Franklin's bumble bee (*Bombus franklini*)

Species Status Assessment

Photo of *Bombus franklini* by Pete Schroeder

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Executive Summary

This document presents the species status assessment (SSA) for Franklin's bumble bee (*Bombus franklini*), completed to characterize the species' overall viability. To characterize viability we use the three conservation biology principles of resiliency, representation, and redundancy. We identify the species' ecological requirements for survival and reproduction at the individual, population, and species levels, and describe risk factors influencing the species' current and future condition.

Bombus franklini has the most restricted range of any North American bumble bee, and possibly the most restricted range of any bumble bee in the world. Historically, *B. franklini* occupied portions of Douglas, Jackson and Josephine Counties in southern Oregon, as well as Trinity and Siskiyou Counties in northern California. Since the late 1990s, *B. franklini* observations have declined significantly, and none have been observed since 2006, despite an expanded and focused survey effort. Very little is known about *B. franklini*; much of the information presented in this SSA is inferred from closely-related *Bombus* species, and we rely heavily on information from species experts. While the decline of *B. franklini* observations is contemporaneous with the decline of other *Bombus* species, the causal factors behind these declines are poorly understood. The species has likely been affected by pathogens, pesticides, and the effects of small population size. The synergistic effects of several stressors to the species have likely exacerbated declines.

Bombus franklini has been found in a wide array of sheltered and exposed habitat types at a broad elevational range, and the species appears to be a generalist forager. Our certainty regarding the species' habitat needs is limited to (1) floral resources for nectaring throughout the colony cycle, and (2) relatively protected areas for breeding and shelter. The habitat elements that *B. franklini* appears to prefer to fulfil those needs are relatively flexible, plentiful, and widely distributed. Despite this fact, no individuals of the species have been found in any habitat since 2006, and therefore we conclude that the resiliency of the species has decreased since the 1990s. Further, no current populations of *B. franklini*, distributed across any level of ecological conditions or spatial extent, are known to exist, and therefore we conclude that genetic and ecological representation as well as redundancy have decreased since the 1990s. Due to the lack of observations of the species since 2006, we did not project anticipated future states of resiliency, redundancy or representation. Although the failure to detect a species during surveys is not equivalent to a conclusive demonstration of its absence and may simply reflect the very low detection probability for rare species, the certain losses in both the number of populations and their spatial extent render *B. franklini* vulnerable to extinction even without further external stressors acting upon the species.

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LIST OF ABBREVIATIONS

SSA	Species Status Assessment
ESA	Endangered Species Act
N	Population Size
N_e	Effective Population Size
λ	lambda, population growth rate
3Rs	Resiliency, Representation, and Redundancy
spp.	Species
USFS	United States Forest Service
FWS	Fish and Wildlife Service
BLM	Bureau of Land Management
EPA	United States Environmental Protection Agency

1.0 Introduction, Analytical Framework, and Methods

1.1 Introduction

This report presents the species status assessment (SSA) conducted for the Franklin's bumble bee (*Bombus franklini*). We, the Fish and Wildlife Service (Service), were petitioned to list *B. franklini* as endangered under the Endangered Species Act of 1973, as amended (ESA), on June 23, 2010, by the Xerces Society for Invertebrate Conservation and Dr. Robbin Thorp, Professor Emeritus from the University of California (Xerces Society and Thorp 2010, p. 2). In September 2011, the Service announced in the Federal Register that the petition presented substantial information indicating that this species may be warranted for listing, and announced the beginning of a status review for the species (Fish and Wildlife Service 2011). This SSA will be the biological underpinning of the status review and the Service's forthcoming 12-month finding on whether *B. franklini* warrants protection under the ESA.

This SSA assesses the viability of *Bombus franklini*; that is, the likelihood that the species will sustain populations over time. To assess *B. franklini's* viability, we used the three conservation biology principals of resiliency, representation, and redundancy (Shaffer and Stein 2000, pp. 308-311). These principals are described in general terms below, and more specifically for *B. franklini* in section 4.0. Our approach for assessing *B. franklini's* viability involved three stages. In Stage 1, we describe the species ecology in terms of the 3Rs, identifying the ecological requirements for survival and reproduction at the individual, population, and species levels. In Stage 2 we use these ecological requirements to determine the baseline condition for the species by assessing the species historical and current condition in relation to the 3Rs, and identifying past and ongoing factors that led to the species current condition. Finally, in Stage 3 we use both the baseline conditions as well as forecasts of the future levels of influence factors to project the future condition of *B. franklini*.

Although there is abundant information available on the sub-genus *Bombus sensu stricto*, there is very limited information available on *B. franklini* in particular. In 2016, the Service completed the Rusty Patched Bumble Bee (*Bombus affinis*) Species Status Assessment (U.S. Fish and Wildlife Service 2016a). *Bombus franklini* shares a close evolutionary relationship and shared natural history traits with *B. affinis* (S. Colla, York University, Toronto, Ontario, Canada, 2018, pers. comm.). We note that despite this taxonomic relationship there are distinct differences between the species, particularly the more restricted range and limited distribution of *B. franklini* compared to *B. affinis*. However, based on the close taxonomic relationship, *B. affinis* has been identified as an acceptable proxy species to use in our assessment of *B. franklini* (R. Thorp, University of California, Davis, California, pers. comm., 2017; Williams et al 2014, p. 114; Goulson 2010, pp. 188-189; Thorp 2004; Schroeder pers. comm. 2017; Hatfield pers. comm. 2017). Due to the limited information on *B. franklini*, and in an effort to avoid duplicating effort when assessing two very similar species, our SSA incorporates a portion of the information and text provided in the rusty patched bumble bee SSA. Additionally, we note that the western bumble bee (*B. occidentalis*) is also a member of the sub-genus *Bombus sensu stricto*, and shares a portion of the range; therefore we also rely on information related to *B. occidentalis* for this assessment of the status of *B. franklini*.

1.2 Analytical Framework

To assess the viability of *Bombus franklini*, we applied the conservation biology principles of resiliency, representation, and redundancy (henceforth, 3Rs). Viability is the likelihood that the species will sustain populations over time. To do this, a species must have a sufficient number and distribution of healthy populations to withstand changes in its biological (*e.g.*, novel diseases, predators) and physical (*e.g.*, climate change) environment, environmental stochasticity (*e.g.*, wet or dry, warm or cold years), and catastrophes (*e.g.*, severe and prolonged droughts). Viability is not a single state — viable or not viable; rather, there are degrees of viability--less to more viable, or low to high viability. As the resiliency, representation, and redundancy of a species increases, the species is better protected against the vagaries of the environment, and thus it can better tolerate stressors (one or more factors that may be acting on the species or its habitat, causing a negative effect). When the 3Rs increase, a species is more able to adapt to future changes, and therefore, it is more viable. The 3Rs framework (assessing the health, number, and distribution of *B. franklini* populations relative to frequency and magnitude of environmental stochasticity and catastrophic events across its historical range of adaptive diversity) is useful for describing the species' degree of viability through time.

1.2.1 Resiliency

Resiliency is the ability of a species to sustain populations in the face of environmental variation and transient perturbations. Environmental variation includes normal year-to-year variation in rainfall and temperatures, as well as unseasonal weather events. Perturbations are stochastic events such as fire, flooding, and storms. To be resilient, a species must have healthy populations that are able to sustain themselves through good and bad years. Resiliency increases as the number of individuals and populations increase, and the amount and distribution of available habitat increases. For many species, resiliency is also affected by the degree of connectivity among populations and the diversity of occupied ecological niches. Connectivity among populations increases the genetic health of individuals (heterozygosity) within a population. Furthermore, by increasing the potential for immigration, connectivity enhances a population's ability to recover from disturbances. Diversity of climate niches improves a species' resiliency by guarding against disturbances and perturbations affecting all populations similarly (*i.e.*, decreases the chance of all populations experiencing bad years simultaneously or to the same extent).

1.2.2 Representation

Species-level representation is the ability of a species to adapt to near and long-term changes in the environment; it is the evolutionary capacity or flexibility of a species. Representation is the range of variation found in a species, and this variation--called adaptive diversity--is the source of species' adaptive capabilities. Representation is therefore measured through the breadth of the species' adaptive diversity. The greater the adaptive diversity, the more responsiveness and adaptability the species will have over time, thereby enhancing its viability. Maintaining adaptive diversity includes conserving both the ecological and genetic diversity of a species. By maintaining these two sources of adaptive diversity across a species' range, the responsiveness

and adaptability of a species over time is preserved. Ecological diversity is the physiological, ecological, and behavioral variation exhibited by a species across its range. Genetic diversity is the number and frequency of unique alleles within and among populations.

In addition to preserving the breadth of adaptive diversity, maintaining evolutionary capacity requires maintaining the evolutionary processes that drive evolution; namely, gene flow, genetic drift, and natural selection. Gene flow is expressed through the physical transfer of genes or alleles from one population to another through immigration and breeding. The presence or absence of gene flow can directly affect the size of the gene pool available. Gene flow will generally increase genetic variation within populations by bringing in new alleles from elsewhere, but decrease genetic variation among populations by mixing their gene pools (Hendry *et al.* 2011, p. 173). Genetic drift is the change in the frequency of alleles in a population due to random, stochastic events. Genetic drift always occurs, but is more likely to negatively affect populations that have a smaller effective population size (N_e) and populations that are geographically spread and isolated from one another. Natural selection is the process by which heritable traits can become more (selected for) or less (not selected for) common in a population, based on the reproductive success of an individual with those traits. Natural selection influences the gene pool by determining which alleles are perpetuated in particular environments. This selection process generates the unique alleles and allelic frequencies reflecting specific ecological, physiological, and behavioral adaptations optimized for survival in different environments.

1.2.3 Redundancy

Species-level redundancy is the ability of a species to withstand catastrophic events. Redundancy protects species against the unpredictable and highly consequential events for which adaptation is unlikely. In short, it is about spreading the risk. Redundancy is best achieved by having multiple populations widely distributed across the species' range. Having multiple populations reduces the likelihood that all populations are affected simultaneously. The more widely distributed populations are, the less likely they are to possess similar vulnerabilities to a catastrophic event. Given sufficient redundancy, single or multiple catastrophic events are unlikely to cause the extinction of a species. Thus, the greater redundancy a species has, the more viable it will be. Furthermore, a greater number of populations and a greater diversity and distribution of those populations, the more likely it is that the adaptive diversity and evolutionary flexibility of the species will be preserved.

1.3 Methods

We gathered information to assess the viability of *Bombus franklini* from a variety of sources, including the information in the 2010 Petition, our previous Federal Register notices, and our files. In addition, we requested information from a diverse but specific audience, seeking information on the species as well as all recent survey data from land managers and Federal agencies. We also conducted a limited expert elicitation to collect more information and solicit opinion on the species' population dynamics. This elicitation included an extended interview with Dr. Robbin Thorp, one of the petitioners and the noted species expert on *B. franklini*. Additionally, we sent a questionnaire to 3 other professionals with experience and knowledge of

B. franklini (Richard Hatfield, Xerces Society; Peter Schroeder, Southern Oregon University; and Pepper Trail, US Fish & Wildlife Service). A copy of the questionnaire can be found in Appendix 4. We incorporated information from this elicitation effort into our analysis.

Building on the occurrence data provided in the 2010 Petition (Xerces Society and Thorp 2010, Appendix 1) and other information gathered during our assessment, we assembled an occurrence table and associated database of all known *Bombus franklini* occurrences, including information provided by the Petitioners; information available in university and museum collections; and in response to our requests (see Appendix 1). The table should not be considered a good representation of actual numbers of *B. franklini* on the landscape because the data used to assemble the occurrence table and database were generally collected through unsystematic, opportunistic surveys and reporting, especially prior to 1998 (Thorp, University of California at Davis, Davis, California, pers. comm. 2017), making it difficult to compare the number of occurrences over time. The SSA for *B. affinis* generated a very rough estimate of the area of habitat required to support a viable population of *B. affinis* by creating a post hoc systematic sampling method (U.S. Fish and Wildlife Service 2016a, p. 11). This method entailed overlaying a 10 km x 10 km grid across the range of the species and assigning a unique numerical identifier and a textual description of the year(s) *B. affinis* were detected within that grid. We do not have sufficient data on *B. franklini* occurrences over a similar spatial and temporal extent to conduct a similarly comprehensive estimate. However, we do draw some conclusions about minimum habitat requirements for *B. franklini*, as described in section 2.2.2.

Although we have evidence of the presence of *Bombus franklini* in certain areas, the lack of systematic surveys across the historic range of the species over time prevents us from using these occurrences to extrapolate reasonable estimates of species abundance or distribution. Many of the occurrence records just provide point data for an occurrence, with no details on the size of the area searched or whether or not the record reflected a comprehensive search of an area. Many records also lack details on the level of survey effort per location (number of searchers, hours of search effort per day, number of days per search effort). Additionally, because bumble bee nest locations vary year-to-year, tracking individual colonies, and thus populations, over time is very difficult. We cannot draw any conclusions on the abundance of *B. franklini* colonies or population overall, since information is not available on how many individuals make up a population (Thorp, pers. comm. 2017; P. Schroeder, pers. comm. 2017; R. Hatfield, Xerces Society, Portland, Oregon, pers. comm. 2017). More targeted surveys were conducted in recent years by those interested in the apparent decline of *B. franklini*, but they were not systematic and only conducted in a limited number of specific sites throughout the species' historic range. More recent search efforts have primarily occurred on Federal land, however surveys have occurred opportunistically on private land when access has been granted. Although it is possible that the species may be extinct (University of California 2009), *B. franklini* colonies could potentially persist in places that have not been systematically surveyed. A close relative, *B. occidentalis,* was recorded in the Ashland, Oregon area in 2010, and not seen again in that area until two individuals were observed during the focused surveys in July 2016 on Mt. Ashland (Thorp, pers. comm. 2017).

2.0 Species Information

2.1 Background

2.1.1Taxonomy and Species Description

All of the approximately 250 species of bumble bees found worldwide (Williams *et al.* 2008, p. 1) belong to the genus *Bombus* (formerly *Bremus*), family Apidae, and order Hymenoptera, and thirty species of *Bombus* are known in the western United States (Koch *et al.* 2012, entire). *Bombus franklini* was first described in 1921, based on the collection of two queen specimens on July 7, and July 8, 1917, in Nogales, Arizona (Frison 1921, pp. 147-148). The description of the species was completed in 1922, based on one worker and one male specimen collected from an unspecified locality in Oregon, and deposited in the United States National Museum (Frison 1923, p. 313-315; Thorp *et al.* 2010, pp. 5, 40). At that time, it was noted that *B. franklini* was one of the rarer species of the widely distributed *Bombus* (*Bremus*) genus (Frison 1923, p. 315).

In 1970, based on museum record research and field studies, the actual location of the Nogales, Arizona collection was called into question, and Gold Hill, Oregon, was proposed instead as the type locality for *Bombus franklini* (Thorp 1970, p. 177-179; Thorp *et al.* 2010, p. 5, 7). Several studies have been published on the taxonomic relationship of *B. franklini* to other bumble bees (Stephen 1957, pp. 79-81; Milliron 1971, pp. 58-67; Plowright and Stephen 1980, pp. 475–479; Thorp *et al*. 1983, pp. 29-30; Scholl *et al*. 1992, pp. 46-51; Cameron *et al*. 2007, p. 173). With the exception of Milliron (1971), who assigned *B. franklini* subspecific status under *B. terricola occidentalis*, all of these studies have accorded *B. franklini* its own specific rank and *B. franklini* is listed in the most recent world checklist of bumble bee species (Williams 1998, p. 129; Thorp *et al*. 2010, p. 5). *Bombus franklini* is also recognized as a valid species in the Integrated Taxonomic Information System (Integrated Taxonomic Information System 2017). For these reasons, we recognize *B. franklini* as a valid species and therefore, a potentially listable entity under the ESA.

As a bumble bee of the subgenus *Bombus sensu stricto*, *B. franklini* is corbiculate (females having pollen baskets on the hind legs) (Williams, *et al.* 2008, entire). In *B. franklini*, the hind leg tibia outer surface (corbicula) is flat with long black fringes at the sides (Williams *et al.* 2014, p. 119). The species is short-tongued with a short head and the cheek (area between the bottom of the compound eye to the insertion of the mandible) is shorter than it is wide (Koch *et al.* 2012, p. 98; Williams *et al.* 2014, p. 119). Shorter faces and tongues are an adaptation to extracting nectar from flowers with short corollas (Koch *et al.* 2012, p. 6). *Bombus* from this sub-genus with short tongues also rob nectar from flowers with longer corollas, by biting holes in the base of the corolla to access the nectar. *Bombus occidentalis*, a closely related species, has mandibles with distinct teeth, possibly to aid in this behavior (Goulson 2010, p. 173). Body size of the queens (22-24 mm, 0.86-0.95 inches) and workers (10-17 mm, 0.40-0.65 inches) is relatively large (Williams *et al*. 2014, p. 119). Males are 13-16 mm (0.50-0.64 inches) in length. In the field, *B. franklini* can most easily be distinguished from other similar species in its range (e.g., *B. occidentalis*, *B. vosnesenskii*, *B. caliginosus*, *B. vandykei*, *B. fervidus*, *B. insularis*, *B. flavidus*), by the inverted U-shape pattern of the yellow hairs on the anterior thorax surrounding a

central black patch and extending beyond the bases of the wings, and the lack of yellow hairs on the abdomen (Thorp *et al.* 2010, p. 5-6; Williams *et al*. 2014, p. 119). In addition, the hairs on the round face are predominantly black, there are yellow hairs on the top of the head, and there are white hairs in two spots at the tip of the abdomen (Thorp *et al.* 2010, p. 5-6). For other diagnostic characters that can be seen in the hand and under the microscope, please see Frison (1921, pp. 147-148; 1922, pp. 313-315), Thorp *et al.* (2010, pp. 5-6), and Williams *et al*. (2014, pp. 119-120).

