated	Extirpated
8.1	Mixed Wood Plains	Yellow	Red	Cross-hatched
8.2	Central USA Plains	Red	Red	Cross-hatched
8.3	Southeastern USA Plains	Red	Extirpated	Extirpated
8.4	Ozark, Ouachita-Appalachian Forests	Yellow	Red	Cross-hatched
8.5	Southeast USA Coastal Plains	Red	Extirpated	Extirpated
9.2	Temperate Prairies	Yellow	Yellow	Red
9.3	West Central Semi-Arid Prairies	Red	Red	Cross-hatched
10.1	Cold Deserts	Red	Red	Cross-hatched

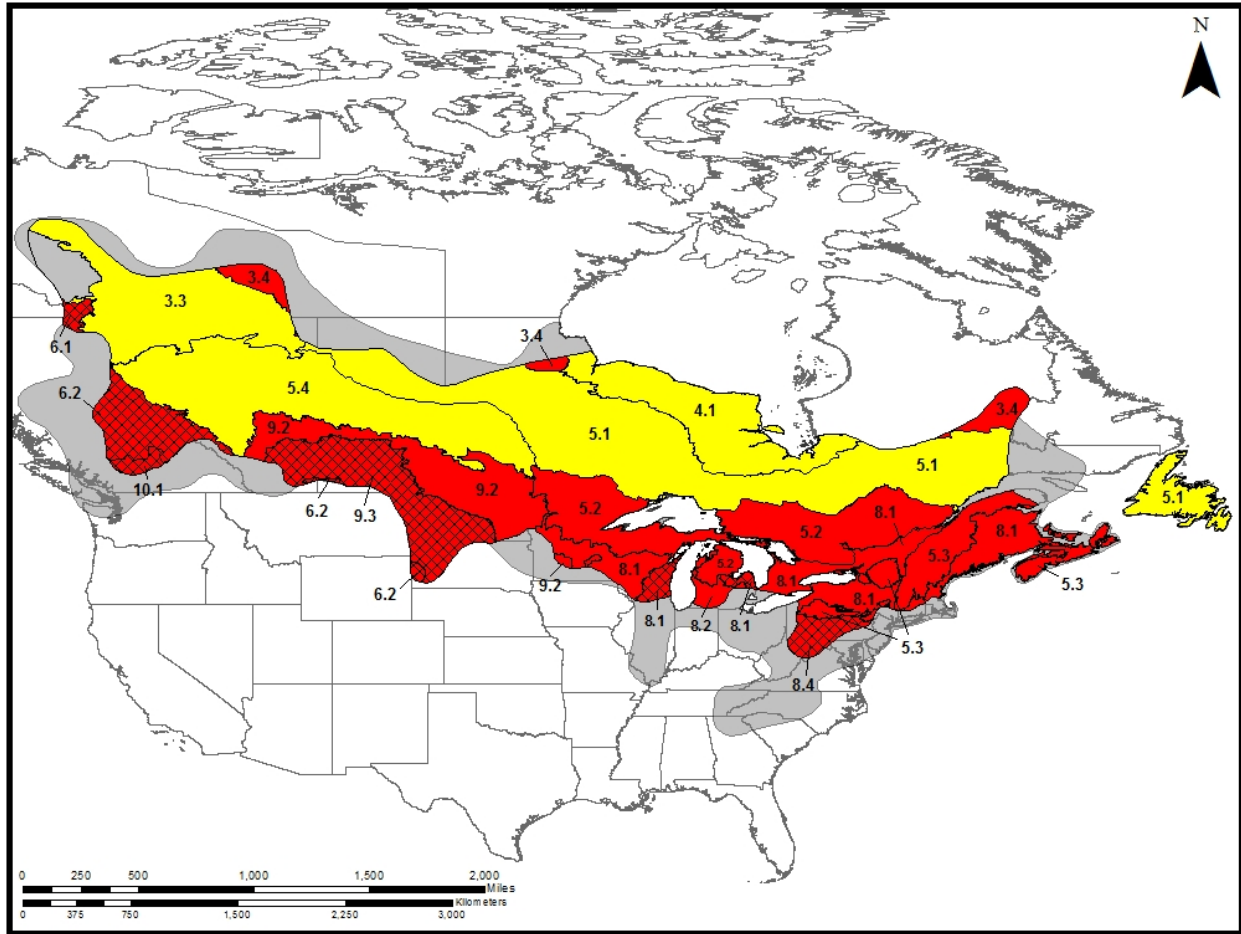


Figure 26. Predicted resiliency of the YBBB by ecoregion under Scenario C. Yellow shading indicates “moderate” resiliency, and red indicates “low” resiliency. Cross-hatching indicates ecoregions where the risk of YBBB extirpation is high and gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated.

Scenario D models the future resiliency of the YBBB based on projected trends in the existing survey data. These data are assumed to reflect the effects of the various stressors to the species. In other words, the model assumes that YBBB occurrence and relative abundance data (and trends) are determined by the effects of the various stressors acting on the species or its habitat, either alone or in combination. Under Scenario D we make no attempt to explicitly assess each stressor independently. However, we note that the effects of some emergent stressors (e.g., the effects of climate change, novel pathogens, and habitat fragmentation) may not be fully reflected in the current data. This may result in the model overestimating YBBB resiliency in the future.

The model uses YBBB and *Bombus* spp. data (YBBB presence in 100 sq km (38.6 sq mi) hexagonal units and individual YBBB counts within ecoregions) which are summarized at the decade scale. Trends in the data were estimated using a generalized linear mixed-effects regression (GLMER) model, which showed a decline in relative abundance in 9 of the 15 (60 percent) extant ecoregions and a decline in occupancy in 11 of 15 (73 percent) extant ecoregions over the next twenty years. At the species level, both relative abundance and occupancy were estimated to be in decline. For a more detailed discussion of the model, see Appendix C.

We used these future projections for YBBB occupancy and relative abundance to estimate the species' resiliency in each ecoregion, as described above in the 3Rs Analysis section. Under Scenario D, the resiliency of the YBBB is predicted to decline in five ecoregions, with one of these likely to become extirpated. Two ecoregions are predicted to improve while the remaining eleven are predicted to remain relatively unchanged (table 18 and figure 27).

Table 18. Predicted resiliency of YBBB by ecoregion under Scenario D with historical and current resiliencies provided for comparison. Green shading indicates "high" resiliency, yellow indicates "moderate" resiliency, and red indicates "low" resiliency. Cross hatching indicates ecoregions where the risk of YBBB extirpation is high.

