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CONSERVATION GENETICS OF A DECLINING BUMBLE BEE IN WESTERN

NORTH AMERICA; THE INFLUENCE OF GEOGRAPHY, DISPERSAL

LIMITATION, AND ANTHOPOGENIC ACTIVITY

by

Ashley T. Rohde

A dissertation submitted in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in

Ecology

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> > 2022

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ABSTRACT

Conservation Genetics of a Declining Bumble Bee in Western North America; the Influence of Geography, Dispersal Limitation, and Anthropogenic Activity

by

Ashley T. Rohde, Doctor of philosophy

Utah State University, 2022

Major Professors: Dr. Karen Mock Department: Wildland Resources

Conservation biology addresses the problem of biological species loss and decline by identifying species in need of protection or recovery. Conservation biology has subfields to better address aspects of biodiversity loss, including conservation genetics, phylogenomics, and sociology. In this dissertation, I used genetic and phylogenomic approaches to assess the conservation status of a bumble bee species of concern, *Bombus occidentalis*, and a sociological approach to measure conservationists' interest in genetics methods for conservation studies.

Bombus occidentalis is a widespread North American bumble bee species that is decreasing in abundance in portions of its range. It is currently under consideration for listing under the Endangered Species Act in the United States and is listed as endangered in parts of its Canadian distribution through the Species At Risk Act. To complicate the problem further, there is debate about whether *Bombus occidentalis* is one species or two. Recent genetic analyses of the mitochondrial *cytochrome oxidase I* (COI) gene indicate

that the group may consist of a northern species (*B. mckayi*) and a southern species (*B. occidentalis*).

I used nuclear (ultraconserved elements, UCE) and mitochondrial (COI) phylogenomic methods to infer maximum likelihood (ML) and Bayesian (BI) phylogenies of the relationship between the two taxa. I used seven species delimitation methods to conduct the most thorough test of the species status of these taxa yet performed. The species delimitation analyses sometimes contradicted one another, but *B. occidentalis mckayi* was consistently recovered as a monophyletic group in both UCE and COI phylogenetic analyses. This analysis provided sufficient evidence to elevate *B. occidentalis mckayi* to the level of species.

I used landscape genetic methods to measure patterns of genetic diversity and structure in *B. occidentalis* and *B. mckayi* from 1960 through 2020, and tested associations with potential environmental drivers of genetic diversity across the landscape. *B. occidentalis* showed patterns of decreasing genetic diversity and increasing genetic structure, but *B. mckayi* did not. The genetic diversity in both species were most strongly influenced by springtime minimum temperatures and proximity to known infections of the fungal parasite *Vairimorpha bombi.*

Finally, I surveyed 974 conservationists from diverse backgrounds to measure their level of understanding, trust, and motivation from conservation genetic studies. The results indicate that lack of understanding, but not trust, may inhibit increased use of molecular methods in conservation.

(353 pages)

PUBLIC ABSTRACT

Conservation Genetics of a Declining Bumble Bee in Western North America; the Influence of Geography, Dispersal Limitation, and Anthropogenic Activity

Ashley T. Rohde

Conservation biology addresses the problem of species loss by identifying species in need of protection. Conservation biology has subfields to address different aspects of biodiversity loss, including genetics and sociology. I used genetic approaches to assess the conservation status of western bumble bees, a bumble bee species of conservation concern.

The western bumble bee is a bumble bee species that ranges from Alaska to New Mexico and as far east as Wyoming and Colorado. This species is disappearing in some places. It may soon be listed as endangered in the United States and is already listed as endangered in parts of its Canadian distribution. To complicate the problem further, the western bumble bee might really be two cryptic species. Recent genetic analyses indicate that there might be a northern species (Mckay's bumble bee) and a southern species (the western bumble bee).

I used DNA from specimens collected across the range and ran genetic analyses to estimate the relationships between western bumble bees and Mckay's bumble bees. This study provided enough evidence to conclude that they are, in fact, two species.

Next, I compared patterns of genetic diversity in the two species to environmental variables to determine how the environment influences how the bees to move across the landscape. I compared patterns of genetic diversity in bees that were collected between

1960 through 2020. Western bumble bees showed patterns of slightly decreasing genetic diversity through time from 1960 to 2019, but Mckay's bumble bee did not. For both species, nighttime temperatures during the spring and proximity to a native fungal parasite were important predictors of differences in genetic diversity among samples. The distance from parasites is probably important because specimens that are near infections are more likely to be infected themselves. Although we found decreases in genetic diversity for western bumble bees, there is still enough genetic diversity in present-day populations for the species to recover if the effects of the drivers of the declines are managed.

Finally, I surveyed 974 conservationists from diverse backgrounds to measure their understanding, trust, and motivation to action from conservation genetic studies. This is important because molecular methods provide important insight into the conservation status of at-risk species, but they are not used very often when land managers make conservation decisions. The results indicate that lack of understanding, but not trust, may be a barrier to increased use of molecular methods in conservation actions.

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CHAPTER I

INTRODUCTION

Biological species loss and decline have been documented in many taxonomic groups around the world and rates of extinction continue to increase in most groups (Spooner et al. 2018, Falaschi et al. 2019, Fisher and Garner 2020, Noske and Briggs 2020, Zattara and Aizen 2020, Bali and Kaleka 2021). Conservation biology addresses this problem by identifying species in need of protection or recovery. Conservation biology is a relatively new field that was formally developed in the mid 1980s, combining resources from many previously established fields to address the apparent world-wide loss of biodiversity in a systematic way (Meine et al. 2006). Of course, the imperative to protect biodiversity is much older than that. In 1863 Alfred Russel Wallace warned that if species were not protected, future generations would "charge us with having culpably allowed the destruction of some of those records of Creation which we had it in our power to preserve; and while professing to regard every living thing… with a strange inconsistency, seeing many of them perish irrecoverably from the face of the earth, uncared for and unknown." (Wallace 1863).

Since its formal conception, conservation biology has developed subfields by incorporating innovative methods and applications to improve the protection of biodiversity. Notably, during the nascent years of conservation biology, genetic methods were developed concurrently with traditional ecological approaches. In particular, the invention of polymerase chain reaction in 1983 allowed for quick and relatively inexpensive amplification of DNA samples (Mullis 1990), initiating an explosion in methods development to quantify evolutionary relationships among species and

population dynamics within species. The co-occurring developments of molecular genetics and conservation biology led to the formation of conservation genetics, a field defined by the application of "genetic principles and methods to advance the preservation of biodiversity" (Kardos 2021) which continues to develop with the increasing use of genomic methods (Allendorf et al. 2010, Véron et al. 2019). Conservation biology also benefited from recognition of the roles of sociology and psychology in conservation actions (Machlis 1992, Saunders 2003, Dunlap 2018). The inherently political nature of conservation decisions to protect biodiversity necessitates wide societal consent. Support for biological conservation is increased by overcoming barriers to understanding and motivation to action among stakeholders (Mascia et al. 2003, Schultz 2007).

Bees (Hymenoptera: Apoidea: Anthophila) pollinate more plant species than any other taxa of pollinators (Ollerton 2017), including approximately 75% of the world's food crops (Klein et al. 2007). There are nearly 20,000 known bee species worldwide (Michener 2000), and more yet to be described, especially in the Neotropics of Central and South America (Freitas et al. 2009) and in parts of Asia (Teichroew et al. 2017). However, bees worldwide are decreasing in abundance and range (Goulson et al. 2015). Bumble bees (*Bombus*) are among the most studied bee genera, largely due to their use as pollinators in agriculture, their relatively large size and characteristic appearance, and their high abundance throughout their distribution. Bumble bees are often the dominant pollinators in cold climate regions, especially early in the active season when nighttime temperatures are relatively low (Goulson 2003). Decreases in abundance and range have been observed in bumble bee species around the world (Goulson et al. 2008, Colla et al. 2012, Cameron and Sadd 2020, Graves et al. 2020).

There are approximately 260 described bumble bee species worldwide, one third of which are under threat of extinction to some extent (Abertman et al. 2017). Bumble bee species in North America are decreasing in abundance and genetic diversity at alarming rates (Cameron et al. 2011, Colla et al. 2012, Abertman et al. 2017). Within the United States several species have been petitioned for listing as endangered through the Endangered Species Act (ESA). In 2016, *Bombus affinis* Cresson 1863 was listed as endangered. This was the first bumble bee species to be listed as endangered in the United States. *Bombus franklini* Frison 1921 was most recently listed as endangered by the ESA in September of 2021 and may already be extinct

(https://www.federalregister.gov, accessed 11:05 a.m., 10/11/2019). Finally, *Bombus occidentalis* Greene 1858 is currently under review for listing as endangered through the ESA (https://ecos.fws.gov/ecp/, accessed 1:39 p.m., 8/13/2018), with a listing decision expected in 2023. This species is also listed as threatened or endangered by the Species At Risk Act (SARA) in portions of its Canadian distribution. Several additional species have been identified as endangered or at risk of decline by SARA and the International Union for Conservation of Nature and Natural Resources (IUCN, www.iucnredlist.org, accessed 11:36 a.m., 8/13/18).

Species in the bumblebee subgenus *Bombus sensu stricto* (s.s.) are economically important for crop pollination and are dominant native pollinators in many ecosystems across the Holarctic region of the world (Goulson 2003, Hines 2008). There are up to 23 identified species in the subgenus worldwide, with as many as eight species native to North America (Williams 2021). However, population distributions and abundances of many *Bombus s.s.* species are decreasing. These species include *B. affinis, B. occidentalis* and *B. franklini* (Cameron et al. 2011, Colla et al. 2012, Abertman et al. 2017). *Bombus occidentalis*, in particular, is the focus of renewed interest among wildlife managers and conservationists due to unresolved taxonomic questions that may influence the upcoming ESA listing decision in the United States. Understanding the genetics and conservation status of *B. occidentalis* are the focus of chapters 2 and 3 of my dissertation.

Bombus occidentalis has a large geographical range. It is found throughout western North America from Alaska to New Mexico and as far east as Wyoming and Colorado, with distributions restricted to high-elevation sites in the southern portion of the range. Regional morphological variation in pyle color historically led to several proposed delimitations of species and subspecies status within *B. occidentalis*, with some taxonomists suggesting that it is conspecific with *B. terricola* Kirby 1837 (Milliron 1971, Poole 1996) and others defining several subspecies (see Sheffiend et al. 2016 for a thorough review). The most recent, and most widely accepted, delimitation indicates two taxa (*B. occidentalis occidentalis* and *B. occidentalis mckayi*), broadly based on the presence or absence of a yellow band of hairs on the abdominal terga. The "un-banded" group extends from the southern edge of the species' range to approximately 55 degrees latitude (though specimens in some parts of this range do have a weak band), and the "banded" group extends from 55 degrees latitude to the northern edge of the range. Whether these taxa represent species or subspecies is a debated topic (Williams et al. 2012, Sheffield et al. 2016, Williams 2021). Identification of the morphotypes within the taxon is challenging, because their definitions are mostly based on pyle color, which is variable among and within geographical regions of the species range (Carolan et al. 2012, Sheffield et al. 2016) and specimens often exhibit intermediate characteristics among the

morphotypes. Geographic collection locations are often used to help define subspecies (Sheffield et al. 2016).

In this dissertation, I integrated methods from across conservation genetics and sociology to contribute to a growing body of literature on the conservation status of bumble bees in North America (Cameron et al. 2011, Colla et al. 2012, Abertman et al. 2017). I focused my genetic research on a widespread but imperiled species native to western North America, *Bombus occidentalis*. In chapter 2 I used phylogenomic methods to clarify the species status and distribution. In chapter 3 I used landscape genetic methods to identify the environmental drivers of diversity loss among populations of the species throughout its geographic range. Finally, in chapter 4 I assessed conservation practitioners' level of understanding and motivation to act on conservation issues in response to the results of conservation genetics studies and conservation biology studies based on more intuitive measurements, such as abundance or fecundity.

In chapter 2, I aimed to resolve the species status of *B. occidentalis* using an integrative approach that combines morphological identification with phylogenetic analysis of nuclear and mitochondrial markers, and automated species delimitation methods*.* This was the first study to use nuclear markers to address this question. I greatly expanded geographical sampling and used more species delimitation methods than any previous analysis of these taxa. I concluded that these two taxa represent true species and, therefore, recommend elevation of *B. occidentalis mckayi* to a species (from here forward referred to as *B. mckayi*). This finding is in agreement with the findings of Williams (2021).

In chapter 3, I used landscape genetic methods to … Landscape genetics is an interdisciplinary field that combines aspects of population genetics, landscape ecology, and spatial statistics to measure genetic discontinuities and diversity patterns across landscapes and to correlate them with environmental features (Manel et al. 2003, Storfer et al. 2007). Landscape genetic techniques can provide insight into questions about potential threats to bumble bees and identify actions that can be taken to protect populations. Measurements of population structure, genetic diversity, and gene flow among populations are important indicators of current conservation status for species. Landscape genetic studies that measure environmental variables as well as gene flow, genetic structure and diversity can indicate which environmental changes have negative effects on bumble bees.

Genetic data and occupancy data are complementary tools for assessing the conservation status of *B. occidentalis* and *B. mckayi*. While patterns of occupancy may indicate where gene flow barriers exist, these patterns are insufficient to predict the causes of gene flow barriers (Roffler et al. 2016). The relationship between occupancy and gene flow could be particularly messy for bumble bees because of their eusocial life history, which dictates that most individuals in the census populations are not reproductive, so they do not contribute directly to gene movement across the landscape.The higher likelihood of observing sterile workers skews occupancy models to identify sites that are adequate or inadequate for colony establishment, rather than gene flow. Therefore, measures of occupancy alone are not enough to determine if gene flow is restricted.

I used microsatellite genetic data from museum specimens to predict current and past genetic structure (samples were collected between 1960 and 2020), genetic diversity, and gene flow patterns in *B. occidentalis* and *B. mckayi*. I used observation data and spatial environmental predictors to predict the influence of environmental variables on occupancy in the two species and to make associations between potential environmental barriers to gene flow and genetic isolation. This is the first landscape genetic study to measure the influence of environmental predictors on occupancy likelihood of *B. mckayi* separately from *B. occidentalis*. This is the first study to identify environmental predictors to gene flow patterns in either species. I detected clear patterns of decreasing genetic diversity and increasing genetic structure in *B. occidentalis*. Patterns of decline were not as strong in *B. mckayi*, but indicate that this species may also be at risk. Springtime minimum temperatures were the most important predictors of occupancy for both species. Proximity to known infections of the fungal parasite *Vairimorpha bombi* was a reliable predictor of genetic differentiation (restricted gene flow). Although decreases in allelic diversity and increases in inbreeding and population structure have been documented in these species, substantial genetic diversity remains in extant populations, which indicates a good opportunity for recovery of the species if the effects of the drivers of the declines are mitigated.

Lastly, in chapter 4 I focus on conservation sociology and use survey methods to determine how different stakeholders feel about molecular methods and results in conservation. Molecular techniques are being used increasingly commonly and to great effect in conservation studies (Abdul-Muneer 2014, Kress 2015, Shafer et al. 2015, Thomsen and Willerslev 2015, Corlett 2017, Holdregger et al. 2019, chapter 1 and 2).

However, these studies are not intuitively easy to understand for practitioners and stakeholders who are not specifically trained to interpret their results. As such, many conservation partners are left out of conversations about these types of studies and the appropriate conservation actions that their results indicate (Keller et al. 2015, Taylor et al. 2017, Sandstrӧm et al. 2019, Klütsch and Laikre 2021). A lack of detailed understanding of the results of genetic conservation studies may lead to a sense of helplessness that undermines motivation for action in some groups. The results of traditional studies more easily overcome the barriers to conservation action than the results of molecular studies (Hoban et al. 2013a, Keller et al. 2015, Shafer et al. 2015, Hoffman et al. 2015, Richardson et al. 2016, Taylor et al. 2017). As a result, the insights provided by genetic studies into distribution and population structure of the target species are not often used to inform conservation decisions (Keller et al. 2015, Shafer et al. 2015, Hoffman et al. 2015, Taylor et al. 2017). This phenomenon is widely known as the conservation genetics gap (Taylor et al. 2017, Britt et al. 2018, Sandström et al. 2019, Klütsch and Laikre 2021). The conservation genetics gap is widely acknowledged and discussed (Cook et al. 2013, Hoban et al. 2013a, Hoban et al. 2013b, McMahon et al. 2014, Hoffman et al. 2015, Keller et al. 2015, Shafer et al. 2015, Haig 2016, Taylor et al. 2017, Aurelle et al. 2018, Britt et al. 2018, Funk et al 2019, Mazel et al. 2019, Sandstrӧm et al. 2019, Klütsch and Laikre 2021), but few studies have directly measured the differences in perception of conservationists between genetic and traditional types of conservation surveys (However see Taylor et al. 2017 and Sandström et al. 2019).

I used a survey to measure the relative understanding, trust, and motivation to action of conservationists from multiple demographics in response to the results of

molecular and traditional conservation studies. I received responses from 974 conservationists from diverse backgrounds. This is the largest and most diverse sample of conservationists ever surveyed to assess attitudes toward conservation genetics. The results indicate that lack of understanding, but not trust, may be a barrier to increased use of molecular methods in conservation actions. However, comparisons of the data presented here to previous studies (Taylor et al. 2017, Sandstrӧm et al. 2019) are hopeful that a shift in perception and increased use of molecular studies may be underway. Previous studies have indicated that increased and improved outreach events among conservation geneticists and other conservation practitioners help to improve understanding of conservation genetics studies among all demographics of conservationists. Also, inclusion of authors who are not genetics experts on publications that include genetics increases the likelihood that those studies will be used to support conservation policy or action, presumably because they ensure the publications contain language that is accessible to a broad audience (Britt et al. 2018).

Taken as a whole, the research presented in this dissertation contributes new information to scientists' growing understanding of the conservation needs of *B. occidentalis* and *B. mckayi* by applying novel methods to the question*.* I present the first analysis of species status using the nuclear genome of the species, as well as a more robust analysis of the mitochondrial COI barcoding gene than has ever been conducted, and a greater variety of automated speciation methods than has ever been applied. I conducted the first population genetics study of the taxa that treats the two species separately, and used novel methods to associate changes in genetic diversity to changes in environmental conditions. Although my analysis of conservation practitioners'

perceptions of conservation genetics does not directly address the problem of the conservation status of *B. occidentalis* and *B.mckayi*, it does inform the choices researchers, policy-makers and conservationists of any background should make to be most effective in communicating the conservation needs of these and any other species.

References

- Abdul-Muneer, P.M. (2014). Application of microsatellite markers in conservation genetics and fisheries management: recent advances in population structure analysis and conservation strategies. Genetics Research International. Article ID 691759, 11 pages.
- Abertman, M.P., Gleiser, G., Morales, C.L., Williams, P. and Aizen, M.A. (2017). Global decline of bumblebees is phylogenetically structured and inversely related to species range size and pathogen incidence. Proceedings of the Royal Society B 284: 20170204. http://dxdoi.org/10.1098/rspb.2017.0204.
- Allendorf, F.W., Hohenlohe, P.A., and Luikart, G. (2010). Genomics and the future of conservation genetics. NAture Reviews Genetics 11:697-709.
- Aurelle, D., Pratlong, M., Haguenauer, A., Brener-Raffalli, K., Toulza, E., Garrabou, J., Pontarotti, P., Linares, C., López-Sendino, P., Montero-Serra, I., Frias-Vidal, S., Ledoux, J.B. (2018). Bridging the gap between evolutionary and conservation biology: the case of a precious octocoral threatened by global change, the Mediterranean red coral. Second Joint Congress of Evolutionary Biology, 19-22 August 2018, Montpellier
- Bali, G.P.K., and Kaleka, A.S. (2021). Potential reasons for insect decline. In Global Decline of Insects. Ed El-Shafie, H. IntechOpen Limited, London, UK.
- Britt, M., Haworth, S.E., Johnson, J.B., Martchenko, D., and Shafer, A.B.A. (2018). The importance of non-academic coauthors in bridging the conservation genetics gap. Biological Conservation 218: 118-123
- Cameron, S.A., Lozier, J.D., Strange, J.P., Koch, J.B., Cordes, N., Solter, L.F., and Griswold, T.L. (2011). Patterns of widespread decline in North American bumble bees. Proceedings of the National Academy of Sciences of the United States of America 108(2):662-667.
- Carolan, J.C., Murray, T.E., Fitzpatrick, U., Crossley, J., Schmidt, H., Cederberg, B., McNally, L., Paxton, R.J., Williams, P.H., and Brown, M.J.F. (2012). Colour

patterns do not diagnose species: quantitative evaluation of a DNA barcoded cryptic bumblebee complex. PLoS ONE 7(1): e29251.

- Cameron, S.A., and Sadd, B.M. (2020). Global trends in bumble bee health. Annual Review of Entomology 65:209-232.
- Colla, S.R., Gadallah, F., Richardson, L., Wagner, D., and Gall, L. (2012). Assessing declines of North American bumble bees (*Bombus* spp.) using museum specimens. Biodiversity and Conservation 21(14):3585-3595.
- Cook, C.N., Mascia, M.B., Schwartz, M.W., Possingham, H.P., and Fuller, R.A. (2013). Achieving Conservation Science that bridges the knowledge-action boundary. Conservation Biology 27(4): 669-678.
- Corlett, R.T. (2017). A bigger toolbox: biotechnology in biodiversity conservation. Trends in Biotechnology 35(1): 55-65.
- Dunlap, R.E. (2018). Environmental sociology. In Companion to Environmental Studies. Eds: Castree, N., Hulme, M., Proctor, J.D. (2018). Routledge, London, UK.
- Falaschi, M., Manenti, R., Thuiller, W., and Ficetola, G.F. (2019). Continental-scale determinants of population trends in European amphibians and reptiles. Global Change Biology 25(10):3504-3515.
- Fisher, M.C., and Garner, T.W.J. (2020). Chytrid fungi and global amphibian declines. Nature Reviews Microbiology 18:332-343.
- Freitas, B.M., Imperatriz-Fonseca, V.L., Medina, L.M., de Matos Peixoto Kleinert, A., Galetto, L., Nates-Parra, G., and Quezada-Euán, J.J.G. (2009). Diversity, threats and conservation of native bees in the Neotropics. Apidologie 40:332-346. DOI:10.1051/apido/2009012.
- Funk, W.C., Forester, B.R., Converse, S.J., Darst, C., and Morey, S. (2019) Improving conservation policy with genomics: a guide to integrating adaptive potential into U.S. Endangered Species Act decisions for conservation practitioners and geneticists. Conservation Genetics 20: 115-134.
- Goulson, D. (2003). Bumblebees: their behavior and ecology. Oxford University Press, Oxford.
- Goulson, D., Lye, G., and Darvill, B. (2008). The decline and conservation of bumblebees. Annual Review of Entomology 53:191-208.
- Goulson, D., Nicholls, E., Botías, and Rotheray, E.L. (2015). Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. Science 347(6229):1255957.
- Graves, T.A., Janousek, W.M., Gaulke, S.A.m Nicholas, A.C., Keinath, D.A., Bell, C.M., Cannings, S., Hatfield, R.G., Heron, J.M., Koch, J.B., Loffland, H.L., Richardson, L.L., Rohde, A.T., Rykken, J., Strange, J.P., Tronstad, L.M., and Sheffield, C.S. (2020). Western bumble bee: declines in the continental United States and rangewide information gaps. Ecosphere 11(6): e03141.
- Haig, S.M., Miller, M.P., Bellinger, R., Draheim, H.M., Mercer, D.M., and Mullins, T.D. (2016). The conservation genetics juggling act: integrating genetics and ecology, science and policy. Evolutionary Applications 9(1): 181-195.
- Hines, H. (2008). Historical biogeography, divergence times, and diversification patterns of bumble bees (Hymenoptera: Apidae: *Bombus*). Systematic Biology 57(1):58- 75.
- Hoban, S.M., Arntzen, J.W., Bertorelle, G., Bryja, J., Fernandes, M., Frith, K., Gaggiotti, O., Galbusera, P., Godoy, J.A., Hauffe, H.C., Hoelzel, A.R., Nichols, R.A., Pérez-Espona, S., Primmer, C., Russo, I.-R.M., Segelbacher, G., Siegismund, H.R., Sihvonen, M., Sjögren-Gulve, Vernesi, C., Vilá, Carles, Bruford, M.W. (2013a). Conservation genetic resources for effective species survival (ConGRESS): bridging the divide between conservation research and practice. Journal for Nature Conservation 21:433-437.
- Hoban, S.M., Hauffe, H.C., Pérez-Espona, S., Arntzen, J.W., Bertorelle, G., Bryja, J., Frith, K., Gaggiotti, O.E., Galbusera, P., Godoy, J.A., Hoelzel, A.R., Nichols, R.A., Primmer, C.R., Russo, I.-R., Segelbacher, G. (2013b). Bringing genetic diversity to the forefront of conservation policy and management. Conservation Genetics Resources 5(2):593-598.
- Hoffman, A., Griffin, P., Dillon, S., Catullo, R., Rane, R., Byrne, M., Jordan, R., Oakeshott, J., Weeks, A., Joseph, L., Lockhart, P., Borevitz, J. and Sgró, C. (2015). A framework for incorporating evolutionary genomics into biodiversity conservation and management. Climate Change Responses 2:1.
- Holderegger, R., Balkenhol, N., Bollinger, J., Engler, J.O., Gugerli, F., Hochkirch, A., Nowak, C., Segelbacher, G., Widmer, A., and Zachos, F.E. (2019). Conservation genetics: linking science with practice. Molecular Ecology 28(17): 3848-3856.
- Kardos, M. (2021) Conservation genetics. Current Biology 31(19): R1185-R1190.
- Keller, D., Holderegger, R., van Strien, M.J., and Bolliger, J. (2015). How to make landscape genetics beneficial for conservation management. Conservation Genetics 16(3): 503-512.
- Klein, A.-M., Vaissiére, Cane, J.H., Steffan-Dewenter, S., Cunningham, S.A., Kremen, C., and Tscharntke, T. (2007). Importance of pollinators in changing landscapes for world crops. Proceedings of the Royal Society B 274(1608):303-313.
- Klütsch, C.F.C, and Laikre, L. (2021). Closing the conservation genetics gap: Integrating genetics knowledge in conservation management to ensure evolutionary potential. In: Ferreira, C.C., Klütsch, C.F.C. (eds) Closing the knowledge-implementation gap in conservation science. Wildlife Research Monographs, vol 4. Springer, Cham. https://doi.org/10.1007/978-3-030-81085-6_3.
- Kress, W.J. (2015). García-Robledo, Uriarte, M., and Erickson, D.L. (2015). DNA barcodes for ecology, evolution, and conservation. Trends in Ecology and Evolution 30(1): 25-35.
- Machlis, G.E. (1992). The contribution of sociology to biodiversity research and management. Biological Conservation 62(3):161-170.
- Manel, S., Schwartz, M.K., Luikart, G., and Taberlet, P. (2003). Landscape genetics: combining landscape ecology and population genetics. Trends in Ecology and Evolution 18(4):189-197.
- Mascia, M. B., Brosius, J.P., Dobson, T.A., Forbes, B.C., Horowitz, L., McKean, M.A., and Turner, N.J. (2003). Conservation and the social sciences. Conservation Biology 17(3): 649-650.
- Mazel, F., Pennell, M.W., Cadotte, M.W., Díaz, S., Riva, G.V.D., Grenyer, R., Leprieur, F., Mooers, A.O., Mouillot, D., Tucker, C.M., and Pearse, W.D. (2018). Prioritizing phylogenetic diversity captures functional diversity unreliably. Nature Communications 10:858.
- McMahon, B.J., Teeling, E.C., and Höglund, J. (2014). How and why should we implement genomics into conservation? Evolutionary Applications 7:999-1007.
- Meine, C., Soulé, M., and Noss, R.F. (2006). "A mission-driven discipline": the growth of conservation biology. Conservation Biology 29(3):631-651.
- Michener, C.D. (2000) Bees of the World. The Johns Hopkins University Press, Baltimore and London.
- Milliron, H.E. (1971). A monograph of the western hemisphere bumblebees (Hymenoptera: Apidae; Bombinae). I. The genera *Bombus* and *Megabombus* subgenus *Bombbias*. Memoirs of the Entomological Society of Canada *No. 82* 80.
- Mullis, K.B. (1990). Target amplification for DNA analysis by the polymerase chain reaction. Annales de Biologie Clinique 48(8):579-582.
- Noske, R.A., and Briggs, A. (2021). Species loss and decline among birds of coastal Central Queensland over 130 years. Pacific Conservation Biology: -, https://doi.org10.1071/PC20081.
- Ollerton, J. (2017). Pollinator diversity: distribution, ecological function, and conservation. Annual Review of Ecology, Evolution, and Systematics 48:353- 376.
- Poole, R.W. (1996). *Nomina insecta nearctica* [sic], a checklist of the insects of North America. Rockville, Maryland. Volume 2: Hymenoptera, Mecoptera, Megaloptera, Neuroptera, Rhaphidioptera, Trichoptera. 793.
- Richardson, J.L., Brady, S.P., Wang, I.J., and Spear, S.F. (2016). Navigating the pitfalls and promise of landscape genetics 25:849-863.
- Roffler, G.H., Schwartz, M.K., Pilgrim, K.L., Talbot, S.L., Sage, G.K., Adams, L.G., and Luikart, G. (2016). Identification of landscape features influencing gene flow: how useful are habitat selection models? Evolutionary Applications 9(6):805-817.
- Sandström, Lundmark, C., Andersson, K., Johannesson, K., and Laikre, L. (2019). Understanding and bridging the conservation-genetics gap in marine conservation. Conservation Biology 33(3): 725-728.
- Saunders, C.D. (2003). The emerging field of conservation psychology. Human Ecology Review 10(2):137-149.
- Schultz, P.W. (2007). Conservation means behavior. Conservation Biology 25(6): 1080- 1083.
- Shafer, A.B.A., Wolf, J.B.W., Alves, P.C., Bergström, L., Bruford, M.W., Brännström, I., Colling, G., Dalén, L., De Meester, L., Ekblom, R., Fawcett, K.D., Fior, S., Hajibabaei, M., Hill, J.A., Hoezel, A.R., Höglund, J., Jensen, E.L., Krause, J., Kristensen, T.N., Krützen, M., McKay, J.K., Norman, A.J., Ogden, R., Österling, E.M., Ouborg, N.J., Piccolo, J., Popović, D., Primmer, C.R., Reed, F.A., Roumet, M., Salmona, J., Schenekar, T., Schwartz, M.K.,Segelbacher, G., Seen, H., Thaulow, J., Valtonen, M., Veale, A., Vergeer, P., Vijay, N., Vilá, C., Weissensteiner, Wennerström, L., Wheat, C.W., Zieliński, P. (2015). Genomics and the challenging translation into conservation practice. Trends in Ecology and Evolution 30(2): 78-87.
- Sheffield, C.S., Richardson, L., Cannings, S., Ngo, H., Heron, J., and Williams, P.H. (2016). Biogeography and designatable units of *Bombus occidentalis* Green and *B. terricola* Kirby (Hymenoptera: Apidae) with implications for conservation status assessments. Journal of Insect Conservation 20:189-199.
- Spooner, F.E., Pearson, R.G., and Freeman, R. (2018). Rapid warming is associated with population decline among terrestrial birds and mammals globally. Global Change Biology 24(10):4521-4531.
- Storfer, A., Murphy, M.A., Evans, J.S., Goldberg, C.S., Robinson, S., Spear, S.F., Dezzani, R., Delmelle, E., Vierling, L., and Waits, L.P. (2007). Putting the 'landscape' in landscape genetics. Heredity 98:128-142.
- Taylor, H.R., Dussex, N., and van Heezik, Y. (2017). Bridging the conservation genetics gap by identifying barriers to implementation for conservation practitioners. Global Ecology and Conservation 10: 231-242.
- Teichroew, J.L., XU, J., Ahrends, A., Huang, Z.Y., Tan, K., and Xie, Z. (2017). Is China's unparalleled and understudied bee diversity at risk? Biological Conservation 210, Part B:19-28.
- Thomsen, P.F. and Willerslev, E. (2015). Environmental DNA- An emerging tool in conservation for monitoring past and present biodiversity 183: 4-18.
- Véron, S., Saito, V., Padilla-García, N., Forest, F., and Bertheau, Y. (2019). The use of phylogenetic diversity in conservation biology and community ecology: a common base but different approaches. The Quarterly Review of Biology 94(2)
- Wallace, A.R. (1863). On the physical geography of the MAlay Archipelago. Journal of the Royal Geographical Society 33.
- Williams, P.H., Brown, M.J.F., Carolan, J.C., An, J., Goulson, D., Aytekin, A.M., Best, L.R., Byvaltsev, A.M., Cederberg, B., Dawson, R., Huang, J., Ito, M., Monfared, A., Raina, R.H., Schmid-Hempel, P., Sheffield, C.S., Šima, P., and Xie, Z. (2012). Unveiling cryptic species of the bumblebee subgenus *Bombus s. str.* worldwide with COI barcodes (Hymenoptera: Apidae). Systematics and Biodiversity 10:21- 56.
- Williams, P.H. (2021). Not just cryptic, but a barcode bush: PTP re-analysis of global data for the bumblebee subgenus *Bombus s. str.* Supports additional species (Apideae, genus *Bombus*). Journal of Natural History 55:271-282.
- Zattara, E.E., and Aizen, M.A. (2020). Worldwide occurrence records suggest a global decline in bee species richness. One Earth 4(1):114-123.

CHAPTER II

GENOME-WIDE MARKERS TEST THE STATUS OF TWO PUTATIVE SPECIES OF NORTH AMERICAN BUMBLE BEES

Abstract

Bombus occidentalis Greene is one of at least three North American bumble bee species within the genus *Bombus* that is decreasing in abundance and range. The historical range of this species extends through western North America from Alaska to New Mexico and as far east as South Dakota (Black Hills) and western Nebraska, with populations restricted to high-elevation sites in the southern portion of the range. Two recent studies used mitochondrial *cytochrome oxidase I* (COI) barcode sequencing and automated species delimitation methods to identify two evolutionarily unique taxa, *B. occidentalis occidentalis* and *B. occidentalis mckayi* within *B. occidentalis,* but the species delimitation used in the studies disagreed on the species status of *B. occidentalis mckayi*. We used nuclear (ultraconserved elements) and mitochondrial (COI) markers to infer maximum likelihood (ML) and Bayesian phylogenies of the relationship between *B. occidentalis occidentalis* and *B. occidentalis mckayi*. We used seven species delimitation methods to conduct the most thorough test of the species status of these taxa yet performed. The phylogenies from our analyses agree that *B. occidentalis mckayi* is a monophyletic clade, but our ML phylogenies (UCE and COI) placed that clade within *B. occidentalis* while our Bayesian phylogeny (COI) resolved the taxa as reciprocally monophyletic. Similarly, the automated species delimitation analyses disagreed between ML and Bayesian phylogenies, with ML analyses lumping the taxa together and Bayesian analyses separating them. Species delimitation analyses based on diversity gaps among

sequences, rather than phylogenies, grouped *B. occidentalis occidentalis* and *B. occidentalis mckayi* together with their sister species *B. terricola*. Despite mixed results from species delimitation methods, we believe that the consistent monophyletic assignment of *B. occidentalis mckayi* specimens represents sufficient evolutionary divergence to elevate *B. occidentalis mckayi* to the level of species.

Introduction

Species in the bumblebee subgenus *Bombus sensu stricto* (s.s.) are economically important for crop pollination and are dominant native pollinators in many ecosystems across the Holarctic region of the world (Goulson 2003; Hines 2008). There are up to 23 identified species in the subgenus worldwide, with as many as eight species native to North America (Williams 2021). However, population distributions and abundances of many *Bombus s.s.* species are decreasing, including at least three species in North America: *Bombus affinis* Cresson 1863*, Bombus franklini* Frison 1921*,* and *Bombus occidentalis* Greene 1858 (Cameron et al. 2011; Colla et al. 2012; Abertman et al. 2017). *Bombus affinis* was the first bumble bee species to be listed as endangered by the Endangered Species Act (ESA) in the United States. *Bombus franklini* has not been observed since 2006, is listed as endangered throughout its range in northern California and southern Oregon (Thorp 2005), and is suspected to be extinct. *Bombus occidentalis* is listed as threatened or endangered by the Species At Risk Act (SARA) in portions of its Canadian distribution and is under consideration for listing by the ESA in the United States. *Bombus occidentalis*, in particular, is the focus of renewed interest among wildlife managers and conservationists due to unresolved taxonomic questions that may influence the upcoming ESA listing decision in the United States. Resolution of the taxonomic
status of *B. occidentalis* could influence the listing decision in the United States by altering the definition of the species boundary, which would influence where land-use restrictions with potential economic repercussions could be enforced (Haig et al. 2006).