2.1.2 Distribution and Known Occurrences/Survey Data

Bombus franklini is thought to have the most limited distribution of all known North American bumble bee species (Plowright and Stephen 1980, p. 479; Xerces Society and Thorp, 2010, p. 6), and one of the most limited geographic distributions of any bumble bee in the world (Frison 1923, p. 315; Williams 1998, p.129). Stephen (1957, p. 81) recorded the species from the Umpqua and Rogue River Valleys in Oregon. Thorp *et al.* (1983, p. 8) also recorded it from northern California and suggested its restriction to the Klamath Mountain region of southern Oregon and northern California. Elevations where it has been observed range from 162 m (540 feet) in the northern part of its range, to over 2,340 m (7,800 feet) in the south of its historical range. All confirmed specimens have been found in an area about 306 km (190 miles) to the north and south, and 70 miles 113 km (70 miles) east to west, between 122° to 124° west longitude and 40° 58’ to 43° 30’ north latitude in Douglas, Jackson, and Josephine counties in southern Oregon, and Siskiyou and Trinity counties in northern California (Thorp 1999, p. 3; Thorp 2005c, p. 1; International Union for Conservation of Nature 2009, p. 1). Twenty three of the 43 sites where *B. franklini* has been located are privately owned, 18 are on Federal land (U.S. Forest Service and Bureau of Land Management), one site is on State land, and one is on municipal land.

Limited occurrence and observation data exist for *Bombus franklini* prior to 1997. Historic observations and occurrence data includes randomly reported observations, student collections, and museum specimens, as well as the collections and notes of interested parties, natural resource managers, and university staff (Xerces Society and Thorp 2010, pp. 34-40). As mentioned in the previous section, *B. franklini* was first observed in 1917 and first described in 1921. Between 1923 and 1992 there were 31 additional occurrences recorded in Oregon, and seven recorded in California. Of the 38 records, 25 noted five or less bees, and only one 1968 record counted more than 12 bees at a single location (Appendix 1). For many of the occurrences between 1923 and 1992, we do not have an understanding of whether or not the surveyors/collectors were noting all of the *B. franklini* observed at that site on that day. No survey methodologies were reported so we do not know how surveyors/collectors looked for the bees at the various sites or how long they spent looking at a given site on a given day. Furthermore, information about search efforts that took place where no bees were detected (negative occurrence data) would not be on record (Thorp, pers. comm. 2017). Therefore, the main information that *B. franklini* records from this period provide is documentation of presence of the species at a given location; they do not provide a clear understanding of historic population abundance across the range. However, Dr. Robbin Thorp has noted that in the 1960’s, when he looked at sites where he thought *B. franklini* might be, he was able to find the species.

He also suspects that if others knew where to look for the species, they would have been likely to find the species at the time (Thorp, pers. comm. 2017).

In 1997, there were two records of *Bombus franklini* in Oregon, each noting two bees counted. Also that year, three queen *B. franklini* and nine workers were observed in the Marble Mountains (Siskiyou County, California) study area, by a master's student from Humboldt State University (M. Brooks, Humboldt State University, Arcata, California, pers. comm. 1997). The study did not survey for *B. franklini* specifically, but was looking at *Bombus* assemblages and flower preferences based on tongue-length. *Bombus franklini* was observed at six specific locations (three locations each, on two of the study's ten total survey areas), between June 6th and August 15th; the bees were observed on lupine (*Lupinus spp*.), mountain monardella (*Monardella odoratissima*), and clover (*Trifolium spp*.) (Brooks pers. comm. 1997; Brooks 1999, p. 11).

A survey effort specifically focused on *Bombus franklini* began in 1998 and continues annually, at sites representing both a subset of historical and potential new localities for the species. According to the information provided in the 2010 petition (Xerces Society and Thorp, 2010, p. 7), between nine and 17 historical sites (averaging 13.8 sites annually), and two to 23 additional sites, were surveyed each year from 1998 until 2010. Some sites were visited more than once per year, or in multiple years, and some historic locations have not been resurveyed since the original observation of *B. franklini* at that location. These surveys were primarily focused on localities in Jackson County, Oregon around the center of the historic range (Xerces Society and Thorp, 2010, p. 9; Thorp, pers. comm. 2017). During the surveys from 1998 to 2006, *B. franklini* was observed at 11 sites, including seven sites where it had not been previously documented. In 1998, 98 individuals were located at eight sites (with 81 of those individuals occurring at two of the eight sites). In 1999, only 20 individual bees were located. Nine individuals were observed in 2000, and one individual in 2001. Although 20 were observed in 2002, only three were observed in 2003 (all at a single locality), and a single worker bee was observed in 2006. There have been no confirmed observations of *B. franklini* since the single worker in 2006. Figure 1 displays a graph of all known presence data for *B. franklini* over time, from the first observation in 1923 until 2017 (2006 was the last documented occurrence). Figure 2 displays a map of all known occurrence data for *B. franklini*, from 1923 to the present.

Figure 1. All known occurrences of *Bombus franklini* by year of observation (Xerces and Thorp 2010, Brooks 1999; Code and Haney 2006, p. 3; Pool 2014, entire; Colyer 2016, entire; Hatfield, pers. comm. 2017; Thorp, pers. comm. 2017)

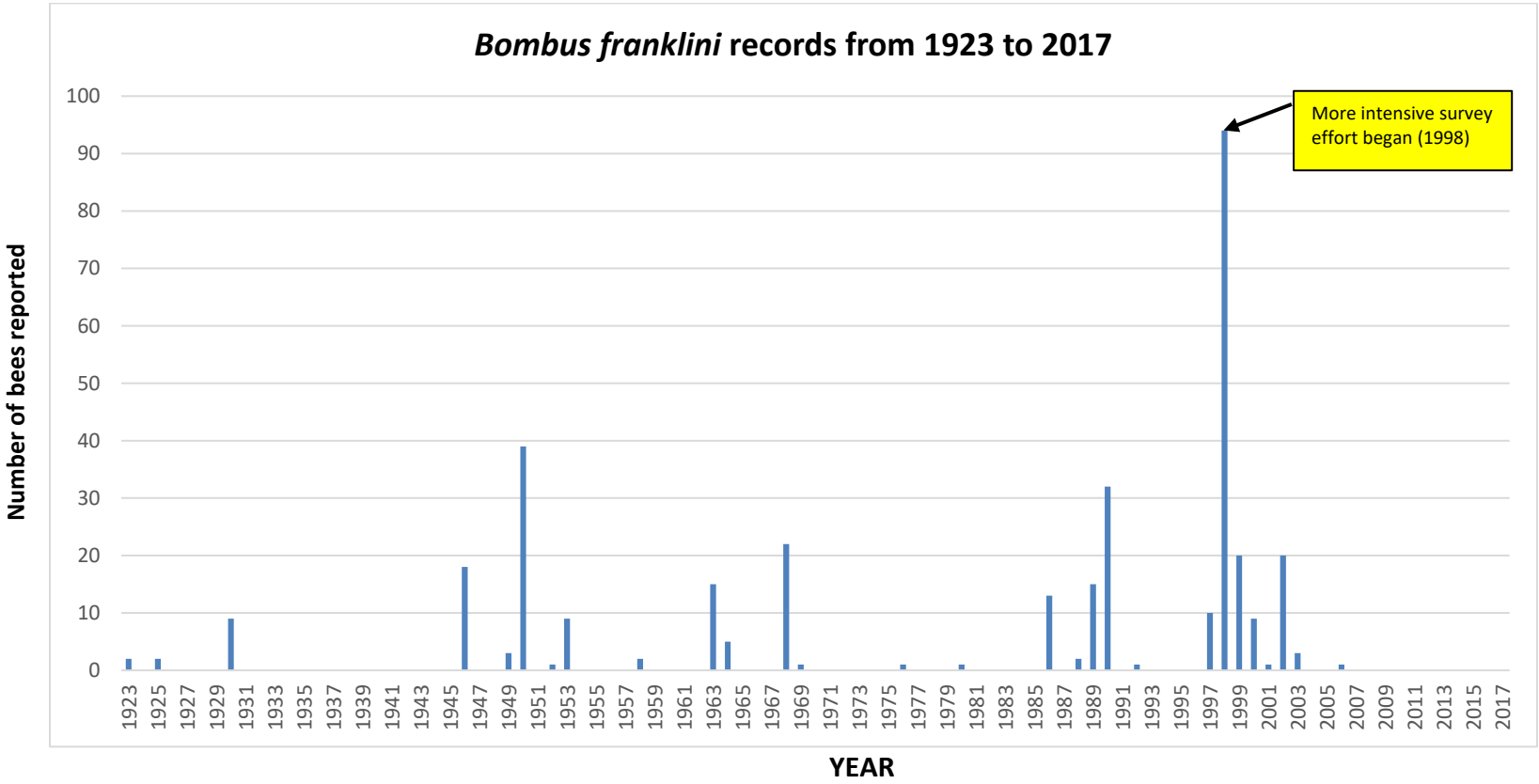
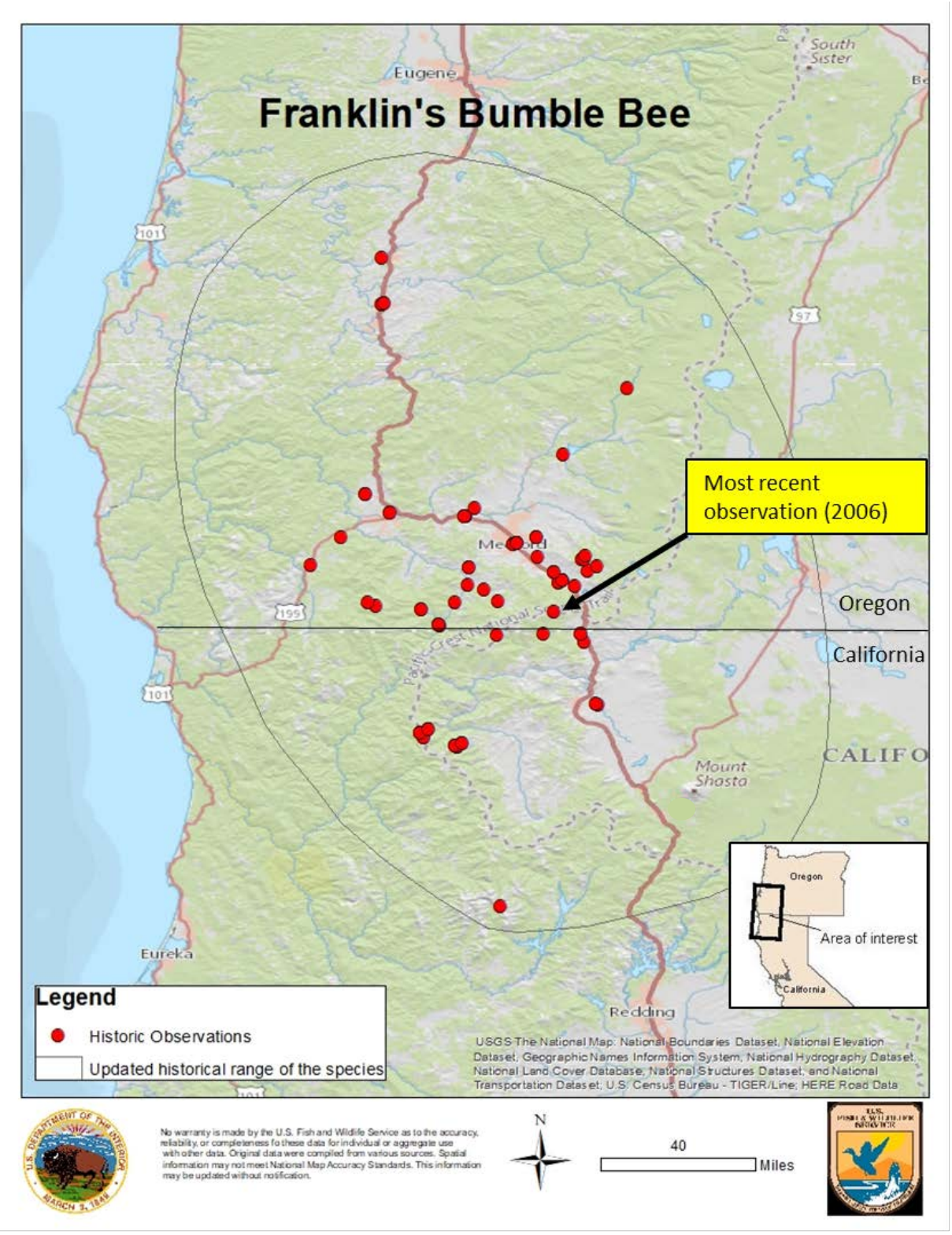


Figure 2. All known occurrences of *Bombus franklini*, from 1923 to 2017.



In 2006, the Bureau of Land Management conducted a survey of 16 sites on the Mt. Ashland resource area in the Medford District that were believed to provide optimal habitat for *Bombus franklini*. Each site was surveyed twice by trained technicians, but no *B. franklini* were found (Code and Haney 2006, p. 3).

Since 2009, a number of targeted surveys have taken place at select locations within the historic range of *Bombus franklini*, in an effort to locate the species and other rare or declining invertebrates (including Western bumble bee (*occidentalis*)). In 2014, the Medford District of BLM conducted a survey for six special status meadow invertebrates, including *B. franklini* and *B. occidentalis*. Surveys were conducted between July and September, with survey locations based on (1) historical occurrence records for private, BLM and USFS lands, and (2) water and floral resources. *Bombus occidentalis* was observed at three locations; no *B. franklini* were found (Pool 2014, entire). The surveys were conducted in areas that appeared to have good quality habitat for *Bombus* (S. Godwin, Bureau of Land Management, Medford, Oregon, pers. comm., 2017).

Surveys targeting *Bombus occidentalis* took place on the Umpqua and Rogue River-Siskiyou National Forests in 2015 and 2016, with trained observers covering dozens of historical locations with a wide variety of habitat types and elevations throughout the flight season. Over a dozen *Bombus* species were recorded including *B. occidentalis*, but no observations of *B. franklini* were made (Colyer 2016, entire). Generally the surveys were conducted in habitat that would be good for *B. franklini*, given that all of the sites had several different *Bombus* spp. detections (S. Colyer, U.S. Forest Service, Prospect, Oregon, pers. comm. 2017).

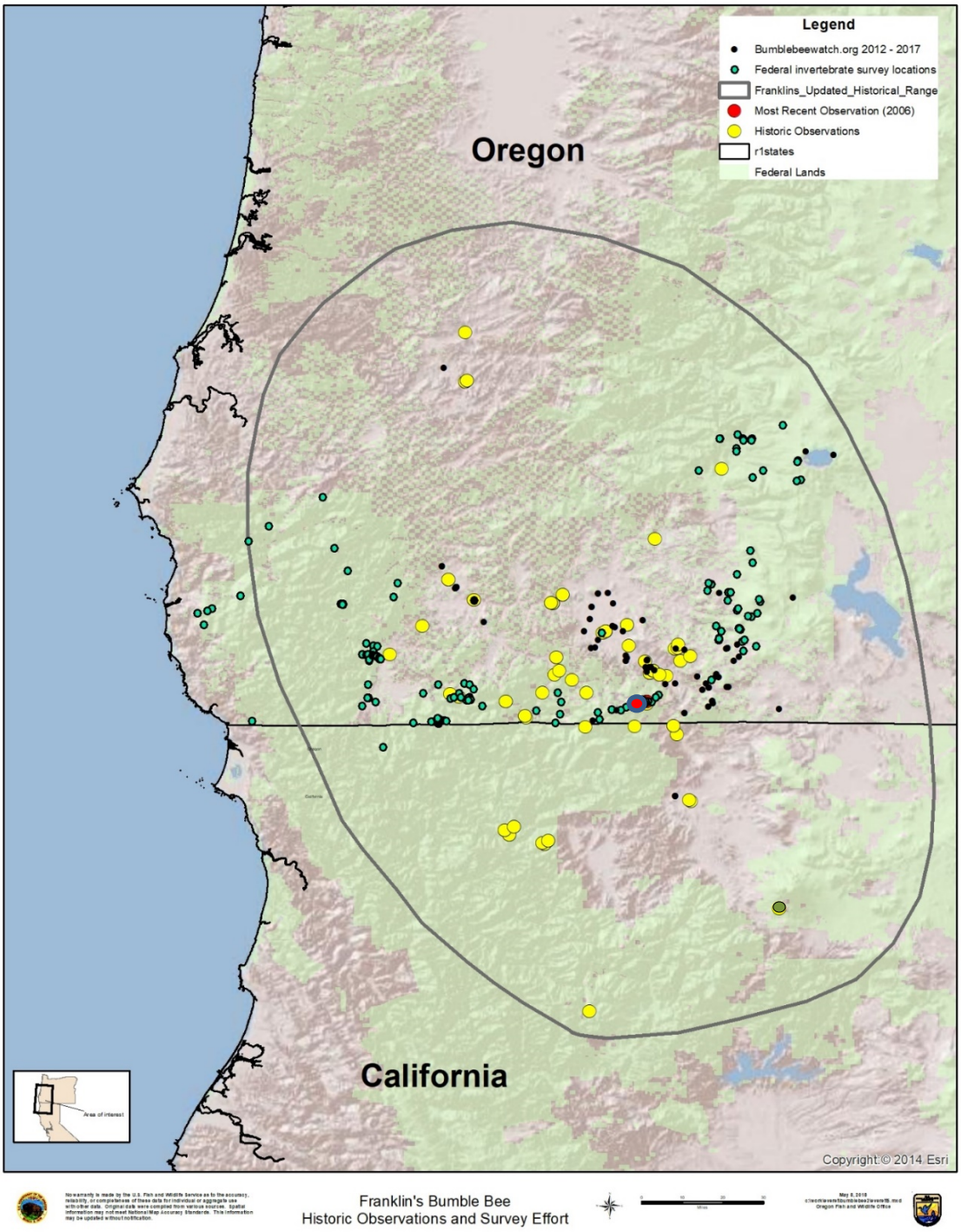
In response to our request for information, the Xerces Society provided records of all *Bombus* observations reported to Bumblebeewatch.org, between 2012 and 2017, within the historical range of *B. franklini*. All reports are from incidental observations and have been confirmed by taxonomic experts. Over 100 observations of *Bombus spp*. were reported from a wide variety of land ownerships, habitats and elevations, and included 18 different *Bombus* species (including *B. occidentalis*), however no *B. franklini* were observed (Hatfield, pers. comm. 2017). While this information is not part of a standardized survey, it does represent some level of opportunistic observation and reporting opportunity available over time within the historical range of the species, and all are observations were verified by experts; thus we feel it is worth mentioning. Again, no new observations of *B. franklini* have been made since 2006.