	Ecoregion	Historical	Current	D
3.3	Taiga Plains	Green	Green	Green
3.4	Taiga Shield	Red	Yellow	Green
4.1	Hudson Plains	Yellow	Green	Yellow
5.1	Softwood Shield	Green	Green	Yellow
5.2	Mixed Wood Shield	Green	Yellow	Green
5.3	Atlantic Highlands	Green	Yellow	Red
5.4	Boreal Plain	Green	Green	Green
6.1	Boreal Cordillera	Red	Red	Red
6.2	Western Cordillera	Red	Green	Green
7.1	Marine West Coast Forest	Red	Extirpated	Extirpated
8.1	Mixed Wood Plains	Yellow	Red	Red
8.2	Central USA Plains	Red	Red	Red
8.3	Southeastern USA Plains	Red	Extirpated	Extirpated
8.4	Ozark, Ouachita-Appalachian Forests	Yellow	Red	Red
8.5	Southeast USA Coastal Plains	Red	Extirpated	Extirpated
9.2	Temperate Prairies	Yellow	Yellow	Red
9.3	West Central Semi-Arid Prairies	Red	Red	Red
10.1	Cold Deserts	Red	Red	Red

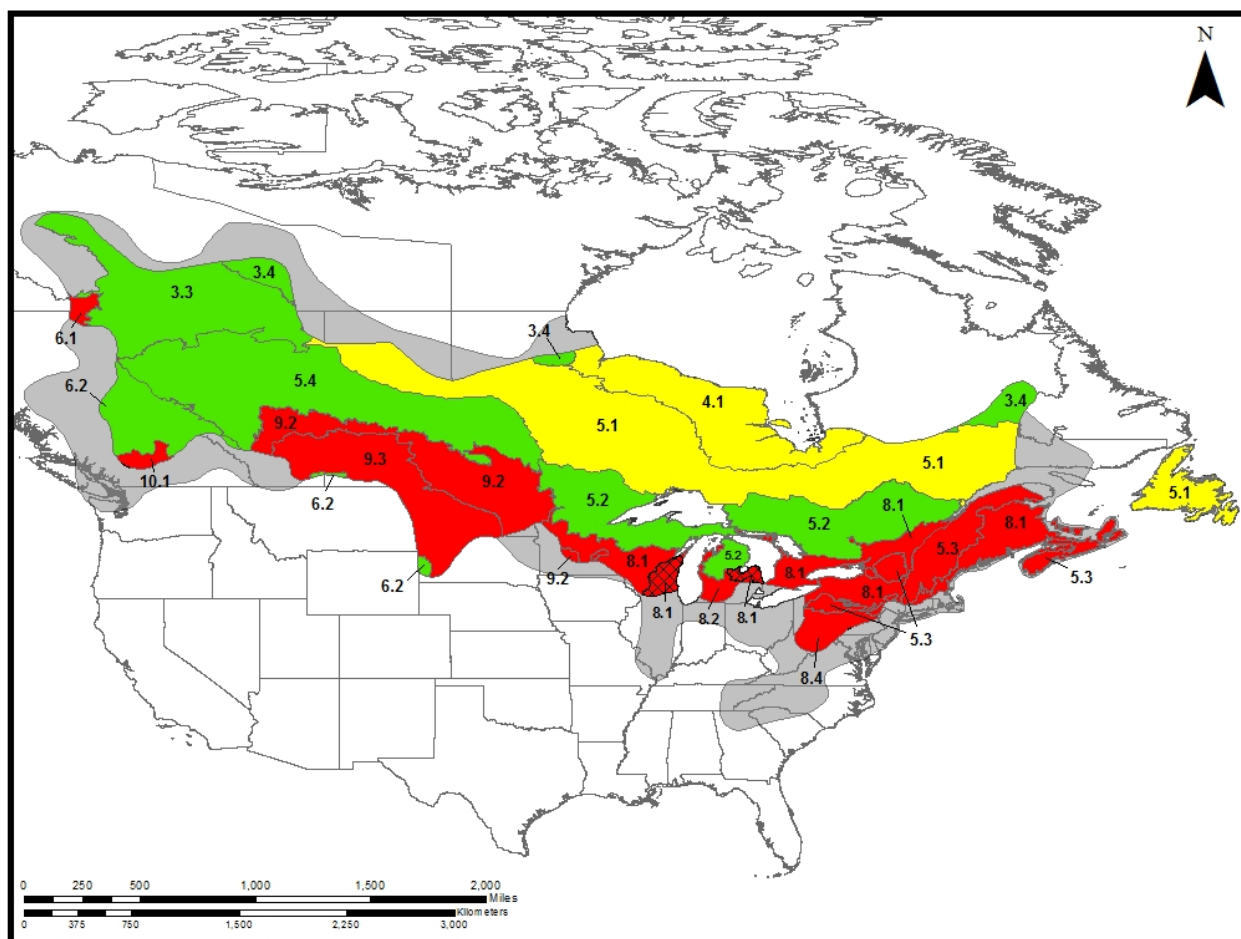


Figure 27. Predicted resiliency of the YBBB by ecoregion under Scenario D. Cross-hatching indicates ecoregions where the risk of YBBB extirpation is high and gray shading represents areas within the historical range of the YBBB where the species is presumed extirpated.

Summary

Under the generally positive assumptions of Scenario A, YBBB resiliency in three ecoregions is predicted to decline, two ecoregions are predicted to improve, and the rest are expected to remain relatively unchanged. Under Scenario B, which assumes climate change will have little effect on YBBB in the northern ecoregions, the species' resiliency is expected to decline in three ecoregions and remain unchanged in the remainder. Under Scenario C, which assumes generally negative changes in the stressors, YBBB resiliency will remain relatively unchanged in one ecoregion and decline in the remaining 14. Notably, under Scenario C the YBBB is likely to be extirpated from six ecoregions where its resiliency is currently low. Under Scenario D, the extrapolation of trends in YBBB data, resiliency is projected to decline in five ecoregions (with the YBBB likely to be extirpated from one), to improve in two, and to remain relatively unchanged in the remaining eight ecoregions. We note, however, that the Scenario D model may underestimate the effects of some stressors.

Under no future scenario do we predict YBBB representation will improve, and given our assessment that some ecoregions will decline in resiliency even under the optimistic scenarios,

representation and redundancy may actually decline too. Also, based on the general trends observed between the historical and current time periods, it is reasonable to conclude that the range of the species will not expand significantly under any future scenario and may in fact continue to contract, especially at the southern boundary. The effect of this range contraction is likely a continued net loss of YBBB populations and overall redundancy.

Table 19. Comparison of YBBB resiliency by ecoregion. Green shading indicates “high” resiliency, yellow indicates “moderate” resiliency, and red indicates “low” resiliency. Cross-hatching indicates ecoregions where the risk of YBBB extirpation is high.