Bombus occidentalis currently has a large geographical range. It is found throughout western North America from Alaska to New Mexico and as far east as the Black hills in South Dakota and western Nebraska, with distributions restricted to highelevation sites in the southern portion of the range. Regional morphological variation in pyle color historically led to several proposed delimitations of species and subspecies status within *B. occidentalis*, with some taxonomists suggesting that it is conspecific with *B. terricola* Kirby 1837 (Milliron 1971; Poole 1996) and others defining several subspecies (see Sheffield et al. 2016 for a thorough review). The most recent, and most widely accepted, delimitation indicates two taxa (*B. Occidentalis occidentalis* and *B. occidentalis mckayi*, Williams et al. 2012), broadly based on the presence or absence of a yellow band of hairs on the abdominal terga. The "un-banded" group extends from the southern edge of the species' range to approximately 55 degrees latitude (though specimens in some parts of this range do have a weak band), and the "banded" group extends from 55 degrees latitude to the northern edge of the range. Whether these taxa represent species or subspecies is still unclear (Williams et al. 2012; Sheffield et al. 2016; Williams 2021). Identification of the morphotypes within the taxon is challenging, because their definitions are mostly based on pyle color, which is variable among and within geographical regions of the taxa's ranges (Carolan et al. 2012; Sheffield et al. 2016) and specimens often exhibit intermediate characteristics among the morphotypes.

Geographic collection locations are often used to help define subspecies (Sheffield et al. 2016).

Given that morphology has proven unreliable to delimit species in *Bombus s.s.*, molecular data are needed to test and refine species boundaries. Molecular data have helped resolve boundaries in a variety of *Bombus* species groups (Lecocq et al. 2015; Lecocq et al. 2019; Williams et al. 2012; Williams et al. 2019; Williams et al. 2020; Ghisbain et al. 2020; Williams 2021), but few studies to date have examined species in *Bombus s.s.* and all have relied upon the single mitochondrial marker *cytochrome oxidase I* (COI), the barcoding gene (Williams et al. 2012; Williams 2021). Using COI data, Williams et al. (2012) found support that *B. occidentalis* is a separate species from *B. Terricola* and that *B. occidentalis* comprises two subspecies, *B. occidentalis occidentalis* in the southern portion of the range and *B. occidentalis mckayi* in the northern portion of the range, possibly with an overlapping distribution between 55 and 60 degrees latitude. Most recently, re-analysis of the same COI barcoding dataset using an alternate molecular delimitation method found support for raising *B. occidentalis mckayi* to species status (Williams 2021). Sharp decreases in geographic range and abundance have been observed primarily in the southern portion of the species range of *B. occidentalis occidentalis* (Evans et al. 2008; Cameron et al. 2011; Graves et al. 2020), with no evidence that populations of *B. occidentalis mckayi* in the northern portion of the range are unstable (Koch and Strange 2012; Pampell et al. 2015). In this case, the species status of *B. occidentalis mckayi* could have a strong influence on policy decisions regarding the conservation of *B. occidentalis* into the future.

Although COI data can be useful for differentiating and identifying species (Williams et al. 2012; Williams et al. 2019; Williams et al. 2020; Nneji et al. 2020; Williams 2021), there are some cases in which the evolution of the mitochondrial COI gene does not concur with the multi-locus nuclear phylogeny (i.e. mito-nuclear discordance, Toews and Brelsford, 2012; Achurra and Eréus 2013; Guening et al. 2020), including some examples in bumble bees (Williams 2021). Reasons for this discordance include incomplete lineage sorting caused by the dramatically smaller effective population sizes of mitochondrial than nuclear genomes within census populations (Funk and Omland 2003; Després 2019), asymmetrical introgression of the two types of genomes across a geographic range after a period of isolation among groups of populations (Després 2019), cytoplasmic bacterial infections (e.g. *Wolbachia*) that may drive fixation of mitotypes in populations (Hurst and Jiggins, 2005), and dissimilarities in how the mitochondrial and nuclear markers are dispersed across the landscape due to sexbased dispersal (mitochondrial genomes are often maternally inherited while nuclear genomes are biparental, Rheind and Edwards 2011). Due to the challenge presented by mito-nuclear discordance, use of multiple nuclear and mitochondrial markers along with multiple species delimitation methods (each with their own strengths and weaknesses), and morphological analysis for species delimitation are necessary to confidently delimit potentially cryptic species (Dupuis et al. 2012; Fujita et al. 2012; Carstens et al. 2013; Hurtado-Burillo et al. 2016; Lukhtanov 2019; Després 2019), with the final delimitation informed by the majority consensus of the markers (Pedraza-Marrón et al. 2019; Després 2019; Gueuning et al. 2020).

Bombus sensu stricto, and the species status of *B. occidentalis* in particular, has proven a difficult group to disentangle using mitochondrial barcoding (Williams 2012), though the recent re-analysis of available mitochondrial data has added some clarity (Williams 2021). The particular difficulties of resolving these species emphasizes the need for both mitochondrial and nuclear markers for species delimitation in this group. The addition of phylogenies based on nuclear markers will greatly improve the confidence of species or subspecies delimitations between *B. occidentalis occidentalis* and *B. occidentalis mckayi* and will contribute substantially to future conservation decisions (Hines et al. 2006; Cameron et al. 2007; Gueuning et al. 2020; Sun et al. 2021).

In this study we tested the taxonomic status of *B. occidentalis occidentalis* and *B. occidentalis mckayi* by analyzing a genome-scale dataset composed of thousands of nuclear ultraconserved element markers and a complementary COI dataset. For both, we tested species boundaries using a variety of species delimitation methods. Ultraconserved elements (UCEs) are highly conserved regions of nuclear DNA found throughout the genome of most eukaryotic species, and many recent studies have used these markers to successfully resolve phylogeny and test species boundaries (Musher and Cracraft 2018; Prebus 2020, Guening et al. 2020; Branstetter and Longino 2022), including in bees and other Hymenoptera. We used next generation sequencing and phylogenomic analyses to build phylogenomic trees representing the relationships among sampled individuals. Additionally, we were able to extract full COI barcodes from our UCE sequences. We used these samples and publicly available COI barcode sequences from the Barcode of Life Database (BOLD) (https://www.boldsystems.org/index.php) to build a gene tree that expands on the geographic sampling of the tree presented in Williams (2012, 2021). We

used the UCE and COI datasets to assess the current species status of *B. occidentalis occidentalis* and *B. occidentalis mckayi*.

Materials and Methods

Acquisition and management of Bombus occidentalis tissue samples

We obtained tissue samples from 102 specimens from across the range of *B. occidentalis occidentalis* and *B. occidentalis mckayi* (Fig. 2.1). Samples were provided by six institutions: the U.S. National Pollinating Insect Collection, the Royal Museum of British Columbia, the University of Alaska Museum of the North, the Essig Museum of Entomology, the University of Calgary Zoology Museum, and the Canadian National Collection. All tissue samples collected for this study were frozen and stored at USDA-ARS Pollinating Insect-Biology, Management, Systematics Research Laboratory (PIRU) in Logan, Utah, USA. DNA extracts were frozen and stored at USDA-ARS PIRU. Specimens owned by each of the respective collections were assigned unique identifiers by those institutions and are permanently stored in those collections (Tables A1, A2, A3, A4).

DNA extraction, UCE enrichment, and sequencing

Methods generally followed those in Branstetter et al. 2021. We extracted DNA from the mid and hind legs of specimens using a Zymo Quick-DNA Miniprep Plus extraction kit and stored extracts in -80°C freezers at the PIRU. Specimens were collected between 1956 and 2017, with one specimen from 1920.

We used a Tapestation 4150 automated electrophoresis system (Agilent, 5301 Stevens Creek Blvd. Santa Clara, CA 95051, USA) to measure the size of DNA

fragments extracted from the specimens and Qubit 3.0 to quantify DNA concentrations. The size of fragments varied among specimens due to their variable ages, collection methods, and storage histories. We sheared the DNA fragments to target fragment sizes of 400 to 600 base pairs using a Q800R2 acoustic sonicator (Qsonica, Newtown, CT, U.S.A.). We varied shearing times from 0 seconds to 120 seconds with a 10 seconds on, 10 seconds off pulsing pattern. Samples with small fragment sizes were sheared for less time and samples with large fragment sizes were sheared for more time. Once sonicated, we purified the DNA samples using a homemade paramagnetic bead solution (Rohland and Reich 2012).

We captured and sequenced UCE loci from our sample specimens following the methods described in Branstetter et al. (2021). We prepared Illumina sequencing libraries using Kapa Hyper prep kits and custom 8 bp dual indexing adapters (Glenn et al. 2019). We amplified the libraries using 12 cycles of PCR, cleaned the amplified DNA using 1.0 to 1.2x SPRI beads to remove contaminants and fragments smaller than 200 bp, and quantified the DNA using Qubit. Samples with low measured volumes of DNA were reamplified for 14 to 16 PCR cycles from an aliquot of the pre-PCR library.

We enriched the samples using an existing UCE bee-ant specific baitset (bee-antspecific Hym-v2, Branstetter et al. 2017; Grab et al. 2019) identified and optimized for use in the order Hymenoptera. The baitset was developed using seven genomes from hymenopteran species, including two species from the bee families Apidae and Halictidae. We enriched the pooled libraries following a combination of the Arbor Biosciences v3.02 protocol (enrichment day 1) and a protocol based on Blumenstiel et al. (2010, available at ultraconserved.org). We pooled up to ten samples per library at

equimolar concentrations for enrichment. Finally, we repeated the PCR amplification, purification, and quantification steps previously described for the pooled enriched samples. Enriched pools were combined into a final sequencing pool and sent to Novogene Inc. for sequencing on an Illumina HiSeq X instrument (PE150).

UCE processing and analysis

We demultiplexed and converted the raw sequences to fasta files using BCL2FASTQ (Illumina, San Diego, CA, USA). In addition to the *B. occidentalis* samples we included one *B. terricola* sequence as an outlier that was extracted and sequenced using the same methods for a previous study. *B. terricola* is the sister species to *B. occidentalis.* We used PHYLUCE version 1.7.1 software (Faircloth 2016) and the associated programs to process the UCE dataset and to generate sequence alignments using the method described by Branstetter et al. (2021). Within the PHYLUCE environment, we used ILLUMIPROCESSOR (Faircloth 2013) to batch process sequences and trim for adaptor contamination using TRIMMOMATIC (Bolger et al. 2014), and assembled contigs *de novo* using SPADES (Bankevich et al. 2012). We used PHYLUCE programs to extract, clean, and align the sequences. We used the match contigs to probes program, which uses LASTZ (Harris 2007), to match the contig sequences to probe sequences and create a database of the fasta files. Finally, we used the get fastas from match counts program to create a monolithic fasta file. Per the recommendation of Branstetter et al. (2021), we set the min-identity and min-coverage to 70 and 75, respectively, to recover the highest number of UCE loci possible. We aligned the UCE loci using MAFFT within the PHYUCE program align seqcap align. We removed poorly aligned regions using GBLOCKS (Talavera and Castresana 2007) within

the PHYLUCE program align_get_gblocks_trimmed_alignments_from_untrimmed with settings of $b1 = 0.5$, $b2 = 0.5$, $b3 = 12$, and $b4 = 7$. Finally, we filtered the alignments to include only those alignments that contained at least 75% of the samples using the PHYLUCE program align_get_only_loci_with_min_taxa program and concatenated the alignments into one phylip file using the align_concatenate_alignments.

A preliminary UCE tree was inferred using IQ-TREE version 2.0 (Nguyen et al. 2015) and visualized using FIGTREE version 1.4.4

(http://tree.bio.ed.ac.uk/software/figtree/). This tree showed that many of the older samples had terminal branches that were longer than expected, potentially skewing their positions in the tree. This is likely due to alignment issues caused by aligning smaller DNA fragments to longer ones. To remove poorly aligned sequences, we used the program SPRUCEUP version 2020.2.19, which is designed to remove outlier sequences from multiple sequence alignments (Borowiec 2019) by removing base pairs on a per sample basis, rather than entire alignment columns. We used an uncorrected distance method with a window size of 20 and an overlap of 15. We iteratively trimmed sequences using user-defined cutoffs for individual samples and compiled intermediate trees until most samples had appropriate branch lengths. Samples that still had exaggerated branch lengths after trimming with SPRUCEUP had relatively short average fragment sizes (<1000 bp), and were removed from the analysis.

After poorly aligned sequences were removed, the concatenated loci were separated back into genes using AMAS (Borowiec 2016) and empty columns were removed from each gene matrix using the custom script remove_empty_columns.py (https://github.com/marekborowiec/remove_empty_columns). The loci were filtered for taxon completeness at 75%, 90%, 95%, and 100% using the

get fastas from match counts program in PHYLUCE. The matrix that required 75% completeness was selected for further analysis. The taxa included in the analysis are very closely related, so there were many loci that were uninformative (had no site differences). The required level of completeness was kept relatively low to include more informative loci. Assessments of the influence of missing data on phylogenomic analyses have produced conflicting results (Phillippe et al. 2004; Thomson and Shaffer 2010; Roure et al. 2012; Sayyari et al. 2017), but careful selection of evolutionary models and inclusion of thousands of genes in analyses likely help to mitigate incorrect taxon placement on phylogenetic trees caused by missing data (Roure et al. 2012; Sayyari et al. 2017) and datasets that contained 30% or less of missing data have been shown to resolve phylogenies correctly (Shah et al. 2021).

We ran a partitioned analysis in IQTREE to produce a final maximum likelihood (ML) species tree. Partitioning creates an analysis that accommodates different substitution patterns in DNA based on the site. We used the general time reversible substitution model with the rate of variation across sites incorporated (GTR $+$ G, Tavaré 1986; Yang 1994). The partitions were derived from the directory of aligned nexus files (one for each locus) produced in PHYLUCE and the best-fit partition scheme was determined using the TESTMERGE option, which uses the greedy algorithm of PartitionFinder (Lanfear et al. 2012) and immediately reconstructs the tree using the best partitioning scheme. We optimized our analysis (sped it up) by choosing some model parameters *a priori*, in place of the model default which runs multiple analyses for each parameter and chooses the best fit. We used the fast relaxed clustering algorithm in place of the slow greedy algorithm with ten percent of the partition pairs to find the best-fit partitions (Lanfear et al. 2017). We also specified AICc (corrected Akaike information criterion) as our optimality criterion, in place of the default of also considering AIC and BIC (Bayesian information criterion). AICc corrects for bias in AIC when sample sizes are small. The correction disappears when sample sizes are large (Hurvich and Tsai 1989; Susko and Roger 2020). We used 1000 replicates of ultrafast bootstrapping (UFB) optimized with nearest neighbor interchange to generate bootstrap supports for the final tree. Additionally, we used 1000 replicates of an approximate likelihood ratio test (Guindon et al. 2010) to provide supports for single branches. We used FigTree to visualize the final species tree.

We used IQTREE to infer gene trees for each locus within the directory of aligned nexus files, which were used in downstream species delineation analyses. We used model testing to select the best substitution model and the -S option in IQTREE to loop through the aligned locus sequences and used 1000 replicates of ultrafast bootstrapping over trees to generate bootstrap support values for the nodes on the gene trees.

COI processing and analysis

We extracted COI barcodes from the *B. occidentalis* and *B. terricola* UCE targeted sequences using the PHYLUCE program assembly match contigs to barcodes and a sequence downloaded from BOLD as a bait sequence (BBHYL247).We aligned the COI sequences using MAFFT within the PHYLUCE program and visually inspected the alignments using the program Mesquite version 3.7.0.

The species delimitation method we used with the COI data (see below) requires equivalent sampling of multiple closely-related species (Taravera et al. 2013) and a less

closely related outgroup species to train the model (as well as the unresolved taxa) to perform reliably. We downloaded all publicly available COI barcode sequences from genbank (Sayers et al. 2022) for *B. occidentalis*, *B. terricola, B. hypocrita* Pérez 1905*,* and *B. jacobsoni* Skorikov 1912, which are closely related, and *B. lucorum* Linnaeus 1761, which is not as closely related but still within the sub-genus, as an outgroup. We combined the sequences with the complete COI barcodes extracted from our samples and re-aligned the entire dataset using PHYLUCE tools as described above.

We performed a partitioned analysis in IQTREE to produce a preliminary ML COI barcoding tree. We used all available *B. occidentalis occidentalis* and *B. occidentalis mckayi* specimens from our sequences and from publicly available sequences in BOLD, plus all publicly available sequences from the closely-related species in BOLD. We used the ModelFinder substitution model (Lanfear et al. 2012) to automatically determine the best-fit model for the data. We used 1000 ultrafast bootstrap replicates and 1000 replicates of an approximate likelihood ratio test to generate bootstrap supports for the tree. We removed all but one specimen that shared the same haplotypes from the *B. occidentalis occidentalis* and *B. occidentalis mckayi* groups and reran the analysis to produce a final ML gene tree. We visualized the final gene tree using FigTree.

We created a Bayesian gene tree using the COI barcoding dataset for use with a downstream species delimitation analysis. We used BEAUti version 1.10.4 (Drummond et al. 2012) to prepare an input file and BEAST version 1.10.4 (Drummond et al. 2012) to infer the tree. We defined taxon sets *a priori* as the previously described species, including *B. occidentalis occidentalis* and *B. occidentalis mckayi* separately. We used the $GTR + G$ substitution model (Hasegawa et al. 1985) with three partitions and a strict

clock (Drummond and Suchard 2010). We set the tree prior to a Yule Process (Drummond et al. 2010). We ran the analysis for 70,000,000 MCMC steps and visualized the resulting traces in TRACER version 1.7.2 (Rambaut et al. 2018) to ensure that the model coalesced. We used TreeAnnotator version 1.10.4 (Drummond et al. 2012) to summarize the data from the replicated trees onto a single target tree. We used 10% of the dataset (2,500 trees) as burn-in, calculated the median of the support values, and mapped them onto the target tree. We visualized the final gene tree in FigTree.

Species delimitation

We tested species boundaries in *Bombus occidentalis* using seven molecular delimitation approaches applied to ML and Bayesian estimates (Table 2.1). These included consideration of species monophyly within phylogenetic reconstructions, Species bOundary Delimitation using ASTRAL (SODA, Rabiee and Mirarab 2020), the Poisson Tree processes (PTP, Zhang et al. 2013), multi-rate PTP (mPTP, Kapli et al. 2017), generalized mixed Yule-coalescent models (GMYC, Pons et al. 2006), Automatic Barcode Gap Discovery (ABGD, Puillandre et al. 2012), and Automated simultaneous analysis phylogenetics (ASAP, Sarkar et al. 2008). PTP, mPTP, ABGD, and ASAP are methods based on the phylogenetic species concept (Baum and Shaw 1995) while SODA analyses are based on the multi-species coalescent model (MSC), which uses the discordance among gene trees to estimate the species tree (Pamilo and Nei 1988; Rannala et al. 2020).

SODA (Rabiee and and Mirarab 2020, https://github.com/maryamrabiee/SODA) was used to delimit species for the UCE dataset. This method is based on the multispecies coalescent model and is similar to the popular program BPP (Yang 2015). It compares

discordance among gene trees which is useful with datasets that contain information from many genes. Genes with no informative sites were removed from species delimitation analyses.

PTP and mPTP use gene or species trees (based directly on sequence substitutions rather than time since divergence) to estimate the number of species in the tree based on branch lengths (Zhang et al. 2013). mPTP is a modification of PTP which incorporates a new algorithm and model to accommodate varying levels of intraspecific genetic diversity among closely related species and sampling bias. PTP was used with COI datasets by Williams (2021) to delimit bumble bee species within the subgenus *Bombus* sensu stricto and by Williams et al. (2020) to delimit bumble bee species within the subgenus *Melanobombus*. We applied both PTP and mPTP to our COI barcode dataset in this study. We did not apply these methods to the UCE dataset because they require a minimum of five well-sampled, related species to train their algorithms (similar to GMYC as described by Taravers et al. 2013). COI sequences for species closely related to our taxa of interest were publicly available via BOLD, but we did not have a comparable dataset for our UCE analyses (UCE data for closely related but non-target species). We filtered haplotypes of the COI barcodes of our target taxa (*B. ocidentalis occidentalis* and *B. occidentalis mckayi*) per the recommendation of Williams et al. (2020), to avoid uneven sampling. One sample from groups with identical haplotypes was chosen to represent that group in the tree based on the length of the haplotypes. If all haplotypes were full barcodes (658 bp), representative samples were chosen to maximize the geographic sampling of the dataset. COI barcodes from the closely-related but nontarget taxa were not filtered for haplotypes. Instead, we selected sequences based on

length and included equivalent numbers of sequences from each species where possible. In the case of *B. jacobsoni*, only three COI barcode sequences were available. PTP species delineations were analyzed using the online PTP web server (https://species.hits.org/ptp/, Zhang et al. 2013) with 500,000 MCMC generations thinned by 100 with a burn-in of 0.1. mPTP species delineations were analyzed using the program mPTP (Kapli et al. 2017, https://github.com/Pas-Kapli/mptp).

GMYC classifies the branches of an ultrametric gene or species phylogenetic tree by maximizing the likelihood of a GMYC model; speciation rates held constant among species without extinction, and panmixia within species (Taravers et al. 2013). GMYC is the only analysis included in this study that requires an ultrametric phylogenetic tree in which the branches represent time, rather than nucleotide substitutions. Due to this requirement, this analysis was only performed on the Bayesian COI barcoding tree. The GMYC analysis was performed using the *splits* version 1.0-20 package.

ABGD sorts aligned barcode sequences (not appropriate for datasets that include multiple genes) into groups based on the ratio of divergence within and among groups. This analysis assumes that genetic divergence among species is greater than genetic divergence within species. The difference between the within-group and among-group diversity is called the 'barcode gap'. It requires two user inputs: P, the prior limit to the expected intraspecific diversity, and X, the minimum gap size between sequence clusters to identify a group (the sensitivity of the analysis). We did not have *a priori* knowledge of the range of intraspecies diversity or gap size among species, so we ran the analysis with a minimum P of 0.001 and a maximum P of 0.1 with steps of 10 at each of four gap

widths, 1, 1.5, 2, 2.5, and compared the results. We used a simple distance measurement for each analysis.

ASAP is similar to ABGD in that it uses a clustering method to sort aligned barcode sequences into groups based on intra- and interspecies diversity and does not build a phylogeny to identify those groups. However, it is different from ABGD in that it uses an updated scoring system that does not require any user defined input estimating the intraspecific diversity. We used a simple distance measurement with this analysis.

Results

We sequenced 102 samples that ranged in age from 3 to 65 years with one sample that was 101 years old. We removed 57 samples because the mean alignment length was below 1,000 bp. The final UCE dataset contained 23 *B. occidentalis occidentalis* specimens and 32 *B. occidentalis mckayi* specimens (Figure 2, Table A5 and A6). The final 75% taxon matrix included 2233 UCE loci (mean sequence length: 1346 ± 334.7), of which 1683 had at least one informative site. Loci with 0 informative sites were removed from the analysis (Table S2). There was a large gap in the geographical coverage of sampling for our UCE dataset in the southern half of British Columbia, Canada. This is partially due to an actual paucity of sampling in that region of the range, but also because the samples we did have for that area had low mean alignment lengths and, therefore, were removed from the analysis.

We mined 34 *B. occidentalis occidentalis* COI barcodes that represented 14 unique haplotypes and 32 *B. occidentalis mckayi* COI barcodes that represented 12 unique haplotypes from our sequences. We also included 12 *B. occidentalis occidentalis* COI barcode sequences and nine *B. occidentalis mckayi* COI barcode sequences from BOLD that represented unique haplotypes.

Phylogenetic reconstruction

The UCE-based phylogenetic tree inferred with IQ-Tree recovered *B. occidentalis mckayi* as reciprocally monophyletic to *B. occidentalis occidentalis* (Table 2.1, Fig. 2.2). However, the support for this relationship was very low, indicating that *B. occidentalis mckayi* is likely a subclade within *B. occidentalis.*Within each group, we examined the results for any evidence of geographic clustering that might indicate phylogeographic structure and did not find any clear patterns. Except for the monophyly of *B. occidentalis mckayi*, samples were generally randomly placed within clades.

The final ML COI barcoding tree indicated that *B. jacobsoni* and *B. hypocrita* were sister species and *B. terricola* was sister to *B. occidentalis* (Table 2.1, Fig. 2.3). All of these relationships were strongly supported, and agree with previous analyses (Cameron et al. 2007; Williams et al. 2012). We found strong support for *B. occidentalis mckayi* as a monophyletic clade, nested within a paraphyletic clade that also included multiple clades of *B. occidentalis occidentalis*, matching the UCE results above (Fig. 2.3).

The topology of the Bayesian COI barcoding tree agreed with the ML tree for all relationships except for *B. occidentalis occidentalis* and *B. occidentalis mckayi*, which it resolved as reciprocally monophyletic with strong support (Table 2.1, Fig. 2.4).

SODA analyses identified 22 species using the UCE phylogeny (Fig. 2.2, Table A7). The analysis correctly identified the *B. terricola* specimen as a separate species and the potential species that it identified were always composed of either *B. occidentalis occidentalis* or *B. occidentalis mckayi,* never mixed (Table 2.1, Table A4).

PTP and mPTP analysis

PTP and mPTP analyses both agreed that the ML COI phylogeny contained five species, with *B. occidentalis occidentalis* and *B. occidentalis mckayi* grouped as a single species and all other previously identified species separated (Table A8, Fig. 2.3). However, mPTP found 7 species using the Bayesian COI phylogeny (Table A9). It split one specimen of *B. hypocrita* into a separate species, but otherwise grouped the species into monophyletic clades and separated *B occidentalis* and *B. mckayi* (Table A8, Fig. 2.2, Fig. 2.4).

ABGD analysis

ABGD analyses were fairly consistent across the four sensitivity levels. Analyses with X (the minimum gap size that identifies a group) = 1 and 1.5 organized the samples into sets of one, four, five, or seven potential species, depending on the assigned intraspecies diversity, and analyses with $X = 2$ and 2.5 found organized samples into sets of four, five, and seven potential species (Table 2.2, Table A8). Analyses that delimited four species grouped *B. terricola* with *B. occidentlalis occidentalis* and *B. occidentalis mckayi* (Table 2.1, Fig. 2.2), but identified *B. lucorum, B. hypocrita,* and *B. jacobsoni* as separate groups. Analyses that delimited five species separated *B. terricola* from *B.*

occidentalis and *B. occidentalis mckayi*, but left the latter two taxa as a single species. Analyses that delimited seven species separated *B. occidentalis occidentalis* and *B.occidentalis mckayi*, but they also identified one specimen of *B. occidentalis occidentalis* from Idaho, USA (BLX2160) as a separate species.

ASAP analysis

ASAP analysis grouped the taxa in the COI barcoding dataset into groups of four, five, six, and seven species. The group of five species agreed with ABGD, PTP, and mPTP (Table A8). The groups of four and seven species agreed with those described by ABGD (Table 2.3, Table A8). The group of six species was identical to the group of seven species except for the specimen *B. occidentalis occidentalis* BLX2160, which was placed into the species with the other *B. occidentalis occidentalis* specimens (it was identified as its own species in the ABGD analysis, Table A8). The support ranking indicated that the four species solution was the most likely (Fig. 2.2), followed by the five species solution, seven species solution, and six species solution. The P value for the partition that includes four species is by far the lowest, which can be interpreted to indicate that it is favored as the most likely number of species. Note that P in the ABGD analysis and p in the ASAP analysis are different metrics.

GMYC analysis

GMYC analyses identified six species using the Bayesian COI barcoding phylogeny. It grouped all of the species as monophyletic groups, including *B. occidentalis occidentalis* and *B. occidentalis mckayi* (Fig. 2.2, Table 2.1, Table A9).

Discussion

Our analysis of the relationship between *B. occidentalis occidentalis* and *B. occidentalis mckayi* is the most thorough yet produced. We conducted the first analyses of nuclear markers to address this question. We expanded geographical sampling and applied more species delimitation methods than any previous analysis of the taxonomic status of the group (Williams et al. 2012; Williams 2021). We compared methods based on the phenic (Michener 1970), monophyletic (Donoghue 1985), diagnosable (Cracroft 1983), diagnosable phylogenetic (Nixon and Wheeler 1990), and multi-species coalescent (Pamilo and Nei 1988) concepts.

Species delimitation methods disagree on the species status of Bombus occidentalis mckayi

Our ML and Bayesian analyses did not agree on the phylogenetic relationship between *B. occidentalis occidentalis* and *B. occidentalis mckayi*. The ML UCE phylogeny found weak support for reciprocal monophyly between *B. occidentalis occidentalis* and *B. occidentalis mckayi*, indicating that *B. occidentalsi mckayi* is probably a subclade within the larger group. SODA analysis of the ML UCE phylogeny split the two taxa into 22 species, which is far more than have been previously suggested (Sheffield et al. 2016). The ML COI phylogeny and the automated species delimitation analyses associated with it agree that *B. occidentalis mckayi* is a monophyletic clade within *B. occidentalis occidentalis*. However, the Bayesian COI phylogeny and the automated species delimitation analyses performed on that dataset strongly support reciprocal monophyly between the taxa. Although reciprocal monophyly is strong evidence for speciation, it is not required (Rieseberg and Brouillet 1994; Knowles 2001;

Hörandl and Stuessy 2010). If *B. occidentalis* mckayi is accepted as a monophyletic species, our ML analyses indicate this would create a paraphyletic species in *B. occidentalis occidentalis*.

The most likely solutions from ABGD and ASAP (not dependent on any phylogeny) not only grouped *B. occidentalis occidentalis* and *B. occidentalis mckayi* together, but also included *B. terricola* as the same species. This is a relationship which has been previously analyzed and discounted (Williams et al. 2012; Williams et al. 2012) and which is not supported by any other analyses in this study. ABGD and ASAP both ranked a five species solution in which *B. terricola* is recognized as a separate species as second-most likely, but they ranked it far below the first-choice four species solution.

Comparison of results among methods and to previous studies

The topology of our ML UCE species tree and the ML COI barcode tree agreed that *B. occidentalis mckayi* is a well-supported monophyletic subclade within *B. occidentalis occidentalis*. However, SODA analysis of the ML UCE data based on the gene trees split the samples into over 20 species, very likely over splitting them. An important aspect of MSC is that individuals of the same species are assumed to have no structure within the species, so their alleles coalesce randomly (Rabiee and Mirarab 2020). SODA exploits this feature by creating a species tree in its sister program ASTRAL (Mirarab et al. 2014) and uses heuristic patterns of quartet trees (unrooted trees that include four taxa, Reaz et al. 2014) to identify areas of the species tree where there is complete coalescence (Rabiee and Mirarab 2020). However, if species do contain substantial structure, SODA may identify that intraspecies structure as species delimitations. UCEs are, as advertised, highly conserved, so one might expect to detect

little within-species genetic diversity using these markers. However, our ML phylogeny of the dataset indicates many highly supported bifurcating nodes within the clades that define *B. occidentalis occidentalis* and *B. occidentalis mckayi*, which is indicative of genetic structure within the taxa, though no geographic pattern of that structure was detected. Although more computationally intensive, Bayesian methods that use an MCMC chain to assign species based on gene trees may be more appropriate for datasets in which intraspecies genetic structure is detected or suspected because plausible species membership can be incorporated into the model based on previous work, geographic distribution, or morphology (e.g. Bayesian Phylogenetics and Phylogeography, Yang and Rannala 2010, Yang 2015).

The results of our Bayesian analyses disagree with the findings of Williams et al. (2012), who found that *B. occidentalis mckayi* is likely an evolutionarily unique taxon within *B. occidentalis* using GMYC modeling. This contradiction is likely due to characteristics inherent to GMYC modeling and the respective datasets. An increase in the number of specimens or haplotypes included in GMYC models is likely to increase the proportion of lineage splits detected (Pentinsaari et al. 2016). Our analysis included 14 haplotypes of *B. occidentalis occidentalis* and 12 haplotypes of *B. occidentalis mckayi* while Williams et al. (2012) included five haplotypes of *B. occidentalis occidentalis* and four haplotypes of *B. occidentalis mckayi*. Also, previous studies indicate that the performance of GMYC increases with the number of species and depth of subclade (Taravers et al. 2013). Our analysis included only six species from the subgenus *Bombus s.s.* (separating *B. occidentalis occidentalis* and *B. occidentalis mckayi*), while Williams et al. (2012) included many more morphologically identified taxa from among the entire

genus *Bombus*. Likely for these reasons, GMYC was not consistent between the two studies. The sensitivity of GMYC to differences in model parameters and input has contributed to its less frequent use in recent years, with PTP succeeding it in popularity for single gene tree and species tree analyses (Zhang et al. 2013; Simon 2020).

In a follow-up analysis, Williams (2021) re-analyzed the same Bayesian COI barcoding phylogeny using PTP analysis, which does not require dating for an ultrametric tree, and determined *B. occidentalis occidentalis* and *B. occidentalis mckayi* to represent two good species. This result agrees with the results of our Bayesian GMYC and mPTP analyses, but contradicts our ML analyses. Bayesian methods have been alternately praised for the easily interpreted support values of posterior probabilities, which describe the proportion of trees that return a clade in an MCMC chain, and criticized for inflated support values and sensitivity to the evolutionary models selected for the analyses (Suzuki et al. 2002; Simon 2020). Likewise, the bootstrap supports associated with ML analyses have been criticized for their unclear definition. They have been described as measures of "precision not accuracy" (Page and Holmes 1998) among other suggested interpretations (Simon 2020). Berry and Gascuel (1996) suggested that 1 minus the bootstrap value is the equivalent to a *p-*value associated with the test of the null hypothesis that a split (branching event) is not really present. The differences in the topology of our ML and Bayesian COI barcode phylogenies may be attributable to relatively inflated support for a split in the Bayesian analysis that was not supported in the ML analysis. As such, the ML and Bayesian analyses may be considered as conservative and liberal estimations respectively of the species status of *B. occidentalsi mckayi*.

Species status

Although our analyses provided some mixed support for the independence of *B. occidentalis mckayi* as a separate species, we conclude that the consistent monophyly of the taxa and the support of the species delimitation analyses of the Bayesian phylogeny are adequate to acknowledge *B. occidentalis mckayi* as a distinct species from *Bombus occidentalis*, *Bombus mckayi*. This result is in agreement with the conclusions of Williams (2021), and will encourage continued research into the conservation status of both species.

Conservation implications

The rank of subspecies is not as clearly defined as higher taxonomic levels, and has been applied inconsistently in the past (Haig et al. 2006; Phillimore and Owens 2006). Subspecies within *B. occidentalis* have previously been described based on morphology (Sheffield et al. 2016) and phylogenies (Williams et al. 2012; Williams 2021), with variable results. Our study confirms the monophyletic status of *B. mckayi* with a robust analysis of the COI barcoding gene and 1683 nuclear genes, though it is still unclear if that clade is within *B. occidentalis* (thereby rendering *B. occidentalis* a paraphyletic species) or if it represents a reciprocally monophyletic sister clade to *B. occidentalis*.

Recognition of these two well-defined taxa has implications for conservation policy in the United States and Canada. Within the bounds of the United States, *B. occidentalis* is found within the contiguous western states and is decreasing dramatically in abundance and range (Graves et al. 2020). *B. mckayi* is found in Alaska where populations appear to be stable at this time (Koch and Strange 2012; Pampell et al. 2015). The Endangered Species Act of the United States allows listing of subspecies (Haig et al. 2006; Waples et al. 2018), which makes it possible to list *B. occidentalis* as endangered without listing *B. mckayi* regardless of species status*.* Endangered species listings are contentious in the United States, as the restrictions placed on habitat often limit use of public and private lands (Haig et al. 2006; Sims and Palikhe 2019). Flexibility to list only the taxa of concern, regardless of species status, eases the burden of such restrictions at the national level. However, protections at the state and municipal level are variable in their taxonomic requirements and subspecies of conservation concern may not garner the attention or resources granted to species. Identification of evolutionarily distinct taxa as species provides a clear and easily understood delimitation for which to create conservation policy.

Both taxa exist within the bounds of Canada and the cryptic morphology and geographical overlap of the taxa make monitoring of population abundances tricky in some regions (Sheffield et al. 2016). Legislation under the Species at Risk Act varies among provinces and territories, which provides flexibility to protect populations in regions where they are in decline. However, this political structure for species protection has also been criticized for inconsistency in standards for listing and compliance with required actions for listed species among provinces and territories (Turcotte et al. 2021). Treatment of the taxa as a single species where their boundaries overlap would provide misleading information about the abundance of the separate taxa and add to confusion about conservation needs for the species. Species delimitation for *B. occidentalis* and *B. mckayi* will help to clarify the need for monitoring of the taxa separately. Continued sampling to monitor abundance and genetic viability of populations at risk is critical to

maintain the species in the Canadian portion of their ranges where they may be confounded.

Future work - genetic differentiation of Bombus occidentalis within a biogeographical context

All of our phylogenies agree that *B. mckayi* is at least a monophyletic group. Historical biogeography may help to explain this observed pattern of differentiation.

The ancestors to extant species of *Bombus sensu stricto* in North America likely entered the continent by crossing the Bering Land Bridge less than five million years ago (Hines 2008). Due to their evolutionary history, these taxa were likely to have been adapted to cold or temperate climates and could have migrated south along the coastal mountain ranges of western North America relatively quickly (Hines 2008). However, approximately 2.6 mya the Cordilleran ice sheet began to form in the Alaskan Range and grew southward (Hidy et al. 2013). At its maximum, this ice sheet stretched as far south as the North Cascades (Seguinot et al. 2016).