As mentioned earlier, *Bombus occidentalis* was recorded in the Ashland, Oregon area in 2010, and not seen again in that area during the annual surveys until two individuals were observed during the focused surveys in July 2016 on Mt. Ashland (Thorp, pers. comm. 2017). *Bombus occidentalis*, like *B. franklini*, is not migratory and therefore must have been present in the survey area, yet remained undetected during the surveys over multiple sequential years. This is indicative of the low detection probability for these rare species, even when focused annual survey efforts by trained observers are taking place.

Figure 3 below displays all the sites on record that have been surveyed for *Bombus franklini*. This includes survey location information from the Bureau of Land Management, U.S. Fish and

Wildlife Service, and the U.S. Forest Service; information from the 2010 Petition (Xerces Society and Thorp 2010, p. 34-40; additional information from Dr. Robbin Thorp (Thorp, pers. comm. 2017) and Brooks (Brooks 1997, p. 4), as well as all *Bombus* observations within the historic range of the species from 2012-2017 as reported in Bumblebeewatch.org (Hatfield pers. comm. 2017).

Figure 3. All sites surveyed for *Bombus franklini* from 1923 to 2017.



2.2 Species Ecology

2.2.1 Individual Level Ecology

The specific life history characteristics or behavior of this rare species have not been studied. As one of the rarest *Bombus* species, *B. franklini* are somewhat enigmatic and a specific habitat study for the species has not been completed. Such a study was initiated in 2006 when *B. franklini* was last seen, but could not continue due to the subsequent absence of the species (Thorp 2017, pers. comm.). While little is known about *B. franklini's* reproductive biology, specific habitat needs or unique behavior, this information is available for *Bombus* in general and for some closely-related species (*B. occidentalis*, *B. affinis*, and *B. vosnesenskii*, among others).

Bombus franklini is a primitively eusocial bumble bee, living in colonies made up of a queen and her offspring – males and workers. The nesting biology of *B. franklini* is unknown (Xerces Society and Thorp 2010, p. 10), but they likely nest underground in abandoned rodent burrows or similar cavities that offer resting and sheltering places, food storage, nesting and room for the colony to grow, as is typical for other eusocial *Bombus* species (Plath 1927, pp. 122-128; Hobbs 1968, p. 157; Thorp *et al*. 1983, p. 1; Thorp 1999, p. 5). It may also occasionally nest on the ground (Thorp *et al*. 1983, p.1) or in rock piles (Plowright and Stephen 1980, p. 475), and has even been found nesting in a residential garage in the city limits of Medford, Oregon (Thorp 2017, pers. comm,).

The flight season of *Bombus franklini* is from mid-May to the end of September (Thorp *et al* 1983, p. 30); a few individuals have been encountered in October (Southern Oregon University Bee Collection records, *in* Xerces Society and Thorp, 2010, Appendix 1 page 39). Colonies of *B. franklini* have an annual cycle, initiated each spring when solitary queens emerge from hibernation and seek suitable nest sites (Thorp, pers. comm. 2017). The queen collects nectar and pollen to support the production of her eggs, which are fertilized by sperm she has stored throughout hibernation since mating the previous fall. In the early stages of colony development, the founding queen (foundress) is responsible for all food collection and care of the eggs and larvae. As the colony grows, workers assume the duties of food collection, colony defense, nest construction, and larval care while the foundress remains within the nest and produces eggs. Colonies of *B. franklini* may contain from 50 to 400 workers, and the founding queen (Plath 1927, pp. 123-124; Thorp *et al* 1983, p. 2; Macfarlane *et al* 1994, p. 7). Two colonies of *B. franklini* initiated in the laboratory and set out to complete development in the field contained over 60 workers by early September, and likely produced over 100 workers by the end of the season (Plowright and Stephen 1980, p. 477).

Near the end of the colony cycle, reproductive queens (gynes) and fertile males are produced. Male bumble bees patrol selected territories, which they mark with queen-attracting scent. Queens locate a territory and remain still until a male finds her. Mating usually takes place on the vegetation on or near the ground. Queens usually mate with only one male, but males may mate with multiple females who enter the territory. After mating, queens feed to build up fat before entering hibernation. At the end of the colony cycle, all the workers and the males die along with

the founding queen; only the inseminated hibernating gynes are left to carry on the line into the following year (Duchateau and Velthius 1988). Over wintering habitat would include micro-habitats such as ground cavities, rotting logs, loose soil and other protected sites for queens to hibernate, with floral resources and suitable nest sites available for the emerging queens the following spring. Mating habitat requirements for most bumble bee species is not known.

Bumble bees are generalist foragers, meaning they gather pollen and nectar from a wide variety of flowering plants (Xerces Society 2013, pp. 27-28). Bumble bees are very efficient at collecting pollen; unlike honey bees, they often vibrate their flight muscles while inside a flower, causing pollen to fall from the plant anthers and stick to the bumble bee's copious body hairs. This behavior of "buzzing" a flower is also known as sonication, and is one of the characteristics of bumble bees that make them particularly attractive for commercial pollination; bumble bees can pollinate flowers hundreds of times faster than honey bees (Williams *et. al*. 2014, p. 16).

Bombus franklini requires a constant and diverse supply of flowers that bloom throughout the colony's life cycle, from spring to autumn (Xerces Society and Thorp 2010, p. 11); these resources would typically be found in open (non-forested) meadows in proximity to seeps and other wet meadow environments. Different *Bombus* species have consistently been observed foraging in the same area visiting similar and different species of flowering plants. During some Oregon surveys, no *Bombus* species was always consistent in the number of different plants species it visited, nor was any *Bombus* species tied to just one plant species (Schroeder, pers. comm. 2017). The nectar from flowers provides carbohydrates and the pollen provides protein. Studies of other *Bombus* species typically exhibit foraging distances of less than 1 km (0.62 miles) from their nesting sites (Knight *et al*. 2005, p. 1816; Wolf and Moritz 2008, p. 422; Dramstad 1996, pp. 163-182; Osborne *et al*. 1999, pp. 524-526; Rao and Strange 2012, pp. 909-911; Hatfield, pers. comm. 2017). *Bombus franklini* may have a foraging distance of up to 10 km (6.2 miles) (Thorp, pers. comm. 2017), but the subgenus' typical dispersal distance is most likely 3 km (1.86 miles) or less (Hatfield, pers. comm. 2017; Goulson 2010, p. 94,). *Bombus franklini* have been observed collecting pollen from lupine (*Lupinus* spp.) and California poppy (*Eschscholzia californica*), and collecting nectar from horsemint or nettle-leaf giant hyssop (*Agastache urticifolia*) and mountain monardella (*Monardella odoratissima*) (Xerces Society and Thorp 2010, p. 11). *Bombus franklini* may also collect both pollen and nectar from vetch (*Vicia* ssp.) as well as rob nectar from it (Xerces Society and Thorp 2010, p. 11). A short-tongued/cheeked bumble bee, *B. franklini* has been found to antagonistically rob nectar from flowering plants that it cannot directly reach with its tongue, by chewing a hole in the host plant where the nectar is located (Pool 2014, p. 3; Schroeder, pers. comm. 2017; Hatfield, pers. comm. 2017). This particular behavior has been known to occur during its visitation to pollinator plants such as *Aconitum*. Table 1 summarizes ecological requirements of *B. franklini* at the individual level.

Table 1. The ecological requisites for survival and reproductive success of *Bombus franklini* individuals.

Life Stage	Winter	Spring	Summer	Autumn
Queen		Diverse floral resources; suitable nest habitat	Diverse floral resources; suitable nest habitat	Diverse floral resources; suitable nest habitat
Worker Females		Diverse floral resources in close proximity to nest	Diverse floral resources in close proximity to nest	Diverse floral resources in close proximity to nest
Males			Diverse floral Resources; suitable mating habitat	Diverse floral resources; suitable dispersal/mating habitat
Gynes (new foundress queens)	Suitable diapause sites		Diverse floral resources	Diverse floral resources; suitable dispersal/mating habitat

In summary, *Bombus franklini* has been found in a wide array of sheltered and exposed habitat types at a broad elevational range, and the species appears to be a generalist forager. Our certainty regarding *B. franklini* habitat needs is limited to (1) floral resources for nectaring throughout the colony cycle, and (2) relatively protected areas for breeding and shelter. The habitat elements that *B. franklini* appears to prefer to fulfil those needs mentioned above are relatively flexible, plentiful, and widely distributed.

2.2.2 Population level ecology

Bombus franklini has long been considered a rare or vary rare species, with a relatively small population size and relatively small colony size compared to other *Bombus* species (Thorp, pers. comm. 2017; Hatfield, pers. comm. 2017). No more than 356 individuals have been observed in total, and no more than 98 total individuals at eight separate locations have been observed in any one year (Xerces Soc. and Thorp 2010, p. 7; Occurrence Table, Appendix 1). We have no definitive information on the minimum number of colonies or minimum habitat patch size for a self-sustaining population of *B. franklini*. As stated above in section 1.3 the assessment for *B. affinis* created a 10 kilometer (km) x 10 km (6.2 miles x 6.2 miles) grid across the range of the species to generate a rough estimate of the area of habitat required to support a viable population of *B. affinis*. The lack of information on *B. franklini* makes it unreasonable to do the same comprehensive exercise for this species, however we can look at some general principles of *B. franklini* life history to provide us with a very rough estimate of minimum habitat requirements for our best guess of what constitutes a population of the species. If we focus on the minimum area of habitat required to allow for individuals from different *B. franklini* colonies to travel their typical foraging distance to forage at a common location and potentially interbreed, we find that

an area 6 km x 6 km might accomplish that. This spatial estimate is appropriate for *B. franklini* for the following reasons*:* (1) the subgenus' typical dispersal distance for *B. franklini* is 3 km (Hatfield, pers. comm. 2017; Goulson 2010); (2) *B. franklini* individuals concurrently visiting a site are often from different colonies (Hatfield, pers. comm. 2017); and, (3) colonies would have to be within dispersal distance of other colonies in order to interbreed and maintain genetic diversity. An area 6 km x 6 km (3.72 miles x 3.72 miles) would allow for the possibility that *B. franklini* from different colonies that are 6 km (3.72 miles) apart could each disperse 3 km (1.86 miles) to a shared foraging location. It is, therefore, reasonable to assume that multiple *B. franklini* detections over time within a 6 km x 6 km (3.72 miles x 3.72 miles) area would likely represent a single population. This measure of 6 km^2 (3.72 square miles) is therefore a reasonable estimate of minimum patch size for a self-sustaining population of *B. franklini.*

Population viability requires healthy demographics and sufficient habitat to support a healthy demography; specifically, viability is a function of population size (N) and its population growth rate (lambda, λ). The population structure of *Bombus franklini* operates similarly to a metapopulation. A metapopulation is an assemblage of interacting subpopulations; a population of *B. franklini* is a collection of interacting colonies. But, whereas a subpopulation is composed of many reproductive individuals, a *B. franklini* colony is founded by a single queen, and thus a colony represents one reproductive unit. The effective population size (N_e) of *B. franklini* is, therefore, the number of successful nests or colonies – not the number of individuals.

Population size also affects population viability through genetic health. Small populations have lower levels of genetic diversity (heterozygosity), which reduces the capacity of a population to respond to environmental change. Inbreeding depression may result, leading to reduced longevity and fecundity and overall population fitness (Darvill *et al.* 2006, p. 602). Populations of monoandrous social species like *Bombus franklini* (colonies headed by a single queen who mates with a single male), are especially vulnerable to inbreeding depression, because the rate of genetic drift in a population is determined by the effective population size (N_e) which is much lower than the number of individuals in an area (Goulson and Darvill 2008, pp. 197-198; Darvill *et al*. 2006, p. 602). The N_e in bumble bees is 1.5 times the number of successful nests, not 2 times, as is the case with diploid-diploid organisms (Goulson and Darvill 2008, pp. 197-198).

The reproductive system of bumble bees renders them particularly sensitive to loss of genetic diversity. *Bombus* species exhibit haplodiploidy (i.e., males are haploid and females are diploid) and exhibited a single locus complementary sex determination (sI-CSD) system (Zayed 2009, p. 238). Typically, heterozygotes at the sex-determining locus develop into diploid females from fertilized eggs, while hemizygotes (a diploid individual with only one allele for a particular gene) develop into haploid males from unfertilized eggs (Zayed 2009, p. 239). In cases, however, where females mate with haploid males that share a sex-determining allele in common (called "matched mating"), half of the females' progeny will be homozygous at the sex-determining locus and will consequently develop into diploid males instead of females. As males do not contribute resources to the colony, homozygosity at the sex-determining locus imposes a cost to the colony by decreasing the number of females produced (Ellis *et al.* 2006, p. 4376). Additionally, diploid males are unviable, or if viable and mate, produce diploid sperm, which will lead to unviable fertilized eggs or sterile triploid daughters (Zayed 2009, p. 239), so those males that are produced are unable to contribute to next year's cohort. Matched mating occurs

most often when allele diversity at the sex-determining locus is low (Ellis *et al.* 2006, p. 4376; Zayed 2009, pp. 239-241). Thus, as N_e decreases, the likelihood of producing diploid males increase, which will further reduce the population size, potentially resulting in a negative, reinforcing downward cycle (*i.e.*, extinction vortex). Zayed and Packer (2005, pp. 10743-10744) found, through modeling simulations, that extinction risks in haplodiploid populations were an order of magnitude higher than probabilities of extinction due to inbreeding depression in diploid populations. They attributed this high extinction risk to the effects of the "diploid male vortex"; a phenomenon where diploid males initiate a positive feedback cycle that leads to rapid extinction. Several species of bumble bee in England have demonstrated a dynamic consistent with this negative, reinforcing pattern. *Bombus subterraneous*, for example, following reduction in population size due to habitat loss eventually went extinct in the United Kingdom despite continued suitability of habitat (Darvill *et al.* 2006, p. 608). Maintaining genetic diversity within populations, thus, requires large N_e and gene flow within and among populations.

The viability of a population is also determined by its long-term lambda; in order for any population to persist over time, its growth rate, λ, must exceed 1.0. Species that fluctuate greatly with environmental conditions, require strong lambdas over time to avoid extirpation. The minimum λ needed to sustain a *Bombus franklini* population over time is unknown, but insects are particularly susceptible to environmental stochasticity. Although bumble bees, because of their relatively larger body size and fuzzy bodies, are not as strongly influenced by environmental conditions as other insects including honey bees, climatic conditions affect the availability of requisite resources, and hence, bumble bee numbers. Pollen and nectar availability, especially in spring and fall when floral resources are scarcer, are influenced by environmental conditions (Holm 1966, pp. 156-157); in years with unfavorable weather, the supply of food is limited, leading to smaller and fewer colonies. Thus, population viability requires occupying areas with a diversity of environmental conditions (spatial heterogeneity) to ensure floral resources are available throughout the season and year-to-year despite variations in climatic variables, such as temperature and precipitation. Similarly, spatial heterogeneity increases the likelihood of asynchrony among colonies, a pre-requisite for metapopulation long-term persistence (Hanski 1999, p. 28). In spatially heterogeneous populations, it is unlikely that the entire population will contemporaneously experience the same environmental conditions, thus ensuring that not all colonies comprising a population will fail due to unfavorable conditions.

In summary, the significant determinants of population-level viability for *Bombus franklini* are a healthy demography and sufficient quality habitat to support this demography. The demography of *B. franklini* populations is a function of its population size (the number of successful nests) and its population growth rate over time. The population size required to support a viable population is likely variable across spatial scales and is unknown, but generally speaking, the larger the population, the more genetically healthy and thus the more robust to extirpation. Similarly, the minimum long-term λ required to sustain a population over time is unknown, but it must exceed 1.0 and likely must be higher, given the susceptibility to environmental stochasticity. Both of these variables, N and λ, are dependent upon the amount and quality of floral resources, nest sites, and overwinter sites across temporal scales (within and among years). A precise estimate of the area of habitat required to support a viable population is dependent on the density and quality of floral resources, but given the large amount of food needed to support

successful colonies, it is reasonable to assume a large area is required. Another important aspect of population viability is connectivity among colonies to ensure mating of unrelated reproductive individuals and connectivity among populations to maintain within-population genetic diversity. Lastly, the degree of spatial heterogeneity across the population area reduces the chances of all colonies failing concurrently due to poor environmental conditions, and thus, is important for long-term persistence. For *B. franklini*, we can estimate that a minimum area of 6 km^2 would allow for individuals from different colonies to travel their maximum foraging distance to forage at a common location and have the opportunity to interbreed. Based on the above, the ecological requirements for successful population of *B. franklini* are listed below in Table 2.

Table 2. The requisites for survival and reproduction success of *Bombus franklini* populations.

Population Health (fitness)	**Element**	**Importance**
Healthy Demography	Large N_e	Multiple, successful colonies
	Patch size at least 6 km^2	Successful colonies, connectivity
	Habitat connectivity	To find unrelated mates
Habitat to support healthy demography	Sufficient floral resources	Adequate quantity of nectar and pollen
	Nesting and overwintering sites	Safe breeding and shelter
	Habitat connectivity	To safely and efficiently find food
	Heterogeneity	Diverse environmental conditions

2.2.3 Synopsis of Species Ecological needs

Viability is the likelihood that a species will sustain populations over time. To do this, *Bombus franklini* needs a sufficient number and distribution of self-sustaining populations to withstand environmental stochasticity (resiliency), adapt to changes in its environment (representation), and withstand catastrophes (redundancy) (Table 3).

