Monarch (*Danaus plexippus*)
Species Status Assessment Report, version 2.1
September 2020

Photo: Kelly Nail

Prepared by:

U.S. Fish and Wildlife Service

Acknowledgments

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Suggested Reference

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*The changes from version 2.0 (July 2020) to 2.1 (Sept 2020) are minor corrections and do not change the results of the SSA analyses.

Executive Summary

The monarch, *Danaus plexippus*, is a species of butterfly globally distributed throughout 90 countries, islands, and island groups. These butterflies are well known for their phenomenal long-distance migration in the North American populations. Descendants of these migratory monarch populations expanded from North America to other areas of the world where milkweed (their larval host plant) was already present or introduced. With the year-round presence of milkweed and suitable temperatures, many of these global monarch populations no longer migrate.

Two North American populations, the migratory populations located east and west of the Rocky Mountains, have been monitored at their respective overwintering sites in Mexico and California since the mid-1990s. While these populations fluctuate year-to-year with environmental conditions, these census data indicate long-term declines in the population abundance at the overwintering sites in both populations (Figure E1). These declining trends led to the petition of the U.S. Fish and Wildlife Service to list the monarch butterfly for protection under the Endangered Species Act of 1973, as amended.

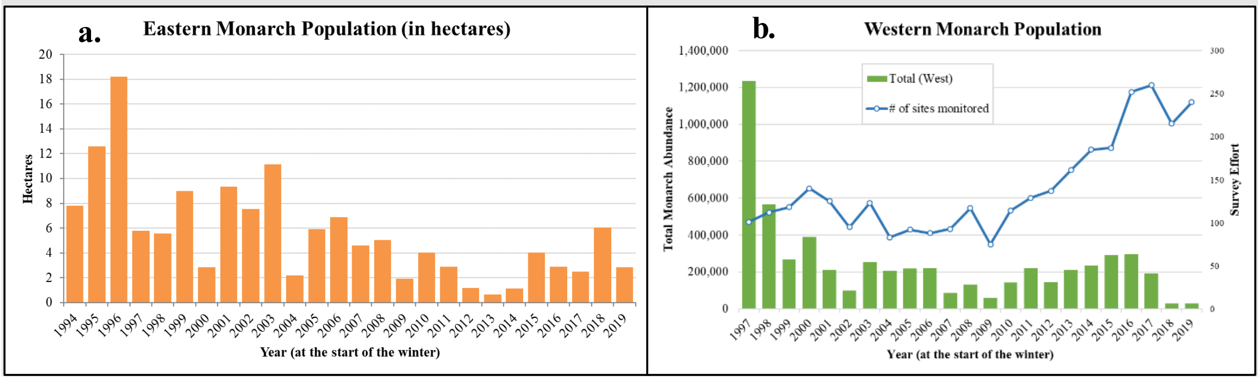

Figure E1. Eastern (a) and western (b) North American monarch population sizes, as measured at overwintering sites in terms of hectares (eastern) and total number (western). The western population count also has a blue line indicating survey effort (number of sites monitored). Horizontal black bars with labels indicate the decadal average population counts.

Using the best available scientific information about monarchs in North America and worldwide, we conducted a species status assessment (SSA). This report summarizes the results of our SSA. We delineated the historical number and distribution of monarch populations, assessed the status and health of the currently extant populations, identified the key drivers of their health, forecasted the likely future change in these drivers and monarch population responses to these changes, and evaluated the consequences of the population responses to monarch viability. Specifically, we evaluated the ability of the monarch to withstand environmental stochasticity (resiliency), catastrophes (redundancy), and novel changes in its biological and physical environment (representation).

We delineated 31 historical populations; of these, 27 are extant and 4 have unknown status. Outside of the 2 migratory North American populations, the health of the remaining 29 populations is undeterminable due to limited information available on population trends and stressors. However, at least 15 of these populations are at risk of extinction due to climate change related sea level rise or unsuitably high temperatures. The results for the two migratory North American populations show that both are facing declining numbers and overall health.