Genomic signals within *B. occidentalis* and *B. mckayi* may provide more information about how the two taxa evolved during the development and recession of this ice sheet across their ranges. If the current distributions of *B. occidentalis* and *B. mckayi* are predictive of the range of the most recent common ancestor before the formation of the Cordilleran ice sheet, that range likely would have been affected by the development of the ice sheet in one of two ways: first, by splitting and reducing the distribution of the ancestor species into isolated northern and southern distributions, or second, by reducing the range completely to a relatively small southern distribution. The ice sheet advanced and receded several times over the course of its existence, likely only receding

sufficiently to allow migration to resume along the pacific coast of North America 20,000 to 17,000 years ago (Pitulko and Pavlova 2020). If there were northern and southern relict populations, such a long separation would have permitted genetic divergence between them. Time since divergence could be estimated using single nucleotide polymorphisms (SNPs) called from the UCE dataset presented in this study (Gutenkunst et al. 2009; Everson et al. 2019) to estimate if the taxa diverged before or after the maximum extent of the ice sheet was established. Additionally, patterns of genetic diversity across the range estimated from the SNPs could indicate if there were two refugia or one that repopulated the previously glaciated portion of the range after the ice sheet retreated (Eckert et al. 2008; Swaegers et al. 2013).

Increased sampling in the geographic region where the subspecies ranges' overlap would provide data that could test the biological speciation concept (Wright and Huxley 1940) for these taxa by detecting (or not detecting) hybrid specimens and testing for reinforcement or introgression between the taxa (Butlin and Smadja; Garner et al. 2018). These analyses would provide additional information about the evolutionary history of the two species.

References

- Abertman, M.P., Gleiser, G., Morales, C.L., Williams, P. and Aizen, M.A. (2017). Global decline of bumblebees is phylogenetically structured and inversely related to species range size and pathogen incidence. Proceedings of the Royal Society B 284: 20170204. http://dxdoi.org/10.1098/rspb.2017.0204.
- Achurra, A. and Erséus, C. (2013). DNA barcoding and species delimitation: the *Stylodrilus heringianus* case (Annelida: Clitellata: Lumbriculidae). Invertebrate Systematics 27(1):118-128.
- Bankevich, A., Nurk, S., Antipov, D., Gurevich, A.A., Dvorkin, M., Kulikov, A.S., Lesin, V.M., Nikolenko, S.I., Pham, S., Prjibelski, A.D., Pyshkin, A.V., Sirotkin,

A.V., Vyahhi, N., Tesler, G., Alekseyev, M.A., and Pvezner, P.A. (2012). Journal of Computational Biology 19(5):455-477.

- Baum, D.A., and Shaw, K.L. (1995). Genealogical perspectives on the species problem. Experimental and molecular approaches to plant biosystematics 53(289-303):123- 124.
- Berry, V., and Gascuel, O. (1996). On the interpretation of bootstrap trees: appropriate threshold of clade selection and induced gain. Molecular biology and Evolution 13:999-1011.
- Blumenstiel, B., Cibulskis, K., Fisher, S., DeFelice, M., Barry, A., Fennell, T., Abreu, J., Minie, B., Costello, M., Young, G., Maquire, J., Kernytsky, A., Melnikov, A., Rogov, P., Gnirke, A., and Gabriel, S. (2010). Targeted exon sequencing by insolution hybrid selection. Current protocols in human genetics 66(1):18.4.1- 18.4.24.
- Bolger, A. M., Lohse, M., & Usadel, B. (2014). Trimmomatic: A flexible trimmer for Illumina Sequence Data. Bioinformatics. http://dx.doi.org/10.1093/bioinformatics/btu170.
- Borowiec, M.L. (2016). AMAS: a fast tool for alignment manipulation and computing of summary statistics. PeerJ 4:e1660.
- Borowiec, M.L. (2019). Spruceup: fast and flexible identification, visualization, and removal of outliers from large multiple sequence alignments. Journal of Open Source Software 4(42):1635.
- Branstetter, M.G., and Longino, J.T. (2022) UCE phylogenomics of New World *Cryptopone* (Hymenoptera: Formicidae) elucidates genus boundaries, species boundaries, and the vicariant history of a temperate-tropical disjunction. Insect systematics and diversity $6(1)$:1-23.
- Branstetter, M.G., Longino, J.T., Ward, P.S., and Faircloth, B.C. (2017). Enriching the tree of life: enhanced UCE bait set for genome-scale phylogenetics of ants and other Hymenoptera. Methods in Ecology and Evolution 8(6):768-776.
- Branstetter, M., G., Müller, A., Griswold, T.L., Orr, M.C., Chao-Dong, Z. (2021). Ultraconserved element phylogenomics and biogeography of the agriculturally important mason bee subgenus *Osmia* (*Osmia*). Systematic Entomology 46:453- 472.
- Butlin, R.K., and Smadja, C.M. (2018). Coupling, reinforcement, and speciation. The American Naturalist 191(2):155-172
- Cameron, S.A., Hines, H.M., and Williams, P.H. (2007). A comprehensive phylogeny of the bumble bees (*Bombus*). Biological journal of the Linnean Society 91(1):161- 188.
- Cameron, S.A., Lozier, J.D., Strange, J.P., Koch, J.B., Cordes, N., Solter, L.F., and Griswold, T.L. (2011). Patterns of widespread decline in North American bumble bees. Proceedings of the National Academy of Sciences of the United States of America 108(2):662-667.
- Carolan, J.C., Murray, T.E., Fitzpatrick, U., Crossley, J., Schmidt, H., Cederberg, B., McNally, L., Paxton, R.J., Williams, P.H., and Brown, M.J.F. (2012). Colour patterns do not diagnose species: quantitative evaluation of a DNA barcoded cryptic bumblebee complex. PLoS ONE 7(1):e29251.
- Carstens, B.C., Pelletier, T.A., Reid, N.M., and Satler, J.D. (2013). How to fail at species delimitation. Molecular Ecology 22(17):4369-4383.
- Colla, S.R., Gadallah, F., Richardson, L., Wagner, D., and Gall, L. (2012). Assessing declines of North American bumble bees (Bombus spp.) using museum specimens. Biodiversity and Conservation 21(14):3585-3595.
- Cracraft, J. (1983). Species concepts and speciation analysis. Current Ornithology 1:159- 187.
- Després, L. (2019). One, two or more species? Mitonuclear discordance and species delimitation. Molecular Ecology 28(71):3845-3847.
- Donoghue, M.J., (1985). A critique of the biological species concept and recommendations for a phylogenetic alternative. Bryologist 88:172-181.
- Drummond, A.J., and Suchard, M.A. (2010). Bayesian random local clocks, or one rate to rule them all. BMC Biology: 8: Article 114.
- Drummond, A.J., Suchard, M.A., Xie, D., and Rambaut, A. (2012). Bayesian phylogenetics with BEAUti and the BEAST 1.7. Molecular Biology and Evolution 29:1969-1973.
- Dupuis, J.R., Roe, A.D., and Sperling, F.A.H. (2012). Multi-locus species delimitation in closely related animals and fungi: one marker is not enough. Molecular Ecology 21(8): 4422-4436.
- Eckert, C.G., Samis, K.E. and Lougheed, S.C. (2008). Genetic variation across species' geographic ranges: the central-marginal hypothesis and beyond. Molecular Ecology 17:1170-1188.
- Evans, E., Thorp, R., Jepsen, S. and Hoffman Black, S. (2008). Status review of three formerly common species of bumble bee in the subgenus *Bombus*.

http://www.xerces.org/wpcontent/uploads/2009/03/xerces_2008_bombus_status_review.pdf

- Everson, K.M., McLaughlin, J.F., Cato, I.A., Evans, M.M., Gastaldi, A.R., Mills, K.K., Shink, K.G., Wilbur, S.M., and Winkler, K. (2019). Speciation, gene flow. And seasonal migration in *Catharus* thrushes (Aves: Turidae). Molecular phylogenetics and Evolution 139:106564.
- Faircloth, B.C. (2013). Illumiprocessor: a trimmomatic wrapper for parallel adapter and quality trimming. https://doi.org/10.6079/J9ILL.
- Faircloth, B.C. (2016). PHYLUCE is a software package for the analysis of conserved genomic loci. Bioinformatics 32:786-788. https://doi.org/10.1093/bioinformatics/btv646.
- Fujita, M.K., Leaché, A.D., Burbrink, F.T., McGuire, J.A., and Moritz, C. (2012). Coalescent-based species delimitation in an integrative taxonomy. Trends in Ecology and Evolution 27(9):480-488.
- Funk, D.J., Omland, K.E. (2003). Speices-level paraphyly and polyphyly: frequency, causes, and consequences, with insights from animal mitochondrial DNA. Annual Review of Ecology, Evolution, and Systematics 34:397-423.
- Garner, A.G., Goulet, B.E., Farnitano, M.C., Molina-Henao, Y.F., and Hopkins, R. (2018). Genomic signatures of reinforcement. Genes 9(4): 191.
- Ghisbain, G., Lozier, J.D., Rahman, S.R., Ezray, B.D., Tian, L., Ulmer, J.M., Heraghty, S.D., Strange, J.P., Rasmont, P., and Hines, H.M. (2020). Substantial genetic divergence and lack of recent gene flow support cryptic speciation in a colour polymorphic bumble bee (*Bombus bifarius*) species complex. Systematic Entomology 45(3):635-652.
- Glenn, T.C., Nilsen, R.A., Kieran, T.J., Sanders, J.G., Bayona-Vásquez, N.J., Finger, J.W., Pierson, T.W., Bentley, K.E., Hoffberg, S.L., Louha, S., Garcia-De Leon, F.J., del Rio Portilla, M.A., Reed, K.D., Anderson, J.L., Meece, J.K., Aggrey, S.E., Rekaya, R., Alabady, M., Belanger, M., Winker, K., and Faircloth, B.C. (2019). Adapterama I: universal stubs and primers for 384 unique dual-indexed or 147,456 combinatorially-indexed Illumina libraries (iTru & iNext). PeerJ 7:e7755
- Goulson, D. (2003). Bumblebees: their behavior and ecology. Oxford University Press, Oxford.
- Grab, H., Branstetter, M.G., Amon, N., Urban-Mead, K.R., Park, M.G., Gibbs, J., Blitzer, E.J., Poveda, K., Loeb, G., and Danforth, B.N. (2019). Agriculturally dominated landscapes reduce bee phylogenetic diversity and pollination services. Science 363(6424):282-284.
- Graves, T.A., Janousek, W.M., Gaulke, S.A.m Nicholas, A.C., Keinath, D.A., Bell,C.M., Cannings, S., Hatfield, R.G., Heron, J.M., Koch, J.B., Loffland, H.L., Richardson, L.L., Rohde, A.T., Rykken, J., Strange, J.P., Tronstad, L.M., Sheffield, C.S. (2020). Western bumble bee: declines in the continental United States and rangewide information gaps. Ecosphere 11(6): e03141.
- Gueuning, M., Frey, J.E., and Praz, C. (2020). Ultraconserved yet informative for species delimitation: ultraconserved elements resolve long-standing systematic enigma in central European bees. Molecular Ecology 29: 4203-4220.
- Guindon, S., Dufayard, J.-F., Lefort, V., Anisimova, M., Hordijk, W., and Gascuel, O. (2010). New algorithms and methods to estimate maximum-likelihood phylogenies: assessing the performance of PhyML 3.0. Systematic Biology 59(3):307-321.
- Gutenkunst, R.N., Hernandez, R.D., Williamson, S.H., and Bustamante, C.D. (2009). Inferring the joint demographic history of multiple populations from multidimensional SNP frequency data. PLoS Genetics 5(10): e1000695.
- Haig, S.M., Beever, E.A., CHambers, S.M., Draheim, H.M., Dugger, B.D., Dunham, S., Elliot-Smith, E., Fontaine, J.B., Kesler, D.C., Knaus, B.J., Lopes, I.F., Loschl, P., Mullins, T.D., and Sheffield, L.M. (2006). Taxonomic considerations in listing subspecies under the U.S. Endangered Species Act. Conservation Biology 20(6):1584-1594.
- Harris, R.S. (2007). Improved pairwise alignment of genomic DNA (order No. 3299002). Available from ProQuest Dissertations & Theses Global. (304835295). Retrieved from https://login.dist.lib.usu.edu/login.
- Hasegawa, M., Kishino, H., and Yano, T. (1985). Dating of the human-ape splitting by a molecular clock of mitochondrial DNA. Journal of Molecular Evolution 22(2):160-174.
- Hidy, A.J., Gosses, J.C., Froese, D.G., Bond, J.D., and Rood, D.H. (2013). A latest Pliocene age for the earliest and most extensive Cordilleran Ice Sheet in northwestern Canada.
- Hines, H.M., Cameron, S.A., and Williams, P.H. (2006). Molecular phylogeny of the bumble bee subgenus *Pyrobombus* (Hymenoptera: Apidae: *Bombus*) with insights into gene utility for lower-level analysis. Invertebrate Systematics 20:289-303.
- Hines, H. (2008). Historical biogeography, divergence times, and diversification patterns of bumble bees (Hymenoptera: Apidae: *Bombus*). Systematic Biology 57(1):58- 75.
- Hörandl, E. And Stuessy, T.F. (2010). Paraphyletic groups as natural units of biological classification. TAXON 59(6):1641-1653.
- Hurst, G.D.D., and Jiggins, F.M. (2005). Problems with mitochondrial DNA as a marker in population, phylogeographic and phylogenetic studies: the effects of inherited symbionts. Proceedings of the Royal Society B 272(1572):1525-1534.
- Hurtado-Burillo, M., May-Itzá, W. de J., Quezada-Eúan, J.J.G., de la Rúa, P., and Ruiz, C. (2016). Multilocus species delimitation in Mesoamerican *Scaptotrigona* stingless bees (Apidae: Meliponini) supports the existence of cryptic species.Systematic Entomology 42(1):171-181.
- Hurvich, C.M. and Tsai, C.L. (1989). REgression and time series model selection in small samples. Biometrika 76(2):297-307.
- Kapli, P., Lutteropp, S., Zhang, J., Kobert, K., Pavlidis, P., Stamatakis, A., and Flouri, T. (2017). Multi-rate Poisson tree processes for single-locus species delimitation under maximum likelihood and Markov chain Monte Carlo. Bioinformatics 33(11): 1630-1638.
- Koch, J.B., and Strange, J.P. (2012). The status of *Bombus occidentalis* and *B. moderatus* In Alaska with special focus on *Nosema bombi* incidence. Northwest Science 86(3):212-220.
- Knowles, L.L. (2001). Genealogical portraits of speciation in montane grasshoppers (genus *Melanoplus*) from sky islands of the Rocky Mountains. Proceedings of the Royal Society B
- Lanfear, R., Calcott, B., Ho, S.Y., and Guindon, S. (2012). Partitionfinder: combined selection of partitioning schemes and substitution models for phylogenetic analyses. Molecular Biology and Evolution 29:1695-1701.
- Lanfear, R., Frandsen, P.B., Wright, A.M., Senfeld, T., and Colcott, B. (2017). PartitionFinder 2: new methods for selecting partitioned modes of evolution for molecular and morphological phylogenetic analyses. Molecular Biology and Evolution 34(3):772-773.
- Lecocq, T., Dellicour, S., Michez, D., Dehon, M., Dewulf, A., Meulemeester, T.D., Brasero, N., Valterová, Rasplus, J.-Y., and Rasmont, P. (2015). Methods for species delimitation in bumblebees (Hymenoptera, Apidae, *Bombus*): towards and intergrative approach. Zoologica Scripta 44(3):281-297.
- Lecocq, T., Biella, P., MArtinet, B., and Rasmont, P. (2019). Too strict or too loose? Integrative taxonomic assessment of *Bombus lapidarius* complex (Hymenoptera: Apidae).Zoologica 49(2):187-196.

Lukhtanov, V.A. (2019). Species delimitation and analysis of cryptic species diversity in the XXI century. Entomological Review 99:463-472.

Michener, C.D. (1970). Diverse approaches to systematics. Evolutionary Biology 4:1-38.

- Milliron, H.E. (1971). A monograph of the western hemisphere bumblebees (Hymenoptera: Apidae; Bombinae). I. The genera *Bombus* and *Megabombus* subgenus *Bombbias*. Memoirs of the Entomological Society of Canada *No. 82* 80.
- Mirarab, S., Rez, R., Bayzid, M.S., Zimmerman, T., Swenson, M.S., and Warnow, T. (2014). ASTRAL: genome-scale coalescent-based species tree estimation. Bioinformatics 17(1):i541-i548.
- Musher, L.J., and Cracraft, J. (2018). Phylogenomics and species delimitation of a complex radiation of Neotropical suboscine birds (*Pachyramphus*). Molecular Phylogenetics and Evolution 118:204-221.
- Nixon, K.C., and Wheeler, Q.D. (1990). An amplification of the phylogenetic species concept. Cladistics 6:211-223.
- Nguyen, L.-T., Schmidt, H.A., von Haeseler, A., and Minh, B.Q. (2015). IQ-TREE: A fast and effective stochastic algorithm for estimating maximum-likelihood phylogenies. Molecular Biology and Evolution 32(1):268-274.
- Nneji, L.M., Adeola, A.C., Ayoola, O.A., Oladipo, S.O., Wang, Y.-Y., Malann, Y.D., Anyaele, O., Nneji, I.C., Rahman, M.M., and Olory, C.S. (2020). DNA barcoding ad species delimitation of butterflies (Lepidoptera) from Nigeria. Molecular Biology Reports 47:9441-9457.
- Page, R.D.M, and Holmes, E.C. (1998). Molecular evolution: a phylogenetic approach. Oxford: Blackwell Science Limited.
- Pamilo, P., and Nei, M. (1988). Relationships between gene trees and species trees. Molecular Biology and Evolution 5(5):568-583.
- Pampell, R., Sikes, D., Pantoja, A., Holloway, P., Knight, C., and Ranft, R. (2015). Bumble bees (*Hymenoptera: Apidae: Bombus* spp.) of interior Alaska: Species composition, distribution, seasonal biology, and parasites. Biodiversity Data Journal 3:e50585.
- Pedraza-Marrón, C. del R., Silva, R., Deeds, J., Van Belleghem, S.M., Mastretta-Yanes, A., Domínguez-Domínguez, O., Rivero-Vega, R.A., Lutackas, L., Murie, D., Parkyn, D., Bullock, L.H., Foss, K., Ortiz-Zuazaga, H., Narváez-Barandica, J., Acero, A., Gomes, G., and Betancur-R, R. (2019). Proceedings of the Royal Society B 286(1900)
- Pitulko, V.V., and Pavlova, E.Y. (2020). Colonization of the Arctic in the New World. *In* Encyclopedia of the World's Biomes, pgs 392-408. Eds. Goldstein, M. and DellaSala, D.A., Elsevier Inc.
- Pentinsaari, M., Vos, R., and Mutanen, M. (2016). Algorithmic single-locus species delimitation: effects of sampling effort, variation and nonmonophyly in four methods and 1870 species of beetles. Molecular Ecology Resources 17(3):393- 404.
- Philippe, H., Snell, E.A., Bapteste, E., Lopez, P., Holland, P.W.H., Casane, D. (2004). Phylogenomics of Eukaryotes: impact of missing data on large alignments. Molecular Biology and Evolution 21(9):1740-1752.
- Phillimore, A.B., and Owens, I.P.F. (2006). Are subspecies useful in evolutionary and conservation biology? Proceedings of the Royal Society B 273(1590):104-1053.
- Pons, J., Barraclough, T.G., Gomez-Zurita, J., Cardoso, A., Duran, D.P., Hazell, S., Kamoun, S., Sumlin, W.D., and Volger, A.P. (2006). Sequence-based species delimitation for the DNA taxonomy of undescribed insects. Systematic Biology 55(4):595-609.
- Poole, R.W. (1996). *Nomina insecta nearctica* [sic], a checklist of the insects of North America. Rockville, Maryland. Volume 2: Hymenoptera, Mecoptera, Megaloptera, Neuroptera, Rhaphidioptera, Trichoptera. 793.
- Prebus, M.M. (2020). Phylogenomic species delimitation in the ants of the *Temnothorax salvini* group (Hymenoptera: Formicidae): an integrative approach. Systematic Entomology 46(2):307-326.
- Puillandre, N., Lambert, A., Brouillet, S., and Achaz, G. (2011). ABGD, Automatic barcode gap discovery for primary species delimitation. Molecular Ecology 21(8):1864-1877.
- Rabiee, M., and Mirarab, S. (2020). SODA: multi-locus species delimitation using quartet frequencies. Bioinformatics 36(24):5623-5631.
- Rambaut, A., Drummond, A.J., Xie, D., Baele, G., and Suchard, M.A. (2018). Posterior summarization in Bayesian phylogenetics using Tracer 1.7. Systematic Biology 67(5):901-904.
- Rannala, B., Edwards, S.V.S.V., Leaché, A., and Yang, Z. (2020). The multi-species coalescent model and species tree inference. Phylogenomics in the Genomic Era. Eds: Scornavacca, C., Delsuc, F., and Galtier, N. Pp. 3.3:1-3.3:21.
- Reaz, R., Bayzid, M.S., ans Sohel Rahman, M. (2014). Accurate phylogenetic tree reconstruction from quartets: a heuristic approach. PLoS ONE 9(8):e104008.
- Rheindt, F.E., and Edwards, S.V. (2011). Genetic introgression: an integral but neglected component of speciation in birds. The Auk 128(4):620-632.
- Riesberg, L.H., and Brouillet, L. (1994). Are many plant species paraphyletic? Taxon 43:21-32.
- Rohland, N. and Reich D. (2012). Cost-effective, high-throughput DNA sequencing libraries for multiplexed target capture. Genome Research 22:939-946. http://doi.org/10.1101/gr.128124.111.
- Roure, B., Baurain, D., and Philippe, H. (2012). Impact of missing data on phylogenies inferred from empirical phylogeomic data sets. Molecular Biology and Evolution 30:197-214.
- Sarkar, I.N., Egan, M.G., Corizzi, G., Lee, E.K., and DeSalle, R. (2008). Automated simultaneous analysis phylogenetics (ASAP): an enabling tool for phylogenomics. BMC Bioinfomratics 9: Article 103.
- Sayers, E.W., Bolton, E.E., Brister, J.R., Canese, K., Chan, J., Comeau, D.C., Connor, R., Funk, K., Kelly, C., Kim, S., Madej, T., MArchler-Bauer, A., Lanczycki, C., Lathrop, S., Lu, Z., Thibaud-Nissen, F., Murphy, T., Phan, L., Skripchenko, Y., Tse, T., Wang, J., Williams, R., Trawick, B.W., Pruitt, K.D. and Sherry, S.T. (2022) Nucleic Acids Research 7(50):D20-D26.
- Sayyari, E., Whitfield, J.B., and Mirarab, S. (2017). Fragmentary gene sequences negatively impact gene tree and species tree reconstruction. Molecular Biology and Evolution 34:3279-3291.
- Seguinot, J., Rogozhina, I., Stroeven, A.P., Margold, M., and Kleman, J. (2016). Numerical simulations of the Cordilleran ice sheet through the last glacial cycle. The Cryosphere 10:639-664.
- Shah, T., Schneider, J.V., Zizka, G., Maurin, O., Baker, W., Forest, F., Brewer, G.E., Savolainen, V., Darbyshire, I., Larridon, I. (2021). Joining forces in Ochnaceae phylogenomics: a tale of two targeted sequencing probe kits. American Journal of Botany 108(7): 1201-1216.
- Sheffield, C.S., Richardson, L., Cannings, S., Ngo, H., Heron, J., and Williams, P.H. (2016). Biogeography and designatable units of *Bombus occidentalis* Green and *B. terricola* Kirby (Hymenoptera: Apidae) with implications for conservation status assessments. Journal of Insect Conservation 20:189-199.
- Simon, C. (2020). An evolving view of phylogenetic support. Systematic Biology 0(0):1- 8.
- Sims, C., and Palikhe, H. (2019). Proposed changes would increase the cost and decrease the benefit of listing species as endangered. Choices 34(2):1-10.
- Sun, C., Huang, J., Wang, Y., Zhao, X., Su, L., Thomas, G.W.C., Zhao, M., Zhang, X., Jungries, I., Kellis, M., Vicario, S., Sharakhov, I.V., Bondarenko, S.M., Hasselmann, M., Kim, C.N., Paten, B., Penso-Dolfin, L., Wang, L., Chang, Y., Gao, Q., Ma, L., Ma, L., Zhang, Z., Zhang, H., Zhang, H., Ruzzante, L., Robertson, H.M., Zhu, Y., Liu, Y., Yang, H., Ding, L., Wang, Q., Ma, D., Xu, W., Liang, C., Itgen, M.W., Mee, L., Cao, G., Zhang, Z., Sadd, B.M., HAhn, M.W., Schaack, S., Barribeau, S.M., Williams, P.H., Waterhouse, R.M., and Mueller, R.L. (2021). Genus-wide characterization of bumblebee genomes provides insights into their evolution and variation in ecological and behavioral traits. Molecular Biology and Evolution 38(2):486-501.
- Susko, E. and Roger, A.J. (2020). On the use of information criteria for model selection in phylogenetics. Molecular Biology and Evolution 37(2):549-562.
- Suzuki, Y., Glazko, G.V., and Nei, M. (2002). Overcredibility of molecular phylogenies obtained by Bayesian phylogenetics. Proceedings of the NAtional Academy of Science of the United States of America 99:16138-16143.
- Swaegers, J., Mergeay, J., Therry, L., Larmuseau, M.H.D., Bonte, D., and Stoks, R. (2013). Rapid range expansion increases genetic differentiation while causing limited reduction in genetic diversity in a damselfly. Heredity 111:422-429.
- Talavera, G. and Castresana, J. (2007). Improvement of phylogenies after removing divergent and ambiguously aligned blocks from protein sequence alignments. Systematic Biology 56:564-577.
- Taravers, G., Dincă, V., Vila, R. (2013). Factors affecting species delimitations with the GMYC model: insights from a butterfly survey. Methods in Ecology and Evolution 4(12):1101-1110.
- Tavaré, S. (1986). Some probabilistic and statistical problems in the analysis of DNA sequences. In *Some Mathematical Questions in Biology - DNA Sequence Analysis*, ed Miura, R.M. (Providence, RI: American Mathematics Society): 57-86.
- Thomson, R.C., and Shaffer, H.B. (2010). Sparse supermatrices for phylogenetic inference: taxonomy, alignment, rogue taxa, and the phylogeny of living turtles. Systematic Biology 59:42-58.
- Thorp, R.W. (2005). Species profile: *Bombus franklin*. In: Shepherd, M.D., Vaughan, D.M., and Balck, S.H. (edS). Red list of pollinator insects of North AMerica. CD-ROM Version 1. The Xerces Society for Invertebrate Conservation, Portland.
- Towes, D.P.L., and Brelsford, A. (2012). The biogeography of mitochondrial and nuclear discordance in animals. Molecular Ecology 21(16):3907-3930.
- Turcotte, A., Kermany, N., Foster, S., Proctor, C.A., Gilmour, S.m., Doria, M., Sebes, J., Whitton, J., Cooke, S.J., and Bennett, J.R. (2021). Fixing the Canadian *Species at Risk Act*: identifying major issues and recommendations for increasing accountability and efficiency. Facets 6(1):240-251.
- Waples, R.S., Kays, R., Fredrickson, R.J., Pacifici, K., and Mills, L.S. (2018). Is the red wolf a listable unit under the US Endangered Species Act? Journal of Heredity 109(5):585-597.
- Williams, P.H., Berezin, M.V., Cannings, S.G., Cederberg, B., Ødegaard, F., Rasmussen, C., Richardson, L.L., Rykken, J., Sheffield, C.S., Thanoosing, C., and Byvaltsev, A.M. (2019). The arctic and alpine bumblebees of the subgenus *Alpinobombus* revised from integrative assessment of species' gene coalescents and morphology (Hymenoptera, Apidae, *Bombus*). Zootaxa 4625(1):1-68.
- Williams, P.H., Brown, M.J.F., Carolan, J.C., An, J., Goulson, D., Aytekin, A.M., Best, L.R., Byvaltsev, A.M., Cederberg, B., Dawson, R., Huang, J., Ito, M., Monfared, A., Raina, R.H., Schmid-Hempel, P., Sheffield, C.S., Šima, P., and Xie, Z. (2012). Unveiling cryptic species of the bumblebee subgenus *Bombus s. str.* worldwide with COI barcodes (Hymenoptera: Apidae). Systematics and Biodiversity 10:21- 56.
- Williams, P.H., Altanchimeg, D., Byvaltsev, A., De Jonghe, R., Jaffar, S., Japoshvili, G., Kahono, S., Liang, H., Mei, M., Monfared, A., Nidup, T., Raina, R., Ren, Z., Thanoosing, C., Zhao, Y., and Orr, M.C. (2020). Widespread polytypic species or complexes of local species? Revising bumblebees of the subgenus *Melanobombus* world-wide (Hymenoptera, Apidae, *Bombus*). European Journal of Taxonomy 719:1-120.
- Williams, P.H. (2021). Not just cryptic, but a barcode bush: PTP re-analysis of global data for the bumblebee subgenus *Bombus s. str.* Supports additional species (Apideae, genus *Bombus*). Journal of Natural History 55:271-282.
- Wright, S., and Huxley, J. (1940). The statistical consequences of Mendelian heredity in relation to speciation. The new systematics. London Oxford Press pgd 161-183.
- Yang, Z. (1994). Maximum likelihood phylogenetic estimation from DNA sequences with variable rates over sites: approximate methods. Journal of Molecular Evolution 39:306-314.
- Yang, Z. and Rannala, B. (2010). Bayesian species delimitation using multilocus sequence data. Proceedings of the National Academy of Science of the United States of America 107(20):9264-9269.
- Yang, Z. (2015). The BPP program for species tree estimation and species delimitation. Current Zoology 61(5):854-865.
- Zhang J., Kapli, P., Pavlidis, P., and Stamatakis, A. (2013). A general species delimitation method with applications to phylogenetic placements. Bioinformatics 29(22):2869-2876.

Tables

Table 2.1. The species delimitation tests used on each dataset included in this study and whether or not the delimitation method identified *B. occidentalis occidentalis* and *B. occidentalis mckayi* as separate species. Speciation tests included Poisson Tree Process (PTP), multi-rate Poisson Tree Process (m-PTP), monophyly within the phylogenetic tree, generalized mixed Yule-coalescent model (GMYC), Automatic Barcode Gap Discovery (ABGD), Automated simultaneous analysis phylogenetics (ASAP), and Species bOundary Delimitation using ASTRAL (SODA). Asterisk indicates that the method identified each taxa as multiple species.

B. occidentalis and *B.*

Table 2.2. The number of potential species identified by Automatic Barcode Gap Discovery (ABGD) using the COI barcoding dataset with variable assigned values of P, the prior limit to the expected intraspecific diversity, and X, the minimum gap size between sequence clusters to identify a group (the sensitivity of the analysis).

0.001	7	7	7	7
0.001668	7	7	7	7
0.002783	5	5	5	5
0.004642	5	5	5	5
0.007743	5	5	4	4
0.012915	4	4	4	4
0.021544	1	1	NA	NA

P $X = 1$ $X = 1.5$ $X = 2$ $X = 2.5$

Table 2.3. The number of potential species identified by Automated Simultaneous Analysis Phylogenetics (ASAP) using the COI barcoding dataset. The ASAP score indicates the level of support for the grouping, p indicates the likelihood that the taxa are actually part of one large panmictic population, and W indicates the size of the diversity 'gap' between the current partition and the one before it.

Number of	ASAP		
Species	score	p (rank)	W (rank)
4	1	0.00157(1)	0.0000383(1)
5	3	0.733(4)	0.00366(2)
6	5.5	0.81(7)	0.000266(4)
7	5	0.747(5)	0.0000234(5)

Figures

Figure 2.1. The distribution of *Bombus occidentalis occidentalis* (green) and *B. occidentalis mckayi* (yellow) samples included in A) the UCE dataset and B) the COI barcoding dataset.

Figure 2.2. Estimate of the species-level maximum likelihood phylogeny based on 1683 UCE loci including *Bombus occidentalis* (green) and *B. mckayi* (gold). Branch lengths represent nucleotide substitutions. Vertical black bars represent the species assignments based on each delimitation method. Some delimitation methods represented here were were conducted on the COI barcode dataset.

Figure 2.3. Estimate of maximum likelihood phylogeny of the COI genes of five closely related species within the subgenus *Bombus sensu stricto*. Branch lengths represent nucleotide substitutions. Numbers at clade nodes represent bootstrap values. Species are color coded:*B. lucorum* (purple), *B. jacobsoni* (orange)*, B.hypocrita* (blue), *B. terricola* (black), *B. occidentalis* (green) and *B. mckayi* (gold). Names of taxa include the length of the barcode sequence, the institutional ID number and the species name, according to the Genbank record. Samples of *B. occidentalis* and *B. mckayi* also include sampling state or province.

Figure 2.4. Estimate of Bayesian phylogeny of the COI genes of five closely related species within the subgenus *Bombus sensu stricto*. Posterior probabilities of well supported clades are shown at the clade nodes. Species as defined by their GenBank records are color coded: *B. lucorum* (purple), *B. jacobsoni* (orange)*, B.hypocrita* (blue), *B. terricola* (black), *B. occidentalis* (green) and *B. mckayi* (gold). Names of taxa include the length of the barcode sequence, the institutional ID number and the species name. Samples of *B. occidentalis* and *B. mckayi* also include sampling state or province.

CHAPTER III

THE INFLUENCE OF GEOGRAPHY, DISPERSAL LIMITATION, AND ANTHROPOGENIC CHANGE ON THE POPULATION GENETIC CHARACTERISTICS OF TWO BUMBLE BEE SPECIES OF CONSERVATION CONCERN (BOMBUS OCCIDENTALIS AND BOMBUS MCKAYI) IN WESTERN NORTH AMERICA

Abstract

Bumble bees are often the dominant insect pollinator species in arctic and highelevation Nearctic ecosystems. *Bombus mckayi* and *B. occidentalis* are two species of montane bumble bees in western North America which were considered a single species until a recent revision of their status. Populations of *B. occidentalis* have been in decline, with decreasing abundance and range, since the mid 1990's, while populations of *B. mckayi* appear to have remained stable. Understanding patterns of population structure and isolation, as well as the environmental factors that drive them, is critical to aid ongoing efforts to provide federal, state, and provincial protections for these species. Here, we examine genetic structure and diversity in *B. mckayi* and *B. occidentalis*, treating them as separate species in a population genetic analysis for the first time*.* Patterns of genetic diversity and genetic structure in *B. mckayi* and *B. occidentalis* were measured using microsatellite markers and specimens collected between 1960 and 2020. Associations between genetic structure and potential environmental drivers, including weather, distribution, habitat change, and exposure to parasites, were tested using structural equation models. *Bombus occidentalis* showed significant but weak patterns of decreasing genetic diversity and increasing genetic structure through time. Detected

patterns of decline were not as strong for *B. mckayi*, but may indicate that this species is also at risk. Historical specimens indicate populations of the two species contained similar levels of allelic richness and structure (FST) prior to recent declines, though the patterns of gene flow across the landscape were not similar between the species. For both species, springtime minimum temperatures were the most important predictor of occupancy likelihood, and proximity to known infections of the fungal parasite *Vairimorpha bombi* was a reliable predictor of genetic differentiation (restricted gene flow). Although decreases in allelic diversity and increases in inbreeding and population structure have been documented in these species, substantial genetic diversity remains in extant populations relative to historical populations, which indicates a good opportunity for recovery of the species if the effects of the drivers of the declines are mitigated.

Introduction

Bees (Hymenoptera: Apoidea: Anthophila) pollinate more plant species than any other taxa of pollinators (Ollerton 2017), including approximately 75% of the world's food crops (Klein et al. 2007). There are nearly 20,000 known bee species worldwide (Michener 2000), and more yet to be described, especially in the Neotropics of Central and South America (Freitas et al. 2009) and in parts of Asia (Teichroew et al. 2017). However, bees worldwide are decreasing in abundance and range (Goulson et al. 2015). Bumble bees (*Bombus*) are among the most studied bee genera, largely due to their use as pollinators in agriculture, their relatively large size and characteristic appearance, and their high abundance throughout their distribution. Bumble bees are often the dominant pollinators in cold climate regions, especially early in the active season when nighttime temperatures are relatively low (Goulson 2003). Decreases in abundance and range have

been observed in bumble bee species around the world (Goulson et al. 2008; Colla et al. 2012; Cameron and Sadd 2020).

There are approximately 260 described bumble bee species worldwide, one third of which are under threat of extinction to some extent (Abertman et al. 2017). At least three bumble bee species in North America are decreasing in abundance and genetic diversity at alarming rates (Cameron et al. 2011; Colla et al. 2012; Abertman et al. 2017). In 2016, *Bombus affinis* was listed as endangered by the U.S. Fish and Wildlife Service through the Endangered Species Act (ESA). This was the first bumble bee species to be listed as endangered in the United States. *Bombus franklini* was most recently listed as endangered by the ESA in September of 2021 and may already be extinct (https://www.federalregister.gov, accessed 11:05 a.m., 10/11/2019). Finally, *Bombus occidentalis* is currently under review for listing as endangered through the ESA (https://ecos.fws.gov/ecp/, accessed 1:39 p.m., 8/13/2018), with a listing decision expected in 2023. Several additional species have been identified as endangered or at risk of decline by the Species at Risk Act (SARA) and the International Union for Conservation of Nature and Natural Resources (IUCN, www.iucnredlist.org, accessed 11:36 a.m., 8/13/18; COSEWIC 2014; Table 3.1).