Ecoregion		Historical Resiliency	Current Resiliency	Future Scenarios			
				A	B	C	D
3.3	Taiga Plains	Green	Green	Green	Green	Yellow	Green
3.4	Taiga Shield	Red	Yellow	Green	Yellow	Red	Green
4.1	Hudson Plains	Yellow	Green	Green	Green	Yellow	Yellow
5.1	Softwood Shield	Green	Green	Green	Green	Yellow	Yellow
5.2	Mixed Wood Shield	Green	Yellow	Yellow	Yellow	Red	Green
5.3	Atlantic Highlands	Green	Yellow	Red	Red	Red	Red
5.4	Boreal Plain	Green	Green	Green	Green	Yellow	Green
6.1	Boreal Cordillera	Red	Red	Yellow	Red	Cross-hatched	Red
6.2	Western Cordillera	Red	Green	Red	Red	Cross-hatched	Green
7.1	Marine West Coast Forest	Red	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated
8.1	Mixed Wood Plains	Yellow	Red	Red	Red	Red	Red
8.2	Central USA Plains	Red	Red	Red	Red	Cross-hatched	Cross-hatched
8.3	Southeastern USA Plains	Red	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated
8.4	Ozark, Ouachita-Appalachian Forests	Yellow	Red	Red	Red	Cross-hatched	Red
8.5	Southeast USA Coastal Plains	Red	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated
9.2	Temperate Prairies	Yellow	Yellow	Red	Red	Red	Red
9.3	West Central Semi-Arid Prairies	Red	Red	Red	Red	Cross-hatched	Red
10.1	Cold Deserts	Red	Red	Red	Red	Cross-hatched	Red

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Appendices

Appendix A: YBBB bee decadal occupancy and relative abundance by U.S. state or Canadian province

Yellow banded bumble bee decadal occupancy by U.S. state or Canadian province. “YBBB+” is the number of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record, “Total” is the number of surveyed hexagons, and “%YBBB” is the percentage of surveyed hexagons where a YBBB was detected. Gray shading indicates no survey data available.

	Pre-1950			1950-59			1960-69			1970-79			1980-89			1990-99			2000-09			2010-17		
	YBBB+	Total	%YBBB	YBBB+	Total	%YBBB	YBBB+	Total	%YBBB	YBBB+	Total	%YBBB	YBBB+	Total	%YBBB	YBBB+	Total	%YBBB	YBBB+	Total	%YBBB	YBBB+	Total	%YBBB
AB	18	71	25%	8	43	19%	6	46	13%	1	27	4%	1	10	10%	0	11	0%	20	43	47%	17	94	18%
BC	9	196	5%	3	83	4%	7	110	6%	5	52	10%	4	77	5%	4	29	14%	7	71	10%	27	159	17%
CT	14	76	18%	1	22	5%	2	53	4%	5	69	7%	7	54	13%	1	14	7%	1	72	1%	0	30	0%
GA	1	14	7%	0	5	0%	0	4	0%	0	1	0%	0	3	0%	0	3	0%	0	1	0%	0	1	0%
IL	4	119	3%	1	27	4%	2	59	3%	0	56	0%	0	61	0%	0	27	0%	0	97	0%	0	63	0%
IN	0	14	0%	2	26	8%	0	18	0%	0	6	0%	0	2	0%	0	1	0%	0	7	0%	0	8	0%
MA	36	80	45%	3	16	19%	3	19	16%	8	29	28%	13	34	38%	8	22	36%	9	90	10%	2	92	2%
MB	30	42	71%	23	29	79%	8	18	44%	3	9	33%	19	20	95%	9	11	82%	6	22	27%	7	26	27%
MD	1	31	3%	0	8	0%	0	18	0%	1	16	6%	1	13	8%	0	4	0%	0	50	0%	2	132	2%
ME	38	67	57%	10	16	63%	9	18	50%	10	17	59%	7	13	54%	3	9	33%	3	34	9%	198	490	40%
MI	140	248	56%	55	113	49%	69	125	55%	19	52	37%	18	37	49%	12	25	48%	0	17	0%	13	114	11%
MN	89	167	53%	38	71	54%	14	47	30%	13	45	29%	10	36	28%	12	52	23%	3	24	13%	21	170	12%
NB,NS,PE	21	46	46%	7	12	58%	14	33	42%	29	49	59%	17	32	53%	69	136	51%	29	106	27%	60	163	37%
NC	8	105	8%	11	63	17%	3	42	7%	5	42	12%	4	29	14%	4	24	17%	4	30	13%	0	26	0%
ND	10	47	21%	1	5	20%	3	20	15%	3	12	25%	6	10	60%	7	11	64%	2	5	40%	1	15	7%
NH	29	55	53%	10	16	63%	11	20	55%	16	30	53%	17	29	59%	4	11	36%	9	37	24%	28	78	36%
NJ	6	98	6%	0	20	0%	1	27	4%	3	42	7%	0	12	0%	0	18	0%	0	65	0%	0	80	0%
NL	10	21	48%	0	1	0%	0	4	0%	4	5	80%	0	0		0	9	0%	5	12	42%	12	23	52%
NT	4	18	22%	0	1	0%	1	16	6%	1	3	33%	0	0		0	0		11	13	85%	5	6	83%
NY	90	252	36%	7	37	19%	30	93	32%	28	92	30%	21	61	34%	5	40	13%	11	97	11%	14	130	11%
OH	1	107	1%	0	25	0%	4	32	13%	0	17	0%	1	10	10%	0	15	0%	0	9	0%	0	49	0%
ON	49	137	36%	30	107	28%	58	150	39%	53	179	30%	37	133	28%	30	95	32%	21	182	12%	66	406	16%
PA	14	77	18%	1	17	6%	4	26	15%	11	26	42%	5	24	21%	0	6	0%	3	72	4%	5	93	5%
QC	13	87	15%	4	18	22%	13	31	42%	4	25	16%	4	17	24%	2	10	20%	11	45	24%	30	99	30%
RI	0	7	0%	0	4	0%	0	2	0%	1	6	17%	0	3	0%	0	5	0%	0	9	0%	0	7	0%
SD	2	76	3%	2	24	8%	1	32	3%	0	18	0%	1	5	20%	0	2	0%	0	12	0%	1	38	3%
SK	11	38	29%	4	11	36%	7	11	64%	0	9	0%	1	6	17%	0	1	0%	3	6	50%	13	42	31%
TN	7	24	29%	4	7	57%	0	2	0%	1	7	14%	4	7	57%	3	6	50%	6	14	43%	0	12	0%
VA	0	19	0%	0	3	0%	0	14	0%	0	12	0%	0	4	0%	1	9	11%	0	15	0%	0	9	0%
VT	22	34	65%	4	13	31%	17	57	30%	24	72	33%	22	60	37%	17	46	37%	62	135	46%	71	229	31%
WA	1	67	1%	0	19	0%	0	42	0%	0	51	0%	0	11	0%	0	10	0%	0	26	0%	0	101	0%
WI	28	64	44%	15	33	45%	25	78	32%	20	38	53%	4	19	21%	6	49	12%	4	38	11%	31	100	31%
WV	0	25	0%	1	8	13%	3	16	19%	2	14	14%	0	10	0%	0	13	0%	0	74	0%	0	83	0%
YT	0	17	0%	0	3	0%	0	19	0%	0	18	0%	0	16	0%	0	0		0	18	0%	2	108	2%
Totals:	706	2,546	28%	245	906	27%	315	1,302	24%	270	1,146	24%	224	858	26%	197	724	27%	230	1,548	15%	626	3,276	19%