The primary drivers affecting the health of the two North American migratory populations are primarily: loss and degradation of habitat (from conversion of grasslands to agriculture, widespread use of herbicides, logging/thinning at overwintering sites in Mexico, senescence and incompatible management of overwintering sites in California, urban development, and drought), continued exposure to insecticides, and effects of climate change. Relative to the recent past, both the eastern and western North American populations have lower abundances and declining population growth rates. Using the best available science, we estimated the probability of the population abundance reaching the point at which extinction is inevitable (pE) for each population given their current abundance and growth rate, as well as under projected future conditions. The pE for the western population is high (60% to 68% chance within 10 years, reaching 99% by year 60) under current conditions and increases under projected future conditions. For the eastern population, the pE in 60 years under current conditions ranges from 48% to 69%, and under the projected future conditions, it ranges from 56% to 74%. The range in the estimates represents the best and worst plausible future state conditions of the primary drivers.

Additionally, at the current and projected low population numbers, both the eastern and western populations are more vulnerable to catastrophic events (e.g., extreme storms at the overwintering habitat) than in the past. These risks, however, are not captured in the pE estimates. Similarly, we found that under different climate change scenarios, the number of days and the area in which monarchs will be exposed to unsuitably high temperatures will increase markedly. We were unable to incorporate the effects of high daily temperatures into the extinction analyses, and thus, these risks as well are not fully captured in the pE estimates.

The extinction of either the western or eastern North American migratory population would increase the risk of losing the North American migratory phenomenon, as its persistence would depend solely upon the continued survival of a single population. Moreover, loss of either population would impair the species' ability to adapt into the future. The North American populations are unique in their long-distance migratory ability, and they represent unique sources of genetic and ecological diversity. Further, these two populations represent the historical and current core of the species and the ancestral lineage of the species. The eastern North American population is by far the largest of all populations (both in number of individuals and range size), and the western North American population encompasses as much as 30% of the geographic range occupied by monarch butterflies in North America. Accordingly, loss of these two ACUs would reduce monarch diversity, rendering the species less able to adapt to novel changes in its environment now and in the future and thereby increasing the extinction risk of the monarch. The chance of *both* populations persisting above the extinction threshold over the next 10 years is 27% to 33% (under future conditions) and drops under 10% within 30 years. Based on this information and other analyses of influences included in this SSA, monarch viability is declining and is projected to continue declining over the next 60 years.

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List of Acronyms

ACU	Adaptive capacity unit
Act	Endangered Species Act of 1973, as amended
CCAA	Candidate Conservation Agreement with Assurances
CDC	Center for Disease Control
CDL	Cropland Data Layer
CMIP5	Coupled Model Intercomparison Project Phase 5
CRP	Conservation Reserve Program
EROS	Earth Resources Observation and Science
FAO	The Food and Agriculture Organization of the United Nations
FSA	Farm Service Agency
IPCC	Intergovernmental Panel on Climate Change
MACA	Multivariate Adaptive Constructed Analogs
MAFWA	Midwest Association of Fish and Wildlife Agencies
MCD	Monarch Conservation Database
MP3	Managed Pollinator Protection Plan
NLCD	National Land Cover Database
NOAA	National Oceanic and Atmospheric Administration
PDSI	Palmer Drought Severity Index
RCP	Representative Concentration Pathways
SSA	Species Status Assessment
USDA	U.S. Department of Agriculture
USDOT	U.S. Department of Transportation
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
VRT	Variable Rate Technology
WAFWA	Western Association of Fish and Wildlife Agencies

Chapter 1: Introduction & Analytical Framework

The Center for Biological Diversity, Center for Food Safety, Xerces Society, and Dr. Lincoln Brower petitioned the U.S. Fish and Wildlife Service (USFWS) to list the monarch (*Danaus plexippus plexippus*) as a threatened species under the Endangered Species Act of 1973, as amended (Act) on August 26, 2014 (Center for Biological Diversity et al. 2014). In December 2014, USFWS found the petition presented substantial scientific or commercial information that indicated listing the monarch may be warranted (79 FR 78775) and initiated a rangewide status review.