Bumble bees are often the dominant insect pollinator species in arctic and high elevation Nearctic ecosystems because they have hairy and robust bodies and they perform a buzzing behavior that vibrates flight muscles to produce heat. These adaptations allow them to tolerate colder temperatures than many other insect species (Heinrich and Kammer 1973; Goulson 2010). Recent distributional changes may limit population sizes and gene flow among populations, especially at the relatively low

elevation portions of their ranges. Distributions of some European bumble bee species have retracted to high-elevation and high-latitude portions of their historical ranges due to changes in land use and climate (Ploquin et al. 2013). The status of North American bumble bees is less well studied than their European counterparts. However, (low) elevational barriers have been shown to limit or direct gene flow in some North American species (Jackson et al. 2018; Koch et al. 2018), indicating that they may be most vulnerable to reductions in abundance and gene flow at the relatively low-elevation portions of their ranges, similar to the patterns found in European species.

A species of particular conservation concern in North America is *Bombus occidentalis*. *B. occidentalis* was once abundant in western North America, with a range that extended from Alaska in the north to New Mexico in the south and as far east as western Nebraska and the Black Hills of South Dakota (Rao and Stephen 2007; Williams et al. 2014). This species was and is an important pollinator in high elevation and high latitude ecosystems in the region. Comparisons of early museum records and studies from before 1997 against recent museum records and collections indicate that populations of *B. occidentalis* have declined dramatically along the west coast and in the Rocky Mountains and Intermountain West since the mid 1990's (Evans et al. 2008; Cameron et al. 2011; Graves et al. 2020). Although a pattern of decrease in abundance for *B. occidentalis* was clearly demonstrated, the cause of the decline remains uncertain (Cameron et al. 2016) and new studies are required to determine if the decline is ongoing.

In addition to declines in abundance and range, the species status of *B. occidentalis* has recently changed. Two morphologically identified subspecies, *B. occidentalis mckayi* and *B. occidentalis occidentalis* have been supported as distinct clades and species in recent analyses of the mitochondrial *cytochrome oxidase I* (COI) barcode region (Williams et al 2012; Williams 2021) and nuclear ultraconserved elements (UCEs). These taxa (hereafter *B. occidentalis* and *B. mckayi*) are geographically separated within the historical range, with *B. mckayi* dominant north of 55[°] latitude and *B. occidentalis* dominant south of 55[°] latitude. There is a region of likely overlap in distribution between the species (possibly a hybrid zone?) near the boundary where differentiating species has proven challenging using morphology (Williams et al. 2012; Sheffield et al 2016; McHugh and Sikes 2016; Williams 2021, Fig. 3.1). Phylogenies based on mitochondrial and nuclear genes clearly group *B. mckayi* specimens into a monophyletic clade, but sampling in the overlapping portions of the ranges is poor (Williams et al. 2012; Williams 2021). The two taxa are likely to be treated as separate species in future conservation decisions (Williams 2021), so we treat each taxon separately throughout the rest of this manuscript. Although there is ample evidence of decreases in population abundance and range within the southern species, *B. occidentalis* (Evans et al. 2008; Cameron et al. 2011; Graves et al. 2020), previous studies indicate that populations and the distribution of the northern species, *B. mckayi*, have remained stable (Koch and Strange 2012; Pampell et al. 2015). However, several years have passed since these data were collected and there has been a call for additional study of the conservation status of both species (McHugh and Sikes 2016; personal correspondence with Jeffery Everett, U.S. Fish and Wildlife Service) to update knowledge of their current distributions, densities, and genetic resiliency.

 In assessing conservation status of *Bombus* species, genetic data are an important adjunct to location records. While patterns of occupancy may indicate where gene flow

barriers exist, these patterns are insufficient to predict the causes of gene flow barriers (Roffler et al. 2016). The relationship between occupancy and gene flow could be particularly messy for bumble bees because of their eusocial life history, which dictates that most individuals in the census populations are not reproductive, so they do not contribute directly to gene movement across the landscape, although they are vital to the survival of their colonies. Reproductive individuals require different resources and are active at different times during the season from non-reproductives. The higher likelihood of observing sterile workers skews maximum entropy models to identify sites that are adequate or inadequate for colony establishment, rather than gene flow. Therefore, measures of occupancy alone are not enough to determine if gene flow is restricted.

Landscape genetics is an interdisciplinary field that combines aspects of population genetics, landscape ecology, and spatial statistics to measure genetic discontinuities and diversity patterns across landscapes and to correlate them with environmental features (Manel et al. 2003; Storfer et al. 2007). This field of study has applications in many aspects of evolution and ecology. Applications in conservation ecology include determinations of population boundaries (Safner et al. 2011), estimates of dispersal ability (Cayuela et al. 2018), estimates of population structure (Jost et al. 2018; Masuda 2018), genetic rescue of high risk and inbred populations (Whiteley et al. 2015; Forsman 2014; Frankham 2015; Hendrick and Garcia-Dorado 2016; Ralls et al. 2017), estimation of metapopulation dynamics (Hand et al. 2015; Salisbury et al. 2016; Hanski et al. 2017), and measurements of the temporal and spatial scale of evolutionary processes (Ellegren and Galtier 2016). Accurate estimates of gene flow and genetic

structure within and among populations are the basic tools needed for land managers to use landscape genetics methods for any of these applications.

Landscape genetic techniques can provide insight into questions about potential threats to bumble bees and identify actions that can be taken to protect populations. Measurements of population structure, genetic diversity, and gene flow among populations are important indicators of current conservation status for species. Landscape genetic studies that measure environmental variables as well as gene flow and genetic structure and diversity can indicate which environmental changes have negative effects on bumble bees. Interest in this work has been piqued in the United States in recent years, especially since the Pollinator Partnership Action Plan (2016) was released by the federal Pollinator Health Task Force, as mandated by President Barack Obama via a presidential memorandum. This plan has increased awareness of decreases in pollinator abundances and ranges and encouraged interest in research into pollinators to inform environmental policy and management.

Here we use observation records, microsatellite genetic data from museum specimens, and spatial environmental data to predict current and past genetic structure, genetic diversity, and gene flow patterns in *B. occidentalis* and *B. mckayi*. We used these data to test two hypotheses: (1) genetic structure among populations has increased and genetic diversity within populations has decreased for *Bombus occidentalis* over time. These population characteristics have remained relatively stable for *Bombus mckayi*, (2) changes in environmental drivers, such as climate, habitat fragmentation, and increased parasite pressures drive genetic diversity and structure in *Bombus occidentalis* and *Bombus mckayi*. We used specimens captured from across the historical range of the

species to measure genetic structure within and among populations and genetic distance among individuals. We used maximum entropy modeling to build resistance landscapes using measured environmental variables and observation records across the range of the species. We used genotypes from museum specimens to estimate gene flow across the resistance landscapes. We used structural equation modeling to quantify the relative influences of the measured environmental variables included in the resistance landscapes on the genetic distances among the genotyped individuals (Wang et al. 2013). We found significant but weak patterns of decreasing genetic diversity and increasing genetic structure in *Bombus occidentalis* through time. Detected patterns of decline were not statistically significant for *B. mckayi*, but may indicate that this species is also at risk. Historical specimens indicate populations of the two species contained similar levels of allelic richness and structure (FST) prior to recent declines, though the patterns of gene flow across the landscape were not similar between the species. For both species, springtime minimum temperatures were the most important predictor of occupancy likelihood, and proximity to known infections of the fungal parasite *Vairimorpha bombi* was a reliable predictor of genetic differentiation (restricted gene flow).

Methods

Study area

Samples used in this study extended across the full extent of the historical range of the target species, *Bombus occidentalis* Greene including *Bombus mckayi* (Williams et al. 2014; Hatfield et al. 2015, Fig. 3.1). This broad area includes two mountain ranges, over 8,000 miles of coastline, and intermountain habitat across 35 degrees of latitude.

Acquisition and management of tissue samples

Bombus occidentalis and *Bombus mckayi* tissue samples were obtained from field-captured bumble bees and museum specimens. Specimens were provided by the U.S National Pollinating Insect Collection, the Royal Museum of British Columbia, the University of Alaska Museum of the North, the University of Alberta E.H. Strickland Entomological Museum, the University of Calgary Zoology Museum, Montana State University Entomology Collection, the Canadian National Collection, The University of California Berkeley Essig Museum of Entomology, the University of Alberta, the U.S. Forest Service, and the U.S. Geological Survey.

Mid-legs of specimens collected in the field specifically for this study (not museum specimens) were frozen and stored at the U.S. Department of Agriculture, Agricultural Research Service Pollinating Insect Research Unit (PIRU). Extracted DNA was frozen at -80^oC and stored on-site at PIRU. Specimens owned by each of the respective collections were assigned unique identifiers by those institutions and are permanently stored in those collections.

DNA extraction and microsatellite amplification

For DNA extraction, one mid-leg or two fore-legs were removed from each specimen and placed into a single well of a 96-well plate. DNA was extracted from tissue samples using a Chelex(R) extraction method (Strange et al. 2009): 150 μ L of 5% Chelex® solution was added to each well with 5 μ L of Proteinase K solution (10 mg/mL) and incubated in a thermocycler at 55^oC for 60 minutes, 99^oC for 15 minutes, 37°C for 1 minute, 99°C for 15 minutes, and cooled to 15°C.

Fifteen microsatellite loci were targeted for amplification using fluorescently tagged primers and amplified using PCR as described by Koch et al. (2017). The loci were identified from the literature (Table 3.2). PCR amplifications were performed in a thermocycler in two multiplex reactions (plex A and plex B), determined by the melting points of the primers (Table 3.2). The heating and cooling cycle for plex A was 95° C for 3 minutes 30 seconds for the initial denaturation, followed by 30 cycles of 95 °C for 30 seconds, 55^oC for 1 minute 15 seconds, and 72^oC for 45 seconds for replication, followed by 72° C for fifteen minutes for final extension before a final chill at 15 $^{\circ}$ C. The heating and cooling cycle for plex B was the same as plex A, except the $55\textdegree C$ step in replication was adjusted to 58^oC. Fragment sizes were measured through capillary electrophoresis using an ABI PRISMTM 3730 DNA Analyzer. Fragment sizes were scored using Geneious v. 7.1. Specimens that successfully amplified seven or more loci were included in downstream analyses.

Bombus occidentalis and Bombus mckayi location records

Location records for *B. occidentalis* from 1960 to 2020 were downloaded from the Global Biodiversity Information Facility (GBIF, September 8, 2020, 2:23 pm). Records were composed of human observations ($N = 543$), material records ($N = 123$), and museum specimens $(N = 17,895)$. Erroneous identification is unlikely from material records and museum specimens, as these records were verified by experts and samples are retained in institutions. The records from in-situ human observations, however, are more likely to be erroneous. The white setae on the terminal abdominal terga of *B. occidentalis* are an easily identifiable and diagnostic character for the species throughout most of its geographic range because it is the only species with this characteristic.

However, portions of the geographic range that overlap with the range of other bumble bees with white setae on the terminal abdominal terga (i.e. *Bombus terricola* in the southeastern portion of the range and *Bombus cryptarum* in the northern portion of the range) could contain records that were mis-identified. These records represent a very small portion of the records for this species $(N < 100)$ and if they are incorrectly identified, they are likely to have a negligible influence on the outcome of our analyses. These records were named prior to the recent phylogenetic work that indicates the taxa is actually two species (Williams 2021, Chapter 1) and consist of both *B. occidentalis* and *B. mckayi*, without differentiation between the taxa. Therefore, all specimens collected north of 55[°] latitude (per the likely range boundary identified by Sheffield et al. 2016) were treated as *B. mckayi* and all specimens collected south of 55[°] latitude were treated as *B. occidentalis.* Only records associated with geographic coordinates were included for these analyses.

Population genetic analyses

I. Definition of populations

The *Buffer* tool in ArcGIS Pro was used to draw a 10 km buffer around the collection location of each genotyped specimen, to define an area slightly larger than the largest expected foraging range for that specimen (Osborne et al. 2001; Westphal et al. 2006; Greenleaf et al. 2007). All specimens that had overlapping buffers were grouped and these groups were defined as geographic clusters for downstream analyses (Table B1). The probability of sibship was analyzed within the geographic clusters in Colony version 2.0.6.7 (Jones and Wang 2010). Sibship analyses were run twice, with different initiation seed numbers (1234 and 4321). All but one specimen from each sibling set

identified by Colony with a probability higher than 0.7 were removed for all downstream analyses, because multiple siblings of eusocial insects in a dataset can lead to underestimates of allelic richness and overestimates of structure at the population level (Table B2). Structure version 2.3.4 (Pritchard et al. 2000) was used to identify genetic structure within each of the species. Admixture models were used and included the geographic clusters as location priors to identify lineages within the species. Location priors in Structure are defined by integers representing groups of samples that were collected at geographically close locations. In this case, where the true number of populations is unknown, the geographic cluster definitions are biologically meaningful. Location priors are useful in analyses when there is population structure in the dataset, but the signal is too weak for standard Structure models to detect (Hubisz et al. 2009). Inclusion of location priors does not tend to cause Structure to find population structure where none is present and, in datasets where population structure is strong, it does not influence population assignments (Hubisz et al. 2009). Initial values for λ (a measure of the independence of markers from one another) and α (the relative admixture levels among populations) were left at the recommended values ($\lambda = 1.0$ and $\alpha = 1.0$) for all Structure analyses. Thirty replicates were run (Porras-Hurtado et al. 2013) for every value from $K = 1$ through the total number of geographic clusters identified for each species, plus ten (*B. mckayi* = 51 and *B. occidentalis* = 90), per the recommendation of Pritchard et al (2009). The initial burn-in length and run length was set to 200,000 and the convergence of summary statistics included in the output of the initial analyses was verified to confirm that the burn-in was long enough to reach convergence.

Structure_threader version 1.3 (Pina-Martins et al. 2017) was used to parallelize the computation of the replicate runs of Structure. Structure_threader also wraps Structure Harvester version 6.94 (Earl and vonHoldt 2012) which automates the summarization of the data and conducts analyses to choose the most likely value of K. The most likely values of K for each dataset were selected based on the output values of Delta K with guidance provided by Porras-Hurtado et al. (2013). Structure Harvester also reformats the data from the Structure output files to Clumpp (Jakobsson and Rosenberg 2007) input files for each value of K. Clumpp version 1.1.2b was used to permute the 30 replicate runs of each of the values of K that were most likely to be real (as determined by the analyses from Structure Harvester), making the values from each replicate run as similar as possible for each value of K, and to derive the median values into a single output matrix. The *Greedy* algorithm was used for values of $K = 15$ and smaller and the *LargeKGreedy* algorithm for values of K larger than 15. These analyses were conducted using the Ceres high performance computing cluster, part of USDA-ARS SCINet (https://scinet.usda.gov)

II. Population structure analyses

GENEPOP version 4.7 was used to test for Hardy-Weinberg equilibrium (HWE) and pairwise genotypic disequilibrium (exact G test) for all loci in all populations. The exact G test tests the null hypothesis that "genotypes are drawn from the same distribution in all populations" (Raymond and Rousset 1995). Rejection of this null hypothesis is an indication of genotypic differentiation among the populations. HP-Rare version 1.0 was used to estimate mean allelic richness across all loci for each population (Kalinowski 2005). The mean allelic richness from all measured loci is often used to

estimate the amount of genetic diversity within populations. However, comparisons of estimates of allelic richness are sensitive to differences in sample size among populations (El Mousadik and Petit 1996; Leberg 2002; Foulley and Ollivier 2006), which are exaggerated in this study due to our dependence on availability of museum specimens (Table B1). HP-Rare uses rarefaction to estimate true allelic diversity in populations with uneven population sizes. Geographic clusters were standardized to 10 individuals for rarefaction. FSTAT version 2.9.4 (Goudet 1995) was used to perform analyses to measure the amount of inbreeding (F_{IS}) within populations. A one-sample t-test was used to identify populations with significantly higher than expected F_{IS} values (Callahan et al. 2013). Pairwise genetic structure (FST) was measured among populations that were sampled at similar times (within six years, five time groups total) and global F_{ST} values were estimated within groups of temporally similar populations using GENEPOP.

An analysis of molecular variance (AMOVA) was performed using the poppr.amova function in the *poppr* package version 2.9.3 (Kamvar et al. 2014) in R to measure the extent of genetic differentiation among geographic clusters and among samples within geographic clusters for both species. Because the populations were sampled in different years, the pairwise genetic differentiation matrix (Nei's genetic distance, Nei 1972) used in the analysis was weighted by subtracting a pairwise matrix of the average number of mutations expected at microsatellite loci $(10^{-4}$ per generation, Estoup and Angers 1998) times the number of loci (15) times the number of years between the pair of populations times two, because there are two populations in each comparison (equation 1).

adjusted genetic distance = (calculated genetic distance $* 0.001$) $* 15 *$ pairwise difference in years $* 2$ [1]

This adjustment estimates the genetic differentiation among samples not accounted for by time since sampling if the loci mutate at the predicted average rate for microsatellite markers. The same adjusted genetic differentiation matrix was used in a Mantel test to measure the relationship between genetic distance and geographical distance of the populations.

Environmental data for occupancy and resistance modeling

I. Elevation

The publicly available world elevation GMTED raster was downloaded from the U.S. Geological Survey (Danielsen and Gesch 2011) through the Living Atlas of the World feature of ArcGIS Pro (version 2.8.2). This raster has a resolution of 250 $m²$ and was used for maximum entropy modeling and structural equation modeling (see below). Additionally, least cost distances were calculated using the R package *leastcostpath* (Lewis 2021) which uses a DEM (digital elevation model) in addition to a continuous landscape resistance raster (McRae 2007; Spear et al. 2010) in this case generated by MaxEnt, see below) to estimate least-cost resistance distance.

II. Weather

Monthly modeled data of maximum, minimum, and mean temperatures and precipitation from across the historical range of *B. occidentalis* from 1960 through 2019 were gathered from PRISM (PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, created 24 August 2021) using the ClimateNA tool (Wang et al. 2016b). The elevation raster described above was resampled to 10 km^2 cells using

the *Resample* tool in ArcGIS Pro version 2.4 and was clipped to the extent of the species range (Hatfield et al. 2015) using the *Clip Raster* tool in ArcGIS Pro. The resulting raster was converted to a table including columns for the latitude, longitude, and cell values using the *Generate Table from Raster Function* tool and the table was exported as a CSV file. This CSV was used as an input for the ClimateNA tool to calculate the monthly weather variables. The results from the ClimateNA tool were exported as one large CSV file including columns for each modeled weather variable at each of the locations (same latitude and longitude values as the input CSV generated from the elevation DEM) during each month of each year. R for Windows version 4.1.1 and R Studio version 1.3.1 were used to extract the data generated for months included in the active season (April through September) and to create monthly tables of maximum temperature, minimum temperature, average (mean) temperature, and precipitation for each location during these months for each year. Additionally, tables of means and standard deviations for all years (1960-2019), approximate years prior to the measured decline in abundance of *B. occidentalis* (1960-1994), and years since the decline (1995-2019) by each month of the active season were created. These tables were exported as CSV files to ArcGIS Pro where the *XY Table to Points* tool and the *Point To Raster* tool were used to create new rasters with the same cell size and orientation for each of the datasets (monthly for each year separately, mean for the entire time frame, standard deviation for the entire time frame) for use with the software MaxEnt (version 3.4.1) in downstream analyses.

III. Land cover (Habitat Quality)

Annual land cover data from 1992 to 2015 were modeled at 300 m^2 resolution across the globe by the European Space Agency (ESA) Climate Change Initiative (Wei et al. 2018) with annual updates at https://www.esa-land cover-cci.org/?q=node/197) and classed into 22 land cover types. Reliable land cover data from across the entire range of this species was not available before 1992. Therefore, analyses for these models do not include land cover data from before 1992. Land cover data was downloaded for each year individually and clipped to the extent of the historical *B. occidentalis* and *B. mckayi* ranges using the *Extract by Mask* tool in ArcGIS Pro.

Land cover data was converted to estimated habitat quality. The *Calculate Combined Table of Raster Values that Intersect Points* tool (Welty et al. 2021) was used with the point locations of all known records of *B. occidentalis* and *B. mckayi* from 1992 to 2015 (GBIF) to extract the land cover types where each record was collected. A weighted rank of habitat quality was calculated by dividing the percentage of the *B. occidentalis* and *B. mckayi* observations that were found within each land cover type by the number of specimens counted in that land cover type. The weighted habitat quality values were finally rescaled from 1 to 100 using the *rescale* function in R. Any Land cover types where *B. occidentalis* or *B. mckayi* have never been collected were assigned a value of zero (Table B2). The rangewide annual land cover layers were reclassified to the weighted and scaled habitat quality values using the *Reclassify* tool in ArcGIS Pro (values were rounded to whole integers). The *Resample Raster* tool was used to resample the resulting rasters from their native cell size of 300 m^2 to 10 km², so they would match the resolution of the elevation and weather rasters. Finally, the *Raster Calculator* tool was used to calculate the change in habitat quality from 1992 to 2015.

IV. Distance to cropland

Data on the density of honey bee hives and pesticide use across the landscape are difficult to obtain and often unreliable from year to year. As a surrogate for these environmental factors, we measured the distance from the collection locations of the genotyped specimens to the nearest croplands, where domesticated bees and pesticides are most likely to be used. Raster cells that represented croplands were extracted from the annual ESA land cover rasters using the *Extract by Attributes* tool and converted to polygons with the *Raster to Polygon* tool in ArcGIS Pro. Euclidean distance between sampling points and the nearest croplands was calculated using the *Near* tool in ArcGIS Pro. As with habitat quality, data was only available for this variable from 1992 to 2015.

Environmental data for structural equation modeling

I. Environmental predictors

The *Calculate Combined Table of Raster Values that Intersect Points* tool (Welty et al. 2021) was used to extract mean, minimum, maximum, and standard deviation temperature and precipitation information from the monthly PRISM rasters described above for the precise collection locations and times of each of the genotyped specimens. Additionally, the habitat quality values, elevations, and latitudes of each specimen's sampling location were extracted from the appropriate rasters using the *Extract Values to Points* tool in ArcGIS Pro.

II. Parasitism

Data on the presence of *Vairimorpha bombi* (*V. bombi*, Tokarev et al. 2020) were provided by bumble bee researchers from published and unpublished datasets (Cameron et al. 2011; Strange and Tripodi, unpublished data). The species was recently moved from

the genus *Nosema* into *Vairimorpha*, and previously published literature refers to it as *Nosema bombi*. *Vairimorpha bombi* is an important species of internal parasite that causes lethargy and death in non-reproductive bumble bees, and reduced fecundity in reproductive castes of bumble bees throughout the range (Gegear et al. 2006; Cameron et al. 2011). *V. bombi* is native in the range of *B. occidentalis*, but frequency of infections in wild-caught *B. occidentalis* specimens increased as population density decreased through time, indicating at least a negative correlation between infection rates and population density (Cameron et al. 2011), and possibly a causative one. Euclidean distance between genotyped bumble bee samples and nearest known *V. bombi* infections from that same year or before was calculated using the *Near* tool in ArcGIS Pro. Any *V. bombi* infections that were detected in specimens collected prior to the sampling year of the genotyped specimens were assumed to be part of a persistent population, and were therefore retained in analyses of bees collected in subsequent years. This sampling scheme is not systematic and likely underestimates the prevalence of *V. bombi* throughout the range of *B. occidentalis* and *B. mckayi*. However, this dataset represents the most complete distribution information currently available for these parasites.

Landscape genetic analyses

I. Genetic distance matrices

The R function *nei.dist* in the package *poppr* (Kamvar et al. 2014) was used to calculate pairwise Nei's standardized genetic distances among sampled individuals from across the species ranges. This statistic describes the pairwise number of nucleotide substitutions between two specimens and is measured on a scale of 0 to 1.

II. Resistance rasters

The elevation, weather, and land cover rasters described above were used along with the location records to create maximum entropy models for both species in the software MaxEnt (version 3.4.0). Twenty-five percent of the data was used as training sets and 75% of the data as the test sets with a logistic output (cell values represent likelihood of occupancy and range from 0 to 1). The monthly and annual weather predictors were likely highly correlated, but these variables were not removed from the analyses because recent empirical and simulation studies found that collinearity (correlation) among predictor variables in training datasets does not influence the outcome of MaxEnt models (De Marco et. al 2018; Feng et al. 2019). However, these studies did indicate an influence of collinearity shift, in which the relationships among predictor variables within geographically separated training and testing datasets were not similar. The training data for each model in this study was taken from across the entire range of the test dataset. Therefore, collinearity shift among predictor correlations for the training and test datasets is not a concern. Most historical records of *B. occidentalis* do not differentiate between *B. mckayi* and *B. occidentalis*, so we were dependent on the ranges described in previously published literature to estimate the likely species identification of the GBIF observation records. The continuous occupancy likelihood raster produced by MaxEnt was inverted using the *Raster Calculator* tool in ArcGIS Pro to estimate a resistance landscape, as described by Wang et al. (2013).

III. *Cost-distance matrice*s

Cost-distance measurements describe the cost to organisms of moving across a landscape, and have been shown to represent the influence of distance on organisms'

movements better than Euclidean distance (McRae 2006; Graves et al. 2014). The *create FETE lcps* function in the R package *leastcostpath* was used with the resistance rasters to create least-cost distance matrices. *Circuitscape* version 4.0.5 (McRae et al. 2013) was used with the resistance rasters to create input files that were used in *Circuitscape.jl* version 0.27 (Anantharman et al. 2020) to calculate pairwise random walk cost-distance matrices. Least-cost and random walk distance measurements mimic the paths traveled by an individual that seeks the route of least resistance across the landscape and an individual that is directed by stochastic movements across the landscape, respectively. These distance measurements represent conservative and liberal estimates of the influence of environmental resistance on the movement of individuals (Marrotte and Bowman 2017).

IV. Structural equation modeling

Structural equation modeling (SEM) can be used to differentiate between the effects of multiple predictor variables on response variables within a complex system using a combination of confirmatory factor analysis (CFA, measurement mode) and path analysis (structural model) to improve the fit of multiple regression models (Grace 2006; Bauer and Curran 2020). An advantage of CFA is that it uses latent variables, which are the variables of interest but are not measured directly. Latent variables are informed using one or more observed variables (Grace 2006). An advantage of path analysis is that it allows the effects of some variables to be mediated by others, which improves the fit of the model to complex hypotheses. Four SEMs were created. Two models measured the relative influences of resistance distance and the measured environmental variables on the pairwise genetic distances among the *B. occidentalis* and the *B. mckayi* samples from

across the entire sampling period. Two additional models used the same predictor variables among samples in subsets of *B. occidentalis* from 1960 to 1994 and 1995 to 2020 to compare the relative influences of the predictor variables before and after the initial observed decrease in abundance and range in the species (Cameron et al. 2011).

SEMs must be overidentified, meaning that there are more variables with known values than variables with unknown values (Grace 2006; Bauer and Curran 2020). In models such as ours, which contain both latent variables and directed paths (hybrid models), the measured model and the structural model can be tested for identification separately. If the two components of the model are identified, the model is identified. The models were estimated using a maximum likelihood estimator in the *Lavaan* package (Yves 2012). Maximum likelihood estimation of SEMs assumes a continuous normal distribution of the endogenous variables. Measurements of distance from *V. bombi* infections, distances from agricultural lands, and latitude of the samples' collection locations were not normally distributed. However, our response variable is the pairwise genetic differentiation among samples, so the predictors had to be pairwise comparisons among samples also (Wang et al. 2013). The pairwise differences of the values of all of the predictor variables except for distance from *V. bombi* infections and distance from agricultural lands were calculated. For distance from *V. bombi* infections and agricultural lands, pairwise sums were used instead of differences. This way, pairs of two specimens that were both far from *V. bomi* infections or agricultural lands had a higher score than pairs with one specimen that was far but one that was close, or two specimens that were close. The values of these pairwise comparisons were normally distributed. Our predictor variables were measured on different scales, which sometimes produced absolute values

that were orders of magnitude different from each other. In order to allow the model to measure differences in variance among the variables, the measured units were adjusted to reduce the disparity between absolute values among variables (Table 3.3) and the standardized regression relationships were reported. This means that each regression value reported indicates the number of units of change in the response variable (genetic differentiation) per (adjusted) unit of change in the predictor variable, under the conditions described by the model.

Direct and indirect regression values were standardized, so the values indicate one unit of change in the predictor resulted in the reported number of units of change in the response. Only regressions for which the response is Nei's genetic differentiation are reported (all other relationships are available in the supplement), so the values are in units of Nei's genetic differentiation (0 to 1, hereafter D). We evaluated the fit of the models using a mix of absolute and relative goodness-of-fit tests, following the recommendations of Bauer and Curran (2020) for fit requirements. To define a model fit as good, we required a non-significant p value from a Chi^2 test, values of 0.9 or higher from the comparative fit index (CFI) and Tucker-Lewis index (TLI), and values less than 0.08 from the root mean square error of approximation (RMSEA) and the standardized root mean square residual (SRMR). We used inferential tests of indirect effects with bootstrapped confidence intervals to infer mediation effects. Significant links in mediational pathways is not sufficient to infer the whole relationship (Bauer and Curran 2020). We used 1000 bootstrapped samples and calculated bias-corrected confidence intervals to infer mediation effects.

Results

Microsatellite amplification and group assignments

We genotyped 1790 *B. mckayi* specimens and 1541 *B. occidentalis* specimens from across their ranges. The number of loci that amplified varied considerably among specimens and was significantly negatively correlated with the age of the specimens in both taxa (*B. occidentalis* t = 6.8, *p* < 0.0001, *B. mckayi* t = 4.5, *p* <0.0001, Fig. 3.2). Three hundred ninety-two specimens of *B. occidentalis* collected between 1963 and 2020 and 568 specimens of *B. mckayi* collected between 1967 and 2019 amplified seven or more loci.

We analyzed 99 geographic clusters of *B. occidentalis* and 55 geographic clusters (representing possible populations, hereafter clusters) of *B. mckayi* specimens with overlapping foraging ranges for sibship. Sample sizes for each cluster (treated as discrete populations) of *B. occidentalis* ranged from one to 45 and *B. mckayi* ranged from 1 to 93. Obviously, we could not perform population genetic analyses on clusters represented by a single specimen, but the mean number of specimens in the clusters that were included in those analyses was 6.1 (standard error = 0.9 , N = 50) for *B*. *occidentalis* and 14.6 (standard error $= 2.4$, $N = 37$) for *B. mckayi*. Within clusters, we identified seven likely *B*. *occidentalis* and four likely *B. mckayi* sibling sets (Table B1). We removed 37 *B. occidentalis* specimens and nine *B. mckayi* specimens from downstream analyses.

Population genetic analyses

We detected significant genotypic disequilibrium (from the G-test) across populations in every locus in the analysis of *B. mckayi* and in every locus except two in *B. occidentalis* (BTERN02 and BTMS0083). We detected significant divergence from expected heterozygosity (Hardy Weinberg Equilibrium, HWE) across populations at every locus for both species. These rangewide differences indicate structure among the sampled clusters. However, average heterozygosity within clusters diverged from HWE in only one population of *B. occidentalis*, which had a measured excess of heterozygosity (sampled in 2018). The cluster was represented by two individuals (Fig. 3.1) and this result is likely due to sampling error associated with the small sample size. No *B. mckayi* populations diverged from HWE. Mean rarefied allelic richness for each locus ranged from 1.62 to 2.41 in *B. occidentalis* and 2.23 to 6.27 in *B. mckayi* (Table 3.4). Rarefied mean allelic richness for all loci in each population of *B. occidentalis* ranged from 2.27 to 43.13 (mean = 16.88, standard error = 1.70) and in *B. mckayi* ranged from 1.67 to 37.93 (mean $= 17.79$, standard error $= 2.40$, Fig. 3.3). Linear regressions of rarefied allelic richness through time indicated a significant decline in allelic richness in *B. occidentalis* $(F_{1,96} = 5.95, p = 0.02)$, and a non-significant trend of decline in *B. mckayi* ($F_{1,52} = 2.67, p$ $= 0.11$). Linear regressions of subsets of the datasets from 1960 to 1994 and from 1995 to 2020 were not significant for *B. occidentalis* (before: $F_{1,24} = 0.58$, $p = 0.45$, after: $F_{1,70} =$ 0.16 *p* = 0.73). However, *B. mckayi* showed a slight significant increase in allelic richness from 1995 to 2020 (before: $F_{1,3} = 2.20$, $p = 0.23$, after: $F_{1,47} = 5.36$ $p = 0.02$). Despite statistically significant relationships, the R^2 values for these relationships indicate that sampling year was a poor predictor of allelic richness.

We identified two as the most likely number of lineages in the Structure analysis of *B. mckayi* (Fig. 3.4). The distribution of these specimens across Alaska and northern Canada indicate an eastern lineage and a western lineage with some overlap in central

and eastern Alaska. Similarly to *B. mckayi*, analyses of *B. occidentalis* specimens indicated that two lineages were most likely (Figure S1). Unlike *B. mckayi*, the members of these two lineages were more geographically mixed and this dataset contained many samples that had mixed ancestry from both lineages (Fig. 3.4). Although the two lineages identified for *B. occidentalis* were quite mixed, geographic groups of clusters dominated by lineage two (Fig. 3.4) are discernable in Wyoming and in southern British Columbia. The strongest support was for two lineages within each taxon, however there was some support for eight lineages within *B. mckayi* and for nine lineages within *B. occidentalis* (Figure B1). This result may indicate that there is hierarchical population structure within the lineages.

AMOVA analyses indicated significant partitioning of genetic variation among geographic clusters for *B. occidentalis* and *B. mckayi* (Table 3.5). Mantel tests indicated a weak but significant positive correlation between Euclidean and genetic distance in *B. occidentalis* ($r = 0.07$, $p = 0.01$) and a stronger relationship in *B. mckayi* ($r = 0.23$, $p =$ 0.0002). The Mantel test using the corrected genetic distance matrices slightly weakened the relationships in both species, but remained significant (*B. occidentalis:* $r = 0.05$, $p =$ 0.04, *B. mckayi:* $r = 0.19$, $p = 0.002$).

Single-tailed one sample t-tests of F_{IS} values indicated that some clusters in both species had significantly higher than expected levels of inbreeding (Fig. 3.5 a and b). Single-tailed one-sample t-tests of global Fs_T values for groups of clusters sampled in similar years indicated that some clusters had higher than expected genetic structure. However, linear regressions of neither species indicated a significant increase in F_{ST} over time (Fig. 3.5 c and d), and linear regression indicated no significant relationship between sampling year and relatively high Fst $(B. \text{ occidentalis: } t = 0.367, p = 0.738, B. \text{ mckavi: } t$ $= 1.525, p = 0.225$, though non-significant trends are observable in the data.

Landscape resistance

We included 10,814 GBIF records of *B. occidentalis* (including *B. mckayi*) observations and collections in the MaxEnt analysis. Maximum entropy produced good fitting models (*B. occidentalis* AUC = 0.857, standard deviation = 0.008, *B. mckayi* AUC $= 0.948$, standard deviation $= 0.008$). Resistance rasters generated from the maximum entropy models indicated that *B. occidentalis* was historically most common along the west coast and in the intermountain west, mostly excluding the Great Basin and Mojave desert, which are arid relative to the rest of the region. *B. mckayi* was most likely to be detected in two general clumps, one in eastern Alaska and one in northern British Columbia and southern Yukon Territory (Fig. 3.6).

The likelihoods of occupancy of *B. occidentalis* and *B. mckayi* across their ranges were most strongly predicted by different variables (Table 3.6). Springtime precipitation and variability in springtime temperatures were strongly influential for *B. occidentalis*, accounting for 63.3% of total variation in that model. Variation in springtime temperature was also an important predictor for *B. mckayi,* but temperature at the end of the active season and variability in temperatures throughout the active season were also influential for that species. Only 3.2% of the variation in the *B. mckayi* model was predicted by variables that measured precipitation. Elevation was not an important predictor for either species in these models.

Resistance rasters generated from the maximum entropy models for *B. occidentalis* in the two time-categories before and after the observed beginning of its decrease in abundance indicate that resistance increased in some portions of the range in the later time category, notably along the coast of California and on the eastern side of the distribution (Fig. 3.7). However, the models also indicate that resistance is lower in more recent years in the intermountain west region of the United States and southwestern Canada.

Analyses of the likelihood of occupancy of *B. occidentalis* before (1960 to 1994) and after (1995 to 2019) the observed decrease in abundance and range indicated that the variability in temperatures in April remains a very important predictor throughout time (Table 3.7). Surprisingly, elevation was an important predictor prior to 1995, but was ranked as one of the least important predictors in more recent years. Additionally, temperature variables at the beginning of the active season and the end of the active season (April and September) explained the most variation in the model from 1960 to 1994, 58.2%. Variation in temperature in the early season remained very important in the 1995 to 2019 time-category, but temperature at the beginning and end of the active season was less influential overall, with a total contribution of 32.5%, while temperatures during mid-season months were more influential during the later time category.