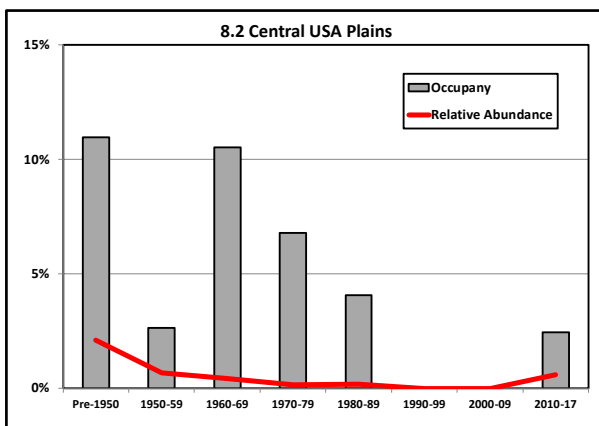
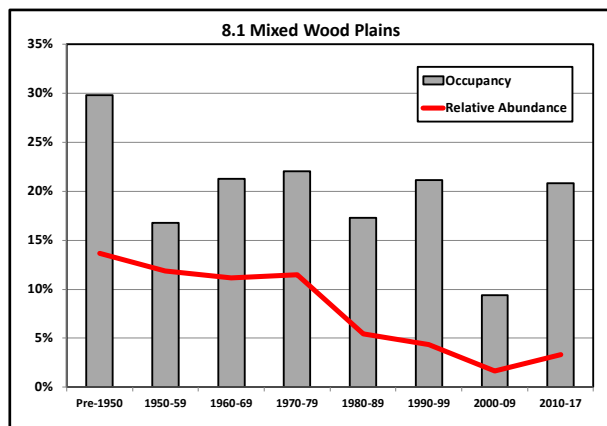
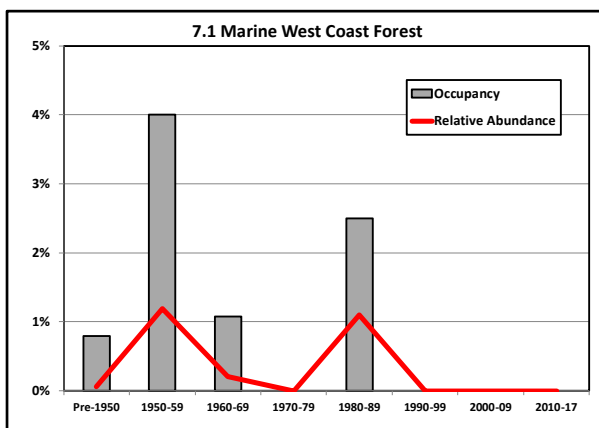
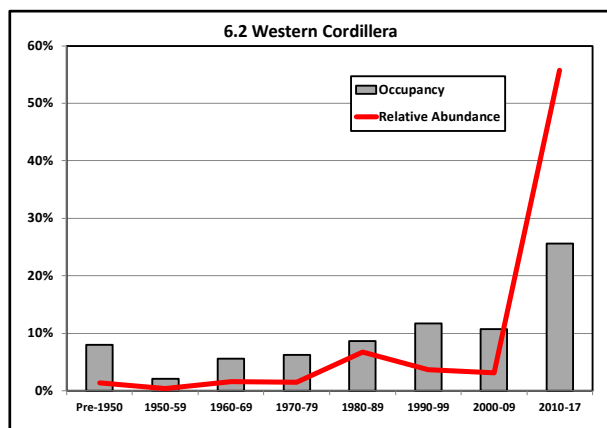
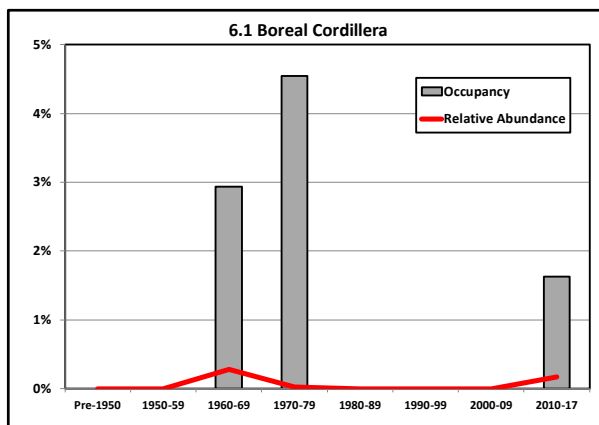
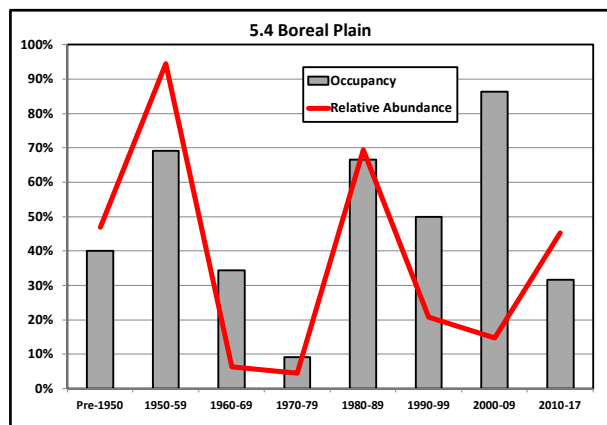
- 1 Yellow banded bumble bee decadal relative abundance by U.S. state or Canadian province. “YBBB” is the number of individual
- 2 YBBB specimens collected, “Total” is the number of all individual *Bombus* spp. specimens collected, and “% YBBB” is the proportion
- 3 of the total number of *Bombus* spp. specimens collected that were YBBB.