Table 3. Ecological requirements for species-level viability in *Bombus franklini*

3Rs	Requisites of long-term viability	Description
Resiliency (able to withstand stochastic events)	Interconnected, healthy populations across a diversity of habitats	Populations with: 1) large N_e, sufficient floral resources in close proximity to nesting and overwintering sites, 2) connectivity among colonies, and 3) spatial heterogeneity; high connectivity among populations dispersed across diverse climatic conditions (spatial heterogeneity)
Representation (to maintain evolutionary capacity)	Maintain adaptive diversity of the species	Healthy populations distributed across areas of unique adaptive diversity
	Maintain evolutionary processes	Maintain evolutionary drivers--gene flow, natural selection, genetic drift- to mimic historical patterns
Redundancy (to withstand catastrophic events	Sufficient distribution of healthy populations	Sufficient distribution to guard against catastrophic events wiping out portions of the species adaptive diversity, i.e., to reduce covariance among populations
	Sufficient number of healthy populations	Adequate number of healthy populations to buffer against catastrophic losses of adaptive diversity

3.0 Factors Influencing the Status of the Species

Factors can influence a species both negatively and positively, as well as in synergy with other factors. We focused our analysis on six primary stressors potentially negatively affecting the species – pathogens, pesticides, habitat loss and degradation, grazing, climate change, and small population dynamics. We then looked at potential synergistic effects between these stressors. Finally, we looked at beneficial actions that may be positively affecting the condition of the species.

3.1 Stressors

The 2010 Petition identified the following factors as stressors on *Bombus franklini* and its habitat: introduced exotic diseases and competition from non-native bees; destruction, degradation and conversion of habitat; pesticides and pollution; inadequacy of current rules, regulations and law; introduction of exotic plant species; increased human use of native habitat; climate change; and alteration of wildfire severity and frequency (Xerces Society and Thorp 2010, p. 4). In our 90-day finding on the 2010 Petition (U.S. Fish and Wildlife Service, 2011), we noted that the petitioners provided substantial information on stressors to *B. franklini* from the destruction, modification or curtailment of habitat (primarily due to the potential impacts of natural or prescribed fire), disease, as well as the inadequacy of existing regulatory mechanisms

and other natural or manmade factors (including pesticides, small population dynamics, competition from non-native bees, and climate change).

In this SSA, we analyzed the factors noted as leading to our substantial 90 day finding. In addition, based on new information received, we looked again at agricultural intensification, urban development, and livestock grazing, as well as synergistic effects of the stressors in combination with each other. We discuss existing regulatory mechanisms and conservation actions in section 3.3, Beneficial Actions.

3.1.1 Pathogens

A number of diseases are known to naturally occur in bumble bee populations. These include the protozoan parasite *Crithidia bombi*, the tracheal mite *Locustacarus buchneri*, and the microsporidium (parasitic fungus) *Nosema bombi*, as well as deformed wing virus. Pathogens and parasites are widespread generalists in the host genus, but affect species differently according to host susceptibility and tolerance to infection (Kissinger *et al*. 2011, p. 221; Malfi and Roulston 2014, p. 18). The host species' life history plays a role in the virulence of a given pathogen; for instance, parasites may have relatively smaller effects on species with shorter colony life cycles and smaller colony sizes (Rutrecht and Brown, 2009, entire).

Pathogen spillover is a process whereby parasites and pathogens spread from commercial bee colonies to native bee populations (Colla *et al*. 2006, p. 461; Otterstatter and Thompson 2008, p. 1). The precipitous decline of certain *Bombus* species from the mid-1990s to present – particularly species in the subgenus *Bombus* sensu stricto (including *B. franklini)* – was contemporaneous with the collapse of commercially bred *B. occidentalis*, which were raised primarily to pollinate greenhouse tomato and sweet pepper crops beginning in the late 1980s (Szabo *et al*. 2012, pp. 232 -233). This collapse was attributed to *N. bombi*. Around the same time, several North American wild bumble bee species – *B. affinis, B. occidentalis, B. terricola* (all in the same subgenus *Bombus sensu stricto*), and *B. pensulvanicus*, also began to decline rapidly (Szabo *et al*. 2012, p. 232).

Bumble bees are very efficient pollinators of a wide variety of crops, including fruits, nuts, and vegetables (Loken 1958; Holm 1966b, Corbet *et al.* 1991, Cane and Payne 1993, MacKenzie and Averill 1995; Goodell and Thomson 1997, Macfarlane and Patten 1997, Mayer and Lunden 1997, Stubbs and Drummond 2001, Thorp 2003). As mentioned in section 2.2.1, bumble bees sonicate or "buzz pollinate" flowers hundreds of times faster than honey bees can. This attribute, combined with their tolerance of temperature extremes and longer foraging seasons, make them ideal for commercial greenhouse crop production (North American Pollinator Protection Campaign 2006, p. 6). Roughly 95 percent of all commercially-reared bumble bee colonies are used in the greenhouse production of tomatoes and sweet peppers (Velthuis and van Doorn 2006, Shipp *et al*. 1994, Ercan and Onus 2003). Commercial bumble bee production started in North America in the early 1990s (Xerces Society and Thorp 2010, p. 15). Queens of both *Bombus occidentalis* and *B. impatiens* were shipped from the United States to rearing facilities in Belgium that were also likely rearing *Bombus terrestris* (a closely related *Bombus* species native to Europe). *Bombus terrestris* was also likely imported to Mexico in 1995 and 1996 for greenhouse tomato pollination (Winter et. al. 2006, p.5).

The commercially-reared colonies produced from these queens were shipped back to the United States between 1992 and 1994. Bumble bee producers experienced major problems with *Nosema bombi* infection in commercial *Bombus occidentalis* in 1997 (Flanders *et al*. 2003, p. 108; Velthius and van Doorn 2006, p. 432), and eventually stopped producing *B. occidentalis*. In addition, the morphology of *N. bombi* found in a native bumble bee in China, *Bombus leucorum*, was found to be the same as that found in *B. terrestris* imported into China from New Zealand (Jilian *et al*. 2005, p. 53), suggesting the disease may have been introduced into native bumble bee populations in China by commercial bees. Studies suggest that disease can be spread from commercial bumble bees to nearby wild bumble bees (Niwa *et al*. 2004, p. 60; Whittington *et al*. 2004, p. 599; Jilian *et al*. 2005, p. 53; Colla *et al*. 2006, p. 461), even when commercial bumble bees are used for pollination in greenhouses. This is because commercial bumble bees frequently forage outside greenhouse facilities and can transmit disease at shared flowers (Xerces Society and Thorp, 2010, p. 15; Whittington *et al*. 2004, p. 599; Colla *et al*. 2006, p. 461). In addition to commercial pollination, *B. occidentalis* colonies were used in field research between 1991 and 2000 in California, Washington, and Alberta, Canada (Mayer *et al*. 1994, p. 21; Mayer and Lunden 1997, p. 283; Richards and Myers 1997, p. 293; Mayer and Lunden 2001, p. 277; Thompson 2004, p. 460).

Nosema bombi is a microsporidium (parasitic fungus) that has been detected in native bumble bees in North America, and has been found to be a part of the natural pathogen load, reported in Canada since the 1940s (Cordes *et al*. 2011, p.7) and appears to have a broad host range in North America (Kissinger *et al*. 2011, p. 222). *Nosema bombi* infections primarily occur in the malpighian tubules (small excretory or water regulating glands), but also in fat bodies, nerve cells, and sometime the trachea (Macfarlane *et al*. 1995). Colonies can appear to be healthy but still carry *N. bombi* and transmit it to other colonies. Transmission of *N. bombi* most likely occurs when spores are fed to larvae (Eijnde and Vette 1993 and Rutrecht 2007, as cited in Meeus *et al*. 2011, p. 666). Murray *et al*. (2013, p. 274 citing Rutrecht *et al*. 2007) notes that *N. bombi* spreads slowly through novel populations with subsequent inter-colony infections through drift of infected adults into non-natal colonies.

The effect of *Nosema bombi* on bumble bees varies from mild to severe (Macfarlane *et al*. 1995; Rutrecht *et al* 2007, p. 1719; Otti and Schmid-Hempel 2008, p. 577). *N. bombi* can have large effects on individual bees. Infected animals may have crippled wings, and queens may have distended abdomens and be unable to mate (Otti and Schmid-Hempel 2007, pp. 122-123). Malfi and Roulston (2014, p. 24) found that *N. bombi* infections are more frequent and more severe in rare species and also that the species with the highest percentages of infected individuals were rare species.

The Petitioners hypothesize that a virulent strain of *Nosema bombi* from *Bombus terrestris* spread to *B. impatiens* and *B. occidentalis* prior to their shipment back into the United States, and once in this country the commercially reared colonies may spread the virulent strain to wild populations of *B. franklini*. In work partially funded by the U.S. Fish & Wildlife Service, surveys for parasites and pathogens in bumble bee populations of the Pacific Northwest and Midwest were conducted by the University of Illinois between 2005 and 2009. The goal was to assess *Bombus* populations for presence and prevalence of pathogens, particularly microsporidia, in an

effort to provide baseline data to assess disease as a potential factor in the decline of *B. franklini*, *B. occidentalis*, and *B. pensylvanicus* (Solter *et al*, 2010, p. 1). The highest prevalence of *N. bombi* was found in *B. occidentalis*, with 26 percent of collected individuals infected. *Crithidia bombi* infections of *B. occidentalis* were 2.8 percent overall. No *B. franklini* were collected during the study. However, Mt. Ashland, Oregon (the last known location for *B. franklini*), was one of only three sites in the Pacific Northwest study area where *N. bombi* infections were found in multiple *Bombus* species *(B. insularis* and *B. bifarius*); the recovery of *N. bombi* infections from multiple *Bombus* species at a site was otherwise rare (Solter *et al* 2010, pp. 3-4). Although Cordes *et al* (2011, p. 7) found a new allele in *N. bombi*, the recent study by Cameron *et al*. (2016) found no evidence of an exotic strain of *N. bombi*. While we have no documentation in our files or evidence of direct effects of a virulent strain of *N. bombi* on *B. franklini*, *N. bombi* has been detected in closely related species in the range of *B. franklini*. Furthermore, *N. bombi* infections in rare species like *B. franklini* are more frequent, more severe and seem to affect a higher percentage of individuals in the species.

Crithidia bombi has been shown to have detrimental effects on colony founding success of queens, the fitness of established colonies, and the survival and foraging efficiency of bumble bee workers (Brown *et al*. 200, p. 421; Brown *et al* 2003, p. 994; Otterstatter *et al*. 2005, p. 388; Gegear *et al*. 2005, p. 1; Gegear *et al*. 2006, p. 1073). Studies suggest that *C. bombi* can spread from commercial bumble bees to nearby wild bumble bees through shared use of flowers when they escape to forage outside and transmit the disease (Durrer and Schmid-Hempel 1994, p. 299; Whittington *et al*. 2004, p. 599; Colla *et al*. 2006, p. 461; Otterslatter and Thompson 2008, p. 1). In fact, *C. bombi* has been shown to be present in higher frequencies in bumble bees near greenhouses where commercial colonies of *Bombus impatiens* are used than in bumble bees remote from these facilities (Colla *et al*. 2006, p. 621).

Although acute mortality is rarely observed, *Crithidia bombi* alters the foraging behavior in host bees by reducing their ability to identify and manipulate nectar flowers. This causes bees with high levels of infection to spend as much as 200 percent more time on flower visits to collect pollen and nectar resources (Gegear *et al*. 2006, Gegear *et al*. 2005). Although *C. bombi* is considered to be a bumble bee parasite, honey bees have also been shown to be possible vectors (Ruiz-Gonzales and Brown, 2006, p. 621).

The extent to which this pathogen occurs within the range of *Bombus franklini* is not known. However, within the historic range of *B. franklini*, *B. impatiens* hives were purchased and installed by a strawberry and vegetable grower to pollinate their crops in Grants Pass, Oregon (Associated Press 2007; Xerces Society and Thorp 2010, p. 18). *Bombus impatiens* is a known vector of *Crithidia bombi*. Experimental evidence shows that bumble bees can contract *C. bombi* while feeding on flowers that have been previously visited by infected bees (Tripodi, pers. comm. 2016 in U.S. Fish and Wildlife 2016, p. 42), and bees from commercial rearing facilities have tested positive for this pathogen upon delivery (Otterstatter *et al*. 2005, p. 388; Murray *et al*. 2013, p. 274). While evidence exist that *C. bombi* does affect *Bombus* spp., we do not have documentation in our files or evidence of direct effects of *C. bombi* on *B. franklini.*

Locustacarus buchneri is a tracheal mite that infects *Bombus* species in Japan, the Netherlands, and Belguim. The specific effects of *L. buchneri* on *Bombus* species, as well as the mechanisms

for spreading the mites, are not well understood. However, Otterstlatter and Whidden (2004, p. 351) and Goka *et al*. (2001) cite studies that found heavy mite infestations can severely injure bumble bees, to the extent that they are no longer able to forage (Goka *et al*. 2001, p. 2098). Ottersatter and Whidden (2004) found that bumble bees containing tracheal mites have significantly reduced lifespans in the laboratory. Commercially raised bumble bees from Europe were found to be infested with tracheal mites at higher rates than detected in wild bees (Goka *et al*. 2001, p. 2098). While evidence exists of *L. buchneri* effecting *Bombus spp*., we do not have documentation in our files or evidence of direct effects of *L. buchneri* on *B. franklini*.

Acute Bee Paralysis was the first honey bee virus to be detected in bumble bee hosts, although its occurrence in natural populations and effects on bumble bee health are unknown. The Black Queen Cell Virus has been the most commonly detected bumble bee pathogen in ongoing surveys, having been found in 31 percent of 559 samples tested to date (Tripodi, pers comm. 2016 in U.S. Fish and Wildlife Service 2016a, p. 42). It should be noted that although 12 *Bombus* species across the United States have tested positive for Black Queen Cell Virus, *B. franklini* has not been evaluated. The effects of this virus, which occurs not only in honey bees and bumble bees but a number of other arthropods, are unknown (Tripodi, pers. comm. 2016 in U.S. Fish and Wildlife Service 2016a, p. 42). We have no documentation in our files or evidence of direct effects of acute bee paralysis on *B. franklini*.

Deformed wing virus (DWV) is a honey bee pathogen that results in crippled and deformed wings. DWV was thought to only affect honey bees, until 2004, when dead *Bombus terrestris* and *B. pascuorum* queens with deformities resembling those in honey bees were observed. Some virus has been shown to be transmitted from honey bees to bumble bees (Singh *et al*. 2010, p. 1; Furst *et al*., 2014, p. 3). Tripodi (pers comm. 2016 in U.S. Fish and Wildlife Service 2016a, p. 42) notes that DWV has been detected in wild and commercially-sources bumble bees. Although virological research focuses on honey bees, many of the 24 viruses isolated from honey bees have a broad host range, infecting some *Bombus* species (Manley *et al*. 2015, p. 2). Commercial bumble bee producers sometimes introduce young honey bees to nesting bumble bees queens to stimulate egg-laying, and commercially raised bumble bee colonies are often fed pollen collected by honey bees, thus providing a potential interface that exposes bumble bees to diseases carried by the honey bees (Genersch *et al*. 2006, pp. 61-62). Infected bees with deformed wings are unable to forage. Bumble bees that were observed with deformities were also not viable (Genersch *et al*. 2006, p. 61). The Petition reports of unpublished personal observations of DWV symptoms in commercially raised *B. impatiens* colonies in North America, but no research is available to determine if other species of bumble bees are also susceptible to this disease (Xerces Society and Thorp 2010, p. 17). While evidence exist that DWV does affect *Bombus* spp., we do not have documentation in our files or evidence of direct effects of DWV on *B. franklini.*

Notwithstanding the studies postulating *Nosema bombi* spillover around commercial greenhouses (such as Colla *et al*. 2006, entire), as well as the timing of commercialization and *Bombus* declines, Szabo *et al*. (2012, p. 237) found that pathogen spillover in this form cannot fully account for the *Bombus* declines. Malfi and Roulston (2014, p. 24) concluded that the evidence linking *N. bombi* to the *Bombus* declines is correlative but does suggest species undergoing range reductions are more susceptible to *N. bombi* infections, while noting that it is nonetheless possible that elevated levels of *N. bombi* are natural in host species. Several experts

have surmised that *N. bombi* may not be culpable (or the only culpable) pathogen in the precipitous decline of wild *Bombus* in North America (e.g., D. Goulson pers. comm. 2016, J. Strange and A. Tripodi (USDA) pers. comm. 2016 in U.S. Fish and Wildlife Service 2016a, p.41). Cameron *et al*. (2011b, p. 662) sum up the likelihood of pathogen spread being a primary cause of *Bombus* declines by stating that higher pathogen prevalence and reduced genetic diversity are realistic predictors of patterns of decline in North American bumble bees, although cause and effect remain uncertain.

Known pathogens occur within the historical range of *Bombus franklini*, and we have evidence of several pathogens infecting closely related species within that range. Although we have no direct evidence of pathogens playing a role in the decline of *B. franklini*, the disappearance of *B. franklini* occurred soon after a period of potential exposure to introduced pathogens, particularly *Nosema bombi* which is known to have a more severe impact on rare species like *B. franklini*. Decline of other closely related pollinators has been associated with these pathogens and it is highly likely the factor has had some negative influence on the health of *B. franklini* populations.