Environmental influences on genetic differentiation

With the help of modification indices built into the Lavaan package, we identified well-fitting SEMs to describe the relationships among our predictor and response variables for *B. mckayi* and *B. occidentalis* from 1960 to 2020 (Table 3.8). We achieved good relative measures of fit, but the one test of absolute fit, the χ^2 test, never achieved non-significance. The χ^2 test compares the variance in the model covariance matrix to the population covariance matrix. If the matrices are not significantly different, this is an
indication that the model describes all of the variance in the dataset (Bauer and Curran 2020). Since the χ^2 test never achieved non-significance, this is an indication that there is variance in the dataset that is not explained by the model. χ^2 tests are sensitive to sample size, and datasets with large samples are often rejected even when they describe relationships well (Schermelleh-Engel and Moosbrugger 2003). My datasets are exceptionally large (*B. mckayi* $n = 153735$ and *B. occidentalis* All Years $n = 56953$, *B. occidentalis* 1960 to 1994 n = 2016, *B. occidentalis* 1995 to 2020 n = 37401), because we used pairwise comparisons of predictor variables for each genotyped specimen. In both cases, the χ^2 statistic was reduced by more than an order of magnitude from the baseline model (*B. mckayi* baseline: 54,769.419, *B. occidentalis* baseline: 354497.631, Table 3.8 for model statistic), but the χ^2 statistic remained significant. However, the relative tests of fit indicate that the predictors in the models do improve model fit and, therefore, are useful to describe changes in the response variables. We was not able to generate models for the datasets that included only data from *B. occidentalis* from 1960 to 1994 and *B. occidentalis* from 1995 to 2020 that fit well enough to be interpreted with any confidence. Therefore, results are only presented for the models with the full datasets (1960 to 2020).

Measurements of habitat quality, maximum monthly temperature, and mean monthly temperature did not improve the model fit to the data when they were included in models for either species, so they were removed from the analyses completely. Temperature was represented in the models by monthly minimum temperatures.

Of all the variables we measured, only distance from a known infection of *V. bombi* had a significant relationship to D in *B. mckayi* (Fig. 3.8). For every 1 kilometer

that a pair of specimens was farther away from known *V. bombi* infections, they were 0.067 D units more closely related (the value in Figure 6 is negative, indicating that genetic distance is decreased). This relationship accounts for the geographic distance between the pair of specimens. This indicates that specimens that were farther from *V. bombi* infections were more likely to exchange genes (have gene flow) than specimens that were closer to *V. bombi* infections, even if the pairs of specimens were the same geographic distance apart*.*

The variable that had the greatest impact on the D of *B. occidentalis* was geographic distance, which is a latent variable that is composed of two variables with different units. The standardized CFA indicates that least cost distance (the more conservative measure) accounted for roughly twice as much variance as random walk distance (the more liberal measure, Table 3.10). For every one standardized unit of geographic distance that a pair of specimens was farther away from each other, they were 0.075 D units less closely related (accounting for the geographic distance of the specimens from each other). Distance from known *V. bombi* infections also influenced genetic differentiation for this species. For every 1 kilometer that a pair of specimens was farther away from *V. bombi* infections, they were 0.027 D units more closely related. Indirect analyses of precipitation and temperature also had significant relationships with D (Table 3.9). The influence of precipitation on D was relatively small, and was primarily driven by precipitation in May, June, and July (Table 3.10). One centimeter increase in the difference in precipitation between the collection locations of the specimens in a pair indicated a decrease of 0.003 D units (more closely related) when mediated by latitude alone and an additional 0.002 D units when mediated by latitude and elevation. Temperature also had a relatively small effect on D. Only springtime minimum temperatures improved the fit of the model, so minimum temperatures from July, August, and September were not included. One degree Celsius increase in the difference in temperature between the collection locations of the specimens in a pair indicated an increase of 0.002 D units (less closely related). However, when mediated by elevation and latitude, that relationship was weakened by 0.009 D units per degree Celsius. Precipitation and temperature had no significant direct effect on D, they were only influential when contextualized by latitude and elevation.

Discussion

B. occidentalis and *B. mckayi* are important and historically abundant pollinators within their geographic ranges (Goulson 2003; Cameron et al. 2011; Koch and Strange 2012). We tested two hypotheses using landscape genetic methods to assess the current and historical genetic diversity and genetic structure within these species: (1) genetic structure among populations has increased and genetic diversity within populations has decreased for *B. occidentalis* over time; these population characteristics have remained relatively stable for *B. mckayi*; and (2) changes in environmental drivers, such as climate, habitat fragmentation, and increased parasite pressures drive patterns of genetic diversity and structure in *B. occidentalis* and *B. mckayi*. Through testing my first hypothesis, we built on the previously published literature that describes the conservation status of these species (Rao and Stephen 2007; Evans et al 2008; Lozier et al. 2011; Colla et al. 2012; Koch and Strange 2012; Williams et al. 2012; Hatten et al. 2015; Pampell et al. 2015; McHugh and Sikes 2016; Sheffield et al. 2016; Graves et al. 2020; Williams 2021), and we presented the first population genetics study to treat these taxa as separate species

(Williams et al 2021, Chapter 1). Through testing my second hypothesis, we used SEMs to directly measure the influence of environmental variables and isolation by distance on the genetic distances between pairs of geographically separated specimens.

Measurements of heterozygosity within populations indicate inbreeding in most geographic clusters

We measured significant genetic disequilibrium in most loci and significantly higher than expected inbreeding in most geographic clusters in both species. However, there was no evidence that populations from recent years were more likely to be inbred than populations from earlier sampling years. We detected higher rarefied allelic richness within most populations of both species than was previously reported for *B. occidentalis* or other North American *Bombus* species within the continental U.S.A. (Lozier et al. 2011). Despite relatively high absolute values of rarefied allelic richness within geographic clusters, we detected a weak statistically significant decline in allelic richness through time in *B. occidentalis* and a non-significant trend of decline in *B. mckayi*. The measured loss in allelic richness is not likely to be biologically meaningful, but signals a need for ongoing monitoring, in case the trend continues. These results may indicate that geographic clusters have been isolated from each other for many generations (at least several generations before 1960), but that genetic diversity within them was slow to decrease. These results may reflect a lag in the loss of allelic diversity due to inbreeding. A simulation study found that in some species (trees), populations with more than 500 (reproductive) individuals had elevated FIS values after only 5 generations, but did not incur substantial decreases in allelic richness until several generations later (Stefenon et al. 2012).

An alternative explanation is that allelic richness and inbreeding are not associated in this dataset. Although we detected statistically significant genetic disequilibrium, the magnitude of the effect may not reflect strong genetic isolation. Allelic richness decreases in inbreeding populations, but it also decreases with reductions in abundance, which are not necessarily associated with reduced gene flow. For example, allelic richness would be expected to decrease suddenly in response to a pathogenic outbreak or large disturbance event that removes many individuals from a population. Such events would not necessarily prevent gene flow among the surviving populations. The detected decrease in allelic richness from 1960 to 2020 was gradual in both species, and did not coincide with the observed sudden decrease in abundance for *B. occidentalis*. Although the allelic richness was lower in the time frame after the decline, it increased slightly during that time for both species (statistically significantly only for *B. mckayi*). Continued monitoring is necessary to determine if *B. occidentalis* or *B. mckayi* will continue to gain allelic diversity in future generations.

Measurements of genetic structure among populations indicate lower levels of gene flow among populations than previously detected

Our analyses of genetic structure among populations of *B. mckayi* and *B. occidentalis* indicate that populations within both species may be more isolated than previous studies have indicated. Cameron et al. (2011) measured the abundance and genetic structure of *B. occidentalis* within the contiguous U.S.A. (CONUS) from 2009 to 2011 and Lozier et al (2011) measured the same characteristics within CONUS and Alaska. While both of these studies found a severe decrease in relative abundance in CONUS, they reported low global F_{ST} values (0.032 from Cameron et al. (2011) and

 0.035 from Lozier et al., including Alaskan populations). Our reported F_{ST} values were much higher than those previously reported, which may indicate less gene flow among populations of *B. occidentalis* and similarly low gene flow among populations of *B. mckayi*. However, there are several other factors that may have inflated our measurements of genetic structure relative to other studies.

One possible explanation for the higher F_{ST} values is that the samples included in our study were taken from across many generations. Microsatellite markers are neutral (not under selection) and have relatively high mutation rates (Estoup and Angers 1998; Gemayel et al. 2012; Vieira et al. 2016) compared to other genetic markers (especially other nuclear markers), which makes them useful for measuring genetic changes within and among populations over short time-scales due to genetic drift. However, this characteristic also means that genetic structure among populations due to population isolation (reduced gene flow) and time (genetic drift) are confounded. To accommodate this limitation, population genetic studies that use microsatellite markers are often conducted using specimens that were collected in the same generation (bumble bees are annual species, so one generation is one year), or as close to the same generation as possible (Cameron et al. 2011; Koch et al. 2017; however see Rosche et al. 2022). This limitation often prevents sampling from across the entire range of widespread species due to the practical limitations of conducting field work across a large geographical area in one or a few years. A goal of this study was to leverage the resource of museum specimens collected from across time to compare population genetic characteristics within and among populations across time. We used a correction to account for genetic differentiation due to time, rather than population isolation, which reduced the amount of

among-population variance detected by AMOVA considerably in both species. To test the change in genetic structure through time, we measured global F_{ST} among populations from similar years (at most seven years per analysis). Although the relationship was not significant, there was an observable increase in F_{ST} values through time, and we suspect that a larger dataset (more populations sampled within each year) may have resulted in statistical significance. Additionally, the global FST value for *B. occidentalis* populations collected from 2007 to 2009 was 0.13, still higher than that reported by Cameron et al. (2011) or Lozier et al. (2011).

A second possible reason for the higher detected genetic structure values in this study could be small sample sizes (largest n for *B. occidentalis*: 45, largest n for *B. mckayi*: 93), although samples of 4 to 6 individuals have been shown to be sufficient to accurately estimate genetic structure using F statistics when the number of loci used for genotyping is high (Willing et al. 2012). Cameron et al. (2011) did not report the number of specimens included in each sampling group and Lozier et al. (2011) collected between 7 and 34 specimens per site (average 17.69 standard error 2.60). Both studies used the same microsatellite loci for genotyping as we did. They amplified between eight and ten loci for each *B. occidentalis* specimen, which is similar to our amplification success. Although the small sample sizes of some of the geographic clusters in our study may have inflated our F_{ST} values relative to those reported by previous studies, that statistical artifact does not account for the magnitude of difference between the measured genetic structure in the studies. Our results indicate that gene flow is lower among populations in *B. occidentalis* and *B. mckayi* than is expected based on results from previous studies*.*

Finally, the analysis conducted by Cameron et al. (2011) focused on *B. occidentalis* populations from a much smaller portion of the species range, mostly from the intermountain west of the United States of America. The subsequent study by Lozier et al. (2011) expanded this data set to include populations from Alaska (now *B. mckayi*) but did not include many Canadian specimens. This difference in sampling range is likely to influence estimates of FST because isolation by distance increases those estimates (van Strien et al. 2015).

No other population genetic study has specifically focused on *B. mckayi* to date (however see Lozier et al. 2011 for a study that includes *B. mckayi* specimens sampled from Alaska), but abundance surveys indicate that it had a stable abundance and distribution as of 2015 (Koch and Strange 2012; Pampell et al. 2015; Sheffield et al. 2016). Structure analyses and the Mantel test both indicated that the structure in *B. mckayi* was more strongly associated with geographic distance than *B. occidentalis*. Also, the relationship of increasing F_{ST} values throughout time that we observed in *B*. *occidentalis* was not as strong in *B. mckayi* (neither relationship was statistically significant). These measurements indicate that populations of *B. mckayi* are exchanging more genes than populations of *B. occidentalis* and more of the genetic structure detected in *B. mckayi* may be associated with natural environmental barriers to gene flow rather than recent population declines. Recent studies indicate that population structure is not necessarily similar across bumble bee species, with some species indicating an effect of isolation by distance (IBD) and other species exhibiting no such patterns, even species that have broadly overlapping ranges (Koch et al. 2017). This finding indicates that population structure may be dependent on species specific niche characteristics. If more

of the genetic structure in *B. mckayi* can be described by natural IBD, the elevated FST values detected in this study may be less of a conservation consideration for *B. mckayi* than they are for *B. occidentalis*, and more of an indication of local adaptation among populations across a heterogeneous landscape.

The influence of environmental variables on maximum entropy models of B. mckayi and B. occidentalis

Until the recent split of *B. mckayi* from *B. occidentalis*, the relationships between the species and their environments were likely confounded, because they were treated as a single species. This study presents the first assessment of a *B. mckayi* ecological niche model and compares the environmental predictors of that species to those of *B. occidentalis*.

Bombus mckayi and *B. occidentalis* were influenced by different environmental predictors in the MaxEnt models. Precipitation was important for *B. occidentalis,* but not *B. mckayi*. This may be because the range of *B. occidentalis* extends into semi-arid and arid habitats, whereas precipitation is relatively abundant throughout the range of *B. mckayi* (Lemmen and Warren 2004; Kharin et al. 2013; Westra et al. 2014)*.* The impact of precipitation (or lack thereof) on *B. occidentalis* is likely to increase into the future, particularly along the southern and eastern edges of the range, as projected climate change for those regions includes hotter, drier summers and longer, more variable springs. While temperature is projected to increase dramatically across the range of *B. mckayi,* precipitation is predicted to remain stable or increase slightly (Lemmen and Warren 2004; Kharin et al. 2013; Westra et al. 2014; IPCC 2018; IPCC 2019).

Variability in springtime temperature was an important predictor for both species. Spring is the time when new queens forage and establish nests. Increased stressors associated with variable springtime temperatures, or potential climatic release of predators or parasites associated with warmer springs (Clare et al. 2016; Gehman et. al 2018; Turner et al. 2020) may have strong impacts on the success or failure of nests. Springs and autumns are predicted to get longer and more variable in temperature and precipitation in the intermountain west of the USA and southern Canadas, and temperatures throughout all seasons are predicted to increase across the range of *B. mckayi* (Cayan et al. 2001; Lemmen and Warren 2004; Melaas et al. 2018).

The ranges of both *B. mckayi* and *B. occidentalis* include mountain ranges throughout, but elevation was an important predictor for only *B. occidentalis*. This is likely because elevation is associated with a greater shift in temperature and precipitation at low latitudes than it is at high latitudes (Minder et al. 2018). *Bombus occidentalis* is more likely to be captured at high elevation sites at the southern end of its range (Fig. 3.1), because high elevation sites have lower temperatures and higher precipitation (Cameron et al. 2011; Notarnicola 2020). However studies indicate that these habitat refugia are warming and drying faster than lower elevation sites (Minder et al. 2018; Notarnicola 2020) indicating that some populations isolated on high elevation sites in the southwestern USA are at highest risk of encountering unsuitable habitat. High elevation sites at northern latitudes are too cold for many bee species, but *B. mckayi* is evolved to thrive in temperate and arctic habitats, and has not been found to be limited by low temperatures within its range (Koch and Strange 2012). However, the short growing season and presence of year-round ice at high elevations at northern latitudes likely

provides some limits, even if elevation is not a factor, and future warming and ice loss may result in a net gain of suitable habitat.

Changes in the drivers of *B. occidentalis* occupancy between the two time-frames (1960 to 1994 and 1995 to 2019) reaffirm the importance of springtime minimum temperatures for this species. This association may be driven in part by minimum thermal tolerance, but is more likely to be associated with changes in vegetation phenology, particularly bloom timing (Cayan et al. 2001), and with competitive pressures due to ecological release (Clare et al. 2016; Gehman et. al 2018; Turner et al. 2020). However, maximum thermal thresholds seem to influence bumble bee movement (Oyen et al. 2016), and temperatures during the hottest time of the year were more influential in the later time frame. Additionally, there was a shift in the importance of elevation, from being ranked as the second most important variable in the early time frame to one of the least important variables in the later time frame. This seems counter-intuitive, because as temperatures change species are expected to move upward in elevation (Pyke et al. 2016) or poleward in latitude (Kerr et al. 2015) to track suitable climate, and some bumble bee species have been shown to do that (Kerr et al. 2015; Pyke et al. 2016). However, *B. occidentalis* has always been a montane species, and it may have already maximized potential elevation gain within the most vulnerable portions of its range (southern edge, Cameron et al. 2011). There is evidence that bumble bees are not shifting their ranges north in latitude to track suitable climate (Kerr et al. 2015; Soroye et al. 2020). If all of these trends hold for *B. occidentalis*, it is possible that elevation is less influential for contemporary populations than it once was, because warming springs have reduced the suitability of high elevation habitat, so that it is no longer optimal.

A possible stressor that was not included in this model was wildfire. Although wildfire has not been identified as a direct stressor to any North American bumble bee species to date, increased fire frequency and size across ranges of both *B. mckayi* and *B. occidentalis* are changing the composition of vegetation on the landscape and are predicted to continue to have major impacts on vegetation in western North America into the future (Kasischke et al. 2010; Wang et al. 2016a; Stralberg et al. 2018; Wotton et al. 2017; Holden et al. 2018; Fusco et al. 2019; Rogers et al. 2020). Such changes may alter or limit the quality and quantity of nesting sites for ground nesting bumble bees, including *B. mckayi* and *B. occidentalis*, in the future. Impacts of wildfire on bumble bee nesting under these changing conditions is an area that needs further study.

Assessment of structural equation model fit to determine suitability for interpretation

There is no single statistical significance test that measures the fit of SEMs, so all goodness-of-fit measures have to be taken together (Bauer and Curran 2020). The only measure associated with a significance test is the χ^2 test (Bauer and Curran 2020), which did not achieve non-significance for any of our models. This result indicates that there is variance in the dataset that is not explained by the model, despite indications of good fit from the other (relative) fit measures. Due to our large datasets, which are known to prevent statistically significant χ^2 results (Schermelleh-Engel and Moosbrugger 2003), and good-fitting relative measures, we accepted the models for *B. mckayi* and *B. occidentalis* from 1960 to 2020 as good-fitting models and interpreted them with confidence. However, we could not identify good-fitting models for *B. occidentalis* from the time frames that roughly represent relationships before (1960-1994) and after (1995- 2020) the observed decrease in abundance and range. The poor fit of these models is

perplexing because they represent subsets of the larger dataset which are represented well by the models. Additionally, even as subsets of the full data, the size of the datasets for these analyses were still quite large (1960 to 1994, $N = 2016$; 1995 to 2020, $N = 37401$).

The most likely explanation for this lack of fit is simply that relationships among samples from these shorter time frames were not strong enough to explain the datasets. Variance within the data was quite high for most predictors, and was exaggerated by the fact that each datapoint represented a comparison of measurements between two points, rather than a single point (though this is also true in Wang et al. 2013). One possible solution to this problem may be to use F_{IS} rather than D as the genetic response variable in models. F_{IS} is a measure of inbreeding derived from measurements of heterozygosity within populations rather than a ratio of amino acid substitutions among individuals (Wright 1951; Holsinger and Weir 2009), so it is not a direct measurement of genetic differentiation. However, F_{IS} increases in populations that are genetically isolated and can be used as an indicator of a lack of gene flow into those populations (Holsinger and Weir 2009). The benefit of using FIS is that it is not a pairwise measurement, so the relationships between the predictor and response variables are more straightforward, and may have lower variance. The drawback of using F_{IS} rather than D is that sampling units become populations (geographic clusters), rather than individuals, which limits sample sizes considerably; in this study *B. mckayi* $N = 36$ and *B. occidentalis* $N = 48$. Additionally, using population genetic measures excludes samples that were not collected as part of a series (that come from geographic clusters with only one specimen). A second possible solution to this problem may be to use a different type of model (in place

of an SEM) to describe genetic differentiation along the genetic gradient, Milligan et al (2018) suggests the spatial Λ -Flemming-Viot model.

The influence of environmental variables on the genetic variation of B. mckayi and B. occidentalis across the landscape (i.e. resistance)

This study is the first attempt to determine relationships between environmental variables and gene flow (or resistence to gene flow) in *B. mckayi* or *B. occidentalis* using SEMs. The loadings for each regression were reported in units of D (0 to 1) and represent the average amount of change in D within pairs of specimens per unit of change in the predictor.

The largest effect was geographic distance. The relationship of distance was significant within *B. occidentalis*, but not within *B. mckayi*, despite being stronger. This is likely due to a high amount of variance in the RandomWalk and LeastCost distance datasets. If the relationship with geographic distance for *B. mckayi* is cautiously interpreted, the strength of the relationship indicates that *B. mckayi* is more strongly influenced by IBD than *B. occidentalis*, which agrees with the results of the Mantel test.

The second most influential relationship for *B. occidentalis* and the most influential statistically significant relationship for *B. mckayi* was with distance from *V. bombi* infections. In both cases, specimen pairs that were collected closer to the locations of known *V. bombi* infections were less genetically differentiated, indicating that they were more closely related. This relationship could be confounded with geographic distance, if the specimens driving the relationship are geographically close to the same detected *V. bombi* infection, and therefore close to each other, but the fit of the model decreased when geographic distance and distance from *V. bombi* infections were allowed to correlate. The correlation between geographic distance within the pair of specimens and distance from a known *V. bombi* infection had to be removed to improve the model fit, indicating that geographic distance was likely not a confounding factor. Koch and Strange (2012) detected similarly high *V. bombi* infection rates in *B. mckayi* and *B. occidentalis* in 2010. Cameron et al. (2016) measured an increase in infection rates in *B. occidentalis* that began in the mid 1990's, but did not report *B. mckayi* separately from *B. occidentalis.* Cameron et al (2016) reported that the four North American *Bombus* species included in the study with high *V. bombi* infection rates were decreasing in abundance and range. It is possible that the effect of the drivers of the reported decline in the four measured species is having the same effect in *B. mckayi,* but later than in the other species or to a moderated extent because of the relative isolation of the species range from other drivers with potentially compounding effects (e.g., agricultural lands, dense urban areas).

Finally, the indirect relationships of precipitation and temperature on the genetic differentiation of *B. occidentalis* were statistically significant. However, the influences were so small, it is likely that they are not biologically meaningful. Habitat quality and distance from agricultural fields were poor predictors that did not improve the fit of any of our models, but may fit better in a model that does not include pairwise comparisons.

Some variables that were not included in the model but that may be barriers to gene flow for these species include competitive interactions with other species, such as domestic honey bees, and density of roads across the landscape. Sufficient data to inform analyses of these variables is not currently available, but there is some evidence to suggest that they have an impact on colony success and reproductive bumble bee

movement across the landscape (Thomson 2004; Keilshon et al. 2018; Fitch and Vaidya 2021), and they merit additional study.

For this system, a combination of SEMs that measure variable influences on D and F_{IS} may be best to resolve relationships between environmental predictors and gene flow. Comparisons of geographic distance to D among pairs of samples is a biologically meaningful way to assess the relative influence of IBD on species. However, measuring the differences of the effect of temperature and precipitation as mediated by elevation and latitude, the differences in habitat quality at collection sites, or combined distance from agricultural lands are regressions that may be too convoluted to describe the true relationships between the variables. Also, significant relationships between distance from *V. bombi* infections and D that were detected in the models presented here may prove to be even stronger when regressed with F_{IS}. Sample size is a concern for models that use populations rather than specimens as sampling units because a lack of degrees of freedom may limit the power of the model. However careful design could mitigate these issues.

Conclusions and implications for conservation

The results of the population genetics analyses partially support my first hypothesis. Both *B. mckayi* and *B. occidentalis* have sustained reductions in genetic diversity over time and currently have moderate levels of genetic structure. Although decreases in allelic diversity, and some evidence of inbreeding have been documented in these species, substantial genetic diversity (Holsinger et al. 2009 for interpretation of F statistics) remains in extant populations, which indicates a good opportunity for recovery of the species if the effects of the drivers of the decline are mitigated.

The results of the SEMs supported my second hypothesis, if somewhat weakly. Improvements on the model design may reveal relationships that were not detected in this study. My results indicated that the two species were influenced by different environmental variables, and were influenced by IBD to different extents. The range of *B. mckayi* is at a higher latitude and has less urban and agriculturally developed land than the range of *B. occidentalis*, characteristics which may buffer *B. mckayi* against some of the factors that drive the decline of *B. occidentalis*. However, the strongest ecological driver of genetic differentiation that I detected was distance from known *V. bombi infections*, a surrogate for the likelihood of infection in our samples, which had a stronger relationship to D in *B. mckayi* than in *B. occidentalis*. Despite more dramatic decreases in abundance in *B. occidentalis*, our results indicate that *B. mckayi* may also be facing increased environmental pressures. Additional monitoring is necessary to ensure populations of *B. mckayi* maintain sufficient population sizes and genetic diversity. Additionally, changes in the importance of environmental variables to predict *B. occidentalis* occupancy reaffirm the influence of changing climate on this montane bumble bee species. Future conservation work should focus on mitigating the impacts of increasing parasite loads and increasing temperatures within the ranges of *B. mckayi* and *B. occidentalis*.

References

Abertman, M.P., Gleiser, G., Morales, C.L., Williams, P. and Aizen, M.A. (2017). Global decline of bumblebees is phylogenetically structured and inversely related to species range size and pathogen incidence. Proceedings of the Royal Society B 284: 20170204. http://dxdoi.org/10.1098/rspb.2017.0204.

- Anatharaman, R., Hall, K., Shah, V.B., and Edelman, A. (2020). Cicuitscape in Julia: high performance connectivity modeling to support conservation decisions. JuliaCon Proceedings 1(1):58.
- Baur, D.J., and Curran, P.J. (2020). Introduction to structural equation modeling, course notes. May 6-8, 2020. Chapel Hill, North Carolina. Curran-Bauer Analystics.
- Callahan, C.M., Rowe, A.C., Ryel, R.J., Shaw, J.D., Madritch, M.D., and Mock, K.E. (2013). Continental-scale assessment of genetic diversity and population structure in quaking aspen (*Populus tremuloides*). Journal of Biogeography 40(9):1780- 1791.
- Cameron, S.A., Lim, Haw Chuan, Lozier, J.D., Duennes, M.A., and Thorp, R. (2016). Test of the invasive pathogen hypothesis of bumble bee decline in North America. Proceedings of the National Academy of Sciences 113(16):4386-4391.
- Cameron, S.A., Lozier, J.D., Strange, J.P., Koch, J.B., Cordes, N., Solter, L.F., and Griswold, T.L. (2011). Patterns of widespread decline in North American bumble bees. Proceedings of the National Academy of Sciences of the United States of America 108(2):662-667.
- Cameron, S.A., and Sadd, B.M. (2020). Global trends in bumble bee health. Annual Review of Entomology 65:209-232.
- Cayan, D.R., Kammerdiener, S.A., Dettinger, M.D., Caprio, J.M., and Peterson, D.H. (2001). Changes in the onset of springs in the western United States. Bulletin of the American Meteorological Society 82(3): 399-414.
- Cayuela, H., Rougemont, Q., Prunier, J.G., Moore, J.-S., Clobert, J., Besnard, A., and Bernatchez, L. (2018). Demographic and genetic approaches to study dispersal in wild animal populations: a methodological review. Molecular Ecology: doi 10.1111/mec.14848.
- Clare, F.C., Halder, J.B., Daniel, O., Bielby, J., Semenov, M.A., Jombart, T., Loyau, A., Schmeller, D.S., Cunningham, A.A., Rowcliffe, M., Garner, T.W.J., Bosch, J., and Fisher, M.C. (2016). Climate forcing of an emerging pathogenic fungus across a montane multi-host community. Philosophical transactions of the Royal Society B 371(1709): 20150454.
- Colla, S.R., Gadallah, F., Richardson, L., Wagner, D., and Gall, L. (2012). Assessing declines of North American bumble bees (*Bombus* spp.) using museum specimens. Biodiversity and Conservation 21(14):3585-3595.
- COSEWIC (2014). COSEWIC assessment and status report on the Gypsy Cuckoo Bumble *Bombus bohemicus* in Canada. Committee on the Status of Endangered

Wildlife in Canada. Ottawa. $ix + 56$ pp. (www.registrelepsararegistry.gc.ca/default_e.cfm).