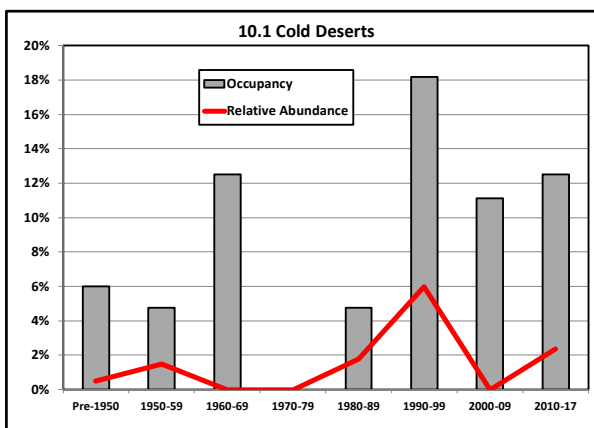
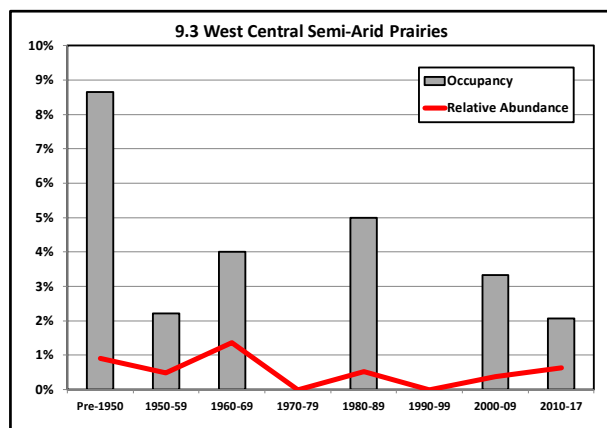
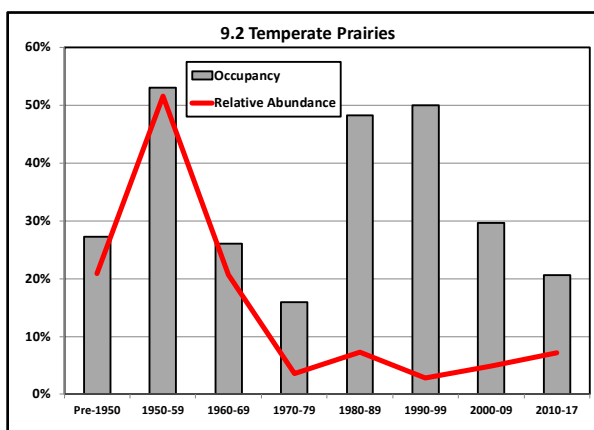
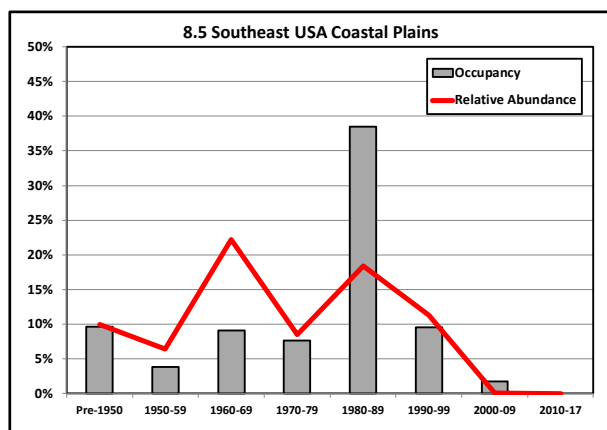
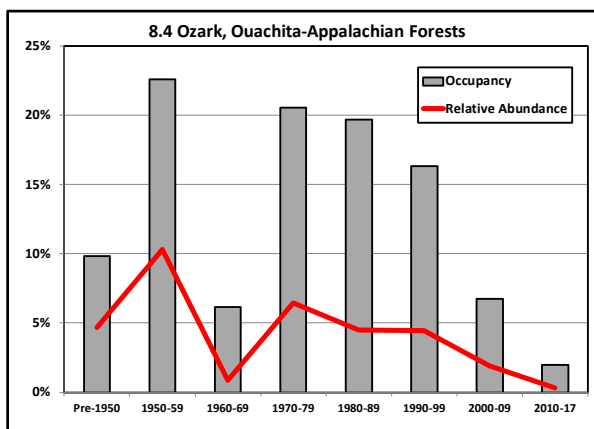
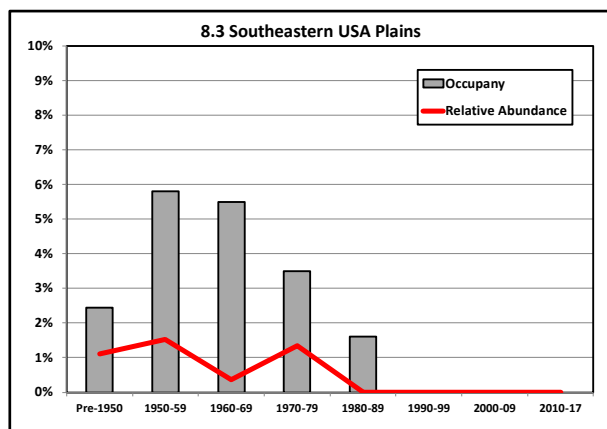
	Pre-1950			1950-59			1960-69			1970-79			1980-89			1990-99			2000-09			2010-17		
	YBBB	Total	%YBBB	YBBB	Total	%YBBB	YBBB	Total	%YBBB	YBBB	Total	%YBBB	YBBB	Total	%YBBB	YBBB	Total	%YBBB	YBBB	Total	%YBBB	YBBB	Total	%YBBB
AB	101	991	10%	24	240	10%	12	223	5%	1	188	1%	1	50	2%	0	44	0%	283	2,094	14%	47	311	15%
BC	16	1,987	1%	5	380	1%	8	613	1%	20	197	10%	9	275	3%	5	96	5%	11	4,510	0.2%	199	656	30%
CT	97	905	11%	3	121	2%	3	291	1%	23	780	3%	47	672	7%	1	26	4%	1	1,550	0.1%	0	658	0%
GA	1	74	1%	0	19	0%	0	14	0%	0	4	0%	0	25	0%	0	4	0%	0	1	0%	0	3	0%
IL	15	1,160	1%	1	114	1%	2	1,180	0.2%	0	1,187	0%	0	292	0%	0	82	0%	0	699	0%	0	227	0%
IN	0	52	0%	2	178	1%	0	58	0%	0	8	0%	0	1	0%	0	1	0%	0	80	0%	0	29	0%
MA	115	915	13%	9	57	16%	12	163	7%	64	732	9%	368	1,375	27%	20	126	16%	15	2,371	1%	2	1,313	0.2%
MB	140	304	46%	222	271	82%	21	280	8%	5	16	31%	95	132	72%	10	14	71%	15	459	3%	14	48	29%
MD	0	172	0%	0	14	0%	0	182	0%	5	227	2%	1	38	3%	0	5	0%	0	1,095	0%	2	2,090	0.1%
ME	155	501	31%	18	41	44%	26	104	25%	152	214	71%	17	45	38%	6	35	17%	13	129	10%	635	9,535	7%
MI	852	1,920	44%	345	712	48%	436	1,778	25%	57	332	17%	527	968	54%	68	153	44%	0	30	0%	14	268	5%
MN	504	1,255	40%	200	507	39%	371	1,151	32%	42	522	8%	72	882	8%	143	3,811	4%	17	151	11%	43	1,283	3%
NB,NS,PE	89	255	35%	12	21	57%	29	105	28%	146	364	40%	52	174	30%	253	1,269	20%	76	1,665	5%	84	915	9%
NC	40	709	6%	22	371	6%	3	281	1%	16	176	9%	5	101	5%	2	49	4%	29	286	10%	0	92	0%
ND	144	906	16%	1	7	14%	0	84	0%	5	167	3%	8	313	3%	20	453	4%	12	263	5%	1	117	1%
NH	476	2,005	24%	20	32	63%	157	502	31%	117	300	39%	77	188	41%	3	32	9%	11	54	20%	97	264	37%
NJ	9	162	6%	0	21	0%	1	34	3%	3	48	6%	0	13	0%	0	21	0%	0	193	0%	0	157	0%
NL	27	79	34%	0	2	0%	0	7	0%	19	32	59%	0	0		42	104	40%	9	32	28%	97	2,046	5%
NT	10	310	3%	0	2	0%	1	75	1%	6	15	40%	0	0		0	0		22	30	73%	13	33	39%
NY	1,083	4,441	24%	13	341	4%	328	1,014	32%	158	540	29%	158	480	33%	16	244	7%	5	2,985	0.2%	41	938	4%
OH	2	1,243	0.2%	0	113	0%	4	647	1%	0	1,431	0%	1	229	0.4%	0	102	0%	0	15	0%	0	198	0%
ON	174	1,439	12%	77	755	10%	746	1,968	38%	713	1,876	38%	595	1,390	43%	806	1,912	42%	37	3,764	1%	146	5,354	3%
PA	23	393	6%	2	40	5%	29	257	11%	18	76	24%	10	116	9%	0	45	0%	5	1,089	0.5%	7	2,532	0.3%
QC	26	591	4%	43	59	73%	20	96	21%	6	31	19%	18	85	21%	8	14	57%	28	434	6%	94	2,711	3%
RI	0	20	0%	0	6	0%	0	13	0%	1	6	17%	0	1	0%	0	4	0%	0	8	0%	0	12	0%
SD	3	585	1%	3	98	3%	1	259	0.4%	0	148	0%	4	16	25%	0	1	0%	1	148	1%	1	873	0.1%
SK	14	143	10%	5	19	26%	19	35	54%	0	27	0%	12	17	71%	0	2	0%	8	20	40%	21	92	23%
TN	15	126	12%	5	19	26%	0	6	0%	1	9	11%	6	33	18%	4	8	50%	7	52	13%	0	19	0%
VA	0	58	0%	0	20	0%	0	63	0%	0	24	0%	0	11	0%	1	44	2%	0	126	0%	0	43	0%
VT	66	222	30%	8	20	40%	85	503	17%	76	650	12%	48	389	12%	40	367	11%	274	1,503	18%	246	15,052	2%
WA	1	903	0.1%	0	189	0%	0	236	0%	0	563	0%	0	132	0%	0	58	0%	0	1,270	0%	0	866	0%
WI	102	480	21%	70	122	57%	140	2,332	6%	157	586	27%	12	122	10%	7	236	3%	5	130	4%	44	334	13%
WV	0	100	0%	1	21	5%	4	69	6%	2	194	1%	0	63	0%	0	51	0%	0	1,303	0%	0	637	0%
YT	0	197	0%	0	6	0%	0	121	0%	0	4,218	0%	0	36	0%	0	0		0	30	0%	2	1,089	0.2%
Totals:	4,300	25,603	17%	1,111	4,938	22%	2,458	14,744	17%	1,813	15,888	11%	2,143	8,664	25%	1,455	9,413	15%	884	28,569	3%	1,850	50,795	4%