3.1.2 Pesticides

Bumble bee exposure to pesticides can occur from direct spray or drift (Johansen and Mayer 1990), or from gathering or consuming contaminated nectar or pollen (Morandin *et al*. 2005, p. 619). Lethal and sublethal effects on bumble bee eggs, larvae, and adults have been documented for many different pesticides under various scenarios (Kevan 1975, p. 301; Johansen 1977, p. 178; Plowright *et al*. 1978, p. 1145; Plowright *et al*. 1980, p. 765; Kearns and Inouye 1997, p. 302; Kearns *et al*. 1998, p. 91–92; Kevan 1999, p. 378; Thompson 2001, p. 305; Gels *et al*. 2002, p. 722; Morandin *et al*. 2005, p. 619; Mommaerts *et al*. 2006, p. 752; Goulson *et al*. 2008, pp. 11.4–11.5). Documented sub-lethal effects to individual bumble bees and colonies include reduced or no male production (Fauser-Misslin *et al*. 2014, pp. 453-454; Feltham *et al*. 2014, p. 320; Gill *et al*. 2012, p. 107; Mommaerts *et a*l. 2006, pp. 3-4; Mommaerts *et al*. 2010, pp. 211-212; Scholer and Krischik 2014, p.7), reduced or no egg hatch (Elston *et al*. 2013, pp. 6-7; Mommaerts *et al*. 2006, pp.3-4), reduced queen production (Fauser-Misslin *et al*. 2014, pp. 453-454; Feltham *et al*. 2014, p. 320; Whitehorn *et al*. 2012, p. 352), reduced queen longevity (Fauser-Misslin *et al*. 2014, pp. 453-454), reduced colony weight gain (Feltham *et al*. 2014, p. 320; Whitehorn *et al*. 2012, p. 351; Scholer and Krischik 2014, p. 6), reduced brood size (Elston *et al*. 2013, p. 6; Feltham *et al*. 2014, p. 320; Gill *et al* 2012, p. 107; Laycock *et al*. 2012, p. 3), reduced feeding (Fauser-Misslin *et al*. 2014, pp. 453-454; Feltham *et al*. 2014, p. 320; Gill *et al*. 2012, p. 107; Gill and Raine 2014, pp. 211-212; Scholer and Krischik 2014, p. 5; Thompson *et al*. 2014, pp. 2-3), impaired ovary development (Laycock *et al*. 2012, pp. 4-5), and an increased number of foragers or foraging trips or duration (interpreted as risky behaviors) (Gill *et al*. 2012, p. 107; Gill and Raine 2014; pp. 5-8; Feltham *et al*. 2014, p. 320).

Studies have also found evidence of adverse impacts to bumble bee habitat associated with pesticides due to changes in vegetation and the removal or reduction of flowers needed to provide consistent sources of pollen, nectar, and nesting material (Johansen 1977, p. 188; Plowright *et al*. 1978, p. 1145; Williams 1986, 54; Kearns and Inouye 1997, p. 302; Smallidge and Leopold 1997, p. 264; Kearns *et al*. 1998, p. 91–92; Shepherd *et al*. 2003). Declines in

bumble bees in parts of Europe have been at least partially attributed to the use of pesticides (Williams 1986, p. 54; Kosior *et al*. 2007, p. 81).

Although the use of land for agricultural purposes has traditionally involved the use of pesticides and other products toxic to bees, one particular class of insecticides known as neonicotinoids have been strongly implicated in the decline honey bees worldwide as well as several *Bombus* species, due to the contemporaneous introduction of neonicotinoid insecticides and the precipitous decline of those species (Pisa *et al*. 2015, p. 69; Goulson 2013, p. 7-8; Colla and Packer 2008, p. 10). Neonicotinoids are based on nicotine compounds; they are systemic insecticides that act as a neurotoxin and varying levels of toxicity, affecting the central nervous system of insects. Laboratory data indicated that neonicotinoids kill insects by interfering with the receptors of the insects' nervous system, causing overstimulation, paralysis, and death. The neonicotinoid family of insecticides includes acetamiprid, clothianidin, imidacloprid, nitenpyram, nithiazine, thiacloprid and thiamethoxam. They are used in a wide variety of agricultural applications.

Imidacloprid became widely used in the United States starting in the early 1990s, followed by clothianidin and thiamethoxam in the early 2000s (Douglas and Tooker 2015, pp. 5091-5092). As of 2013, nearly all corn planted in the United States was treated with neonicotinoids and various fungicides (Stokstad 2013, p. 675) and as of 2014 approximately one-third of the soybean acreage in the United States was planted with neonicotinoid-coated seeds (Douglas and Tooker 2015, p. 5090; U. S. Geological Survey National Pesticide Synthesis 2016). Imidacloprid is one of the most widely used insecticide in the world (Yamamoto and Casida 1999).

Most studies examining the effect of neonicotinoids on bees have been conducted using the European honey bee (*Apis mellifera*), and a handful of *Bombus* species including *B. terrestris*, *B. impatiens,* and *B. affinis* (Lundin *et al.* 2015, p. 7), but there have been no studies on *B. franklini* (Lundin *et al.* 2015, p. 7). We infer, however, that studies of the effect of pesticides to other *Bombus* species will likely reflect their effects on *B. franklini* because these species have similar life history traits (e.g., generalist foragers collecting pollen from same food sources). Bumble bees may, in fact, be more vulnerable to pesticide exposure than honey bees. Bumble bees are more susceptible to pesticides applied early in the year than are honey bees, because for one month every year the entire bumble bee population depends on the success of the queens to forage and establish new colonies. Also, because most bumble bees have smaller colonies (N=~ several hundred to a thousand) than honey bees (N=~30,000), a single bumble bee worker is more important to the survival of the colony than a single honey bee worker (Thompson and Hunt 1999, p. 155; Sponsler *et al.*. 2017, p. 30). Furthermore, since bumble bees nest underground, they are additionally exposed to pesticide residues in the soil, specifically when the application of a pesticide overlaps with colony establishment in the spring (Arena and Sgolastra 2014, p. 333). Moreover, bumble bee larvae consume large amounts of unprocessed pollen, and therefore, are much more exposed to pesticide residues in pollen (Arena and Sgolastra 2014, p. 333).

Studies (*e.g*., Piiroinen and Goulson 2016, entire) are now emerging that have simultaneously documented effects to bumble bees and honey bees at field-realistic levels. As generalist foragers, both honey bees and bumble bees are often collecting from the same pollen sources (E.

Evans, pers. comm. 2016, in U.S. Fish and Wildlife Service 2017, p. 45). Based on detected concentrations in the wild and the results of toxicity test, as well as the frequency of hives across the landscape, Sanchez-Bayo and Goka (2014, pp. 12-14) predicted that exposure to thiamethoxam, imidacloprid, and clothianidin (along with two organophosphates--phosmet and chloropyrifos) pose the greatest risk to honey bees at a global scale. However, the additive and synergistic effects of exposure to multiple pesticides and multiple times may exacerbate the toxicity of exposure to any single pesticide, and thus, additional pesticides in combination with others, may pose risks to bees as well. Several studies have revealed that bees are often chronically exposed to a cocktail of pesticides throughout their lifetime (Sanchez-Bayo and Goka 2014, p. 5; Chauzat *et al.* 2006, pp. 256-257; Mullin *et al.* 2010, pp. 3-8; Krupke *et al.* 2012, pp. 3-5). For example, Sanchez-Bayo and Goka (2014, p. 5) detected 161 different pesticides at honey bee colonies.

The effects of chronic exposure to multiple pesticides are poorly understood and are not regularly examined in risk assessments (Goulson 2016, p. 4), and thus, the toxicity results, may underestimate the actual risks posed to bees. Furthermore, pesticide formulations typically contain less than 50 percent active ingredients with the remainder being surfactants (surface active agent that reduces the surface tension of water) and solvents (collectively, referred to as adjuvants). As bees forage, they are exposed to many adjuvants as well as active ingredients (Mullin *et al.* 2015, p. 7). Adjuvants, however, are not typically included in risk assessments that are required for pesticide registration (Mullin *et al.* 2015, p. 2), and are therefore, less studied, but can be as or more toxic to bees as the active ingredients (Mullin *et al.* 2015, p.4). For example, bumble bees are highly susceptible to emulsifiers such as perfluorooctane sulfonic acid (Mommaerts *et al*. 2011, pp. 450-452). Goodwind and McBrydie (2010, p.232) found that four of 11 commercially available spray adjuvants were toxic to honey bees at field rates. Furthermore, active ingredients and inert ingredients may interact synergistically, causing impacts that would not occur by exposure to the active ingredients alone (Mullin *et al.* 2015, p. 3). Lastly, bees are exposed to a number of significant and interacting stressors (Goulson *et al.* 2015, entire), which can compound the effects of pesticides. Exposure to fungicides greatly increased the toxicity of insecticides in honey bees (Schmuck *et al.* 2003, pp. 82-85; Iwasa *et al.* 2004, p. 376; Piling and Jepsen 1993, pp. 295-296; Mullin *et al.* 2015, p. 4). Honey bees exposed to fungicides had reduced colony nutrition and higher virus levels to fungicides (DeGrandi-Hoffman *et al.* 2015, pp. 2523-2524). Pettis *et al.* (2013, p. 4), for example, found increased probability of *Nosema* infection in honey bees feeding on pollen with high fungicide loads. Several studies found exposure to insecticides reduced resistance to diseases (Fauser Misslin *et al.* 2014, pp. 454-455, Pettis *et al.* 2013, p. 4), and exposure to dietary related stresses (*e.g.*, short-term starvation) reduced the ability of bees to cope with toxins (Brown *et al.* 2000, p. 424; Tyler *et al.* 2006, p. 2; Moret and Hempel 2000, p. 1167). Piiroinen and Goulson (2016, pp. 3-5) found that exposure to *N. ceranae* reduced learning in honeybees and bumble bees, but both species reacted differently to the combination of pathogen plus pesticide exposure.

Determining the extent of bee fatality caused by pesticides is difficult due to the myriad of other potential stressors (*e.g.*, pathogens, parasitoids, and diseases) and possible synergistic effects of these sources. There are known instances where neonicotinoids such as clothianidin have adverse effects to immunity and promote replication of viral pathogens in bees (*e.g.*, DiPrisco *et al.* 2013,

p. 3). The interruption or disruption of endocrine functions is related to the function of species' immune systems and the application of neonicotinoids may exacerbate the effects of pathogens.

To assess the perceived cause and effect relationship between neonicotinoid application levels and *Bombus franklini* declines, we gathered available data on pesticide use for a subset of chemicals and charted the application trend over time throughout the range of *B. franklini*. Specifically, using pesticide application rate data collected from 1995 to 2015 (United States Geological Survey National Pesticide Synthesis, accessed November 2017), we examined the trend in use of three prevalent neonicotinoids; imidacloprid, clothianidin, and thiamethoxam over time in 5 counties with recent (since 1995) *B. franklini* occurrences. Limited information on neonicotinoid application in California is available in this dataset, so in addition we received data from the California Pesticide Information Portal (accessed December 2017). All three chemicals were added for each year to get a total application rate of imidacloprid, clothianidin, and thiamethoxam combined. While we chose to focus these trend analyses on three commonly used and studied neonicotinoids, we recognize that there are a myriad of pesticides, inactive ingredients, and other chemicals that have documented negative effects on bees (as discussed above) and could be similarly analyzed for application rate trends in our study area. Furthermore, the vast majority of neonicotinoids are used as seed treatments on grains and other field crops (Oregon Department of Agriculture 2018, pers. comm.). The National Pesticide Synthesis data are both a high and low estimate of application rates based on the best available data – see Appendix 3 for more information on the sources, assumptions, and use limitations of the data. We also recognize that the timing, location, and methods of pesticide application play a role in their effectiveness on target species (i.e., aerial spraying of row crops vs. placement of ant traps). For our study area (Jackson, Douglas, and Josephine Counties in Oregon as well as Trinity and Siskiyou Counties in California), the first reported use of Imidacloprid was in 1996; thiamethoxam first reported in 2001, and clothianidin in 2004. Total estimated neonicotinoid applications increased from 53.35 pounds/acre (24.19 kilograms/acre) in 1996 to 1,144.128 pounds/acre (518.86 kilograms/acre) in 2014.

While the rapid decline of *Bombus franklini* observations occurred shortly after the introduction of neonicotinoid pesticide use within the historic range of the species, the exponential growth of neonicotinoid applications starting in 2011 took place five years after the last observation of the species so it is unlikely that the introduction and use of neonicotinoid pesticides alone can account for the decline in *B. franklini*. There have been no studies on the effects of pesticide use on *B. franklini*, no documented discoveries of any *B. franklini* injured or killed by pesticides. Furthermore, the species is a habitat generalist and is not known to have a close association with agricultural lands so it may have less exposure to pesticides than some other *Bombus* species. However, pesticide use does occur in the range of *B. franklini* and confirmed effects to honey bees and other *Bombus* species suggests that pesticide use could have been a factor in the decline of *B. franklini*. The similarity in foraging traits that *B. franklini* has with both honey bees and the other *Bombus* species (e.g., generalist foragers collecting pollen from similar food sources) allows us to infer that that *B. franklini* would suffer exposure to and impacts from pesticides in similar measure to other *Bombus* species when *B. franklini* is in areas where pesticides are applied.

3.1.3. Habitat Loss and Degradation

Habitat loss and degradation was identified by the Petitioners as a threat contributing to the decline of *Bombus franklini* (Xerces Society and Thorp 2010, p. 4). Habitat loss and degradation can be manifest in many forms over different spatial and temporal scales. In this section we look at habitat loss and degradation in the range of *B. franklini* and its potential effects on the species through agricultural intensification, natural and introduced fire, and urbanization; livestock grazing and climate change are discussed later in separate sections.

Conversion of natural habitat that is rich in flowers to farmlands, urban and suburban areas, and other uses is a primary cause of bumble bee habitat loss (Goulson *et al*. 2015, p.2). Agricultural intensification can result in habitat loss for bumble bees, as these practices often result in the planting of monocultures, which tend to provide floral resources for a limited period of time, rather than throughout the colony life cycle. Studies have confirmed that agricultural intensification can negatively impact wild bees by reducing floral resource diversity and abundance (Johansen 1997, p. 177; Williams 1986, p. 57; Kearns *et al*. 1998, p. 89; Hines and Hendrix 2005, p. 1477; Carvell *et al*. 2006, p. 481; Diekotter *et al.* 2006, p. 57; Fitzpatrick *et al*. 2007, p. 185; Kosior *et al.* 2007, pp. 81, 84-86; Ockinger and Smith 2007, pp. 50; Goulson *et al.* 2008, p. 11.1; International Union for Conservation of Nature 2009, p. 2; Le Feon *et al.* 2010, p. 143). Agricultural intensification was determined to be a primary factor leading to the local extirpation and decline of Illinois bumble bees (Grixti *et al*. 2009, p. 75). An increase use of herbicides often accompanies development and agricultural intensification, and the wide-spread use of herbicides in agricultural, urban and even natural landscapes has led to decreases in flowering plants (Potts *et al*. 2010, p. 350).

Douglas, Jackson and Josephine Counties in Oregon, and Siskiyou and Trinity Counties in California, are generally characterized as rural, agriculturally based counties with large proportions of public land and which lack the larger population centers found to the north and south of the historical range of *Bombus franklini*. Information specific to agricultural intensification within the historic range of *B. franklini* is not available at the spatial and temporal scales needed to quantify this threat (United States Department of Agriculture – National Agriculture Statistics Service, pers. comm. 2017). However, between 1997 and 2012, Oregon saw a decrease in both the overall number of farms and ranches, as well as a decrease of more than a million acres of land in agriculture (United States Department of Agriculture – National Agriculture Statistics Service 2015, p. 6). Within the historic range of *B. franklini*, Douglas, Jackson and Josephine Counties all saw a similar decrease of total acres in agricultural cropland, as summarized in Table 4 below (United States Department of Agriculture – National Agriculture Statistics Service, pers. comm. 2017). While the total number of acres of agricultural cropland is not synonymous with agricultural intensification (specifically, the expansion of monocultures), a decrease in total acres of agriculture leads us to conclude that agricultural intensification is not likely a major threat to *B. franklini.* We have no documentation in our files or any direct evidence that agricultural intensification has contributed to the decline of *B. franklini*.

Table 4. Acres of Agricultural Cropland in Douglas, Jackson, and Josephine Counties in Oregon
Data: USDA-NASS 2015, 2017

	Acres of Agricultural Cropland			
	1997	**2002**	**2007**	**2012**
Douglas	123,133	107,503	73,559	49,222
Jackson	71,251	67,762	56,530	32,765
Josephine	17,767	15,860	17,389	8,365
Oregon	1997	2002	2007	2012
Number of farms and ranches	39,975	40,033	38,533	35,439
Total land in agriculture (millions of acres)	17.7	17.2	16.4	16.3

Forty-two percent of the sites where *Bombus franklini* have been located (18 of 43) occur on federally owned land, primarily the U.S. Forest Service and Bureau of Land Management. This notable proportion could be due to a good percentage of *B. franklini* occurring on Federal land or simply that searches for *B. franklini* often occur on Federal land. Very little habitat on these ownerships has been permanently altered or lost through development or agricultural intensification.

Fire caused by both natural and manmade factors has been an important change agent on the landscape in the range of *Bombus franklini.* Because fire reduces natural succession of forests through the burning of encroaching woody plants, fire is a primary factor in the maintenance of grassland and meadow habitat that supports *Bombus* species (Shultz and Crone 1998, p. 244; Huntzinger 2003). With the increase in human development came fire suppression to limit damage to manmade structures. Fire suppression allows woody encroachment to occur and the diverse landscape created by fire (open areas mixed within forested areas) is being replaced by increasing areas of denser forested habitat; the open areas that facilitated the growth of diverse understory plant communities are greatly reduced from their historical condition (Ruchty 2011, p. 26). Conifer species now cover much of the area that was previously open meadow habitat in the range of *B. franklini* (Panzer 2002; Shultz and Crone 1998, p. 244). This loss of habitat by fire suppression likely played some role in the decline if *B. franklini* by limiting the availability and diversity of floral resources and nest and overwintering habitat. However, because there is still healthy meadow habitat located in areas where *B. franklini* were previously found, we do not believe that loss of habitat from fire suppression was a major factor in the decline of the species, particularly the precipitous decline that occurred after 1998.

The increased fuel loads from fire suppression increase the potential for catastrophic, large scale, and high temperature wildfires. Any *Bombus* colonies in the path of this type of fire would be at risk of extirpation. Wildfire may have played a role in the decline of *B. franklini* by extirpating some historical populations in the range, but we have no information confirming this. We have no information that suggests that any known *B. franklini* occurrence sites were in the path of catastrophic wildfires at the time they were occupied. Controlled burning became a management tool for reducing potential fuel loads for wildfire; controlled burning and other fuel reduction

activities are carried out by Federal land management agencies including the US Forest Service and Bureau of Land Management in the range of *B. franklini*. The effects of fire on invertebrates depends greatly on the biology of the specific taxa (Gibson *et al*. 1992) and in the case of *B. franklini*, controlled burns could certainly cause death of individual bees and negative effects to a colony. However, we have no information to indicate that controlled burns were a factor in the decline of *B. franklini.*

Ongoing urbanization also contributes to the loss and fragmentation of natural habitats. Urban gardens and parks may provide habitat for some pollinators including bumble bees (Frankie *et al*. 2005, McFrederick and LeBuhn 2006) but they tend not to support the species richness of bumble bees that can be found in nearby undeveloped landscapes (Xerces Society and Thorp 2010, p. 13), or that which was present historically (McFrederick and LeBuhn 2006). *Bombus franklini* and *B. occidentalis* have both been observed in urban areas of Ashland, Oregon, but not since 2002. A study in Boston, Massachusetts, concluded that human built structures, such as roads and railroads, can fragment plant populations and restrict bumble bee movement (Bhattacharya *et al.* 2003, p. 37). Urban development may also lead to direct mortality, i.e. through vehicle collisions (Goulson *et al* 2015, p. 2). Another study of the factors adversely affecting bumble bees and cuckoo bees in Europe found the expansion of urban areas to be an important driver of pollinator loss in approximately half of the countries examined (Kosior *et al.* 2007, p. 81).