- Danielson, J.J., and Gesch, D.B. (2011). Global Multi-resolution Terrain Elevation Data 2010 (GMTED2010). U.S. Geological Survey Open-File Report 2011-1073, 26 p.
- De Marco, P., Corrêa Nóbrega, C. (2018). Evaluating collinearity effects on species distribution models: An approach based on virtual species simulation. PLOS ONE 13(9):e0202403.
- Earl, D.A., and vonHoldt, B.M. (2012). STRUCTURE HARVESTER: a website and program for visualizing STRUCTURE output and implementing the Evanno method. Conservation Genetics Resources 4:359-361.
- El Mousadik, A., and Petit, R.J. (1996). High level of genetic differentiation for allelic richness among populations of the argantree (*Argania spinosa* (L.) Skeels) endemic to Morocco. Theoretical and Applied Genetics 92:832-839.
- Ellegren, H., and Galtier, N. (2016). Determinants of genetic diversity. Nature Reviews Genetics 17:422-433.
- Estoup, A., Solignac, M., Cornuet, J.M., Goudet, J., and Scholl, A. (1996). Genetic differentiation of continental and island populations of *Bombus terrestris* (Hymenoptera: Apidae) in Europe. Molecular Ecology 5(1):19-31.
- Estoup, A., Angers, B. (1998). Microsatellites and minisatellites for molecular ecology: theoretical and experimental considerations. In: Carvalho G. (ed) Advances in Molecular Ecology. NATO Press: Amsterdam.
- Evans, E., Thorp, R., Jepsen, S. and Hoffman Black, S. (2008). Status review of three formerly common species of bumble bee in the subgenus *Bombus*. http://www.xerces.org/wpcontent/uploads/2009/03/xerces_2008_bombus_status_review.pdf
- Feng, X., Park, D.S., Liang, Y., Pandey, R., Papeş, M. (2019)> Collinearity in ecological niche modeling: Confusions and challenges. Ecology and Evolution 9(18):10365- 10376.
- Fitch, G., and Vaidya, C. (2021). Roads pose a significant barrier to bee movement, mediated by road size, traffic and bee identity. Journal of Applied Ecology 58(6):1177-1186.
- Frankham R. (2015). Genetic rescue of small inbred populations: meta-analysis reveals large and consistent benefits of gene flow. Molecular Ecology 24(11):2610-2618.
- Freitas, B.M., Imperatriz-Fonseca, V.L., Medina, L.M., de Matos Peixoto Kleinert, A., Galetto, L., Nates-Parra, G., and Quezada-Euán, J.J.G. (2009). Diversity, threats and conservation of native bees in the Neotropics. Apidologie 40:332-346. DOI:10.1051/apido/2009012.
- Forsman, A. (2014). Effects of genotypic and phenotypic variation on establishment are important for conservation, invasion, and infection biology. Proceedings of the National Academy of Sciences of the United States of America 111(1):302-307.
- Foulley, J.-L. and Ollivier, L. (2006). Estimating allelic richness and its diversity. Livestock Science 101:150-158.
- Fusco, E., Finn, J.T., Balch, J.K., Nagy, C., and Bradley, B.A. (2019). Invasive grasses increase fire occurrence and frequency across US ecoregions. Proceedings of the National Academy of Sciences of the United State of America 116(47):23594- 23599.
- Gegear, R.J., Otterstatter, M.C., and Thomson, J.D. (2006). Bumble-bee foragers infected by a gut parasite have an impaired ability to utilize floral information. Proceedings of the Royal Society B 273(1590):1073-1078.
- Gehman, A.-L.M., Hall, R.J., and Byers, J.E (2018). Host and parasite thermal ecology jointly determine the effect of climate warming on epidemic dynamics. Proceedings of the National Academy of Sciences of the United States of America 115(4):744-749.
- Gemayel, R., Cho, J., Boeynaems, S., and Verstrepen, K.J. (2012). Beyond junk-variable tandem repeats as facilitators of rapid evolution of regulatory and coding sequences. Genes 3(3):461-480.
- Goudet, J. (1995). FSTAT version 1.2: a computer program to calculate F-statistics. Journal of Heredity 86:485-486.
- Goulson, D. (2003). Bumblebees: their behavior and ecology. Oxford University Press, Oxford.
- Goulson, D. (2010). Bumblebees: behavior, ecology, and conservation. Second edition. Oxford University Press Inc., New York.
- Goulson, D., Lye, G.C., and Darville, B. (2008). Decline and conservation of bumble bees. Annual Review of Entomology 53:191-208.
- Goulson, D., Nicholls, E., Botías, and Rotheray, E.L. (2015). Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. Science 347(6229):1255957.
- Grace, J.B. (2006). Structural Equation Modeling and Natural Systems. Cambridge University Press. ISBN 1139457845, 9781139457842.
- Graves, T.A., Chandler, R.B., Royle, J.A., Beier, P., and Kendall, K.C. (2014). Estimating landscape resistance to dispersal. Landscape Ecology 29:1201-1211.
- Graves, T.A., Janousek, W.M., Gaulke, S.A.m Nicholas, A.C., Keinath, D.A., Bell,C.M., Cannings, S., Hatfield, R.G., Heron, J.M., Koch, J.B., Loffland, H.L., Richardson, L.L., Rohde, A.T., Rykken, J., Strange, J.P., Tronstad, L.M., and Sheffield, C.S. (2020). Western bumble bee: declines in the continental United States and rangewide information gaps. Ecosphere 11(6): e03141.
- Greenleaf, S.S., Williams, N.M., Winfree, R., and Kremen, C. (2007). Bee foraging ranges and their relationship to body size. Plant Animal Interactions 153: 589- 596.
- Hand, B.K., Muhlfeld, C.C., Wade, A.A., Kovach, R.P., Whited, D.C., Narum, S.R., Matala, A.P., Ackerman, M.W., Garner, B.A., Kimball, J.S., Stanford, J.A., and Luikart, G. (2015). Climate variables explain neutral and adaptive variation within salmonid metapopulations: the importance of replication in landscape genetics. Molecular Ecology 25(3):689-705.
- Hanski, I., Schulz, T., Chong Wong, S., Ahola, V., RUokolainen, A., and Ojanen, S.P. (2017). Ecological and genetic basis of metapopulation persistence of the Glanville fritillary butterfly in fragmented landscapes. Nature Communications 8:14504.
- Hatfield, R., Jepsen, S., Thorp, R., Richardson, L., Colla, S., Folz Jordan, S. (2015). Bombus occidentalis. The IUCN Red List of Threatened Species.
- Hatten, T.D., Strange, J.P., and Maxwell, J.M. (2015). Late-season survey of bumble bees along Canadian highways of British Columbia and Yukon territories. Western North American Naturalist 75(2):170-180.
- Hendrick, P.W., Garcia-Dorado, A. (2016). Understanding inbreeding depression, purging, and genetic rescue. Trends in Ecology and Evolution. 31(12):940-952.
- Heinrich, B., and Kammer, A.E. (1973). Activation of the fibrillar muscles in the bumblebee during warm-up, stabilization of thoracic temperature and flight. Journal of Experimental Biology 58:677-688.
- Holden, Z.A., Swanson, A., Luce, C.H., Jolly, W.M., Maneta, M., Oyler, J.W., Warren, D.A., Parsons, R., and Affleck, D. (2018.) Decreasing fire season precipitation increased recent western US forest wildfire activity. Proceedings of the National Academy of Science of the United States of America 115(36) E8349-E8357.
- Holsinger, K.E. and Weir, B.S. (2009). Genetics in geographically structured populations: defining, estimating and interpreting F_{ST}.
- Hubisz, M.A., Falush, D., Stephens, M., and Pritchard, J.K. (2009). Inferring weak population structure with the assistance of sample group information. Molecular Ecology 9:1322-1332.
- IPCC, 2018: Summary for Policymakers. In: *Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty* [Masson-Delmotte, V., P. Zhai, H.-O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J.B.R. Matthews, Y. Chen, X. Zhou, M.I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, and T. Waterfield (eds.)]. *World Meteorological Organization, Geneva, Switzerland, 32 pp.*
- IPCC, 2019: Summary for Policymakers. In: *Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems* [P.R. Shukla, J. Skea, E. Calvo Buendia, V. Masson-Delmotte, H.- O. Pörtner, D. C. Roberts, P. Zhai, R. Slade, S. Connors, R. van Diemen, M. Ferrat, E. Haughey, S. Luz, S. Neogi, M. Pathak, J. Petzold, J. Portugal Pereira, P. Vyas, E. Huntley, K. Kissick, M. Belkacemi, J. Malley, (eds.)]. In press*.*
- Jackson, J.M., Pimsler, M.L., Oyen, K.J., Koch-Uhaud, J.B., Herndon, J.D., Strange, J.P., Dillion, M.E., and Lozier, J.D. (2018). Distance, elevation and environment as drivers of diversity and divergence in bumble bees across latitude and altitude. Molecular Ecology 27:2926-2942.
- Jakobsson, M. and N.A. Rosenberg (2007). CLUMPP: a cluster mathing and permutation program for dealing with label switching and multimodality in analysis of population structure. Bioinformatics 23(14):1801-1806.
- Jones, O., and Wang, J. (2010) COLONY: A program for parentage and sibship inference from multilocus genotype data. Molecular Ecology Resources 10(3):551-555.
- Jost, L., Archer, F., Flanagan, S., Gaggiotti, O., Hoban, S., and Latch, E. (2018). Differentiation measures for conservation genetics. Evolutionary Applications 11(7):1139-1148.
- Kalinowski S.T. (2005). HP-Rare: a computer program for performing rarefaction on measures of allelic diversity. Molecular Ecology Notes 5:187-189.
- Keilsohn, W., Narango, D.L., and Tallamy, D.W. (2018). Roadside habitat impacts insect traffic mortality. Journal of Insect Conservation 22:183-188.
- Kamvar ZN, Tabima JF, Grünwald NJ. (2014) Poppr: an R package for genetic analysis of populations with clonal, partially clonal, and/or sexual reproduction. PeerJ 2:e281.
- Kasichke, E.S., Verbyla, D.L., Rupp, T.S., McGuire, A.D., Murphy, K.A., Jandt, R., Barnes, J.L., Hoy, E.E., Duffy, P.A., Calef, M., and Turetsky, M.R. (2010). Alaska's changing fire regime- implications for the vulnerability of its boreal forests. Canadian Journal of Forest Research.
- Kerr, J.T., Pindar, A., Galpern, P., Packer, L., Potts, S.G., Roberts, S.M., Rasmont, P., Schweiger, O., Colla, S.R., Richardson, L.L., Wagner, D.L., Gall, L.F., and Sikes, D.S. (2015). Climate change impacts on bumblebees converge across continents. Science 349(6244):177-180.
- Kharin, V.V., Zwiers, F.W., Zhang, X., and Wehner, M. (2013). Changes in temperature and precipitation extremes in the CMIP5 ensemble. Climate Change 119(2):345- 357.
- Klein, A.-M., Vaissiére, Cane, J.H., Steffan-Dewenter, S., Cunningham, S.A., Kremen, C., and Tscharntke, T. (2007). Importance of pollinators in changing landscapes for world crops. Proceedings of the Royal Society B 274(1608):303-313.
- Koch, J.B., Looney, C., Sheppard, W.S., and Strange, J.P. (2017). Patterns of population genetic structure and diversity across bumble bee communities in the Pacific Northwest. Conservation Genetics 18(3):507-520.
- Koch, J.B., and Strange, J.P. (2012). The status of *Bombus occidentalis* and *B. moderatus* In Alaska with special focus on *Nosema bombi* incidence. Northwest Science 86(3):212-220.
- Koch, J.B., Vandame, R., Mérida-Rivas, Sagot, P., and Strange, J. (2018). Quaternary climate instability is correlated with patterns of population genetic variability in *Bombus huntii*. Ecology and Evolution 8(16):7849-7864.
- Leberg, P.L. (2002). Estimating allelic richness: effects of sample size and bottlenecks. Molecular Ecology 11:2445-2449.
- Lemmen, D.S., and Warren, F.J. (2004). Climate change impacts and adaptation: A Canadian perspective. http://hdl.handle.net/10214/15055.
- Lewis, J. (2021) leastcostpath: Modelling Pathways and Movement Potential Within a Landscape (version 1.8.5. Available at: https://cran.rproject.org/web/packages/leastcostpath/index.html
- Lozier, J.D., Strange, J.P., Stewart, I.J., and Cameron, S.A. (2011). Patterns of rangewide genetic variation in six North American bumble bee (Apidae: *Bombus*) species. Molecular Ecology 20:4870-4888.
- Manel, S., Schwartz, M.K., Luikart, G., and Taberlet, P. (2003). Landscape genetics: combining landscape ecology and population genetics. Trends in Ecology and Evolution 18(4):189-197.
- Marrotte, R.R. and Bowman, J. (2017) The relationship between least-cost and resistance distance. PLoS ONE 12(3): e0174212. https://doi.org/10.1371/journal.pone.0174212.
- Masuda, R. (2018). Status and perspective of the population-based on genetic diversity: Introduction. *In* Biodiversity conservation using umbrella species. *Eds*
- McRae, B.H., V.B. Shah, and T.K. Mohapatra. (2013) Circuitscape 4 User Guide. The Nature Conservancy. http://www.circuitscape.org.
- McHugh, M., and Sikes, D. (2016). *Bombus occidentalis* in Alaska and the need for future study (Hymenoptera: Apidae). Newsletter of the Alaska Entomological Society 9(1):2-5.
- McRae, B.H. (2007). Isolation by resistance. Evolution 60(8):1551-1561.
- Melaas, E.K., Sulla-Menashe, D., and Friedl, M.A. (2018). Multidecadal changes and interannual variation in springtime phenology of North American temperate and boreal deciduous forests. Geophysical Research Letters 45(6):2679-2687.
- Michener, C.D. (2000) Bees of the World. The Johns Hopkins University Press, Baltimore and London.
- Milligan, B.G., Archer, F.I., Ferchaud, A.-L., Hand, B.K., Kierepka, E.M., and Waples, R.S. (2018). Disentangling genetic structure for genetic monitoring of complex populations. Evolutionary Applications 11(7):1149-1161.
- Minder, J.R., Letcher, T.W., and Liu, C. (2018). The character and causes of elevationdependent warming in high-resolution simulations of rocky mountain climate change. Journal of Climate 31(6):2093-2113.
- Nei, M. (1972) Genetic Distance between Populations. American Naturalist, 106, 283- 292. http://dx.doi.org/10.1086/282771
- Notarnicola, C. (2020). Hotspots of snow cover changes in global mountain regions over 2000-2018. Remote Sensing of Environment 243: 111781.
- Ollerton, J. (2017). Pollinator diversity: distribution, ecological function, and conservation. Annual Review of Ecology, Evolution, and Systematics 48:353- 376.
- Osborne, J.L., Clark, S.J., Morris, R.J., Williams, I.H., Riley, J.R., Smith, A.D., Reynolds, D.R., and Edwards, A.S. (2001). A landscape-scale study of bumble bee foraging range and constancy, using harmonic radar. Journal of Applied Ecology 36(4):519-533.
- Oyen, K.J., Giri, S., and Dillon, M.E. (2016). Altitudinal variation in bumble bee (*Bombus*) critical thermal limits. Journal of Thermal Biology 59:52-57.
- Pampell, R., Sikes, D., Pantoja, A., Holloway, P., Knight, C., and Ranft, R. (2015). Bumble bees (*Hymenoptera: Apidae: Bombus* spp.) of interior Alaska: Species composition, distribution, seasonal biology, and parasites. Biodiversity Data Journal 3:e50585.
- Pina-Martins, F., Silva, D.N., Fino, J., Paulo, O.S. (2017). Structure_threader: an improved method for automation and parallelization of programs structure, fastStructure and MavericK on multicore CPU systems. Molecular Ecology Resources 17:e268-e274.
- Ploquin, E.F., Herrera, J.M., Obeso, J.R. (2013). Bumblebee community homogenization after uphill shifts in montane areas of northern Spain. Oecologia 173:1649-1660.
- Pritchard, J.K., Stephens, M., and Donnelly, P. (2000). Inference of population structure using multilocus genotype data. Genetics 155:945-959.
- Pollinator Partnership Action Plan (2016). https://www.whitehouse.gov/sites/whitehouse.gov/files.images/Blog/PPAP_2016. pdf.
- Porras-Hurtado, L., Ruiz, Y., Santos, C., Phillips, C., Carracedo, Á, and Lareu, M.V. (2013). An overview of STRUCTURE: applications, parameter settings, ans supporting software. Fronteirs in Genetics 4(98):1-13.
- Pyke, G.H., Thomson, J.D., Inouye, D.W. and Miller, T.J. (2016). Effects of climate change on phenologies and distributions of bumble bees and plants they visit. Ecosphere 7(3):e01267.
- Ralls, K., Ballou, J.D., Dudash, M.R., Eldridge, M.D.B., Fenster, C.B., Lacy, R.C., Sunnucks, P., and Frankham, R. (2017). Call for a paradigm shift in the genetic management of fragmented populations. Conservation Letters 11(2):e12412.
- Rao, S. and Stephen, W.P. (2007). *Bombus* (*Bombus*) *occidentalis* (Hymenoptera: Apiformes): in decline or recovery? The Pan-Pacific Entomologist 83:360-362.
- Raymond, M., Rousset, F. (1995). An exact test for population differentiation. Evolution 49(6): 1280-1283.
- Reber-Funk, C., Schmid-Hempel, R., and Schmid-Hempel, P. (2006). Microsatellite loci for *Bombus* spp. Molecular Ecology Notes 6:83-86.
- Roffler, G.H., Schwartz, M.K., Pilgrim, K.L., Talbot, S.L., Sage, G.K., Adams, L.G., and Luikart, G. (2016). Identification of landscape features influencing gene flow: how useful are habitat selection models? Evolutionary Aapplications 9(6):805- 817.
- Rogers, B.M., Balch, J.K., Goetz, S.J., Lehmann, C.E.R., and Turetsky, M. (2020). Focus on changing fire regimes: interactions with climate, ecosystems, and society. Environmental Research Letters 15:030201.
- Rosche, C., Baasch, A., Runge, K., Brade, P., Träger, S., Parisod, C., and Hensen, I. (2022). Tracking population genetic signatures of local extinction with herbarium specimens. Annals of Botany: mcac061.
- Safner, T., Miller, M.P., McRae, B.H., Fortin, M.-J., and Menel, S. (2011). Comparison of Bayesian clustering and edge detection methods for inferring boundaries in landscape genetics. International Journal of Molecular Sciences 12(2):865-889.
- Salisbury, S.J., McCracker, G.R., Keefe, D., Perry, R., and Ruzzante, D.E. (2016). A portrait of a sucker using landscape genetics: how colonization and life history undermine the idealized dendritic metapopulation. Molecular Ecology 25(17):4126-4145.
- Schermelleh-Engel, K., and Moosbrugger, H. (2003). Evaluating the fit of structural equation models: tests of significance and descriptove goodness-of-fit measures. Methods of Psychological Research Online 8(2):23-74.
- Sheffield, C.S., Richardson, L., Cannings, S., Ngo, H., Heron, J., and Williams, P.H. (2016). Biogeography and designatable units of *Bombus occidentalis* Greene and *B. terricola* Kirby (Hymenoptera: Apidae) with implications for conservation status assessments. Journal of Insect Conservation 20: 189-199.
- Soroye, P., Newbold, T., and Kerr, J. (2020). Climate change contributes to widespread declines among bumble bees across continents. Science 367(6478):685-688.
- Spear, S.F., Balkenhol, N., Forti, M.-J., McRae, B.H. and Scribner, K. (2010). Use of resistance surfaces for landscape genetic studies: considerations for parameterization and analysis. Molecular Ecology 19(17):3576-3591.
- Stefenon, V.M., and Severo da Costa, L. (2012). A simulation stuy on the behavior of allelic richness and inbreeding coefficient over generations in fragmented populations of tree species. Annals of Forest Research 55(1):3-10.
- Storfer, A., Murphy, M.A., Evans, J.S., Goldberg, C.S., Robinson, S., Spear, S.F., Dezzani, R., Delmelle, E., Vierling, L., and Waits, L.P. (2007). Putting the 'landscape' in landscape genetics. Heredity 98:128-142.
- Stralberg, D., Wang, Xianli, Parisien, M.-A., Robinne, F.-N., Sólymos, P., Mahon, C.L., Nielsen, S.E., Bayne, E.M. (2018). Wildfire-mediated vegetation change in boreal forests of Alberta, Canada. Ecosphere 9(3): e02156.
- Strange, J.P., Knoblett, J., and Griswold, T. (2009). DNA amplification from pinmounted bumble bees (*Bombus*) in a museum collection: effects of fragment size and specimen age on successful PCR. Apidologie 40:134-139.
- Stolle, E., Rohde, M., Vautrin, D., Solignac, M., Schmid-Hempel, P., Schmid-Hempel R., and Moritz, R.A. (2009). Novel microsatellite DNA loci for *Bombuis terrestris* (Linnaeus, 1758). Molecular Ecology Notes 9(5):1345-1352.
- Teichroew, J.L., XU, J., Ahrends, A., Huang, Z.Y., Tan, K., and Xie, Z. (2017). Is China's unparalleled and understudied bee diversity at risk? Biological Conservation 210, Part B:19-28.
- Thomson, D. (2004). Competitive interactions between the invasive European honey bee and native bumble bees. Ecology 85(2):458-470.
- Tokarev, Y.S., Huang, W.F., Solter, L.F., Malysh, J.M., Becnel, J., and Vossbrinck, C.R. (2020). A formal redefinition of the genera *Nosema* and *Vairimorpha* (Microsporidia: Nosematidae) and reassignment of species based on molecular phylogenetics. Journal of Invertebrate Pathology 169:107279.
- Turner, M.G., Calder, W.J., Cumming, G.S., Hughes, T.P., Jentsch, A., LaDeau, S.L., Lenton, T.M., Shuman, B.N., Turetsky, M.R., Ratajczak, Z., Williams, J.W., Williams, A.P., and Carpenter, S.R. (2020). Climate change, ecosystems, and abrupt change: science priorities. Philosophical Transactions of the Royal Society B 375: 20190105.
- van Strien, M.J. Holderegger, R., and Van Heck, H.J. (2015). Isolation-by-distance in landscapes: considerations for landscape genetics. Heredity 114:27-37.
- Vieira, M.L.C., Santini, L., Diniz, A.L., and Munhoz, C.d.F (2016). Microsatellite markers: what they mean and why they are so useful. Genetics and Molecular Biology 39(3) https://doi.org/10.1590/1678-4685-GMB-2016-0027.
- Wang, I.J., Glor, R.E., and Losos, J.B. (2013). Quantifying the roles of ecology and geography in spatial genetic divergence. Ecology Letters 16:175-182.
- Wang, J., Santiago, E., and Caballero, A. (2016a) Prediction and estimation of effective population size. Heredity 117:193-206.
- Wang, T., Hamann, A., Spittlehouse, D., Carroll, C. (2016b). Locally downscaled and spatially customizable data for historical and future periods for North America. PLoS ONE 11(6):e0156720. doi:10.1371/journal.pone.0156720.
- Welty, J.L., Jefferies, M.I., Arkle, R.S., Pilliod, D.S., and Kemp, S.K. (2021). GIS clipping and summarization toolbox: U.S. Geological Survey Software Release, https://doi.org/10.5066/P99X8558.
- Westphal, C., Steffan-Dewenter, I., and Tscharntke, T. (2006). Bumblebees experience landscapes at different spatial scales: possible implications for coexistence. Community Ecology 149:289-300.
- Westra, S., Fowler, H.J., Evans, J.P., Alexander, L.V., Berg, P., Johnson, F., Kendon, E.J., Lenderink, G., and Roberts, N.M. (2014). Future changes to the intensity and frequency of short-duration extreme rainfall. Reviews of Geophysics 52(3):522- 555.
- Williams, P.H., Thorp, R., Richardson, L., and Colla, S. (2014). Bumble Bees of North America. An Identification Guide. Pp 115. Princeton University Press. Princeton, New Jersey.
- Williams, P.H. (2021). Not just cryptic, but a barcode bush: PTP re-analysis of global data for the bumblebee subgenus Bombus s. str. Supports additional species (Apideae, genus Bombus). Journal of Natural History 55:271-282.
- Williams, P.H., Brown, M.J.F., Carolan, J.C., An, J., Goulson, D., Aytekin, A.M., Best, L.R., Byvaltsev, A.M., Cederberg, B., Dawson, R., Huang, J., Ito, M., Monfared, A., Raina, R.H., Schmid-Hempel, P., Sheffield, C.S., Šima, P., and Xie, Z. (2012). Unveiling cryptic species of the bumblebee subgenus Bombus s. str. worldwide with COI barcodes (Hymenoptera: Apidae). Systematics and Biodiversity 10:21- 56.
- Willing, E.-M., Dreyer, C., and van Ooosterhout, C. (2012). Estimates of genetic differentiation measured by F_{ST} do not necessarily require large sample sizes when using many SNP markers. LPoS ONE 7(8):e42649.
- Whiteley, A.R., Fitzpatrick, S.W., Funk, W.C., and Tallmon, D.A. (2015). Genetic rescue to the rescue. Trends in Ecology and Evolution 30(1):42-49.
- Wotton, B.M., Flannigan, M.D., and Marshall, G.A. (2017). Potential climate change impacts on fire intensity and key wildfire suppression thresholds in Canada. Environmental Research Letters 12:095003.
- Wright, S. (1951). The genetical structure of populations. Annals of Eugenics 15:323- 354.
- Yves, R.l (2012). lavaan: An R Package for Structural Equation Modeling. Journal of Statistical Software, 48(2), 1-36. URL http://www.jstatsoft.org/v48/i02/.

Data Accessibility

All data generated for this study will be submitted for publication through the US Geological Survey. When the publication is complete, it will be assigned a DOI number and will be publicly available.

Benefit-sharing Statement

This study contributes to the body of information that will inform the upcoming Endangered Species Act listing decision in the USA. It is imperative that this decision is based on the best available science, as it is likely to have ecological and economic impacts throughout the range of the species.

This study would not have been possible without the generosity of the institutions in the USA and Canada who shared their *Bombus occidentalis* specimens with us. This study is an example of the powerful datasets that can be produced through such collaborations. Additionally, this study is a clear example of the value of developing and maintaining museum collections of biological specimens.

Author Contributions

Ashley Rohde conceived of the study, collected samples, analyzed data, and wrote the manuscript as part of her dissertation research. James Strange, Michael Branstetter, Karen Mock and Thomas Edwards provided guidance during the development and execution of the project through their roles on the dissertation committee. James Strange and Michael Branstetter secured funding for the support of Ashley Rohde and for research materials.

Tables

Table 3.1. The listed status of North American bumble bees of conservation concern. Bumble bees with an asterisk are parasitic bees from the subgenus *Psytherus*, whose declines parallel the decline of their host species, bumble bees from the subgenus *Bombus sensu stricto*.

Plex A			Plex B
Locus	Source	Locus	Source
B ₁₂₄	Estoup et al. 1995	B126	ü et al. 1995
B96	Estoup et al. 1996	BL ₁₃	Reber-Funk et al. 2006
BT30	Reber-Funk et al. 2006	BTERN02	Reber-Funk et al. 2006
BT28	Reber-Funk et al. 2006	BTMS0062	Stolle et al. 2009
BTERN01	Reber-Funk et al. 2006	BTMS0066	Stolle et al. 2009
BT10	Reber-Funk et al. 2006	BTMS0086	Stolle et al. 2009
BTMS081	Stolle et al. 2009	BTMS0059	Stolle et al. 2009
		BTMS0083	Stolle et al. 2009

Table 3.2. Fifteen microsatellite primers were identified for use from previously published literature.

Table 3.3. The measured and adjusted units of variables included in the structural equation models.

	Bombus occidentalis			Bombus mckayi		
Locus	χ^2	р	AR	χ^2	р	AR
B124	178.89	< 0.0001	2.59	231.1	< 0.0001	2.83
BTERN01	163.81	< 0.0001	3.41	146.82	< 0.0001	4.99
BT28	150.39	< 0.0001	2.1	207.39	< 0.0001	2.23
BT10	209.92	< 0.0001	2.98	109.16	< 0.0001	3.36
B96	53.35	0.005	2.01	51.71	0.044	2.9
BT30	172.46	< 0.0001	2.22	168.32	< 0.0001	2.02
BTMS081	158.75	< 0.0001	2.12	86.66	0.021	2.65
BTMS0066	118.79	< 0.0001	3.18	149.91	< 0.0001	4.47
BTMS0083	75.43	0.007	2.98	83.24	< 0.0001	3.52
B126	171.22	< 0.0001	2.81	144.03	< 0.0001	3.95
BTMS0062	223.35	< 0.0001	3.27	260.19	< 0.0001	6.27
BTERN02	177.5	< 0.0001	3.4	195.23	< 0.0001	5.11
BTMS0086	141.59	< 0.0001	1.62	218.73	< 0.0001	2.38
BL13	88.52	< 0.0001	1.76	233.41	< 0.0001	3.19
BTMS0059	86.87	< 0.0001	2.5	178.88	< 0.0001	3.65

Table 3.4. X^2 test of genotypic disequilibrium and rarefied allelic richness in the fifteen microsatellite loci used to genotype bumble bee specimens.

Table 3.5. Results of AMOVA analysis of genetic structure among geographic clusters for both species, including analyses using uncorrected Nei's genetic distance matrices and matrices corrected for time between sample collection. *p* is the significance of the relationship as described by a Monte-Carlo randomization test.

Table 3.6. Percent contribution of the measured environmental variables to the occupancy likelihood of *Bombus mckayi* and *Bombus occidentalis.* Only variables that contributed a minimum of one percent are listed. Percent

Bombus mckayi	Contributio n	Bombus occidentalis	Percent Contribution
mean max temp in September	25.7	mean precip in April	36.5
variation in min temp in April	18.5	variation in min temp in April	26.8
variation in max temp in September	12.9	mean precip in July	8
variation in max temp in June	12.5	variation in min temp in August	4.1
variation in min temp in May	6.3	variation in min temp in June	3.4
variation in max temp in May	5.1	variation in max temp in September	2.9
variation in precip in April	3.7	mean habitat quality	2.5
variation in precip in June	2.8	mean max temp in September	2.4
mean precip in July	2.1	elevation	2.1
mean precip in September	1.1	variation in min temp in September	1.7
elevation	1	variation in max temp in June	1.4
		variation in max temp in April	1.1
		variation in min temp in July	1
		mean precip in August	1
		mean average temp in July	1
Bombus occidentalis 1960 to 1994	Percent Contribution	Bombus occidentalis 1995 to 2019	Percent Contribution
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variation in min temp in April	34.2	variation in min temp in April	26.4
elevation	12.9	max temp in June	20.8
max temp in September	8.6	variation in max temp in June	15.7
variation in mean temp in April	8.4	variation in mean temp in July	7.7
variation in max temp in September	8.3	variation in mean temp in June	6.1
max temp in July	7.9	variation in min temp in August	3.8
variation in min temp in September	7.1	variation in mean temp in April	3
variation in max temp in May	2.7	variation in min temp in September	2.7
Variation in max temp in July	1.2	variation in max temp in August	2
variation in min temp in May	1.1	max temp in September	1.8
variation in mean temp in August	1.1	variation in max temp in May	1.7
		variation in max temp in September	1.6
		variation in min temp in May	1.5
		elevation	1.1

Table 3.7. Percent contribution of the measured environmental variables to the occupancy likelihood of *Bombus occidentalis* in two time categories, 1960 to 1994 and 1995 to 2019. Only variables that contributed a minimum of one percent are listed.

Model	$\chi^2(p)$	CFI	TLI	RMSEA	SRMR
Bombus mckayi	2249.50 (< 0.001)	0.960 0.933		0.067	0.044
Bombus occidentalis All Years	27805.4 (< 0.001) 0.922 0.886			0.069	0.050
Bombus occidentalis 1960 to 1994	3174.377 (< 0.001) 0.834 0.765			0.128	0.082
Bombus occidentalis 1995 to 2020	61785.502 (0.001)	0.677	0.603	0.142	0.108

Table 3.8. Scores of goodness of fit for each of the four structural equation models used in this analysis.

Table 3.9. Indirect influences of predictor variables on Nei's genetic differentiation. Paths from predictors through latitude and through elevation and latitude were measured. Variables with values of NA were not included in the model, variables with values in gray were not significant at $p = 0.05$.

Geographic Distance		Precipitation		Minimum Temperature	
Variable	Value	Variable	Value	Variable	Value
Least Cost distance	0.4	April	0.567	April	1.158
Random Walk resistance 0.199		May	0.477	May	1.077
		June	0.508	June	0.917
		July	0.037		
		August	0.022		
		September	0.462		

Table 3.10. Factor loadings of measured variables on latent variables in the model of *Bombus occidentalis* 1960 to 2020.

Figures

Figure 3.1. The historical ranges and recorded observations of *Bombus mckayi* (yellow range, grey observation records) and *Bombus occidentalis* (green range, black observation records).

Photo credits: *Bombus mckayi* Alistair Fraser, *Bombus occidentalis* Joyce Knoblett.

Figure 3.2. The number of microsatellite loci that successfully amplified in specimens of A) *Bombus occidentalis* and B) *Bombus mckayi* collected between 1904 and 2020. Small gray circles represent individual specimens, variably-sized black circles represent the mean number of loci amplified for all of the specimens from the year, the solid blue line represents the linear relationship between the number of loci amplified and the collection year, and the dashed red line is the cutoff of seven successfully amplified loci that was determined to represent a useful genotype for downstream analyses. The red n value is the number of specimens that had seven or more amplified loci.

Figure 3.3. Rarefied allelic richness of geographic clusters of *Bombus occidentalis* and *Bombus mckayi* throughout time. A and B) samples from 1960 to 2020 included in the linear regression. C and D) regressions from 1960 to 1994 (blue) and 1995 to 2020 (red).

Figure 3.4. The geographic distribution of genotyped specimens with seven or more microsatellite loci amplified, grouped into clusters (populations). Each pie represents a single cluster. Color divisions within pies represent the proportion of the clusters that was derived from one of two identified likely lineages. Top: *Bombus mckayi* (blues) and bottom: *Bombus occidentalis* (oranges). The *B. occidentalis* cluster framed in black had an excess of heterozygotes. Sizes of the pies are scaled to the average rarefied allelic richness across all loci within each cluster. Also included are bar graphs of the lineage assignments of individuals, each bar represents the likely lineages of an individual. Gray lines on the graphs demarcate clusters. Clusters were arranged from north to south (top to bottom) for *B. occidentalis* and from west to east (left to right) for *B. mckayi*.

Figure 3.5. A and B) FIS values of clusters of *Bombus occidentalis* and *Bombus mckayi* averaged across all loci and distributed throughout time from 1960 to 2020. C and D) global FST values of *B. occidentalis* and *B. mckayi* averaged across all loci among groups of clusters that were sampled within similar years. Samples were compared by singletailed one sample t-tests, red points represent clusters (F_{IS}) or groups of clusters (F_{ST}) that had values significantly higher than expected and black points represent clusters or groups of clusters with values that are not significantly different than expected for alpha $= 0.05$. The black lines, R2 values, and p values represent linear regressions of values throughout time for each statistic and species. Only populations that were represented with samples greater than one individual were able to be included in these analyses (*B. occidentalis* $N = 48$, *B. mckayi* $N = 36$).

Figure 3.6. Resistance distributions based on distribution of *Bombus mckayi* and *Bombus occidentalis* across elevation and averaged weather variables from 1960 through 2019. These time categories roughly represent before and after the observed decrease in abundance and range of *B. occidentalis*.

Figure 3.7. Resistance distributions based on the distribution of *Bombus occidentalis* across elevation and averaged weather variables in two time categories, 1960 to 1944 and 1995 to 2019. These time categories roughly represent before and after the observed decrease in abundance and range of *B. occidentalis*.

Bombus mckayi 1960 to 2020

Figure 3.8. Graphical representation of the structural equation models used to quantify the relative influences of environmental variables on genetic divergence among individuals across the range of *Bombus mckayi* (top) and *Bombus occidentalis* (bottom). Variables in boxes are measured and variables in ovals are latent. Thin black lines moving from latent variables to measured variables represent confirmatory factor analyses to define latent variables, thick solid black lines represent direct regressions, dotted gray lines represent non-significant direct regressions, colored dashed lines represent path analyses, and colored dotted lines represent non-significant indirect regressions. Colors of the indirect paths correlate with the labels in Table 3.9. Direct regressions onto variables other than genetic differentiation are provided in the supplemental materials.

MEASUREMENTS OF ATTITUDES AMONG STAKEHOLDERS TOWARD MOLECULAR TECHNIQUES IN BIOLOGICAL CONSERVATION STUDIES

CHAPTER IV

Abstract

Molecular methods are important tools for addressing biodiversity loss by identifying species and populations within species that have low or decreasing genetic diversity. However, adoption of molecular methods to inform conservation decisions has been slow in some areas, while traditional, often abundance-based, methods are used more frequently. This disparity is often called the conservation genetics gap. In order to bridge the gap, practitioners must overcome barriers to understanding and use of molecular methods in conservation studies and policy decisions. In this study, 974 conservationists from diverse backgrounds were surveyed to determine their levels of understanding and trust in molecular and traditional conservation studies that may represent barriers (or motivation) to action. This is the largest and most diverse sample of conservationists ever surveyed to assess attitudes toward conservation genetics. The results indicate that lack of understanding, but not trust, may be a barrier to increased use of molecular methods in these types of studies. However, comparisons of the data presented here to previous studies are hopeful that a shift in perception and increased use of molecular studies may be underway. Establishment of additional centralized conservation genetic research centers in underserved regions of the world, increased incentivization for conservation genetic researchers to communicate their science to

practitioners, and research targeting regionally important topics would improve understanding and utilization of molecular studies in conservation.

Introduction

Conservation science is an amalgamation of data-driven biological studies and social actions that attempt to address degradation in the conservation status of taxa or habitats, which is often caused by human activities (Robertson and Hull 2002; Mascia et al. 2003). However, in order to bridge the gap between data and action, practitioners must overcome three barriers: **understanding** of the results of studies, **trust** that the results are reliable, and **motivation** to change the current conservation status of the target taxa (Mascia et al. 2003; Schultz 2007).

Molecular techniques are being used increasingly often and are gaining impact in conservation studies by providing vital information about the standing genetic diversity within and gene flow among populations of at-risk species (Abdul-Muneer 2014; Kress et al. 2015; Shafer et al. 2015; Thomsen and Willerslev 2015; Corlett 2017; Holdregger et al. 2019). However, these studies are not intuitively easy to understand for practitioners and stakeholders who are not specifically trained to interpret their results. As such, many conservation partners are left out of conversations about these types of studies and the appropriate conservation actions that their results indicate (Keller et al. 2015; Taylor et al. 2017; Sandstrӧm et al. 2019; Klütsch and Laikre 2021). A lack of detailed understanding of the results of genetic conservation studies may lead to a sense of mistrust or helplessness that undermines motivation for action in some groups (Taylor et al. 2017; Sandstrӧm et al. 2019).

Applications of genetic tools have been shown to be under-utilized in conservation decisions (Hoban et al. 2013a and b; Shafer et al. 2015; Taylor et al. 2017; Sandström et al. 2019; Klütsch and Laikre 2021), especially outside of the United States (Taylor et al. 2017; Sandstrӧm et al. 2019) and despite the fact that conservation genetic studies are published at similar rates for all continents except Antarctica (Klütsch and Laikre 2021). Shafer and colleagues (2015) described categories of conservation applications for genetic tools and their relative level of use for conservation activities. They indicated that genetic tools are often used to measure population history, taxonomic identification, inbreeding detection, population census (abundance estimates), and kinship. However, studies of adaptive and quantitative genetic variation, population viability and genetic monitoring are rarely used for conservation studies (Shafer et al. 2015). These latter applications are most relevant to landscape-scale genetic questions that might inform species-level conservation decisions. Hoban et al. (2013a) found that only 24.8% of articles published in the scientific journal Conservation Genetics between 2000 and 2013 contained the word "management" and only 0.8% contained the word "policy." Taylor et al. (2017) found that genetics are only mentioned in conservation policy documents 50% of the time in Australia, 30% of the time in Canada, and 18% of the time in South Africa. Also, when genetics are mentioned in conservation policy documents, genetic diversity is the most commonly mentioned concept, and it is described so generically that it is effectively meaningless (Taylor et al. 2017). Similarly, Sandstrӧm et al. (2019), found that genetics were often mentioned in international and national policies on biodiversity conservation in the Baltic Sea, but were rarely mentioned in regional documents. Regional managers in the area indicated that

insufficient explanation of the policies, insufficient resources to implement them, and ambivalence toward the results of genetic studies were all barriers to their implementation.

Measurements of genetic diversity, gene flow or effective population size are less intuitively understandable than traditional studies of conservation status, which often involve simple counts of individuals or populations. Additionally, conservation practitioners who do not have experience with molecular studies or who have had experience with only poorly executed molecular studies may not trust the results of studies that use these methods for conservation. A possible lack of trust in molecular methods has been implied in previous studies, but never directly measured (Taylor et al. 2017; Sandstrӧm et al 2019; Klütsch and Laikre 2021). The results of traditional studies more easily overcome the barriers to conservation action than the results of molecular studies (Hoban et al. 2013a; Keller et al. 2015; Shafer et al. 2015; Hoffman et al. 2015; Richardson et al. 2016; Taylor et al. 2017). As a result, the insights provided by genetic studies into the likelihood of recovery and best recovery actions for the target species are not often used to inform conservation decisions (Keller et al. 2015; Shafer et al. 2015; Hoffman et al. 2015; Taylor et al. 2017). This phenomenon is widely known as the conservation genetics gap (Taylor et al. 2017; Britt et al. 2018; Sandström et al. 2019; Klütsch and Laikre 2021). While many studies acknowledge the conservation genetics gap (Haig et al. 2016; Britt et al. 2018; Klütsch and Laikre 2021) and offer suggestions for improving communication between researchers and other conservation practitioners, few actually measured the opinions of conservation practitioners about molecular and traditional methods (however see Taylor et al. 2017 and Sandström et al. 2019). Those

that did interview conservationists focused on professional resource managers in particular geographical regions or fields of interest; a small segment of the broader group represented by the broad category of conservationist. None addressed trust as a possible barrier to implementation separately from understanding (Taylor et al. 2017; Sandström et al. 2019).

The conservation genetics gap leaves practitioners with molecular results that are difficult to interpret and translate into action (Taylor et al. 2017; Sandström et al. 2019; Klütsch and Laikre 2021). This is opposed to traditional abundance or population counts, which are relatively easily interpreted and can be used to justify activities such as breeding programs (Harley et al. 2018), habitat protection (Dunk et al. 2019), or cessation of collection (Campbell et al. 2020) for rare species or lethal or non-lethal controls for invasive species (Fonner and Bohara 2017; Green and Grosholz 2020). Relatively abstract measurements of genetic diversity may decrease the understanding, trust, and motivation that are required to achieve conservation action within communities of stakeholders that actively participate in conservation actions or that are directly affected by conservation actions (Hoban et al. 2013a and 2013b; Keller et al. 2015; Funk et al. 2019, Holderegger et al. 2019; Klütsch and Laikre 2021). This disconnect between research and implementation in conservation is a great loss to conservation efforts, especially since the scientific community will undoubtedly continue to use and develop molecular techniques into the future (Hoban et al. 2013a and 2013b; Holdregger et al. 2019; Funk et al. 2019; Klütsch and Laikre 2021).

This study used a survey to measure the relative understanding, trust, and motivation to action of conservationists from multiple demographics in response to the results of molecular and traditional conservation studies. It bypassed the question of whether or not the general public is interested in or motivated to act on conservation issues; that topic has been well-studied in other places (Fischer and Young 2007; Schultz 2007, Byg et al. 2017). Instead, only members of organizations that are directly involved in or affected by conservation decisions were included in the study. Participants included professional researchers and resource managers, as well as volunteer conservationists and industry professionals who are affected by conservation policy. In this way, the pool of respondents was limited to individuals who have already demonstrated the understanding and motivation necessary to participate in conservation decisions and activities. This study compared the understanding and resulting action response to molecular and traditional studies by the largest and most diverse group of conservationists ever sampled about their perceptions of the value of molecular and traditional methods for conservation science. Differences in the application of results between genetic and genomic studies was not measured, though genomic studies have been shown to be applied to conservation questions less frequently than genetic studies (McMahon et al. 2014; Taylor et al. 2017). I addressed two hypotheses: First, stakeholders with little understanding of molecular techniques trust the results of molecular studies more than they trust traditional studies. Stakeholders with training in molecular techniques are more critical of the results of molecular studies than stakeholders with little or no training. Second, stakeholders with little understanding of molecular techniques are less likely to be motivated to action based on the results of molecular studies than they would be by traditional studies because they do not understand them as well. Stakeholders with training in molecular techniques are more likely to act on the results of molecular studies than stakeholders

without molecular training based on their relatively strong understanding of the conclusions of these studies.

Methods

Creation of a survey questionnaire

The survey questions were drafted using the technique described by Dillman (1978; 2000). The survey was multi-mode (administered via tablets and online) and contained both quantitative and qualitative questions (Table C1). Qualtrics survey management software was used to create and format the questionnaire. The questionnaire was approved by the Utah State University Institutional Review Board as protocol # 9724.

Solicitation of respondents

Organizations whose missions are directly affected by conservation actions and professionals who work in conservation science were contacted. Each identified organization representative was contacted twice by email, two weeks apart, or by telephone or social media if email was not available. The organization was not contacted again if there was no response after two attempts. If the organization representatives agreed to participate, they were asked to share the questionnaire with the members of their organization. The questionnaire was also submitted twice to potential participants, two weeks apart. Recommendations from the members of responsive organizations were requested for other groups that have an interest in conservation studies. Organization representatives were also encouraged to share the survey with anyone they thought might be interested in participating.

Questionnaire design

Questionnaires were anonymous. However, respondents were asked to disclose their organizational affiliations. The survey consisted of 16 multiple choice questions, 1 short answer question, and one descriptive question (Table C1).