Appendix B: Trends in YBBB occupancy and relative abundance by level II ecoregion

Occupancy is the proportion of surveyed hexagons ((100 sq km)(38.6 sq mi)) with at least one YBBB record. Relative abundance is the proportion of the total number of *Bombus* spp. specimens collected that were YBBB.







Appendix C: Modeling Yellow Banded Bumble Bee (YBBB) Count Data

Purpose

We modeled the trends in counts of YBBB from the 1950s to the present, and then used those models to forecast future counts over the next 3 decades. However, only 2 decades were able to be modelled with sufficient certainty for inclusion in the YBBB's future scenario.

Data

The data are counts of collected YBBB and other *Bombus* species, which have been summarized at the decade scale. Spatially, the data are summarized within large analytical units (AUs), which represent Level-II eco-regions. There are two types of data: 1) counts of occupied hexagons and 2) counts of individual specimens. Because collection effort varied among years, counts of YBBB were analyzed relative to counts of all *Bombus* species.

Models

We used generalized linear mixed-effects regression (glmer) to estimate the trend in YBBB counts at two spatial scales. The regression model estimates an overall trend based on all the data across the YBBB range and then also estimates trends for each AU. We based out analyses on the following glmer equation and the glmer model looks like this (Bates *et al.* 2015)

$$y_{ij} = (\beta_0 + b_{0i}) + (\beta_1 + b_{1i}) \times \text{Decade}_{ij} + \epsilon_{ij}$$

where y_{ij} is the response variable (count in this case), β_0 and β_1 are the intercept and slope for the overall trend, b_{0i} and b_{1i} are the deviations from the overall trend for the intercept and slopes for the i^{th} AU, Decade is an integer representing the 10-year period with the time series, and ϵ_{ij} is random error.

There are some options for the form of the regression model depending on assumed probability distribution for the response variable, which will determine what is called the link function for the model. Recall that the counts of YBBB are conditional on counts of all *Bombus* species either as occupied hexagons or as individual specimens. The data can be thought of as observations of the event that a *Bombus* occupied hexagon or individual specimen is also a YBBB. In that case, we would be interested in modeling the probability of that event, and logistic regression with a “logit” link function would be a common approach. In the logistic regression case, the above equation would be

$$\log\left(\frac{\pi_{ij}}{1 - \pi_{ij}}\right) = (\beta_0 + b_{0i}) + (\beta_1 + b_{1i}) \times \text{Decade}_{ij} + \epsilon_{ij}$$

where π is the probability of the event or as observed, it is $\pi = Y/T$, the proportion of *Bombus* counts that are YBB. For logistic regression, the predictions are probabilities that a *Bombus* observation is YBBB regardless of how many *Bombus* observations there are. Because data are counts, the results might be more interpretable if the predictions are also counts. The predicted probabilities from logistic regression can be multiplied by the count of all *Bombus* to calculate predicted counts of YBBB.

An alternative approach is to use poisson regression with the “log” link function and an offset term, $\log(T)$, to acknowledge that YBBB count cannot exceed the *Bombus* count. Either logistic or poisson regression will work for the YBBB data at the hexagon or individual specimen levels. We fit both and the results in terms of fit and inference were consistent; so, we present only the logistic regression results. Logistic regression was used by Colla *et al.* (2012) in their assessment of *Bombus* declines.

The models were used to forecast or project into future decades. Heuristically, it seemed reasonable to project 3 decades; however, depending on the AU, the uncertainty in the projection became quite large beyond 2 decades. To standardize collection effort, we calculated the median *Bombus* counts within each AU and applied the median to compute retrospective counts of YBBB over the past 8 decades and prospective counts of YBBB over the next 3 decades.

Software packages lme4 within R (Bates *et al.* 2015, R Core Team 2018) were used to fit the models and to conduct diagnostic assessment of the goodness-of-fit. Prediction uncertainty was estimated using a boot-strapping procedure available in the merTools package (Knowles and Frederick 2016).