Table 5 shows the 1995 and 2017 populations for Douglas, Jackson, and Josephine Counties, as well as the population for Ashland, OR. Table 5 also shows the population growth estimates that were completed in 2015 for each of the counties.

Table 5. Human population growth estimates for Douglas, Jackson, and Josephine Counties in Oregon and Ashland, Oregon.

	1995 population	**2017 population***	**2035 estimate**	**2065 estimate**
Douglas County	98,820	111,180	129,910	152,910
Jackson County	167,330	216,900	246,575	306,575
Josephine County	71,290	85,650	99,720	121,720
Ashland, Oregon	17,985	20,700		
*2017 data preliminary				

For *Bombus* in general, loss and degradation of habitat is known to reduce both bee diversity and abundance (Potts *et al.* 2010, p. 348-349). Habitat fragmentation can alter pollinator community composition, change foraging behavior of bumble bees, and reduce bee foraging rates, and is believed to be one of the factors contributing to the decline of several bumble bee and cuckoo bee species in Europe (Kearns and Inouye, 1997, p. 299; Ockinger and Smith 2007, p. 50; Rusterholz and Baur 2010, p. 148; Kosior *et al* 2010, pp. 81). Bumble bees have been found to be susceptible to the disruption of healthy metapopulation structures due to fragmentation, and may decrease source populations of bumble bees for recolonization (National Research Council 2007, p. 93; Goulson *et al.* 2008, p. 11.7). Other studies have suggested that fragmented bumble bee populations can suffer from inbreeding depression as a result of geographic isolation (Darvill *et al*. 2006, p. 601, Goulson *et al.* 2008, p. 11.7) (see sections 2.2.2 and 3.1.6 for more on genetic impacts from small population sizes).

Although habitat loss and fragmentation has established negative effects on bumble bees (Goulson *et al*. 2008; Williams and Osborne 2009, pp. 371-373), many feel it is unlikely to be a main driver of the recent, widespread North American bee declines (Szabo *et al*. 2012, p. 236; Colla and Packer 2008, p. 1388; Cameron *et al*. 2011b, p. 665). Further, habitat remains generally intact and in good condition throughout the known historic *Bombus franklini* locations and all of the recent focused survey areas, with the notable exceptions being the creation of Lake Applegate upon the completion of Applegate Dam in the fall of 1980 and a report of soil modification on a portion of the Gold Hill site. The Applegate Dam project inundated two historic *B. franklini* locations (Copper and 2 miles north of Copper), with historic observations from 1963 and 1968 (Xerces Society and Thorp 2010, p. 13; Thorp, pers. comm. 2017). The Petition noted that in 2004, soil had been excavated and deposited in a portion of the Gold Hill area (Xerces Society and Thorp 2010, p. 13). The last observation of *B. franklini* at Gold Hill was in the year 2000, and the site was revisited 14 times over the next three years with no observations of the species. At both Lake Applegate and Gold Hill, we don't know if the species was still using the habitat by the time the activities took place. Overall, many feel that habitat loss and fragmentation are not a main driver of the decline of *B. franklini*, particularly since many other *Bombus* species have been recorded in the habitat where searchers have looked for *B. franklini* (Thorp, pers. comm. 2017; Godwin, pers. comm. 2017; Colyer, pers. comm. 2017).

3.1.4 Livestock Grazing

Livestock grazing occurs on public land on much of the historic range of *Bombus franklini*. The Petition stated that livestock grazing may adversely impact bumble bee populations by: (1) depleting food resources (Morris 1967, p. 472; Sugen 1985, p. 299; Kruess and Tscharntke 2002b, p. 1570; Vazquez and Simberloff 2003, p. 1081; Hatfield and LeBuhn 2007, p. 150); (2) trampling nesting sites (Sugden 1985, p.299); and (3) negatively impacting ground-nesting rodents (Johnson and Horn 2008, p. 444; Schmidt *et al*. 2009, p. 1), which may in turn reduce the number of nest sites available for bumble bees (Xerces Society and Thorp 2010, p. 13). The Petition also stated that livestock grazing has differing impacts on flora and fauna based on the type, habitat, intensity, timing and length of grazing (Gibson *et al*. 1992, p. 174; Carvell 2002, p. 44; Kreuss and Tscharntke 2002a, p. 293; Kruess and Tscharntke 2002b, p. 1577; Xerces Society and Thorp 2010, p. 13). Several studies of livestock grazing impacts on bees suggest increased intensity of livestock grazing affects the species richness of bees (Morris 1967, p. 473; Sugden 1985, p. 309; Vazquez and Simberloff 2003, p. 1080; Hatfield and LeBuhn 2007, p. 156). In contrast, one study cited in the Petition suggests that grazing, especially by cattle, can play a key positive role in maintaining the abundance and species richness of preferred bumble bee forage (Carvell 2002, p. 44).

Overgrazing by sheep between 1890 and 1920, resulted in trampling vegetation and denuding soils, and is currently evident today in the continuing erosion of the granitic soils of the McDonald Basin, Siskiyou Gap, Mt. Ashland, and the Siskiyou Crest (LaLande 1995, p. 31; T. Atzet, Siskiyou Field Institute, Selma, Oregon, pers. comm. 2017). While sheep overgrazing likely has degraded *B. franklini* habitat, we have no specific information on the effects of this habitat loss and fragmentation. Evidence of livestock grazing has been observed interspersed within abundant floral resources in *B. franklini* habitat during several recent targeted survey efforts (Brooks 1997 pers. comm.; U.S. Fish and Wildlife Service 2016; U.S. Fish and Wildlife

Service 2017; P. Trail, U. S. Fish and Wildlife Service, Ashland, Oregon, pers. comm. 2017). However, no specific information is available on the impacts of livestock grazing on *B. franklini* making it impossible to connect the activity to any specific species response. The number of grazing allotments have decreased on The Rogue-River Siskyou NF, particularly on the Siskiyou Mountains Ranger District (Applegate) in the last 20 years; the grazing on High Cascades Ranger District (near Prospect, OR), have not changed in the last 20 years (J. von Kienast, pers. comm. 2018). Generally the dates that cows are on the allotments start on June 15th and stay on until October but dates vary by allotment. Most of the locations for *B. franklini* on the Rogue River-Siskiyou NF overlap with grazing allotments. Cattle grazing has been observed at all *Bombus* survey locations on the High Cascades Ranger District. (S. Colyer pers. comm. 2018). Outside of these areas, we have no new information that the timing, location, intensity, or duration of grazing has changed, with the exception of the Cascade-Siskiyou National Monument, where most grazing has been retired on the Cascade-Siskiyou National Monument (Trail, pers. comm. 2017) (See Figure 1).

3.1.5 Climate Change

Global climate change was identified in the Petition as a threat to *Bombus franklini* (Xerces Society and Thorp 2010, pp. 20-21). Climate change may cause shifts in the range of host plant species, which can be especially detrimental to dependent pollinators when combined with habitat loss (Xerces Society and Thorp 2010, p. 20; Schroeder, pers. comm. 2017). Specific impacts of climate change on pollinators are not well understood; most of the existing information on climate change impacts to pollinators comes from studies on butterflies – studies specifically relating to bumble bees are scant, and we found no climate change information specific to *B. franklini*.

Climatic changes in temperature, precipitation, and the increased frequency of storm events (Intergovernmental Panel on Climate Change 2013, entire) can affect population sizes directly, affecting survival and reproduction (Bale *et al.* 2002, p. 11; Roland and Matter 2016, p. 22). These climatic changes can also affect populations indirectly, by altering resource availability and species interactions (Boggs and Inouye 2012, p. 505; Hoye *et al.* 2013, p. 762; Kudo and Ida 2013, p. 2319). Some studies suggest that bumble bee populations are responding to climate change with recent latitudinal and elevational range shifts (Ploquin *et al* 2013, p. 9; Pyke *et al* 2016, p. 11). Some bumble bee populations are active earlier in the season than in the past (Bartomeus *et al* 2011, p. 20646). Ogilvie *et al.* (2017, p. 1) found that bumble bee abundances were driven primarily by the indirect effects of climate on the temporal distribution of floral resources.

The changes in climate likely to have the greatest effects on bumble bees in general include increased drought, increased flooding, increased storm events, increased temperature and precipitation events, early snow-melt, late frost, and increased variability in temperature and precipitation. These climate changes may lead to decreased resource availability (due to mismatches in temporal and spatial co-occurrences), decreased availability of nesting habitat (due to higher temperatures), and increased pressures from pathogens and non-native species (Goulson *et al*. 2015, p. 4; Goulson, pers. comm. 2016 in U.S Fish and Wildlife 2016, p. 52; Kerr

et al. 2015, pp. 178-179; Potts *et al*. 2010, p. 351; Cameron *et al.* 2011a, pp. 35-37; Williams and Osborne 2009, p. 371).

Climate variability may lead to range shifts, such that there is spatial mismatch among plants and their pollinators (Memmott *et al*. 2007, p. 712). While this has been demonstrated in butterflies (Forister *et al*. 2010, pp. 2088-2089; Hickling *et al*. 2006, p. 452), it may be less of a factor for bumble bees. As generalist foragers, they do not require synchrony with a particular plant species. However, elevational range shifts have been documented in some bumble bees (e.g., Pyke *et al*. 2016, pp. 8-10; Kerr *et al* 2015, p. 179). Temporal mismatches may be more of an issue for bumble bees due to their long active season, during which they require consistent access to floral resources. Floral resource availability in early spring is particularly crucial for bumble bees, as that is when they first emerge and initiate nests. Thus, temporal asynchrony could lead to diminished resource availability at times that are critical to bee development and colony success. Other potential effects from climate change include increased flooding and storm events, which may directly reduce available nesting habitat and hibernating habitat by inundating those areas (Goulson *et al*. 2015, p. 4). Changes in rodent populations due to climate change may also reduce nesting habitat, as bumble bees often use rodent burrows as nesting areas. Furthermore, bumble bees are poorly adapted to high temperatures, and thus are vulnerable to increased stress from overheating.

Several of the targeted *Bombus franklini* and *B. occidentalis* survey reports between 2015 and 2017 include mention of widespread hot, dry climate affecting timing and abundance of floral resources during the surveys (Bureau of Land Management 2015; Trail, pers. comm. 2017), indicating that at least at local scales in recent years, changing climate conditions may have affected resources available to *Bombus* colonies. Although the Olgilvie *et al.* study as well as the survey reports suggest potential indirect effects of climate change on *Bombus*, we have no information to indicate that the effects of climate change were connected to the decline of *B. franklini;* numerous *Bombus* species persist in areas considered to maintain good quality habitat for *B. franklini* (Pool 2014, entire; Colyer 2016, entire).

3.1.6 Small Population Dynamics

Small population size has been identified as a threat to *Bombus franklini*. The Petition (Xerces Society and Thorp, p. 20) states that *B. franklini* is rare and has always had very small populations (relative to other similar, native bumble bees in the western United States), and likely have low genetic diversity, making the species more vulnerable to habitat change or loss, parasites, diseases, stochastic events, and other natural disasters such as droughts (Xerces Society and Thorp 2010, p. 20).

As stated in section 2.1.2, between 1998 and 2006, the number of *Bombus franklini* observations declined from a high of 98 at 8 locations, to a lone individual in 2006. No observations of *B. franklini* have occurred since 2006 despite an increase in the survey effort.

As mentioned in section 2.2.2, bumble bees exhibit a haplodiploidy sex determination system. In these systems, unfertilized (haploid) eggs become males that carry a single set of chromosomes, and fertilized (diploid) eggs become females that carry two sets of chromosomes. This may

result in lower levels of genetic diversity than the more common diploid-diploid sex determination system, in which both males and females carry two sets of chromosomes. Haplodiploid organisms (such as bumble bees) may be more prone to population extinction than diploid-diploid organisms, due to their susceptibility to low population levels and loss of genetic diversity (Packer and Owen 2001, p. 26; Zayed and Packer 2005, p. 10742; Darvill *et al.* 2006, p. 601, Ellis *et al*. 2006, 4375; Goulson *et al*. 2008, p. 11.7-11.9). Inbreeding depression in bumble bees can led to the production of sterile diploid males (Goulson *et al*. 2008, p. 11.7) and has been shown to negatively affect bumble bee colony size (Herrman *et al*. 2007, p. 1167), which are key factors in a colony's reproductive success. Diploid male production has been detected in naturally occurring populations of bumble bees, and recent modeling work has shown that diploid male production, where present, may initiate a rapid extinction vortex (a situation in which genetic traits and environmental conditions combine to lead a species to extinction) (Goulson *et al*. 2008, p. 11.8). *Bombus franklini* is a haplodiploid organism with a relatively small population size compared to other *Bombus* species. A haplodiploid genetic system makes bees very vulnerable when populations get small because of inbreeding and the production of sterile males (Colla, 2018, pers. comm.). Although we have no direct evidence that small population size or a rapid extinction vortex contributed to the decline of the species, the genetic system and historically small population size of *B. franklini* likely heightened the species' vulnerability to other stressors in the environment.

3.1.7 Competition from non-native bees

The European honey bee (*A. mellifera*), was first introduced to eastern North America in the early 1620s, and introduced to California in the early 1850s (Xerces Society and Thorp, p.21). The resources of *A. mellifera* and native *Bombus* species may overlap resulting in the potential for increased competition for resources (Thomson 2004, p. 458; Thomson 2006, p. 407; Thomson 2016, p. 1247). Decreased foraging activity and lowered reproductive success of *Bombus* colonies have been noted near *A. mellifera* hives (Evans 2001, p. 32–33; Thomson 2004, p. 458; Thomson 2006, p. 407). Additionally, the size of workers of native *Bombus* species were noticeably reduced where *A. mellifera* were present, which may be detrimental to *Bombus* colony success (Goulson and Sparrow 2009, p. 177). As noted in the 2010 Petition, is likely that the effects discussed in these studies are local in space and time, and most pronounced where floral resources are limited and large numbers of commercial *A. mellifera* colonies are introduced (Xerces Society and Thorp, p. 21). We could not find information to indicate that any area of *B. franklini* habitat in the range of the species has limited floral resources and large numbers of *A. mellifera*. We have no information related to the specific placement of commercial honey bee colonies in or near *B. franklini* habitat. Furthermore, *A. mellifera* have been present without noticeable declines in *Bombus* populations over large portions of their ranges (Xerces Society and Thorp, p. 21) and we have no new information that connects competition from *A. mellifera* to the decline of *B. franklini*, particularly the noticeable decline after 1998.

There is potential for non-native commercially raised bumble bees to naturalize and outcompete native bumble bees for limited resources such as nesting sites and forage areas. Five commercially reared *Bombus impatiens* workers and one queen were captured in the wild near greenhouses where commercial bumble bees are used, suggesting this species may have naturalized outside of its native range. In this study, *B. impatiens*, which has a native range in

eastern North America, was detected in western Canada (Ratti and Colla 2010, pp. 29–31). A study in Japan found that non-native *B. terrestris* colonies founded by bees that had escaped from commercially produced colonies had over four times the mean reproductive output of native bumble bees (Matsumura et al. 2004, p. 93). A study in England found that commercially raised *B. terrestris* colonies had higher nectar-foraging rates and greater reproductive output than a native subspecies of *B. terrestris* (Ings et al. 2006, p. 940). The 2010 Petition noted that *B. impatiens* colonies were imported to pollinate agricultural crops and strawberries in Grants Pass, Oregon, in the range of *B. franklini* (Xerces Society and Thorp, p. 18). Although non-native *Bombus* species in the range of *B. franklini* could outcompete *B. franklini* for floral resources and nesting habitat, we could not find any information to definitely connect competition with non-native bumble bees to the decline of *B. franklini*. Furthermore, invertebrate surveys in *B. franklini* habitat continue to show evidence of healthy populations of other native *Bombus* species unaffected by competition from non-native bees (Pool 2014, entire; Colyer 2016, entire).

3.2 Synergistic Effects

It is likely that several of these risk factors are acting additively and synergistically on *Bombus* species (Goulson *et al.* 2015, p. 5) and the combination of multiple stressors is likely more harmful than a stressor acting alone (Gill *et al.* 2012; Coors and DeMeester 2008; Sih *et al.* 2004). There is recent evidence that the interactive effects of pesticides and pathogens could be particularly harmful for bumble bees (Fauser-Misslin *et al.* 2014, pp. 453-455; Baron *et al.* 2014, pp. 463-465) and other bees (Alaux *et al.* 2010, pp. 775-777; Pettis *et al.* 2012, pp. 155-156; Vidau *et al.* 2011, pp. 3-5; Aufavre *et al.* 2012, pp. 2-3). Nutritional stress may compromise the ability of bumble bees to survive parasitic infections as evidenced by a significant difference in mortality in bumble bees on a restricted diet than well fed bees infected with *Crithidia bombi* (Brown *et al.* 2000, pp. 424-425). Bumble bees with activated immunity may have metabolic costs, such as increased food consumption (Tyler *et al.* 2006, p. 2; Moret and Schmid-Hempel 2000, pp. 1166-1167). Additionally, exposure to pesticides may increase with increased food consumption in infected bees (Goulson *et al.* 2015, p. 5). There is evidence that activating immunity impairs learning in bumble bees (Riddelland Mallon 2006, Alghamdi *et al.* 2008, p. 480). Impaired learning is thought to reduce the ability of bees to locate floral resources and extract nectar and pollen, therefore, exacerbating nutritional stresses (Goulson *et al.* 2015, p. 5). Further, evidence of the relationship between low genetic diversity and disease susceptibility was discussed in Cameron *et al.* (2011b, p. 665), who stated that declining North American species with low genetic diversity have higher prevalence of the pathogen *N. bombi*. Therefore, pathogens in combination with pesticides, and pathogens in combination with the effects of small population size likely hastened and amplified the decline of *B. franklini* to a greater degree than any one of the three factors would cause on its own.