Question 2 on the survey assessed the accuracy of the assumption that members of the organizations included in the sampling pool are interested in conservation activities, regardless of study-type. Answers of "Not very important" and "I do not know" were interpreted as low interest, all other answers were interpreted as high interest. Only respondents that indicated high interest were included in downstream analyses. The remaining questions assessed the respondents' opinions about the relative importances of species, their experience with molecular and traditional conservation studies, their formal education in conservation science, how they learn about conservation issues, and how they participate in conservation actions.

Data analyses

All of the quantitative data collected in this survey were categorical. Binomial exact tests were used to compare binomial responses, always with a predicted success rate of 0.5 and a confidence level of 0.95. Pearson's χ^2 tests were used to compare counts of response choices for nominal multiple-choice questions. A log-linear model was used to compare the levels of experience with molecular and traditional studies between respondents from the U.S.A. and from outside of the U.S.A. This was in response to a previous report which indicated that national policies in the U.S.A. promote the use of molecular methods more than other parts of the world (Taylor et al. 2017). Cochran's *Q* test was used to compare counts of the proportional categorical ranked question.

Understanding of conservation studies was estimated by summing the ranked scores from two questions (for traditional and molecular studies separately): the type of experience and level of training. Trust in the studies was determined by if the respondent believes the studies should be used to make conservation decisions. Motivation to action was determined using the number of conservation studies and the number of conservation meetings the respondents participated in (Table C1). Our measurement of motivation to action may be confounded with opportunity to participate in actions (or lack thereof). By restricting our survey pool to conservationists who are members of conservation organizations we hoped to minimize the effect of restrictions from opportunities to participate in actions, but that restriction may still exist for some individuals. Spearman's rank correlation coefficient (Spearman's *p*) was used to make correlations among variables to address the hypotheses.

Results

Survey respondent demographics

Overall, 974 respondents from 333 organizations returned surveys that contributed to this dataset (maximum from a single organization: 62, minimum: 1, mean: 2.5, standard deviation: 5.7). Surveys were collected primarily from organizations in the United States of America, which were targeted by the sampling scheme. However, respondents were encouraged to invite their collaborators to submit questionnaires, which resulted in 89 respondents associated with organizations from outside of the U.S.A (Figure 4.1, Table C2). Comparisons of the use of molecular and traditional conservation methods among countries was not the goal of this study, but previous literature have

indicated that national policies in the U.S.A encourage the use of molecular methods for conservation studies more strongly than policies in other countries (Taylor et al. 2017; Sandström et al. 2019). If this pattern existed in the dataset, uneven sampling between the U.S.A and other countries could produce misleading conclusions. However, a log-linear model indicated that frequencies of respondents reporting experience with different types of conservation studies (see Fig. 4.1 for options for types of experience) was not significantly different between the USA and all other countries $(z = -0.975, p = 0.330)$. Responses from all countries were analyzed together in downstream analyses.

Although our solicitation scheme targeted the leadership of organizations, who then disseminated our survey to members of their organizations, some respondents reported membership in organizations for which leadership did not explicitly agree to participate. Therefore, it is important to note that respondents were solicited to participate in this study through their memberships in conservation organizations because they are likely to have a stronger interest in biological conservation than the public at large, but their opinions are their own and do not represent the organizations with which they are associated. Response rates were consistently high for multiple-choice questions (max: 958, min: 882, mean: 921.8 standard deviation: 25.8). Eight hundred and seventy respondents answered the short answer question and 538 respondents answered the descriptive question.

Respondents to the survey were diverse in their experience and motivation for participating in conservation (Fig. 4.2). All respondents in the survey indicated that they were interested in biological conservation by rating it as very important $(N = 916)$ or somewhat important $(N = 58)$. Six hundred and thirty-eight respondents reported having

experience collecting data for traditional conservation studies (probability of an answer of yes: 0.67, *p* < 0.0001) and 332 respondents reported having experience collecting data for molecular conservation studies (probability of an answer of yes: 0.35 , $p < 0.0001$). Respondents had variable levels of training and participated in variable roles within conservation (Figure 2). Respondents who chose "other" for this question listed general interest, private land-owner and science administrator most often (Table C3). Significant χ^2 tests of the number of respondents that identified different roles in conservation, and different levels of experience and training demographics indicated uneven sampling across the measured demographics (Type of Experience: $\chi^2 = 254.4$, $p < 0.0001$, Training: $\chi^2 = 413.7$, $p < 0.0001$, Conservation Role: $\chi^2 = 167.6$, $p < 0.0001$, Figure 4.2, Table C4 for pairwise comparisons). In particular, many respondents had high levels of training in conservation or related biological sciences and few respondents participated in only molecular studies. One of the drawbacks of collecting data through voluntary surveys is that sampling is often uneven. Some respondents reported holding multiple roles in conservation. In this case, the first role reported was included in analyses.

Learning, interests, and actions taken by respondents

Most respondents reported that the primary way they learn about conservation science is through scientific papers (χ^2 = 600.45, *p* < 0.0001, Fig. 4.3). However, this statistic is skewed by the large group of respondents who hold graduate degrees. Training on how to access, read, and write scientific papers is often taught at the graduate level, and therefore is more likely to be an important source of information for respondents who hold a graduate degree. When respondents with graduate degrees were removed from the analysis $(\chi^2 = 94.45, p < 0.0001)$, scientific papers, scientific presentations, and popular

science articles were ranked as the most common ways to learn about conservation and were not significantly different from one another (for pairwise comparisons see Table C4). Respondents who chose "other" for this question got their information through direct correspondences with scientists, land managers or, in one case, a conservation consultant.

Respondents expressed diverse values when they were asked what characteristics make a species valuable. Six hundred and ninety-one respondents (72.9%) indicated that some species are more valuable than others and 256 respondents (25.6%) indicated that all species are equally valuable (probability of an answer of yes: 0.73 , $p \le 0.0001$). Respondents were asked to rank the importance of three characteristics in deciding which species to conserve, and most of them were in agreement that species that play an important role in their environments should be prioritized, followed by species that are unique. Most respondents did not think the value of species to humans should be the top priority for selecting species to conserve, but they did value it as second-most important in most cases (Fig. 4.4). The differences in the rankings of the importance of the three characteristics were significant $(Q = 918.0, p \le 0.0001)$.

Respondents were interested in conservation studies that focused on a wide variety of taxa (Fig. 4.5). Five hundred and eighty-six respondents described a conservation study in which they had participated. The most common taxa of interest by far were plants, while reported studies that focused on fungi and lichens were rare. The respondents' descriptions of the goals of the studies could be grouped into several classes based on life history characteristics of the taxa of interest. For example, some descriptions emphasized native or invasive species. However, 31 of the studies described

did not focus on any taxa in particular, but were landscape scale studies. Seven of those studies focused on habitat assessments, 11 focused on habitat restoration, 13 focused on water quality, and 15 were general biological diversity surveys (not included in Fig. 4.5). The differences in the number of studies that focused on each class of taxa (including landscape scale studies as a class) was significant (χ^2 = 745.88, *p* < 0.0001) including most of the pairwise comparisons (Table C4). The number of respondents that participated in studies that focused on different taxa characteristics was also significantly different $(\chi^2 = 1437, p < 0.0001,$ Table C3), with rarity ranked as the most common characteristic of interest and medicinal value ranked as the least common characteristic of interest.

A large majority of respondents indicated that both molecular and traditional studies should be used to inform conservation decisions at least some of the time (molecular: 98.2%, traditional: 99.3%). Sixteen respondents indicated that molecular studies should never be used for conservation decisions and 6 indicated that traditional studies should never be used, which was significantly different from the counts recorded for the other two options (molecular: $\chi^2 = 402.85$, $p < 0.0001$, traditional: $\chi^2 = 498.31$, $p <$ 0.0001). Significantly more respondents answered "yes" than "sometimes" to whether or not traditional studies should be used ("Yes" $N=518$, "Sometimes" $N = 374$), indicating a relatively strong level of trust in this type of study. Fewer respondents answered "yes" than "sometimes" for molecular studies ("Yes $N = 415$, "Sometimes" $N = 461$), but this relationship was not statistically significant (Table C4 for pairwise relationships).

Influence of understanding and motivation on conservation actions

Measures of understanding of traditional and molecular studies were positively correlated with respondents' level of motivation to participate in conservation actions (traditional: $r = 0.33$, $p < 0.0001$, molecular: $r = 0.17$, $p = 0.002$), though the effect sizes were not strong, especially for molecular studies. The amount that respondents trusted the results of conservation studies was not correlated with understanding or motivation to action for either type of study. The distribution of data points driving the correlations indicate different relationships between understanding and motivation for the study types (Fig. 4.6). Respondents that had high scores for understanding molecular studies had variable levels of motivation to action, while respondents that had high scores for understanding of traditional studies had mostly high motivation to action.

Discussion

Respondents experience with molecular and traditional conservation studies

Respondents reported having participated in more molecular conservation studies than has been previously described. The most frequently reported type of experience was with traditional studies, but the second most reported option was experience with both types. Many respondents who identified as traditional researchers reported having experience working on molecular studies. Also, many of the respondents who selfidentified as land managers indicated that they had worked with both molecular and traditional studies. These results indicate a shift in perception from previous studies, which found that the results of molecular research were far less likely to be implemented by managers or used by policy-makers than the results of traditional studies (Keller et al. 2015; Taylor et al. 2017; Sandström et al. 2019). The trends presented here may indicate

a shift away from the previously documented perception of molecular studies as financially out-of-reach, or unhelpful to address immediate conservation needs (Sandström et al. 2019; Klütsch and Laikre 2021) and towards normalization of the inclusion of molecular methods in conservation studies. However, previous studies included many fewer respondents and focused primarily on resource managers. Our results may simply indicate a different pattern in the broader conservation community that we sampled. Additionally, even respondents who reported the highest level of understanding of molecular studies reported mixed levels of motivation to action based on that understanding, which was in stark contrast to respondents with high levels of understanding of traditional studies, who reported being highly motivated to action (Fig. 4.6).

Most respondents who reported experience working on studies that use molecular methods held graduate degrees or 4-year degrees in conservation science. Almost no respondents with 2 years or fewer of formal training or who identified as nonprofessional conservationists had participated in molecular studies. It is possible that this contrast between professional and non-professional respondents' experience is driven by the necessity of training to work in genetics laboratories, which is often taught in 4-year programs and above. However, genetic studies require DNA samples from target taxa, which are collected using similar field methods to many traditional studies and can often be performed by technicians or volunteers with minimal training. For example, volunteers have been successfully deployed to collect environmental DNA in many studies (Julian et al. 2019; Larson et al. 2020; Meyer et al. 2021; Lavin 2022).

Respondents also indicated that they participated in conservation studies that focus on diverse taxa and various aspects of conservation (e.g. species rarity, invasive species, etc.). By far, the most commonly studied taxon that respondents mentioned was plants. Respondents were most likely to work on studies that focused on biological communities or rare species. Tissue or pollen samples could easily be collected from plants for use in molecular studies by conservationists with all levels of training. As the use of molecular methods in conservation science continues to grow, steps must be taken to overcome barriers to participation in these studies for conservationists from all backgrounds if the conservation genetics gap is to be closed. These steps may include ongoing education for conservationists without formal genetics training and including conservationists without genetics training at the sample collection stage of molecular studies, which is often similar to sample collection for traditional studies and also provides opportunity for communication and ongoing education about the uses of molecular methods.

Methods for learning about conservation

Unsurprisingly, most respondents with graduate degrees reported primarily learning about conservation research through scientific literature. However, when this group was removed from the analysis, scientific presentations and popular science articles became equally important sources of information as scientific papers. This result, taken in consideration with the finding that respondents with less than 4 years of formal training are unlikely to participate in molecular studies, indicates that diversification of the methods by which researchers disseminate their findings could improve understanding, especially for molecular studies.

Within professional science communities, a culture of "publish or perish" is pervasive (Moosa 2018; Kiai 2019; Coriat 2019; Chatterjee 2019; van Dalen 2021). However, the data presented here indicate that scientific papers may not be the only way or the best way to facilitate learning among all demographics of conservationists. A survey of science faculty at land grant universities across the U.S.A. indicated that academic researchers were interested in participating in science communication with diverse audiences (outside of their own colleagues), but that they were not encouraged and often felt indirectly discouraged from doing so by colleagues and administrators at their institutions (Rose et al. 2020). A call for a shift in academic culture to incentivize public science communication has been made and clearly documented (Bickford et al. 2012; Cook et al. 2013; Rose et al 2020), and we echo that call here. Also, including nonacademic authors on scientific papers has been shown to increase the likelihood that studies provide direct solutions to conservation problems or advice to end-users (Britt et al. 2018). A combination of scientific papers that include authors with diverse experiences, publicly accessible scientific presentations (carefully crafted for a lay audience, Bullock et al. 2019), and increased exposure to novel scientific ideas in popular media may contribute to increased understanding and motivation among conservationists to participate in studies that use conservation genetics.

Conservation priorities among conservationists

Even when respondents demonstrated a strong understanding of conservation methods, their motivation to action varied between molecular and traditional methods. This difference in response may be influenced by their personal values. The characteristics that respondents reported as important in species indicate a diversity of

values among them, which may present both challenges and opportunities to policymakers whose decisions attempt to reflect those values. The characteristics that respondents valued when choosing species to conserve coincided with several conservation frameworks (van Eeden et al. 2020) that were previously defined. Twenty seven percent of respondents indicated that all species were equally important. This belief coincides with compassionate conservation (Wallach et al. 2018), which combines aspects of the animal rights movement with conservation. Sixty seven percent of respondents agreed that the importance of a species to its environment is its most valuable characteristic. This philosophy coincides with traditional conservation (Soulé 1985), which focuses on maintaining native ecosystems due to their inherent value. Twenty eight percent of respondents ranked the uniqueness of a species as the most valuable characteristic. This philosophy represents conservation based on critical faunas analysis (Vane-Wright et al. 1991; Diaz et al. 2013), which prioritizes conservation of species which are evolutionarily or functionally unique within their ecosystems. This framework is still being developed (Cadotte et al. 2011; Mazel et al. 2018; Mazel 2019), but is also being used with increasing frequency to identify species for conservation (Cadotte et al. 2011; Aurelle et al. 2018; Hoelzel et al. 2019). Very few respondents, 4%, ranked importance to humans as the most valuable characteristic. However, most respondents (81%) ranked it as the second most valuable characteristic, indicating that it is highly valued (but not the top concern) for most respondents. This option represents functionalist conservation (Callicott et al. 1999) which focuses on maintaining the function of ecosystems so humans can continue to use them.

Recognition of the diversity of worldviews and interests among conservationists requires a re-contextualization of conservation problems to a viewpoint that considers the plurality of values and objectives represented in conservation communities (Pascual et al. 2017). Incorporation of multiple worldviews into conservation learning, actions, and policy-making will encourage ongoing learning and participation in studies that include both traditional and novel (often molecular) methods.

Trust in the results of conservation science

Most of the respondents indicated that they think the results of both molecular and traditional studies should be used at least some of the time. Previous studies that measure the causes of the conservation genetics gap did not specifically measure the impact of conservationists' trust of molecular study results on their motivation to participate in the studies or to incorporate results from them into policy decisions (Taylor et al. 2017; Sandström et al. 2019; Klütsch and Laikre 2021). The relatively strong trust of the results of conservation research expressed by conservationists of diverse backgrounds and priorities in this study is an encouraging result, and provides a firm foundation from which to continue work toward closing the conservation genetics gap through continued education.

Relationships among understanding, trust, and motivation in conservation

Our results do not support hypothesis 1, that trust of molecular and traditional studies influences conservationists' motivation to action. They do, however, support hypothesis 2, that level of understanding of molecular and traditional studies influences motivation to action. The relationship was stronger for traditional studies than molecular

studies (higher r^2 value, though neither was extremely high), indicating that there are still more barriers to motivating conservationists to participate in molecular studies than there are to motivating them to participate in traditional studies. One possible nuance that wasn't explored in this study is a differentiation between understanding of the meaning of the metrics used in molecular methods and understanding of the potential applications of those data to conservation issues. Previous studies found that one barrier to inclusion of the results of molecular studies in policy documents is the perception that the patterns documented in genetic studies are long-term, and therefore not relevant to immediate conservation decisions (Haig et al. 2006; Taylor et al. 2017; Klütsch and Laikre 2021). It is possible that respondents indicated a high level of understanding for molecular studies because they were familiar with the methods and metrics of genetics, but still did not understand the usefulness of these methods within the context of conservation. It is also possible that these people have a strong understanding of the methods, but prioritize relatively meager funds dedicated to conservation for other tasks over genetic work, such as habitat conservation or restoration. In subsequent surveys on this topic, direct measurement of conservationists' perceptions of the immediate applicability of the results of conservation genetics studies and their perceptions of the value of genetics studies relative to other conservation work when funding is limited would provide interesting insight into the drivers of the conservation genetics gap.

Recommendations for improving understanding of molecular methods

Suggestions have been made to improve understanding of conservation genetics tools in previous studies, and address similar themes. Taylor et al. (2017) suggest improved communication among conservation genetic researchers and policy-makers through networking events, improved scientific literacy among policy-makers through training events, and adjusted hiring practices to create a workforce of professional conservationists who are literate in the most current scientific methods. Sandström et al. (2019) recommended increased focus on integrating conservation genetics into regional conservation (as opposed to national or international policy), modifications to policy frameworks to specifically mandate conservation genetics, and financial resources specifically for implementation of conservation genetic studies. Both of these studies, as well as Haig et al. (2016) called for the creation of centralized units (e.g. conservation genetics hubs, national conservation genetics laboratories) where conservation genetics research is conducted and where managers and conservationists can easily obtain information (including raw genetic data and help understanding and interpreting results) that they need to use that research efficiently.

Taylor et al. (2017) and Sandström et al. (2019) identified the U.S.A. as the country most likely to employ molecular methods to make national conservation decisions. They credit this advancement to direct mandates from the U.S. Endangered Species Act to include the "best available science" in listing decisions, and to the U.S. Fish and Wildlife Service Conservation Genetics Laboratory. While it is true the U.S.A. has a strong national framework for including molecular data in conservation actions, international efforts in other parts of the world must not be discounted. Efforts through

the United Nations Convention on Biological Diversity (https://www.cbd.int/) call for political awareness of the value of genetic information for conservation, outline general applications for genetic information to conservation questions, and encourage international resource sharing. The European Molecular Biology Laboratory (https://www.embl.org/) conducts molecular biology, including research focusing on biodiversity, and researchers associated with the laboratory recently published a call for world-wide genetic data-sharing through a similar framework as their European Bioinformatics Institute (Scholz et al. 2022). Academic societies dedicated to conservation genetics exist and societies that focus on biodiversity more broadly often have subsections that focus on conservation genetics. Additionally, universities around the world host research programs that contribute directly to conservation genetics. All of these organizations fill the role of producing conservation research, possibly fulfilling the need for additional centralized conservation genetic research facilities in some places. However, these resources are not equitably distributed around the world (Culley et al. 2021; Titley et al 2021). If additional centralized conservation genetics research laboratories are to be established, as previous studies have recommended, resources should be focused on establishing them in underserved areas. Although there are many organizations that produce conservation research, they may not fill the role of providing a platform for interactions between researchers and conservationists who play different roles in conservation (e.g. managers, volunteers, private land-owners). Incentivizing these interactions at existing research institutions may help to improve understanding among diverse conservationists while leveraging resources available through those institutions and changing the perception among researchers that science communication is less

valuable than direct research. These interactions could also help researchers to identify species or ecosystems of particular interest to regional resource managers, which increases the likelihood that the results from those studies will be incorporated into onthe-ground conservation actions (Sandström et al. 2019).

This study concluded that understanding, and not trust, is likely to be the greatest barrier to the use of molecular information in conservation decisions. Ongoing efforts to narrow or close the conservation genetics gap should include improved education of conservation stakeholders about molecular methods. This improvement will require increased access to learning opportunities for decision-makers and conservation practitioners without molecular training, and increased incentivization for molecular researchers to engage in outreach.

References

- Abdul-Muneer, P.M. (2014). Application of microsatellite markers in conservation genetics and fisheries management: recent advances in population structure analysis and conservation strategies. Genetics Research International. Article ID 691759, 11 pages.
- Aurelle, D., Pratlong, M., Haguenauer, A., Brener-Raffalli, K., Toulza, E., Garrabou, J., Pontarotti, P., Linares, C., López-Sendino, P., Montero-Serra, I., Frias-Vidal, S., Ledoux, J.B. (2018). Bridging the gap between evolutionary and conservation biology: the case of a precious octocoral threatened by global change, the Mediterranean red coral. Second Joint Congress of Evolutionary Biology, 19-22 August 2018, Montpellier
- Bickford, D., Posa, M.R.C., Qie, L., Campos-Arceiz, A., Kudavidanage, E.P. (2012). Science communication for biodiversity conservation. Biological Conservation 151(1): 74-76.
- Britt, M., Haworth, S.E., Johnson, J.B., Martchenko, D., and Shafer, A.B.A. (2018). The importance of non-academic coauthors in bridging the conservation genetics gap. Biological Conservation 218: 118-123
- Bullock, O.M., Amill D.C., Shulman, H.C., Dixon, G.N. (2019). Jargon as a barrier to effective science communication: evidence from metacognition. Public Understanding of Science 28(7): 845-853.
- Byg, A., Martin-Ortega, J., Glenk, K., and Novo, P. (2017). Conservation in the face of ambivalent public perceptions- the case of peatlands as 'the good, the bad, and the ugly'. Biological Conservation 206: 181-189.
- Cadotte, M.W., Carscadden, K., and Mirotchnick, N. (2011). Beyond species: functional diversity and the maintenance of ecological processes and services. Journal of Applied Ecology. 48(5): 1079-1087.
- Callicott, J.B., Crowder, L.B., Mumford, K. (1999). Current normative concepts in conservation. Conservation Biology 13: 22-35.
- Campbell, S.J., Darling, E.S., Pardede, S., Ahmadiyya, G., Mangubhai, S., Amkieltiela, Estradivari, Eva, M. (2020). Fishing restrictions and remoteness deliver conservation outcomes for Indonesia's coral reef fisheries. Conservation Letters 13(2): e12698.
- Chatterjoee, D. (2019) Bean counting ignores structural inequalities. Nature human behavior 3: 1003-1004.
- Cook, C.N., Mascia, M.B., Schwartz, M.W., Possingham, H.P., and Fuller, R.A. (2013). Achieving Conservation Science that bridges the knowledge-action boundary. Conservation Biology 27(4): 669-678.
- Coriat, A.-M. (2019). PhD merit needs to be defined by more than just publications. Nature Human Behaviour 3: 1007.
- Corlett, R.T. (2017). A bigger toolbox: biotechnology in biodiversity conservation. Trends in Biotechnology 35(1): 55-65.
- Culley, T.M., Philpott, M., Tunison, R., Merritt, B.J., Sanchez, J.M.B., Wafer, A., and Holdren, R. (2021). Research inequity in the plant sciences. Applications in Plant Sciences 9(4): e11417.
- Díaz, S., Purvis, A., Cornelissen, J.H.C., Mace, G.M., Donoghue, M.J., Ewers, R.M., Jordano, P., and Pearse, W.D. (2013). Functional Traits, the phylogeny of function, and ecosystem service vulnerability. Ecology and Evolution 3(9): 2958- 2975.
- Dillman, D.A. (1978). Mail and telephone surveys: the total design method. New York: Wiley-Interscience.
- Dillman, D.A. (2000). Mail and internet surveys: the tailored design method, $2nd$ edition. New York: John Wiley and Sons, Inc.
- Dunk, J.R., Woodbridge, B., Schumacher, N., Glenn, E.M., White, B., LaPlante, D.W., Anthony, R.G., Davis, R.J., Halupka, K., Henson, P., Marcot, B.G., Merola-Zwartjes, M., Noon, B.R., Raphael, M.G., Caicco, J., Hansen, D.L., Mazurek, M.J., and Thrailkill, J. (2019). Conservation planing for species recovery under the Endangered Species Act: a case study with the Northern Spotted Owl. PLoS ONE 14(!): e0210643. https://doi.org/10.1371/journal.pone.0210643.
- Fischer, A. and Young, J.C. (2007). Understanding mental constructs of biodiversity: implications for biodiversity management and conservation. Biological Conservation 136(2): 271-282.
- Fonner, R., and Bohara, A.K. (2017). Optimal control of wild horse populations with nonlethal methods. Land Economics 93(3):390-412.
- Funk, W.C., Forester, B.R., Converse, S.J., Darst, C., and Morey, S. (2019) Improving conservation policy with genomics: a guide to integrating adaptive potential into U.S. Endangered Species Act decisions for conservation practitioners and geneticists. Conservation Genetics 20: 115-134.
- Green, S.J., and Grosholz, E.D. (2020). Functional eradication as a framework for invasive species control. Frontiers in Ecology and the Environment 19(2):98-107.
- Haig, S.M., Miller, M.P., Bellinger, R., Draheim, H.M., Mercer, D.M., and Mullins, T.D. (2016). The conservation genetics juggling act: integrating genetics and ecology, science and policy. Evolutionary Applications 9(1): 181-195.
- Harley, D., Mawson, P.R., Olds, L., McFadden, M., and Hogg, C. (2018). The contribution of captive breeding in zoos to the conservation of Australia's threatened fauna. In Recovering Australian Threatened Species. Eds: Garnett, S., Latch, P., Lindenmayer, D., and Woinarski, J. Csiro Publishing. Locked Bag 10, Clayton South VIC 3169, Australia.
- Hoban, S.M., Arntzen, J.W., Bertorelle, G., Bryja, J., Fernandes, M., Frith, K., Gaggiotti, O., Galbusera, P., Godoy, J.A., Hauffe, H.C., Hoelzel, A.R., Nichols, R.A., Pérez-Espona, S., Primmer, C., Russo, I.-R.M., Segelbacher, G., Siegismund, H.R., Sihvonen, M., Sjögren-Gulve, Vernesi, C., Vilá, Carles, Bruford, M.W. (2013a). Conservation genetic resources for effective species survival (ConGRESS): bridging the divide between conservation research and practice. Journal for Nature Conservation 21:433-437.
- Hoban, S.M., Hauffe, H.C., Pérez-Espona, S., Arntzen, J.W., Bertorelle, G., Bryja, J., Frith, K., Gaggiotti, O.E., Galbusera, P., Godoy, J.A., Hoelzel, A.R., Nichols, R.A., Primmer, C.R., Russo, I.-R., Segelbacher, G. (2013b). Bringing genetic

diversity to the forefront of conservation policy and management. Conservation Genetics Resources 5(2):593-598.

- Hoelzel, A.R., Bruford, M.W., and Fliescher, R.C. (2019). Conservation of adaptive potential and functional diversity. Conservation Genetics 20: 1-5.
- Hoffman, A., Griffin, P., Dillon, S., Catullo, R., Rane, R., Byrne, M., Jordan, R., Oakeshott, J., Weeks, A., Joseph, L., Lockhart, P., Borevitz, J. and Sgró, C. (2015). A framework for incorporating evolutionary genomics into biodiversity conservation and management. Climate Change Responses 2:1.
- Holderegger, R., Balkenhol, N., Bollinger, J., Engler, J.O., Gugerli, F., Hochkirch, A., Nowak, C., Segelbacher, G., Widmer, A., and Zachos, F.E. (2019). Conservation genetics: linking science with practice. Molecular Ecology 28(17): 3848-3856.
- Julian, J.T., Glenney, G.W., Rees, C. (2019). Evaluating observer bias and seasonal detection rates in amphibian pathogen eDNA collections by citizen scientists. Diseases of Aquatic Organisms 134: 15-24.
- Keller, D., Holderegger, R., van Strien, M.J., and Bolliger, J. (2015). How to make landscape genetics beneficial for conservation management. Conservation Genetics 16(3): 503-512.
- Kiai, A. (2019). To protect credibility in science, banish "publish or perish". Nature Human Behavior 3: 1017-1018.
- Klütsch, C.F.C, and Laikre, L. (2021). Closing the conservation genetics gap: Integrating genetics knowledge in conservation management to ensure evolutionary potential. In: Ferreira, C.C., Klütsch, C.F.C. (eds) Closing the knowledge-implementation gap in conservation science. Wildlife Research Monographs, vol 4. Springer, Cham. https://doi.org/10.1007/978-3-030-81085-6_3.
- Kress, W.J, García-Robledo, Uriarte, M., and Erickson, D.L. (2015). DNA barcodes for ecology, evolution, and conservation. Trends in Ecology and Evolution 30(1): 25-35.
- Larson, E.R., Graham, B.M., Achury, R., Coon, J.J., DAniels, M.K., Gambrell, D.K., Jonasen, K.L., King, G.D., LaRacuente, N., Perrin-Stowe, T.I.N., Reed, E.M., Rice, C.J., Ruzi, S.A., Thairu, M.W., Wilson, J.C., and Suarez, A.V. (2020). From eDNA to citizen science: emerging tools for the early detection of invasive species. Frontiers in Ecology and the Environment 18(4): 194-202.
- Lavin, J. (2022). Environmental DNA metabarcoding and citizen science as a costeffective and rapid tool for monitoring terrestrial mammalian species. MSc by research thesis, University of Salford, The Crescent, Salford, UK.
- Manel, S., Schwartz, M.K., Luikart, G., and Taberlet, P. (2003). Landscape genetics: combining landscape ecology and population genetics. Trends in Ecology and Evolution 18(4):189-197.
- Mascia, M. B., Brosius, J.P., Dobson, T.A., Forbes, B.C., Horowitz, L., McKean,M.A., and Turner, N.J. (2003). Conservation and the social sciences. Conservation Biology 17(3): 649-650.
- Mazel, B.K. (2019). Conservation of adaptive potential and functional diversity: integrating old and new approaches. Conservation GEnetics 20: 89-100.
- Mazel, F., Pennell, M.W., Cadotte, M.W., Díaz, S., Riva, G.V.D., Grenyer, R., Leprieur, F., Mooers, A.O., Mouillot, D., Tucker, C.M., and Pearse, W.D. (2018).
- McMahon, B.J., Teeling, E.C., and Höglund, J. (2014). How and why should we implement genomics into conservation? Evolutionary Applications 7:999-1007.
- Meyer, R.S., Ramos, M.M., Lin, M., Schweizer, T.M., Gold, Z., Ramos, D.R., Shirazi, S., Kandlikar, G., Kwan, W.-Y., Curd, E.E.,Freise, A., Parker, J.M., Sexton, J.P., Wetzer, R., Pentcheff, N.D., Wall, A.R., Pipes, L., Garcia-Vedrenne, A., Mejia, M.P., Moore, T., Orland, C., Ballare, K.M., Worth, A., Beraut, E., Aronson, E., Nielsen, R., Lewin, H.A., Barber, P.H., Wall, J., Kraft, N., Shapiro, B., Wayne, R.K. (2021). The CALeDNA program: Citizen scientists and researchers inventory California's biodiversity. California Agriculture 75(1).
- Moosa, I.A. (2018) Publish or perish: perceived benefits versus unintended consequences. Edward Elgar Publishing, Northampton, MA, U.S.A.
- Pascual, U., Balvanera, P., Díaz, S., Pataki, G., Roth, E., Stenseke, M., Watson, R.T., Dessane, E.B., Islar, M., Kelemen, E., Maris, V., Quaas, M., Sunramanian, S.M., Wittmer, H., Adlan, A., Ahn, A., Al-HAfedh, Y.S., Amankwah, E., Asah, S.T., Berry, P., Bilgin, A., Breslow, S.J., Bullock, C., Cáceres, D., Daly-Hassen, H., Figueroa, E., Golden, C.D., Gómez-Baggethun, González-Jiménez, D., Houdet, J., Keune, H., Kumar, R., Ma, K., May, P.H., Mead, A., O'Farrell, P., Pandit, R., Pengue, W., Pichis-Madruga, R., Popa, F., Preston, S., Pacheco-Balanza, D., Saarikoski, H., Strassburg, B.B., van den Belt, M., Verma, M., Wickson, F., Yagi, N. (2017). Valuing nature's contributions to people: the IPBES approach. Current Opinion in Environmental Sustainability 26-27: 7-16.
- Richardson, J.L., Brady, S.P., Wang, I.J., and Spear, S.F. (2016). Navigating the pitfalls and promise of landscape genetics 25:849-863.
- Robertson, D.P. and Hull, R.B. (2002). Beyond biology: toward a more public ecology for conservation. Conservation Biology 15(4): 970-979.
- Rose, K.M., Markowitz, E.M., and Brossard, D. (2020). Scientists' incentives and attitudes toward public communication. Proceedings of the National Academy of Science 117(3): 1274-1276.
- Sandström, Lundmark, C., Andersson, K., Johannesson, K., and Laikre, L. (2019). Understanding and bridging the conservation-genetics gap in marine conservation. Conservation Biology 33(3): 725-728.
- Scholz, A.H., Freitag, J., Lyal, C.H., Sara, R., Cepeda, L., Cancio, I., Sett, S., Hufton, A.L., Abebaw, Y., Bansal, K., Benbouza, H., Boga, H.I., Brisse, S., Bruford, M.W., Clissold, H., Cochrane, G., Coddington, J.A., Deletoille, A.-C., García-Cardona, F., Hamer, M., Hurtado-Ortiz, R., Miano, D.W., Nicholson, D., Oliveira, G., Bravo, C.O., Rohden, F., Seberg, O., Segelbacher, G., Shouche, Y., Sierra, A., Karsch-Mizrachi, I., de Silva, J., Hautea, D.M., de Silva, M., Suzuki, M., Tesfaye, K., Tiambo, C.K., Tolley, K.A., Varshney, R., Zambrano, M.M., and Overman, J. (2022). Multilateral benefit-sharing from digital sequence information will support both science and biodiversity conservation. Nature Communications 13: 1086.
- Schultz, P.W. (2007). Conservation means behavior. Conservation Biology 25(6): 1080- 1083.
- Shafer, A.B.A., Wolf, J.B.W., Alves, P.C., Bergström, L., Bruford, M.W., Brännström, I., Colling, G., Dalén, L., De Meester, L., Ekblom, R., Fawcett, K.D., Fior, S., Hajibabaei, M., Hill, J.A., Hoezel, A.R., Höglund, J., Jensen, E.L., Krause, J., Kristensen, T.N., Krützen, M., McKay, J.K., Norman, A.J., Ogden, R., Österling, E.M., Ouborg, N.J., Piccolo, J., Popović, D., Primmer, C.R., Reed, F.A., Roumet, M., Salmona, J., Schenekar, T., Schwartz, M.K.,Segelbacher, G., Seen, H., Thaulow, J., Valtonen, M., Veale, A., Vergeer, P., Vijay, N., Vilá, C., Weissensteiner, Wennerström, L., Wheat, C.W., Zieliński, P. (2015). Genomics and the challenging translation into conservation practice. Trends in Ecology and Evolution 30(2): 78-87.
- Soulé, M.E. (1985). What is conservation biology? BioScience 35: 727-734.
- Taylor, H.R., Dussex, N., and van Heezik, Y. (2017). Bridging the conservation genetics gap by identifying barriers to implementation for conservation practitioners. Global Ecology and Conservation 10: 231-242.
- Titley, M.A., Butchart, S.H.M., Jones, V.R., Whittingham, M.J., and Willis, S.G. (2021). Global inequities and political borders challenge nature conservation under climate change. Proceedings of the NAtional Academy of Sciences of the United States of America 118(7): e2011204118.
- Thomsen, P.F. and Willerslev, E. (2015). Environmental DNA- An emerging tool in conservation for monitoring past and present biodiversity 183: 4-18.
- Van Dalen, H.P. (2021). How to publish-or-perish principle divides a science: the case of economists. Scientometrics 126:1675-1694.
- van Eeden, L.M., Newsome, T.M., Crowther, M.S., Dickman, C.R., Bruskotter, J. (2020). Diverse public perceptions of species' status and management align with conflicting conservation frameworks. Biological Conservation 242: 108416.
- Vane-Wright, R.I., Humphries, C.J., and Williams, P.H. (1991). What to protect? Systematics and the agony of choice. Biological Conservation: 235-254.
- Wallach, A.D., Bekoff, M., Batavia, C., Nelson, M.P., and Ramp, D. (2018). Summoning compassion to address the challenges of conservation. Conservation Biology 32: 1255-1265.

Figures

Figure 4.1. The locations of individuals who submitted a questionnaire.

Figure 4.2. Measures of the diversity of experience of the respondents in the survey. A) The conservation role that the respondents reported holding, and B) the highest level of conservation or biological science training that the respondents reported having. Each bar is proportionately filled with the type of experience that the respondents reported.

Figure. 4.3. The primary way that conservationists learn about conservation issues.

Figure 4.4. Respondents' opinions of the importance of species characteristics in determining their conservation value.

Figure 4.5. Taxonomic groups of organisms that were included in respondents' descriptions of conservation studies in which they had participated.

Figure 4.6. Spearman's rank correlation of understanding and motivation to action of a) molecular and B) traditional conservation studies. Bubble size corresponds to the number of respondents with the same response (largest = 228 smallest = 1).

CHAPTER V

DISCUSSION

Bees are important pollinators world-wide and bumble bees in particular are important pollinators in high elevation and temperate biological communities. The importance of *Bombus occidentalis* as a pollinator within those communities is amplified by its expansive range and historically high abundance throughout that range. The observed decrease in abundance and range of *B. occidentalis* has the potential to cause cascading effects through the biological communities that it once inhabited (Tepedino 1979; Rollin et al. 2013; Parrey et al. 2021). The additional complexity associated with the unresolved taxonomic status of the species impedes attempts to assess the severity of the threat to the species continuation (Koch and Strange 2012; Williams et al. 2012; Sheffield et al. 2016; Williams 2021).