Results

Examination of the distribution of residuals, their relationship to fitted values, and the sensitivity of model results to high leverage observation indicate that the logistic and poisson versions of the glmer provided good fits to the data, and the model results were qualitatively consistent. Code and extensive results are two R markdown files “YBBB.glmer.Individual.html” and “YBBB.glmer.hexagon.html”. Here we summarize results from the logistic regression model.

Individual Specimens

The population-averaged trend (i.e., β_1 from above equations) indicated an overall decline in individual specimens ($\beta_1 = -0.12$, $SD = 0.092$). At the AU level, the trends varied, but 11 out of the 18 AUs had negative slopes (Table 1). The AUs with the highest individual specimens had negative slopes (Figure 1).

The counts of YBBB individual specimens standardized by the median count of *Bombus* declined range-wide and is expected to continue to decline (Figure 2).

Occupied Hexagons

The population-averaged trend indicated an overall decline in occupied hexagons ($\beta_1 = -0.06$, $SD = 0.050$). At the AU level, the trends varied, but 11 out of the 18 AUs had negative slopes (Table 2). The AUs with the highest YBBB-occupied hexagons had negative slopes (Figure 3). The counts of YBBB-occupied hexagons standardized by the median count of *Bombus*-occupied hexagons declined range-wide and is expected to continue to decline (Figure 4).

Comparison of Individual and Hexagon Counts

The patterns in the counts of individuals or occupied-hexagons were qualitatively consistent in direction and magnitude. However, the significance of the declines was stronger for individual-specimen counts.

Predictions

Predicted counts with 90 percent confidence intervals for the next 3 decades for each AU are in “YBB.lmer3binpred.hex.csv”.

References

Bates D, Maechler M, Bolker B, Walker S. 2015. Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software*, 67(1), 1-48. <doi:10.18637/jss.v067.i01>.

Colla SR, Gadallah F, Richardson L, Wagner D, Gall L. 2012. Assessing declines of North American bumble bees (*Bombus* spp.) using museum specimens. *Biodiversity and Conservation* 21:3585-3595.

Knowles JE, Frederick C. 2016. merTools: Tools for analyzing mixed effect regression models [R package version 0.2. 1] <https://CRAN.R-project.org/package=merTools>.

R Core Team. 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>

Table 1. Coefficients for each analytical unit (ecoregion) from logistic generalized linear mixed model regression based on counts of individual specimens relative to counts of all *Bombus* specimens over 8 decades. Estimates are means, medians, and standard deviations (SD) based on 1000 bootstrapped samples.

Analytical unit label	Mean	Median	SD
AU10.1	0.2862	0.2892	0.1016
AU3.3	0.5250	0.5257	0.0835
AU3.4	0.5635	0.5607	0.0872
AU4.1	0.1714	0.1708	0.0823
AU5.1	-0.2157	-0.2165	0.0740
AU5.2	-0.0810	-0.0823	0.0735
AU5.3	-0.2658	-0.2681	0.0735
AU5.4	-0.1274	-0.1280	0.0739
AU6.1	0.2440	0.2468	0.1348
AU6.2	0.8366	0.8372	0.0757
AU7.1	-0.2086	-0.2032	0.1074
AU8.1	-0.1371	-0.1384	0.0736
AU8.2	-0.4690	-0.4693	0.1031
AU8.3	-0.4159	-0.4156	0.1012
AU8.4	-0.1014	-0.1036	0.0759
AU8.5	-0.4073	-0.4082	0.0785
AU9.2	-0.2415	-0.2424	0.0762
AU9.3	0.0553	0.0520	0.0795

Table 2. Coefficients for each analytical unit from logistic generalized linear mixed model regression based on counts of occupied hexagons relative to counts of all *Bombus* specimens over 8 decades. Estimates are means, medians, and standard deviations (SD) based on 1000 bootstrapped samples.

Analytical unit label	Mean	Median	SD
AU10.1	0.1361	0.1341	0.0829
AU3.3	0.2896	0.2913	0.0909
AU3.4	0.3043	0.3041	0.1017
AU4.1	-0.0125	-0.0122	0.0947
AU5.1	0.0367	0.0365	0.0610
AU5.2	-0.2082	-0.2087	0.0484
AU5.3	-0.0337	-0.0343	0.0475
AU5.4	0.1029	0.1021	0.0602
AU6.1	-0.0320	-0.0293	0.1094
AU6.2	0.2651	0.2691	0.0649
AU7.1	-0.1331	-0.1334	0.1234
AU8.1	-0.0064	-0.0076	0.0465
AU8.2	-0.1914	-0.1899	0.0762
AU8.3	-0.2148	-0.2103	0.0895
AU8.4	-0.0573	-0.0551	0.0572
AU8.5	-0.1032	-0.1052	0.0785
AU9.2	0.0199	0.0194	0.0541
AU9.3	-0.1136	-0.1124	0.0741

Figure 1. Analytical unit level trend in individual specimens related to median YBBB count.

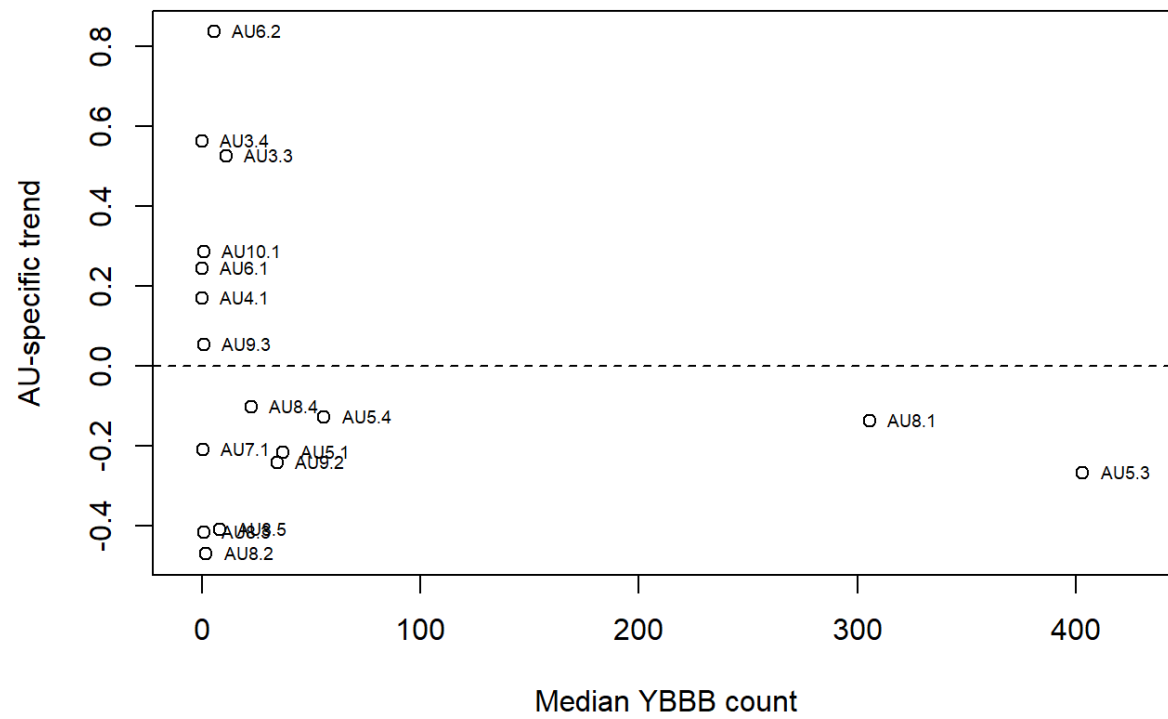


Figure 2. Predicted counts of individual YBBB standardized by the median *Bombus* count = 13170 over the eight decades that data were collected. The solid line shows the median predicted count and the dashed lines show the 90% confidence interval.