3.3 Beneficial Actions

We are aware of no conservation efforts or beneficial actions specifically taken to address the stressors to *Bombus franklini*. Oregon does not include invertebrates on their state endangered species list (ODFW 2018) and California has no bees on its list of Threatened and Endangered Invertebrates (CDFW 2018). California has the species listed on its list of Terrestrial and Vernal

Pool Invertebrates of Conservation Priority but has no required actions or special protections associated with the listing (CDFW 2017, p. 10). *Bombus franklini* is on the species index for the U.S. Forest Service and Bureau of Land Management Interagency Special Status /Sensitive Species Program (ISSSSP) (ISSSSP 2018). Though the agencies do include the species in survey efforts and conduct general meadow enhancement activities like reducing conifer encroachment, there are no actions resulting from the ISSSSP classification that reduce or ameliorate known threats to *B. franklini*.

The U.S. Forest Service is working to include a section in all biological evaluations to address the effects from agency actions on pollinators. In addition, the Rogue River-Siskiyou National Forest is currently implementing projects and mitigations to create and enhance pollinator habitat (S. Colyer, pers. comm. 2018) The Oregon Department of Agriculture restricts some potential sources of *N. bombi* from entering the state for agricultural uses, including commercially-produced colonies of *Bombus impatiens*; only *Bombus* species native to Oregon are permitted for commercial pollination purposes (Oregon Department of Agriculture 2017, p. 5). California does however allow for the importation of *B. impatiens*, and other species such as the Blue Orchard Bee (*Osmia lignaria*) for pollination services with appropriate permits in both Oregon and California (California Department of Food and Agriculture 2017; Oregon Department of Agriculture 2017).

General awareness of honey bee colony losses and increase of conservation efforts for pollinators in general has likely had limited, indirect effects. Stemming from this general awareness is a reduction in the use of some pesticides throughout North America. Some local municipalities have enacted legislation against aerial pesticide applications but similar efforts have not been adopted at the state or range-wide scales (Powell 2017, p. 1; City of Portland 2015, p. 2). However, in the 2017 legislative session, Oregon passed an Avoidance of Adverse Effects on Pollinating Insects law (ORS 634.045) that is providing enhanced training of licensed and unlicensed pesticide applicators in the state (A. Melathopoulos, pers. comm. 2018). In January 2017, the U.S. Environmental Protection Agency's Office of Pesticide Programs published their *Policy to Mitigate the Acute Risk to Bees from Pesticide Products*, which recommended new labeling statements for pesticide products including warnings for pesticides with a known acute toxicity to bees including neonicotinoids (specifically including imidacloprid, clothianidin, and thiamethoxam) (United States Environmental Protection Agency 2017, p. 31). In addition, EPA is working with state and tribal agencies to develop and implement local pollinator protection plans, known as Managed Pollinator Protection Plans (MP3s). EPA is promoting MP3s to address potential pesticide exposure to bees at and beyond the site of the application. However, states and tribes have the flexibility to determine the scope of pollinator protection plans that best responds to pollinator issues in their regions. For example, state and tribal MP3s may address pesticide-related risks to all pollinators, including managed bees and wild insect and non-insect pollinators (United States Environmental Protection Agency 2018).

4.0 Analysis of Current Condition

As described in section 1.2, we applied the conservation biology principles of resiliency, representation, and redundancy (the 3Rs) as a framework to assess the viability of *Bombus*

franklini. For a species to sustain populations over time it needs a sufficient number and distribution of viable populations to withstand environmental stochasticity (resiliency), catastrophes (redundancy), and changes in its environment (representation). To assess resiliency and redundancy, we evaluated the change in *B. franklini* occurrences (populations) over time. To assess representation (as an indicator of adaptive capacity) of *B. franklini*, we evaluated the spatial extent of occurrences over time.

Resiliency is the ability to sustain populations in the face of environmental variation and transient perturbations. In section 2.2.3 we described that *Bombus franklini* requires the following for resiliency: (1) populations with large N_e, (2) sufficient floral resources in close proximity to nesting and overwintering sites, (3) connectivity among colonies and populations, and (4) spatial heterogeneity. Historically, the species has always been rare and has one of the narrowest distributions of any *Bombus* species in the world. Even so, the abundance and distribution of *B. franklini* has declined significantly (U.S. Fish and Wildlife Service 2018, pp. 10-14); the species has not been observed since 2006 despite an intensive survey effort in some areas of the historical range. Prior to 1998, search efforts for the species were varied in timing, scope, intensity, and methodology. During the more intensive surveys from 1998 until the last observation in 2006, *B. franklini* was observed at 11 sites, including seven locations where it had not been previously documented. In 1998, 98 bees were found among eight locations. Searchers found fewer and fewer bees after that even though they continued extensive searches in multiple locations with the highest likelihood of finding the species. Twenty bees were located in 1999, nine individuals were observed in 2000, and one individual in 2001. Although 20 *B. franklini* were observed in 2002, only three were observed in 2003 (all at a single locality), and a single worker bee was observed in 2006. Despite continued intensive search efforts through 2017, there have been no confirmed observations of *B. franklini* since 2006. There are currently no known healthy *B. franklini* individuals and therefore no known healthy colonies or populations of *B. franklini*. Despite the fact that some high quality habitat with diverse floral resources and available nesting and overwintering sites appears to be available in the historic range of *B. franklini*, no individuals of the species have been found in any habitat since 2006. The resiliency of *B. franklini* has declined significantly since the late 1990's.

Representation is the ability to adapt to changing environmental conditions; it is the species' evolutionary capacity or flexibility. In section 2.2.3 we described that *Bombus franklini* requires the following for representation: healthy populations distributed across areas of unique adaptive diversity (i.e., ecoregions) to maintain evolutionary drivers (gene flow, natural selection, genetic drift) to mimic historical patterns. *Bombus franklini* is rare and has always had very small populations (relative to other similar, native bumble bees in the western United States), and likely have low genetic diversity, making the species more vulnerable to environmental factors. As a haplodiploid organism, *B. franklini* may be more prone to population extinction than diploid-diploid organisms, due to its susceptibility to low population levels and loss of genetic diversity. No *B. franklini* have been observed since 2006 despite an intensive survey effort and therefore we cannot identify any current populations of *B. franklini* distributed across any level of ecological conditions. The vulnerability resulting from *B. franklini's* genetic system and the loss in the spatial extent of its populations suggest the representation of *B. franklini* has declined significantly since the late 1990's.

Redundancy protects species against the unpredictable and highly consequential events for which adaptation is unlikely. In section 2.2.3 we described that *Bombus franklini* requires the following for redundancy: sufficient distribution to guard against catastrophic events wiping out portions of the species adaptive diversity, i.e., to reduce covariance among populations, and an adequate number of healthy populations to buffer against catastrophic losses of adaptive diversity. *Bombus franklini* has the smallest geographic distribution of any North American bumble bee and possibly the world (Williams 1998, as cited in Xerces Society and Thorp 2010, p. 6), and thus likely had low redundancy prior to its decline. When we look at occurrence data for the species and overlay it with our 6 km^2 grid estimating minimum habitat patch to estimate the number of populations present on the landscape, we find that each site where *B. franklini* has ever been observed could potentially reflect a population. Therefore, data allow us to estimate 43 potential populations of *B. franklini* since records have been kept. From 1998 to 2006, 14 potential populations could be identified and no *B. franklini* have been observed since 2006 despite a more intensive survey effort in some areas of the historic range. We cannot identify any current healthy populations distributed across any spatial extent. The losses in both the number of populations and spatial extent indicate that the redundancy of *B. franklini* has declined significantly since the late 1990's.

5.0 Analysis of Future Condition

Due to the lack of observations of the species since 2006, we did not project anticipated future states of resiliency, redundancy or representation. Numerous survey efforts for invertebrate pollinators have occurred since 2006 in high quality habitat where *Bombus franklini* have been historically observed. During these efforts by Xerces Society, USFS, BLM, FWS, classes at Southern Oregon University and many private individuals, several species of *Bombus* have consistently been observed, but *B. franklini* has never been found. Although the failure to detect a species during surveys is not equivalent to a conclusive demonstration of its absence and may simply reflect the very low detection probability for rare species, the certain losses in both the number of populations and their spatial extent render *B. franklini* vulnerable to extinction even without further external stressors acting upon the species.

Several conservation measures (as described in section 3.3) could be applied to important *Bombus* habitats within the historic range of *B. franklini*, which would be beneficial to other *Bombus* species (notably *B. occidentalis*) and any existing but unknown populations of *B. franklini*. These include but are not limited to reductions in herbicide and pesticide applications and restrictions on the importation and use of commercially produced bees. Expanded and standardized surveys for *B. franklini*, *B. occidentalis*, and other special-status invertebrates would improve knowledge of species abundance, distribution, and habitat conditions.

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7.0 Appendices

Appendix 1: Occurrence Table

Appendix 2: Dear Interested Party Letter

Appendix 3: Neonicotinoid Pesticide Application Data by County, 1995-2015

Appendix 4: Expert Elicitation Questionnaire

Appendix 1: Occurrence Table- Table 1: Known occurrences of Bombus franklini

(Q=Queen, W= worker, M= male, U= individual of unknown type; number in parentheses indicates number of days the site was visited that year)

		YEAR																																										
	Landowner	1923	1925	1930	1946	1949	1950	1952	1953	1958	1963	1964	1968	1969	1976	1980	1986	1988	1989	1990	1992	1994	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
OREGON																																													
Unknown Location	unknown	1 W, 1 M																																										
Douglas County																																												
Sutherlin (3 mi W of)	Private																							**1 U** (1)	0 (1)	0 (1)	0 (2)	0 (1)		0 (2)	0 (3)		0 (1)											
Roseburg	Private		1 W	2 M, 1 U																																								
Jackson County																																												
Ashland	Private									2 W						1 U		1 U		4 Q, 1 W, 1 Q	1 U																								
Ashland (8mi ENE)	Private										3 W, 6 M																																	
Ashland (Ashland pond)	City of Ashland																						2 W																					
Ashland (Mistletoe Rd)	Private																								0 (1)	0 (2)	**1 U** (3)		0 (4)	0 (7)	0 (5)	0 (2)											
Ashland (SOU Roca Canyon)	State (no public access)																							2 Q																				
Buncom (E of)	Private																								**1 U** (1)	0 (3)	0 (1)	0 (1)															
Central Point	Private																			2 W																								
Copper	USFS												7 U			**Inundated by Lake Applegate after completion of Applegate Dam in Fall of 1980**																														
Copper (2 mi N)	USFS										2 M, 2 W																																	
Copper (14 mi N)	BLM										1 W																																	
Copper (8 mi W of)	USFS												2 U																															
Copper (nr- see 17 mi. W of Ruch)	USFS												12 Q (plus 207 other specimens)																																
Gold Hill	Private		1 W	3 W			1 W										7 W			12 W																								
Gold Hill (3 mi E of)	Private																							**44 U** (4)	0 (2)	**5 U** (7)	0 (7)	0 (3)	0 (4)	0(2)	0 (4)	0 (2)	0 (2)	**Soil modification at a portion of the site in 2004.**									
Grizzley Peak	BLM																					1 Q		0 (2)	0 (2)	0 (1)	0 (2)	0 (2)	0 (2)	0 (2)	0 (3)	0 (1)	0 (2)											
Jackson campground	USFS																							**2 U**(2)	0 (2)	0 (1)		0 (1)			0 (1)												
Kenney Meadows	BLM																							**3 U** (2)	0 (2)	0 (2)	0 (2)	0 (1)	0 (1)		0 (1)												
Lost Creek Reservoir	Private																							**1 Q**	0 (1)		0 (1)			0 (1)	0 (1)												
Medford	Private				8 W, 10 M	2 W, 1 M	2 Q, 12 W, 24 M (7)	1 M	5 W, 4 M																																		
Medford (Roxy Ann Peak)	Private																						2 Q																				
Mt. Ashland	USFS											4 W (over 2 days)	1 Q (plus 207 other specimens)											**37 U** (3)	**19 U** (6)	**2** U (7)	**1 U** (5)	**19 U** (10)	**3 U** (9)	0 (13)	0 (11)	**1 W** (8)	0 (7)									
Phoenix (E of)	Private																								0 (1)	0 (2)																	
Ruch	Private																6 W			12 W				**3 U** (3)	0 (2)	**1** U (2)	0 (1)	0 (2)		0 (2)												
Ruch (SSE of)	Private																								0 (2)	**1** U (3)	0 (2)	0 (1)	0 (2)		0 (1)												
Shale City Rd to Grizzley Peak	Private																							**1 Q**																			
Union Creek	USFS														1 Q										0 (1)																		
Josephine County																																												
Grants Pass	Private			3 W (2)														1 W																										
Merlin	Private																																										
Selma (S of)	Private																							**2 U** (1)	0 (1)	0 (1)																	
Wonder (W of)- reported as historical record but no date of occurance record reported	Private																									0 (1)																	
CALIFORNIA																																												
Siskiyou County																																												
Hilt	Private										1 M	1 Q								12 W				**2** U (2)	0 (3)	0 (3)	0 (1)	0 (2)	0 (1)	0 (1)	0 (2)	0 (2)	0 (1)											
Everitt Mem. Hwy	USFS																			1 W																								
Marble Mtn Wilderness (Back Meadows, southpe Boulder Creek	USFS																						? U																				
Marble Mtn Wilderness Bear Valleyrea 5500ft	USFS																						? U																				
Marble Mtn Wilderness Big Meadows ^500 ft	USFS																						? U																				
Marble Mtn Wilderness LoweWright Lake 7000 ft	USFS																						? U																				
Marble Mtn Wilderness Pacific Crest Trail (.5 mi S Paradise Lake))	USFS																						? U																				
Marble Mtn Wilderness Upper Kelsy Creek 5600 ft	USFS																						? U																				
Montegue	Private																								0 (1)					0 (1)												
Yreka	Private																			2 W																								
Trinity County																																												
Willo Creek, Trinity Alps, 17 mi N Weaverville	USFS													? U																													
TOTAL:		**2**	**2**	**9**	**18**	**3**	**39**	**1**	**9**	**2**	**15**	**5**	**22**	**?**	**1**	**1**	**13**	**2**	**15**	**32**	**1**	**2**	**10**	**98**	**20**	**9**	**1**	**20**	**3**	**0**	**0**	**1**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**

Appendix 2: Dear Interested Party Letter

United States Department of the Interior

FISH AND WILDLIFE SERVICE
Oregon Fish and Wildlife Office
2600 SE 98th Avenue, Suite 100
Portland, Oregon 97266
Phone: (503) 231-6179 FAX: (503) 231-6195

Reply To: 8185.0153
File Name: DIP Franklin bumble bee.doc
TS Number: 17-576

Dear (Interested Party),

The U.S. Fish and Wildlife Service (Service) is evaluating the status of Franklin's Bumble Bee (*Bombus franklini*) to determine the need for potential listing as an endangered or threatened species under the Endangered Species Act of 1973, as amended (16 U.S.C 1531 *et seq.*; Act). We initiated this process following our receipt of a petition dated June 23, 2010, from the Xerces Society for Invertebrate Conservation, and Dr. Robbin Thorp (petitioners). The petitioners requested listing of Franklin's bumble bee as an endangered species and that critical habitat be designated for the species. On August 16, 2010, we provided the petitioners with our determination that an emergency listing was not warranted based on our assessment of the immediacy of possible threats to Franklin's bumble bee as presented in the petition. We also informed the petitioners that at that time, we would not be able to further address the petition due to requirements to complete a significant number of listing and critical habitat designations. In September 2011, we published 90-day finding in the Federal Register (76 FR 56381), wherein we determined that the petition presented substantial information indicating that the listing of the Franklin's bumble bee may be warranted. We also requested scientific and commercial data and other information regarding this species at that time. With this letter, we are providing early notification to our conservation partners that we are continuing with this status review process (as initiated with the 90-day finding on September 13, 2011 (76 FR 56381).

Franklin's bumble bee has been found in an area of about 190 miles north-south and 70 miles east-west in Douglas, Jackson and Josephine counties in southern Oregon and in Siskiyou and Trinity counties in northern California, which is the most limited distribution known of any bumble bee species in North America, and perhaps the world. Franklin's bumble bee is a eusocial bumble bee, and each colony goes through an annual cycle and only the queen lives through the winter. The nesting biology of Franklin's bumble bee is unknown, but like other *Bombus* species, it is thought to nest underground in grassy areas, presumably in abandoned rodent burrows. The flight season of Franklin's bumble bee is from mid-May, when the queen emerges from hibernation, to the end of September. Franklin's bumble bee requires habitat with a rich supply of floral resources that bloom continuously from spring to autumn. Bumble bees are generalist foragers, meaning that they gather pollen and nectar from a wide variety of flowering plants. Unlike honey bees, bumble bees do not produce honey for winter nutrition –

rather, nutrition is derived from nectar, which provides carbohydrates, and from pollen provides protein.

According to the petition, the primary threats to Franklin's bumble bee in Oregon and California, according to the petitioners, include exotic diseases introduced from commercial bumble bees used for greenhouse pollination of tomatoes and field pollination of a variety of crops; habitat loss due to destruction, degradation and conversion; pesticides and pollution; and inadequacy of current rules, regulations and law. The petitioners also identified the following additional threats: small population size, exotic plant species introduction, increased human use of native habitat, climate change affecting alpine habitat, and alteration of wildfire severity and intensity.