Through this dissertation, I contributed to the body of knowledge that informs ongoing attempts to conserve and protect *B. occidentalis* through four major findings: 1) *B. occidentalis* and *B. mckayi* are separate species, 2) genetic structure among populations of *B. occidentalis* has increased over time, but structure in *B. mckayi* remains stable, 3) minimum temperatures in the springtime and proximity to known infections from a fungal parasite influence genetic differentiation throughout the ranges of the twos species, and 4) conservationists whose goal is to preserve these and other at-risk species require on-going education to make the best conservation and management decisions possible.

My finding that the taxa previously recognized as *B. occidentalis* are actually two separate species is in agreement with the findings of Williams (2021), who used

mitochondrial *COI* barcoding to revise the species delimitations in the subgenus *Bombus sensu stricto*. However, my addition of expanded geographic sampling for the mitochondrial analyses and use of ultraconserved elements (UCEs) to compare the nuclear genomes strengthened the evidence for the described relationship and improved the level of confidence in this conclusion. Previous publications have indicated that there may be a hybrid zone where the ranges of the two species overlap (Sheffield et al. 2016). There was no evidence of hybridization in my analyses, but this could be due to undersampling of that geographic region. Unfortunately, sampling in that portion of the range is sparse, likely due to the relative remoteness of the region. Additional studies directly designed to search for hybridization of these closely-related species where the ranges overlap is an area of exciting research that could provide broad insights into the process of speciation in bumble bees.

My findings that genetic structure among populations of *B. occidentalis* is increasing over time, but that *B. mckayi* is currently relatively stable, is in agreement with previous studies (Lozier et al. 2011; Koch and Strange 2012). The task of tracking the gene flow among populations of an at-risk species requires ongoing sampling, and this study provided an update on previous findings. Additionally, this study expanded the geographic range of previous studies and compared measurements of genetic structure across a larger time frame than has been previously been included.

My findings that levels of genetic differentiation among specimens of both species were most strongly influenced by differences in springtime minimum temperatures and likelihood of exposure to the fungal parasite *Vairimorpha bombi* are supported by findings in previous studies (Cameron et al. 2016, Rohde and Pilliod 2021). However, previous studies made associations between abundance and these environmental factors, not genetic differentiation. This is the first study to detect direct causal relationships between the environmental variables and genetic differentiation for either variable in these species.

Finally, my survey of conservationists from around the world indicated that conservationists require additional education and training to better understand conservation studies that use molecular methods. Contrary to my expectation, there was no significant difference in the amount of trust that the respondents feel for the results of studies that use molecular methods compared to more traditional, often abundance-based, studies. This result may seem to wander from the theme of the other work in this dissertation, but application of results is a crucial component of conservation science. Adoption of molecular methods, such as the ones used in chapters 2 and 3, to inform conservation decisions has been slow in some areas, while traditional, often abundancebased, methods are used more frequently. This disparity is often called the conservation genetics gap. This study is one of only few (Taylor et al. 2017; Sandström et al. 2019) that directly measured the opinions of conservation practitioners about their understanding of molecular methods. It is by far the largest ($n = 974$), including people from diverse backgrounds and locations from around the world. It is also the first to measure the influence of trust, as well as understanding, on conservationists opinions of molecular studies.

The overarching goal of this dissertation was to assess the conservation status of *B. occidentalis* and *B. mckayi* using various molecular methods and to communicate the findings of that assessment to a community of interested conservation practitioners. My

research will contribute to an ever-growing body of literature that will, with a bit of luck,

provide the knowledge and motivation needed to protect these important species.

References

- Cameron, S.A., Lim, Haw Chuan, Lozier, J.D., Duennes, M.A., and Thorp, R. (2016). Test of the invasive pathogen hypothesis of bumble bee decline in North America. Proceedings of the National Academy of Sciences 113(16):4386-4391.
- Koch, J.B., and Strange, J.P. (2012). The status of *Bombus occidentalis* and *B. moderatus* In Alaska with special focus on Nosema bombi incidence. Northwest Science 86(3):212-220.
- Lozier, J.D., Strange, J.P., Stewart, I.J., and Cameron, S.A. (2011). Patterns of rangewide genetic variation in six North American bumble bee (Apidae: Bombus) species. Molecular Ecology 20:4870-4888.
- Parrey, A. H., Raina, R. H., Saddam, B., Pathak, P., Kumar, S., Uniyal, V. P., Gupta, D. and Khan, S. A. (2021). Role of bumblebees (Hymenoptera: Apidae) in pollination of high land ecosystem: a review. Agricult Rev https://doi. org/10.18805/ag.
- Rohde, A.T. and Pilliod, D.S. (2021). Spatiotemporal dynamics of insect pollinator communities in sagebrush steppe associated with weather and vegetation. Global Ecology and Conservation 29: e01691.
- Rollin, O., Bretagnolle, V., Decourtye, A., Aptel, J., Michel, N., Vaissiére, B.E. and Henry, M. (2013). Differences of floral resource use between honey bees and wild bees in an intensive farming system. Agriculture, Ecosystems, & Environment 179(1): 78-86.
- Sandström, Lundmark, C., Andersson, K., Johannesson, K., and Laikre, L. (2019). Understanding and bridging the conservation-genetics gap in marine conservation. Conservation Biology 33(3): 725-728.
- Sheffield, C.S., Richardson, L., Cannings, S., Ngo, H., Heron, J., and Williams, P.H. (2016). Biogeography and designatable units of *Bombus occidentalis* Green and *B. terricola* Kirby (Hymenoptera: Apidae) with implications for conservation status assessments. Journal of Insect Conservation 20:189-199.
- Taylor, H.R., Dussex, N., and van Heezik, Y. (2017). Bridging the conservation genetics gap by identifying barriers to implementation for conservation practitioners. Global Ecology and Conservation 10: 231-242.
- Tepedino, V.J. (1979). The importance of bees and other insect pollinators in maintaining floral species composition. Great Basin Naturalist Memoirs 3:139-150.
- Williams, P.H., Brown, M.J.F., Carolan, J.C., An, J., Goulson, D., Aytekin, A.M., Best, L.R., Byvaltsev, A.M., Cederberg, B., Dawson, R., Huang, J., Ito, M., Monfared, A., Raina, R.H., Schmid-Hempel, P., Sheffield, C.S., Šima, P., and Xie, Z. (2012). Unveiling cryptic species of the bumblebee subgenus *Bombus s. str.* worldwide with COI barcodes (Hymenoptera: Apidae). Systematics and Biodiversity 10:21- 56.
- Williams, P.H. (2021). Not just cryptic, but a barcode bush: PTP re-analysis of global data for the bumblebee subgenus *Bombus s. str.* Supports additional species (Apideae, genus *Bombus*). Journal of Natural History 55:271-282.

APPENDICES

APPENDIX A

Chaprter II supplemental tables and figures

Table A1. Collection and institutional information associated with the bumble bee specimens (1 of 4).

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Table A2. Collection and institutional information associated with the bumble bee specimens (2 of 4).

ExtractionID/ BOLD ID	COI Sample Name	UCE Sample Name
632 mckayi Alaska	632 mckayi Alaska	
657 mckayi Alaska	657 mckayi Alaska	
657 mckayi Alaska	657 mckayi Alaska	
ACUF11823-15	658 ACUFI1823-15 lucorum	
ACUF11824-15	658 ACUFI1824-15 lucorum	
ACUF11825-15	658 ACUFI1825-15 lucorum	
ACUF11833-15	658 ACUFI1833-15 lucorum	
ACUF11837-15	658 ACUFI1837-15 lucorum	
BBBO066-10	658 BBBO066-10 hypocrita	
BBBO252-10	658 BBBO252-10 hypocrita	
BBBO336-10	658 BBBO336-10 hypocrita	
BBBO372-10	658 BBBO372-10 hypocrita	
BBBO394-11	426 BBBO394-11 jacobsoni	
BBBO396-11	425 BBBO396-11 jacobsoni	
BBBO397-11	425 BBBO397-11 jacobsoni	
BBHEC139-09	658 BBHEC139-09 terricola	
BBHEC143-09	658 BBHEC143-09 terricola	
BBWP556-10	658 BBWP556-10 hypocrita	
BBWP557-10	658 BBWP557-10 hypocrita	
BEECD384-09	615 BEECD384-09 terricola	
BEECD387-09	633 BEECD387-09 terricola	
BEECD396-09	630 BEECD396-09 terricola	
BEECD397-09	648 BEECD397-09 terricola	
BEECD399-09 BEECD400-09	658 BEECD399-09 terricola 631 BEECD400-09 terricola	

ExtractionID/ BOLD							Author
ID	Species Name Bombus	Genus	Subgenus	Species	Author (Species)	Subspecies	(Subspecies) Ashmead
632 mckayi Alaska	occidentalis mckayi	Bombus		occidentalis	Greene, 1858	mckayi	1902
	Bombus						Ashmead
657 mckayi Alaska	occidentalis mckayi Bombus	Bombus		occidentalis	Greene, 1858	mckayi	1902 Ashmead
657 mckayi Alaska	occidentalis mckayi	Bombus		occidentalis	Greene, 1858	mckayi	1902
ACUF11823-15	Bombus lucorum	Bombus		lucorum	Linnaeus, 1761		
ACUF11824-15	Bombus lucorum	Bombus		lucorum	Linnaeus, 1761		
ACUFI1825-15	Bombus lucorum	Bombus		lucorum	Linnaeus, 1761		
ACUF11833-15	Bombus lucorum	Bombus		lucorum	Linnaeus, 1761		
ACUF11837-15	Bombus lucorum	Bombus		lucorum	Linnaeus, 1761		
BBBO066-10	Bombus hypocrita	Bombus		hypocrita	Pérez, 1905		
BBBO252-10	Bombus hypocrita	Bombus		hypocrita	Pérez, 1905		
BBBO336-10	Bombus hypocrita	Bombus		hypocrita	Pérez, 1905		
BBBO372-10	Bombus hypocrita	Bombus		hypocrita	Pérez, 1905		
BBBO394-11	Bombus jacobsoni	Bombus		jacbosoni	Skorikov, 1912		
BBBO396-11	Bombus jacobsoni	Bombus		jacbosoni	Skorikov, 1912		
BBBO397-11	Bombus jacobsoni	Bombus		jacbosoni	Skorikov, 1912		
BBHEC139-09	Bombus terricola	Bombus		terricola	Kirby, 1837		
BBHEC143-09	Bombus terricola	Bombus		terricola	Kirby, 1837		
BBWP556-10	Bombus hypocrita	Bombus		hypocrita	Pérez, 1905		
BBWP557-10	Bombus hypocrita	Bombus		hypocrita	Pérez, 1905		
BEECD384-09	Bombus terricola	Bombus		terricola	Kirby, 1837		
BEECD387-09	Bombus terricola	Bombus		terricola	Kirby, 1837		
BEECD396-09	Bombus terricola	Bombus		terricola	Kirby, 1837		
BEECD397-09	Bombus terricola	Bombus		terricola	Kirby, 1837		

Table A3. **Collection and institutional information associated with the bumble bee specimens (3 of 4).**

	Collection			Latitude	Longitude
ExtractionID/ BOLD ID	Year	Country	Full Locality	(DD)	(DD)
632 mckayi Alaska	2001	Canada	Nova Scotia		
657 mckayi Alaska	2005	Canada	Alberta		
657 mckayi Alaska	2002	Canada	Nova Scotia		
ACUFI1823-15	2009	Finland	Karelia ladogensis	61.475	29.578
ACUFI1824-15	2004	Finland	Nylandia	60.152	24.214
ACUFI1825-15	2001	Finland	Tavastia australis	61.448	23.555
ACUFI1833-15	2009	Finland	Nylandia	60.456	26.245
ACUFI1837-15	1998	Finland	Regio aboensis	60.243	21.305
BBBO066-10	1990	Russia	Ulaanbaatar		
BBBO252-10	2006	Japan	Hokkaido		
BBBO336-10	1989	Japan			
BBBO372-10	2009	Japan			
BBBO394-11	2008	China			
BBBO396-11	2007	China			
BBBO397-11	2007	China			
BBHEC139-09	2009	Canada	Newfoundland and Labrador	48.492	-54.022
BBHEC143-09	2009	Canada	New Brunswick	45.657	-65.015
BBWP556-10	2001	Russia	Sakhalinskaya	47.2553	142.806
BBWP557-10	2003	Russia	Oblast		
			Northwest		
BEECD384-09	2005	Canada	Territories Northwest	62.433	-114.35
BEECD387-09	2005	Canada	Territories	61.583	-117.149
BEECD396-09	2006	Canada	British Columbia	55.175	-126.361

Table A4. Collection and institutional information associated with the bumble bee specimens (4 of 4).

Locus	Gaps	A Count	C Count	G Count	T Count
$uce-3$	10353	14841	18432	14014	18040
$uce-4$	10856	16374	23831	21224	22043
$uce-5$	11094	18577	12758	15569	15362
uce-6	7786	18990	8606	9718	18124
uce-7	25509	29918	9848	14589	23904
$uce-8$	9767	19812	12440	14404	22537
$uce-9$	21362	31155	15796	12703	29640
$uce-10$	12109	18716	11273	11019	17891
$uce-11$	13739	20525	19386	20240	16998
$uce-12$	10784	14747	16306	17865	16626
$uce-13$	11080	26661	9715	10472	27136
$uce-18$	7848		8559		12291
		11069		11803	
$uce-19$	12312	21780	14531	15122	19085
$uce-20$	6635	12582	9817	11331	11940
$uce-22$	9511	16635	10319	11062	16835
$uce-23$	11707	14136	14309	13039	17113
$uce-24$	10735	17753	10289	12092	17115
$uce-27$	14959	16027	7797	8481	16296
$uce-29$	16064	19951	12617	13176	16816
$uce-30$	12626	21477	11296	15237	17876
$uce-31$	13971	19431	10744	10186	18356
$uce-32$	14367	24544	12537	13924	21820
$uce-33$	5929	15576	12121	12047	19511
$uce-34$	19714	28394	22042	22221	26685
$uce-35$	7364	17358	7616	6174	17270
$uce-36$	13775	16834	16487	15471	18017
$uce-37$	7979	13448	13982	13171	14588
$uce-38$	9883	25124	11094	12553	24282
$uce-39$	22616	25294	22369	16134	22619
$uce-40$	27216	27872	21773	23382	27437
$uce-42$	9977	17063	17819	14847	16174
$uce-43$	13737	16849	11690	15777	12897
$uce-44$	12128	21851	12658	15544	16667
$uce-46$	15920	24160	10512	12212	21924
$uce-47$	15528	18467	17077	13079	21921
$uce-48$	15864	23151	14794	17707	24300
$uce-49$	10965	18986	15359	13595	19439
$uce-50$	8371	16943	10899	10811	17695
$uce-51$	16675	29410	17641	25091	17639
$uce-54$	17242	19970	24119	18597	29888
$uce-55$	11433	16963	18386	16493	21150
$uce-57$	9283	16443	15960	20417	15177

Table A6. UCE summary statistics (1 of 2).

Table A7. SODA species assignments based on the maximum likelihood UCE phylogeny.

Specimen	Subspecies	Species Assignment	
Bombus occidentalis BLX1983	mckayi	0	
Bombus occidentalis BLX1987	mckayi	0	
Bombus occidentalis BLX1677	mckayi	$\mathbf{1}$	
Bombus occidentalis BLX1977	mckayi	1	
Bombus occidentalis BLX1679	mckayi	$\overline{2}$	
Bombus occidentalis BLX1715	mckayi	\overline{c}	
Bombus occidentalis BLX1736	mckayi	3	
Bombus occidentalis BLX1740	mckayi	3	
Bombus occidentalis BLX1739	mckayi	4	
Bombus occidentalis BLX1988	mckayi	4	
Bombus occidentalis BLX1743	mckayi	5	
Bombus occidentalis BLX1979	mckayi	5	
Bombus occidentalis BLX1674	mckayi	6	
Bombus occidentalis BLX1980	mckayi	6	
Bombus occidentalis BLX1742	mckayi	7	
Bombus occidentalis BLX1978	mckayi	7	
Bombus occidentalis BLX1671	mckayi	8	
Bombus occidentalis BLX1744	mckayi	8	
Bombus occidentalis BLX1735	mckayi	9	
Bombus occidentalis BLX1738	mckayi	9	
Bombus occidentalis BLX1986	mckayi	9	
Bombus occidentalis BLX1668	occidentalis	10	
Bombus occidentalis BLX1716	occidentalis	10	
Bombus occidentalis BLX1724	occidentalis	10	
Bombus occidentalis BLX1731	occidentalis	11	
Bombus occidentalis BLX1733	occidentalis	11	
Bombus occidentalis BLX1669	occidentalis	12	
Bombus occidentalis BLX1722	occidentalis	12	
Bombus occidentalis BLX1687	occidentalis	13	
Bombus occidentalis BLX1717	occidentalis	13	
Bombus occidentalis BLX1719	occidentalis	14	
Bombus occidentalis BLX1729	occidentalis	14	
Bombus occidentalis BLX1718	occidentalis	15	
Bombus occidentalis BLX1726	occidentalis	15	
Bombus occidentalis BLX1721	occidentalis	15	
Bombus occidentalis BLX1732	occidentalis	15	
Bombus occidentalis BLX1666	occidentalis	15	
Bombus occidentalis BLX1975	occidentalis	15	
Bombus occidentalis BLX1667	occidentalis	16	
Bombus occidentalis BLX1720	occidentalis	16	
Bombus occidentalis BLX1730	occidentalis	16	
Bombus occidentalis BLX1723	occidentalis	16	

Table A8. PTP, mPTP, ABGD and ASAP group assignments for maximum likelihood COI barcoding sequences four, five, six and seven species are identified. PTP and mPTP found solutions for five species. ABGD found solutions for four, five and seven species, with four as the most likely. ASAP found solutions for four, five, six, and seven species, with four identified as the most likely.

Table A9. mPTP and GMYC species assignments based on the Bayesian COI barcoding phylogeny.

APPENDIX B

Chapter III supplemental tables and figures

Table B1. The geographical clusters included in the population genetics analyses. GeoCluster is the cluster number, *N* is the number of specimens included in the cluster, *N* no siblings is the number of individuals in the cluster after all but one sibling from each sibling set was removed, SibSets is the number of sibling sets within each geographic cluster, and Species is the species that the cluster represents.

Number of	specimens Land cover Class Description	Absolute Rank	Weighting factor	Weighted Rank
2156	Tree cover, needleleaved, evergreen, closed to open (>15%)	1	0.4546	980.0413
898	Herbaceous cover	$\overline{2}$	0.1893	170.0198
586	Mosaic tree and shrub (>50%)/herbaceous cover (<50%)	3	0.1236	72.4006
337	Shrubland	4	0.0711	23.9445
158	Grassland	5	0.0333	5.2633
119	Urban Areas	6	0.0251	2.9857
98	Tree cover, needleleaved, evergreen, closed (>40%)	7	0.0207	2.0249
94	Tree cover, flooded, fresh or brakish water	8	0.0198	1.863
83	Bare areas	9	0.0175	1.4525
63	Water bodies	10	0.0133	0.8368
35	Mosaic natural vegetation (tree, shrub, herbaceous cover)(>50%)/cropland(<50%)	11	0.0074	0.2583
34	Tree cover, mixed leaf type(broadleaved and needleleaved)	12	0.0072	0.2437
29	Tree cover, broadleaved, deciduous, closed to open (>15%)	13	0.0061	0.1773
28	Cropland, rainfed	14	0.0059	0.1653
11	Shrub or herbaceous cover, flooded, fresh/saline brakish water	15	0.0023	0.0255
5	Sparse vegetation (tree, shrub, herbaceous cover) (<15%)	16	0.0011	0.0053
3	Tree cover, Broadleaved, deciduous, closed (>40%)	17	0.0006	0.0019
$\overline{2}$	mosaic cropland (>50%)/natural vegetation (tree, shrub, herbaceous cover) (<50%)	18	0.0004	0.0008
2	Tree cover, needleleaved, deciduous, closed to open (>15%)	18	0.0004	0.0008
1	Tree cover, broadleaved, evergreen, closed to open (>15%)	19	0.0002	0.0002

Table B2. Weighted ranks of Land cover values used to create habitat quality values.

Figure B1. The Delta K values reported for each tested value of K for the three Structure analyses.Asterisks highlight values of K that had relatively high probabilities of representing the true structure within each analysis. The value of K with the highest support was accepted for these analyses, but the other highlighted values may represent within-species hierarchical structure.

APPENDIX C

Chapter IV supplemental tables and figures

Table C1. The questions and response options included in the questionnaire.

16. If none of the above, please explain Short answer None

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Organization	# of surveys
Acadiana Native Plant Project	3
Acushnet Conservation Commission	1
Affiliation not provided	17
Agricultural University of Tirana, Albania	1
Agriculture and Agri-Food Canada	$\mathbf{1}$
Alaska Center for Conservation Science	1
Alaska Department of Fish and Game	1
Alaska Plant Materials Center	1
Allegheny Land Trust	1
Altar Valley Conservation Alliance	2
Arizona Game and Fish Department	1
Arizona Master Naturalist Association	$\mathbf{1}$
Arizona Native Plant Society	62
Arizona Site Steward Program	$\mathbf{1}$
Arizona State University	1
Arizona-Sonora Desert Museum	2
Arkansas Department of Transportation	1
Arkansas Master Naturalist	2
Arkansas Native Plant Society	30
Arkansas Natural Heritage Commission	2
Audubon Society	2
Australian Institute of Agricultural Science	1
Avalonia Land Conservancy, Inc.	3
Baton Rouge Audubon Society	1
bear river land conservancy	1
Bee Monitoring RCN	1
Biological Society of Western Pennsylvania	1
Boise Foothills Learning Center	1
Bolton Land Trust	1
Botanical Society of Western Pennsylvania	16
Bournemouth University	1
Boyd Woods Audubon Sanctuary (belongs to Litchfield Hills Audubon Society)	1

Table C2. The number of respondents from each organization that participated in the survey.

Table C3. Self-identified conservation roles of respondents that selected "other" on their questionnaire.

Table C4. The pairwise correlations associated with the Pearson's rank correlations reported in the main text. A) Role in conservation, B) training in conservation, C) type of experience, D) conservation topic of interest, E) type of experience, F) way of learning about conservation (respondents with graduate degrees included), G) way of learning about conservation (respondents with graduate degrees excluded), H) conservation decisions based on molecular studies, I) conservation decisions based on traditional studies.

A) Role in Conservation

B) Training in Conservation

C) Taxonomic Groups of Interest

D) Conservation topic of interest

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E) Type of Experience

F) Way of Learning about Conservation (Respondents with graduate degrees included)

Other	No Response	4.36		0.11
Television or Radio	No Response	0.49		0.56
Television or Radio	Other	7.69	1	0.022
Popular Articles	No Response	66.28	1	${}_{0.0001}$
Popular Articles	Other	39.06	1	${}_{0.0001}$
Popular Articles	Television or Radio	76.27	$\mathbf{1}$	${}_{0.0001}$
Scientific Presentations	No Response	83.10	1	${}_{0.0001}$

G) Way of learning about Conservation (Respondents with graduate degrees excluded)

H) Conservation decisions based on molecular studies

Group 1	Group 2	γ^2	df	Adjusted p
N ₀	Sometimes 415.15			${}_{0.0001}$
N ₀	Yes	369.38		${}_{0.0001}$
Sometimes	Yes	2.42		0.12

I) Conservation decisions based on traditional studies

CURRICULUM VITAE

Ashley T. Rohde

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Education:

- Bachelor of Arts, Department of Biology, Kalamazoo College, Kalamazoo, MI Major: Biology Minor: Spanish Literature Graduation Date: June 2006
- Masters of Science, Department of Biological Sciences, Boise State University, Boise, ID Defense Date: March 2014 Graduation Date: May 2014
- Doctorate of Philosophy, Department of Wildland Resources, Utah State University, Logan, UT

Defense Date: April 2022

PhD Dissertation Research:

Conservation genetics of a declining bumble bee in western North America; the influence of geography, dispersal limitation, and anthropogenic activity

Landscape genetics is an interdisciplinary field that combines aspects of population genetics, landscape ecology, and spatial statistics to measure genetic discontinuities and diversity patterns across landscapes and to correlate them with environmental features. Over the course of two independent studies, I used landscape genetic techniques to measure the gene flow and genetic structure across the species-wide distribution of a native bumble bee species of conservation concern in North America (*Bombus occidentalis*, the western bumble bee). In a third study, I surveyed over 1,000 conservationists from across the United States and Canada to determine how the results of molecular ecology studies (for any species) are understood and applied to conservation actions and policy decisions.

Chapter 1: Genome-wide nuclear markers reveal patterns of genetic structure in putative subspecies of a bumble bee species of conservation concern (*Bombus occidentalis* **Greene)**

Bombus occidentalis was once common in western North America, but comparisons of early museum records and studies from before 1997 against recent museum records and collections indicate that populations have declined dramatically along the west coast and in the Rocky Mountains and Intermountain West since the mid 1990's. To complicate the problem further, previous studies are conflicted about the species status of *Bombus occidentalis* Greene with up to five subspecies proposed, and two subspecies (*B. occidentalis occidentalis* and *B. occidentalis mckayi*) most commonly accepted. The two most commonly recognized subspecies are broken primarily into northern (*B. occidentalis mckayi*) and southern (*B. occidentalis occidentalis*) taxa, with a geographic overlap in the distributions in northern British Columbia, Canada. A recent phylogenomic study using COI-barcoding found support to elevate *B. occidentalis mckayi* to species status. Species definition for these taxa is

particularly important because it is currently under consideration for listing as endangered by the Endangered Species Act in the USA and has some protection under the Species At Risk Act in parts of Canada. Decisions regarding appropriate protections for *Bombus occidentalis* may be influenced by the inclusion or exclusion of *B. occidentalis mckayi* in those decisions, as evidence suggests decreases in population abundances and ranges are primarily occurring within the range of *B. occidentalis occidentalis*, while populations of *B. occidentalis mckayi* appear to be stable at this time. I genotyped 100 specimens of *B. occidentalis* from across the range of the species, including both putative subspecies, using ultra-conserved elements (UCEs). These nuclear markers are found throughout the genomes of most plants and animals, and provide a complementary dataset to the existing COI-barcode dataset, which is derived from a single, quickly evolving gene. Analysis of this dataset is ongoing, but preliminary results support two distinct clades that represent *B. occidentalis occidentalis* and *B. occidentalis mckayi*.

Chapter 2: The influence of geography, dispersal limitation, and anthropogenic change on the population genetic characteristics of a bumble bee of conservation concern (*Bombus occidentalis***) over 80 years in western North America**

Although a pattern of decrease in abundance for *B. occidentalis* has been clearly demonstrated in previous studies, the cause of the decline remains uncertain and new studies are required to determine if the decline is ongoing. I used observation records, museum specimens from 13 independent institutions, and spatial environmental data to measure changes in genetic structure, genetic diversity, and patterns of gene flow of *Bombus occidentalis* from 1960 to 2020. I genotyped over 2,000 specimens collected from across the entire range of *B. occidentalis*, from Alaska to New Mexico using microsatellite ISSR analysis. I compared measures of genetic inbreeding among populations throughout time to determine if populations are becoming more isolated. I used spatial models along with environmental predictor variables including weather, geography, and land use data to estimate landscape resistance among populations. Finally, I used structural equation models to compare genetic distances among populations to resistance distances and estimate the relative influence of the predictor variables on gene flow across the landscape. Analysis of this dataset is ongoing.

Chapter 3: Measuring attitudes among stakeholders toward molecular techniques in conservation studies

Conservation science is an amalgamation of data-driven biological studies and social actions. In order to bridge the gap between data and action, practitioners must overcome two barriers: understanding of the results of studies and motivation to change the current conservation status of the target taxa. Molecular techniques are being used increasingly commonly and to great effect in conservation studies. However, these studies are not intuitively easy to understand for practitioners and stakeholders who are not specifically trained to interpret their results. A lack of detailed understanding of the results of genetic conservation studies may lead to a sense of helplessness that undermines motivation for action in some groups. I surveyed conservationists from government and non-government conservation organizations in the USA and Canada to determine how members learn about conservation issues, how well they understand molecular methods used in conservation studies, and how scientists can better communicate the results of these studies. Analysis of this dataset is ongoing.

Other Ongoing Research:

Analysis of population structure of several widespread bumble bee species across their ranges determines current conservation status and potential for future decline

In collaboration with Dr. James Strange of the Ohio State University, I worked on a team that collected bumble bees from across the western United States and Canada in the summers of 2017 through 2019, visiting historical and previously unsampled sites. Dr. Strange and I will genotype these specimens, measure changes in population structure, and use community genetics methods to measure genetic interactions among species within communities.

Development of a novel method to detect environmental DNA left on flowers by insect pollinators

In collaboration with Dr. David Pilliod and Matthew Laramie at the USGS Forest and Rangeland Ecosystem Science Center, as well as a large network of collaborators from multiple federal agencies, I am working to develop methods for collecting insect pollinator DNA from previously visited flowers. We are currently conducting field and controlled garden studies to optimize collection methods and compare detection probabilities using qPCR methods (to detect single target species) and metabarcoding methods (broad taxonomic identification).

Master's Thesis Research:

Influence of wildfire disturbance and post-fire seeding on vegetation and insects in sagebrush habitats.

I conducted the first investigation of insect community response to post-fire seeding on public rangelands by comparing the composition of insect communities at burned-and-seeded and burned-and-unseeded sagebrush-steppe ecological sites in southwestern Idaho to unburned areas. I captured and identified 24,862 insects to the level of family (129 families) at three burned areas over two years. Insect communities in burned plots were not similar to those in unburned plots, regardless of treatment. Treated plots had insect communities with greater inter-annual variability in composition, suggesting they may be less stable than communities in unburned or burned-and-unseeded plots. This study was published in the journal Insect Conservation and Diversity (see publications below).

Work Experience:

Post-doctoral Researcher: April 2022 to present, 40 hours per week New Mexico State University

 Development and execution of original research related to the conservation status of pollinating insects

Student Trainee*: June 2020 to December 2020, 40 hours per week United States Geological Survey *Continuation of my PhD dissertation work started in 2017 under the title of Research Assistant at Utah State University

Research Assistant: January 2017 to June 2020, 40 hours per week January 2021 to April 2022, 20 hours per week Utah State University and USDA-ARS Pollinating Insect Laboratory

- Use of microsatellite markers and UCEs to address hypotheses regarding the conservation status of *Bombus occidentalis*, a bumble bee of conservation concern in the USA and Canada
- Use of specialized softwares including Geneious (microsatellite marker scoring), ArcGIS (mapping), Survey123 (data collection) MaxEnt (Species distribution modeling), R (statistical analyses, particular packages of interest include ResistanceGA, Spagedi, and LAVANT), Phyluce (analysis of UCE datasets), and various softwares used to clean and analyze microsatellite genotype datasets (Colony, HP-Rare, BOTTLENECK, FSTAT, GENEPOP, MaxEnt)
- DNA extraction and benchtop laboratory genetic sample preparation including amplification using PCR
- Organization and preparation of collecting permits, field computers, forms, and gear in 2017 and 2018
- Field collection of bees and vegetation data, summer 2017 and 2018
- Organization of insect samples, including labeling, identification, and databasing into the USDA Pollinating Insect Collection Database
- Procurement of loaned bumble bee specimens from 13 institutions across the United States and Canada for use in genetic analyses
- Preparation and submission of manuscripts for publication in peer-reviewed scientific journals
- Preparation and submission of grant proposals
- Coordinator of the 2017 BOMBUSS conference, sponsored by USDA-ARS (BOMBUSS)

Science Reporter: August 2018 to August 2020, 10 hours per week Utah Public Radio

- Interviews of scientists and other subjects for science stories
- Production of two-minute science news segments for radio broadcast
- Production of six-minute feature science news segments for radio broadcast and for original series including Women 20/20, Diagnosed, and Driven to Succeed
- Winner of four first-place awards from 2019 to 2021 from the Society of Professional Journalists in various categories for science-based stories
- All stories can be accessed at www.upr.org

Ecologist: May 2014 to December 2017, 40 hours per week

United States Geological Survey

- Design and implementation of protocols for field data collection
- Insect and vegetation sample management and identification
- Direction of biological technicians
- Data Analysis using programs such as R, SAS, PCOrd, and HyperNiche
- Assistance in writing proposals, reports, and peer reviewed papers

Ecological SCEP Student: August 2009 to May 2014, 30 hours per week from Sept. through May, 40 hours per week from May through August United States Geological Survey

- Identification of terrestrial and aquatic arthropods
- Technical writing including proposals and study plans
- Vegetation identification and arthropod trapping

Relevant Work Experience Prior To August 2009:

Quality Management Microbiologist: August 2008 to May 2009 IEH Technologies

Biological Technician: May 2008 to August 2008 and May 2009 to August 2009 United States Geological Survey

Chemistry Technician: August 2007 to February 2007 Cephalon Pharmaceuticals, Quality Control Laboratory

Biological (June Sucker) Technician: June 2007 to July 2007 Utah State Dept. of Natural Resources, Wildlife Resources

ESR Technician/Fuels Reduction Technician: June 2006 to May 2007 Eastern Nevada Landscape Coalition

Publications:

Rohde, A.T., and Pilliod, D.S. (2021). Spatiotemporal dynamics of insect pollinator communities in sagebrush steppe associated with weather and vegetation. Global Ecology and Conservation (accepted, in press).

Graves, T.A., Janousek, W.M., Gaulke, S.M., Nicholas, A.C., Keinath, D.A., Bell, C.M., Cannings, S., Hatfield, R.G., Heron, J.M., Koch, J.B., Loffland, H.L., Richardson, L.L., **Rohde A.T.**, Rykken, J., Strange, J.P., Tronstad, L.M., and Sheffield, C.S. (2020). Western bumble bee: declines in the continental United States and range-wide information gaps. Ecosphere 11(6):e03141.

Rohde, A.T., D.S. Pilliod, and Novak, S.J. (2019). Insect communities in big sagebrush habitat are altered by wildfire and post-fire restoration seeding. Insect Conservation and Diversity 12:216-230.

Pilliod, D.S., **Rohde, A.T.**, Charnley, S., Davee, R.R., Dunham, J.B., Gosnell, H., Grant, G.E., Hausner, M.B., Huntington, J.L., Nash, C. (2017). Survey of beaver-related restoration practices in rangeland streams of the western USA. Environmental Management 61(1):58- 68.

Pilliod, D.S., and **Rohde, A.T.** (2016). Insect community responses to climate and weather across elevation gradients in the Sagebrush Steppe, eastern Oregon: U.S. Geological Open-File Report 2016–1083, 50 p., https://doi.org/10.3133/ofr20161183.

Rohde, A.T. (2014). Influence of wildfire disturbance and post-fire seeding on vegetation and insects in sagebrush habitats (master's thesis). Retrieved from ScholarWorks (accession number 832). https://scholarworks.boisestate.edu/td/832/.

Grants and Awards:

2020; "Status and Conservation of the Western Bumble Bee, *Bombus occidentalis*" Funding source: U.S. Geological Survey/U.S. Fish and Wildlife Service Science Support Partnership Program

2018; Utah State University Ecology Center Graduate Student Research Award

2013; Northwest Climate Science Center Early Career Scientist Climate Boot Camp Nominee/Attendee

2012; Pilliod, D.P. and Rohde A.T. "Forecasting insect community responses to changes in climate in great basin sagebrush steppe." Funding source: Oregon State Bureau of Land Management

2010; USGS STAR Award

Selected Recent Presentations:

Rohde, A.T., Everett, J., Pilliod, D.S., and Strange, J.P. "Genetic measurements of the conservation status of a North American Bumble bee pollinator in decline, *Bombus occidentalis*" Oregon Chapter of the Wildlife Society, Eugene, Oregon, February 5-7, 2020.

Rohde, A.T., Pilliod, D.S., and Evers, L. "Spatio-temporal dynamics of insect pollinator communities in sagebrush-steppe." Joint meeting of the Entomological Society of America and the Entomological Society of Canada, Vancouver, BC, November 11-14, 2018.

Rohde, A.T., Knoblett, J., and Strange, J. "Microsatellite marker development in *Osmia lignaria*, a common and ecologically important North American pollinator." National meeting of the Entomological Society of America, November 5-8, 2017.

Rohde, A.T., Pilliod, D.P., and Halford, A. "Diverse pollinator assemblages use planted forb "island" restoration treatments within burned areas in shrub-steppe habitats." Pacific Branch Entomological Society of America. Portland, OR, April 2-5, 2017.

Rohde, A.T., Pilliod, D.S., and Halford, A. "Effects of Shrubland Techniques on Insect Pollinators and Communities." Natural Areas Association. Fort Collins, CO, October 10-12, 2017.

Rohde, A.T., and Strange, J.P. "Microsatellite marker development in *Osmia lignaria*, a common and ecologically important North American bee pollinator." Entomological Society of America. Fort Collins, CO, November 5-8, 2017.