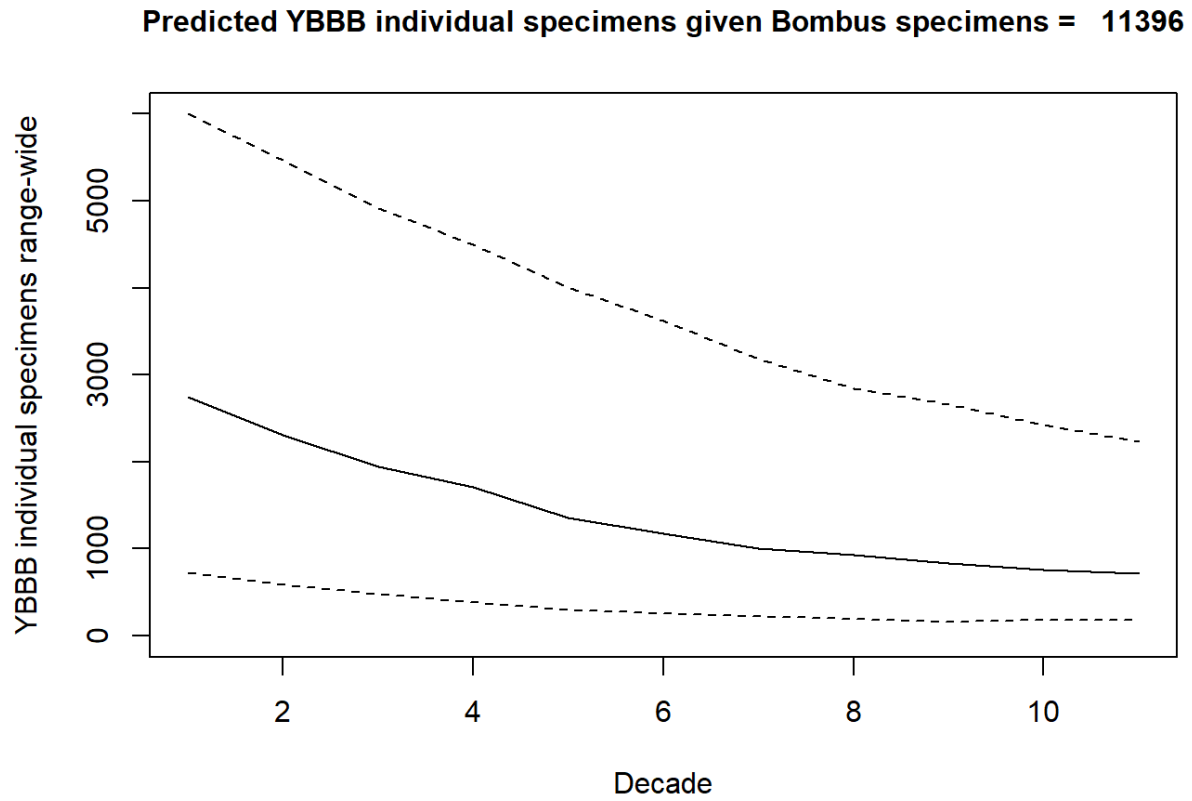


Figure 3. Analytical unit level trends in YBBB-occupied hexagons related to median YBBB count.

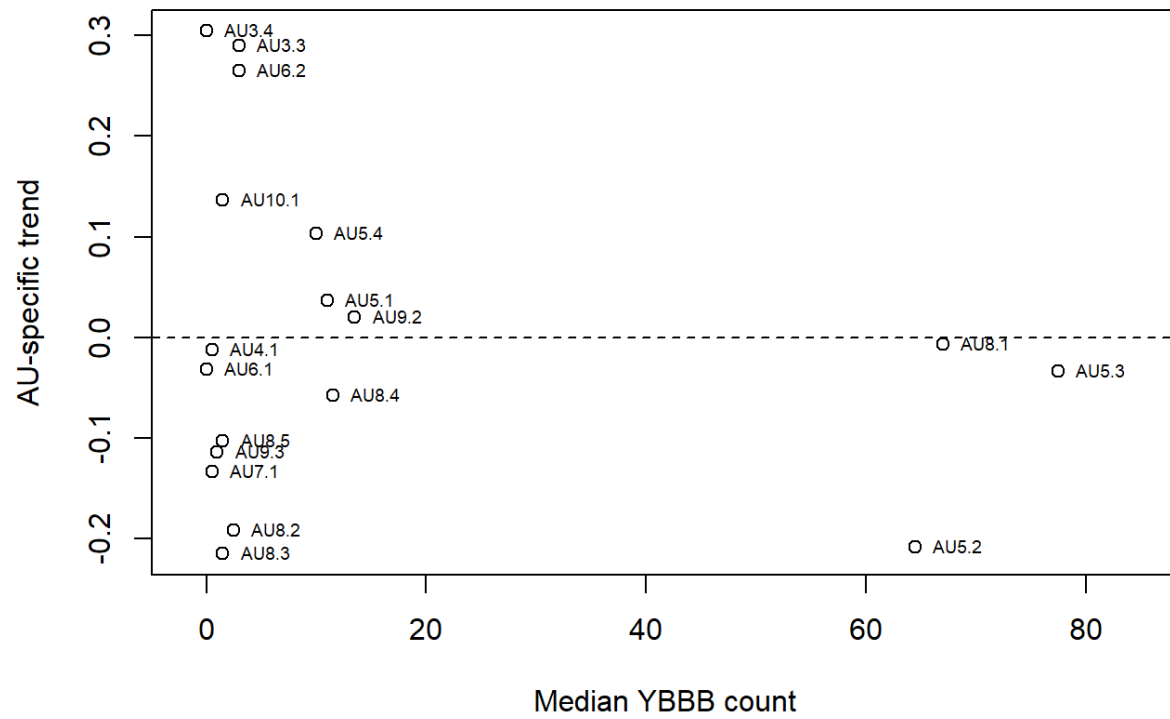


Figure 4. Predicted counts of YBBB-occupied hexagons standardized by the median number of *Bombus*-occupied hexagons = 1405 over the eight decades that data were collected. The solid line shows the median predicted count and the dashed lines show the 90% confidence interval.