Our status review includes consideration of all of the best scientific and commercial data available to us regarding Franklin's bumble bee populations, and is not limited to the information provided in the petition. Over the next several months, we will be gathering and analyzing available information as part of our evaluation of the species' status. We are required to use the best scientific and commercial data available in the development of our finding to ensure our analysis and finding is as accurate as possible. We are seeking your input to ensure we have the best scientific data available to inform our finding.

We are particularly seeking information and data for Franklin's bumble bee throughout its range in Oregon and California regarding the following:

- Biology, range, and population trends, including:
    - Habitat requirements for feeding, breeding, and sheltering;
    - Genetics and taxonomy of the population;
    - Historical and current range including distribution patterns, and presence or absence of physical, physiological, or behavioral barriers to movement between populations;
    - Historical and current population levels, and current and projected trends; and
    - Past and ongoing conservation measures for the species, its habitat, or both.

- The factors that are the basis for making a listing determination for a species under section 4(a) of the Act, which are:
    - The present or threatened destruction, modification, or curtailment of its habitat or range;
    - Overutilization for commercial, recreational, scientific, or educational purposes;
    - Disease or predation;
    - The inadequacy of existing regulatory mechanisms; or
    - Other natural or manmade factors affecting its continued existence.

While we will accept new information throughout this process, we request that you provide us with any pertinent information by August 3rd, 2017, to ensure we have adequate time to consider it during development of our finding. If you have already provided us with any data recently, thank you; there is no need to resubmit that information, as it will be fully considered in our status review.

Information should be submitted to Jeff Everett of our Oregon Fish and Wildlife Office at (503) 231-6952 (Jeff_Everett@fws.gov). Please be aware that all data and information

submitted to us, including names and addresses, will become part of the decision record for this package and will be available for public inspection.

Thank you for your interest in the conservation of Franklin's bumble bee. If you would like additional information about the listing process, please contact Rebecca Migala of our Portland Regional Office at (503) 231-2011 (Rebecca_Migala@fws.gov). Additional information on the listing process is available online at our website at *http://www.fws.gov/endangered/what-we-do/listing-overview.html*.

Sincerely,

Paul Henson
State Supervisor

Appendix 3: Neonicotinoid Pesticide Data by County, 1995-2015

EPEST_LOW all counties

	Josephine County, OR			Jackson County, OR			Douglas County, OR			Siskiyou County, CA			Trinity County, CA		
	Imidacloprid	Thiamethoxam	Clothiandin	Imidacloprid	Thiamethoxam	Clothiandin	Imidacloprid	Thiamethoxam	Clothiandin	Imidacloprid	Thiamethoxam	Clothiandin	Imidacloprid	Thiamethoxam	Clothiandin
1995	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1996	45.7	0	0	1.5	0	0	2.4	0	0	0	0	0	0	0	0
1997	197.5	0	0	0.7	0	0	3	0	0	0	0	0	0	0	0
1998	320.5	0	0	0.8	0	0	5.2	0	0	0	0	0	0	0	0
1999	1.6	0	0	0.7	0	0	1.2	0	0	0	0	0	0	0	0
2000	8.8	0	0	3.1	0	0	5.2	0	0	0	0	0	0	0	0
2001	92.8	73.8	0	3.6	0.1	0	7.4	1.2	0	0	0	0	0	0	0
2002	18	99	0	6.6	0.2	0	11.3	1.7	0	0	0	0	0	0	0
2003	69.3	16	0	0.9	0	0	2.5	0.3	0	14.9	0	0	0	0	0
2004	14.5	0	0	4.5	0	0	9.1	0	0	4.6	0	0	0.1	0	0
2005	8.7	5.4	0	2.6	0	0	6.2	0.1	0	24.4	0	0	0	0	0
2006	2.5	76.1	0	0.7	0.3	0	2.6	1.2	0	13.2	0	0	0	0	0
2007	11.4	12	0	2.8	0.1	0	7.9	0.2	0	16.7	2.5	0	0	0	0
2008	1.6	0.3	0.2	1	0.2	0	2.3	0.1	0	11.7	0	0	0	0	0
2009	3.5	54.5	0.2	1.3	0.3	0	4.1	1.1	0	37.1	0	0	0	0	0
2010	3.1	1.2	0.3	1.1	0.4	0.1	2.3	0.2	0	0	0	0	0	0	0
2011	9.7	0.3	1	3.3	0.1	0.4	8.2	0.1	0	0	4.8	0	0	0	0
2012	4.7	11.4	1	2.4	0.1	0.5	5.7	3	0	132.3	3.4	0	0	0	0
2013	22.4	0.9	3.4	10.6	0.3	1.8	25.2	0.2	0	487.6	14.1	0	0	0	0
2014	152.8	30	3.5	78.8	0.6	1.8	155.6	7.9	0	276.7	12.4	0	0	0	0
2015	436.8	0	0	37.9	0	0	19.8	0	0	64.1	0	0	0	0	0

Siskiyou County, CA CalPiP Data

	Imidacloprid	Thiamethoxam	clothiandin
1995	n/a	n/a	n/a
1996	3.75	0	0
1997	0.16	0	0
1998	0.239	0	0
1999	n/a	n/a	n/a
2000	3.05625	0	0
2001	0.374	0	0
2002	18.035	0	0
2003	33.618	0	0
2004	10.289	0	0
2005	56.928	0	0
2006	31.547	0	0
2007	39.344	5.427	0
2008	26.063	0	0
2009	64.672	0.08	0
2010	15.281	19.268	0
2011	2.219	10.657	0
2012	305.535	7.498	0
2013	473.582	31.156	1.25
2014	685.758	27.37	0
2015	160.033	0	0

EPEST_HIGH all counties

	Josephine County, OR			Jackson County, OR			Douglas County, OR			Siskiyou County, CA			Trinity County, CA		
	Imidacloprid	Thiamethoxam	Clothiandin	Imidacloprid	Thiamethoxam	Clothiandin	Imidacloprid	Thiamethoxam	Clothiandin	Imidacloprid	Thiamethoxam	Clothiandin	Imidacloprid	Thiamethoxam	Clothiandin
1995	0.7	0	0	1.8	0	0	0.9	0	0	0	0	0	0	0	0
1996	2.2	0	0	47.3	0	0	3.9	0	0	0	0	0	0	0	0
1997	1.7	0	0	199.9	0	0	4.6	0	0	0	0	0	0	0	0
1998	1.4	0	0	322	0	0	6	0	0	0	0	0	0	0	0
1999	0.9	0	0	2.2	0	0	1.5	0	0	0	0	0	0	0	0
2000	2.8	0	0	316.1	0	0	8.7	0	0	0	0	0	0	0	0
2001	3.6	0.2	0	92.8	73.9	0	7.4	1.3	0	0	0	0	0	0	0
2002	6.7	0.2	0	18.6	99	0	11.5	1.7	0	0	0	0	0	0	0
2003	1.4	0.1	0	70.8	16.2	0	3.4	0.4	0	14.9	0	0	0	0	0
2004	4.7	0.3	0	15.3	116.4	0	9.5	1.9	0	4.6	0	0	0.1	0	0
2005	2.9	0	0.2	40.4	5.4	0.9	7	0.1	0.9	24.4	0	0	0	0	0
2006	0.8	0.3	0.2	2.8	76.1	52.4	2.8	1.2	0.8	13.2	0	0	0	0	0
2007	3.6	0.1	1.5	57.6	12	352.7	9.4	0.2	5.2	16.7	2.5	0	0	0	0
2008	1.1	0.4	0.3	18.3	49.3	63.4	2.7	0.8	1	11.7	0	0	0	0	0
2009	1.4	0.3	0.3	32.5	54.8	50.6	4.7	1.1	0.8	37.1	0	0	0	0	0
2010	2.8	0.6	0.1	16.4	45.5	0.3	6.6	1	0	0	0	0	0	0	0
2011	3.7	0.1	0.4	65.1	13.8	1	10.7	0.3	0	0	4.8	0	0	0	0
2012	2.8	0.4	0.5	64	68.5	1	8.6	4.1	0	132.3	3.4	0	0	0	0
2013	11.2	0.8	1.8	110.7	97.2	9.6	27.6	2.6	0.1	487.6	14.1	0	0	0	0
2014	79.1	0.9	1.8	215.4	81.4	3.5	157.8	8.8	0	276.7	12.4	0	0	0	0
2015	38	0.3	0.1	436.9	47.5	18.8	80.2	0.9	0.3	64.1	0	0	0	0	0

Trinity County, CA CalPiP Data

	Imidacloprid	Thiamethoxam	clothiandin
1995	n/a	n/a	n/a
1996	0	0	0
1997	0	0	0
1998	0	0	0
1999	n/a	n/a	n/a
2000	3.422	0	0
2001	1.312	0	0
2002	22.335	0	0
2003	0	0	0
2004	0.23	0	0
2005	0.738	0	0
2006	0.946	0	0
2007	1.81	0	0
2008	0.884	0	0
2009	1.04	0	0
2010	0	0	0
2011	0.654	0	0
2012	0.117	0	0
2013	3.594	0.065	0
2014	2.494	0	0
2015	0.362	0	0.02

USGS Data from https://water.usgs.gov/nawqa/pnsp/usage/maps/about.php.
For all States except California, proprietary farm survey pesticide-use data are aggregated and reported at the multi-county Crop Reporting District level.
Harvested crop acreage data by county from the US Department of Agriculture Census of Agriculture are used to calculate the median pesticide-by-crop use rates for each CRD
Estimates for California are obtained from annual Department of Pesticide Use Reports. Methods for generating county level pesiticide use estimates are described in Thelin and Stone, 2013, and Baker and Stone, 2015
These data are estimates - please refer to the USGS website for more detailed information on how the estimates are generated and important limitations on data use
California Pesticide Information Portal can be found at http://calpip.cdpr.ca.gov
CalPip database does not have results for 1995 and 1999.
2013 raw Calpip data includes a single application of 620.423 pounds of imidacloprid on 44 acres of potatos in siskiyou county; not included here as suspected erroneous data

Appendix 4: Expert Elicitation Questionnaire

Franklin's bumble bee (*Bombus franklini*)- Questions from USFWS 10/27/2017

In our status assessment of *B. franklini*, we need to articulate the habitat requirements for a viable population of the species; we look at viability in terms of what a reasonable naturalist would consider a stable, self-sustaining population. In order to get a better understanding of the species' population structure and habitat requirements, as well as the factors that might have influenced *B. franklini* populations, we have the following questions:

1. What would the number of individual *B. franklini* recorded at a site indicate?
   - Could you draw any conclusions on the number of colonies represented at the site? (e.g. 3 bees= 1 colony, or 44 bees= more than 1 colony?)

   - How big is the typical (most likely) colony foraging area (in km^2) of *B. franklini* or other *Bombus* species that have similar foraging behavior?

     What is the largest?

     Smallest?

   - What is the probability (high, medium, or low) that *Bombus* colonies overlap in their foraging areas? (0-32% = low, 33-65% = medium, 66-100% = high)

   - What is the probability (high, medium, low) that at sites where multiple bees were counted over several visits (in all historical records and surveys), that the same individual bee might have been counted twice? (0-32% = low, 33-65% = medium, 66-100% = high)

2. How many colonies would make up a viable population (as defined above in bold) of *B. franklini* or a similar species?

   How large (in km^2) would a habitat patch have to be to support a viable population?

3. In the past (pre- 1998), when people went out to collect native bumble bees including *B. franklini*, what was the probability (high, medium, low) that a specimen of *B. franklini* was included in the day's collection? (0-32% = low, 33-65% = medium, 66-100% = high)

    - Could people usually collect a Franklin's specimen if they knew where to find them or was it always a challenge to find them even at sites where they previously occurred?

4. Can we infer from the historical data that a higher count of *B. Franklini* at a site was due to higher abundance of bees? Or… should we not infer this because of the varying intensity of survey efforts (i.e. maybe they found more bees that year because they looked harder for the bees)?

5. Can you suggest a species of *Bombus* that is similar to *B. franklini* that has been studied more (a species we can use as a surrogate for information on population structure and habitat requirements)? If so, are there any caveats or considerations we should keep in mind when using this species as a surrogate?

6. Would you consider any of the sites where *B. franklini* were found in the past (Table 1 below) extirpated or no longer viable? Why or why not? How confident are you in your answer? (highly confident, confident, minimally confident)

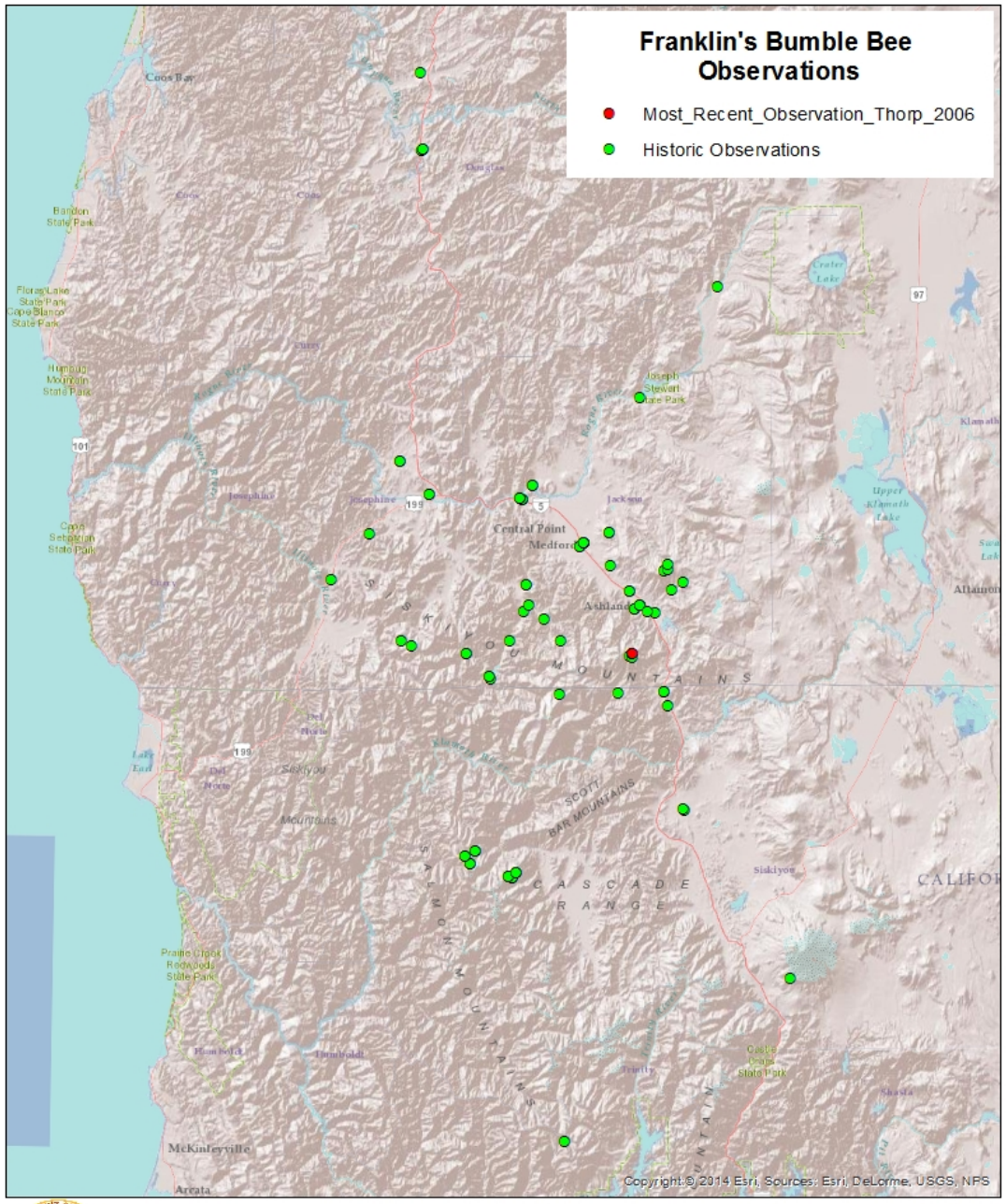
7. In Table 1, please fill in any site-specific information you have on the following potential stressors at last known occurrence sites of *B. franklini*. Do you think there are any other factors that may have led to the decline of *B. franklini* at any of these sites or elsewhere in the historic range (Douglas, Jackson, and Josephine Counties, Oregon, and Siskiyou and Trinity Counties, California)?

Table 1: Potential stressors at known occurrence sites of *Bombus franklini* from 1997-2006 (occurrence site information taken from 2010 Petition to List Franklin's Bumble Bee (Thorp et al. 2010, p. 9 and Appendix 1)).

	Pesticide Use	Grazing	Proximity to commercially raised bees	Wildfire	Changes in bloom time of forage Vegetation	Invasive Species	Changes in habitat (development or other)
Jackson County, Oregon							
Sutherlin (3 miles West of)							
Ashland							
Ashland Pond							
Ashland (SOU_ Roca Canyon)							
Buncom (1.5 miles East of)							
Gold Hill (3 miles East of)							Significant excavation and deposited soil in 2004- altered 50% of bumble bee habitat
Grizzley Peak/ Shale City Road							
Jackson campground							
Kenney Meadows							
Lost Creek Reservoir							
Medford- Roxy Ann Peak							
Mt. Ashland							

	Pesticide Use	Grazing	Proximity to commercially raised bees	Wildfire	Changes in bloom time of forage Vegetation	Invasive Species	Changes in habitat (development or other)
Phoenix (E of)							
Ruch							
Ruch (4 miles SSE of)							
Josephine County, Oregon							
Selma (South of)							
Siskiyou County, California							
Hilt							

8. In looking at the draft distribution map of known occurrences of *B. franklini* (below), are there areas in Douglas, Jackson, Josephine, Siskiyou, and Trinity Counties in addition to these occurrence sites that might contain its known foraging plants (and therefore potential *B. franklini* habitat): lupine (*Lupinus spp*.), California poppy (*Eschscholzia californica),* horsemint or nettle-leaf giant hyssop (*Agastache urticifolia*) and mountain monardella (*Monardella odoratissima)?*



No warranty is made by the U.S. Fish and Wildlife Service as to the accuracy, reliability, or completeness fo these data for individual or aggregate use with other data. Original data were compiled from various sources. Spatial information may not meet National Map Accuracy Standards. This information may be updated without notification.

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