This report summarizes the results of a species status assessment (SSA) conducted for the monarch butterfly, and it is intended to provide the biological support for the decision on whether the monarch warrants listing under the Act. Importantly, the SSA report is not a decisional document; rather it provides a review of available information strictly related to the species' biological status. The USFWS will make a listing determination after reviewing this document and all relevant laws, regulations, and policies, and will announce the results of the determination in the *Federal Register*, with appropriate opportunity for public input. This report has undergone peer and state review and incorporates the best available scientific data.

This chapter describes the analytical framework and the conservation principles used to assess monarch viability over time. Chapter 2 summarizes the ecological requirements for survival and reproduction at the individual, population, and species levels. Chapter 3 details the methods underlying our analyses. Chapters 4 and 5 summarize the historical and current conditions of monarch, respectively, and identifies the key factors (referred to as influences) that contributed to the species' current condition. Chapter 6 describes the projected changes in these key influences. Chapter 7 summarizes the projected future condition of the monarch given the plausible projections of the key influences. Lastly, Chapter 8 synthesizes the above analyses and describes how the consequent change in the number, health, and distribution of monarch populations influence monarch viability over time. In this final chapter, we also describe sources of uncertainty and the implications of this uncertainty. Additionally, we include four appendices providing further information on taxonomy, methodology, results, and other drivers considered.

Analytical Framework

Viability is the ability of a species to maintain populations in the wild over time. To assess viability, we use the conservation biology principles of resiliency, redundancy, and representation (Shaffer and Stein 2000, pp. 308-311). Meaning, to sustain populations over time, a species must have a sufficient number of populations distributed throughout its geographic range to withstand:

(1) environmental stochasticity and disturbances (Resiliency),
(2) catastrophes (Redundancy), and
(3) novel changes in its biological and physical environment (Representation).

Viability is a continuous measure of the likelihood of sustaining populations over time and can be defined in relative terms, such as "low" or "high" viability. A species with a high degree of resiliency, representation, and redundancy (the 3Rs) is generally better able to adapt to future changes and to tolerate catastrophes, environmental stochasticity, and stressors, and thus, typically has high viability.

Resiliency is the ability of the species to withstand and sustain populations through environmental stochasticity (normal, year-to-year variations in environmental conditions, such as temperature or rainfall), periodic disturbances (fire, floods, storms, etc.), and anthropogenic stressors (factors that cause a negative effect to a species or its habitat) (Redford et al. 2011, p. 40). Simply stated, resiliency refers to a species' ability to sustain populations through favorable and unfavorable environmental conditions and anthropogenic impacts.

Resiliency is multi-faceted. First, it requires having healthy populations demographically (robust survival, reproductive, and growth rates), genetically (large effective population size, high heterozygosity, and gene flow between populations), and physically (good body condition). Second, resiliency also requires having healthy populations distributed across heterogeneous environmental conditions (referred to as spatial heterogeneity; this includes factors such as temperature, precipitation, elevation, and aspect). Because environmental stochasticity can operate at regional scales (Hanski and Gilpin 1997, p. 372), populations tend to fluctuate in synchrony over broad geographical areas (Kindvall 1996, pp. 207, 212; Oliver et al. 2010, pp. 480-482). Spatial heterogeneity induces asynchronous fluctuations among populations, thereby guarding against concurrent population declines. Lastly, resiliency often requires connectivity among populations to maintain robust population-level heterozygosity via gene flow among populations and to foster demographic rescue following population decline or extinction due to stochastic events.

Redundancy is the ability of a species to withstand catastrophes; defined here as highly consequential events (cause population extinction) for which adaptation is unlikely (Mangal and Tier 1993, p. 1083). For all species, a minimal level of redundancy is essential for long-term viability (Shaffer and Stein 2000, pp. 307, 309-310; Groves et al. 2002, p. 506). Reducing the risk of extinction due to a single or series of catastrophic events requires having multiple populations widely distributed across the species' range, with connectivity among groups of locally adapted populations to facilitate demographic rescue following population decline or extinction. This provides a margin of safety to reduce the risk of losing substantial portions of genetic diversity or the entire species to a single or series of catastrophic events.

Representation is the ability of a species to adapt to both near-term and long-term novel changes in the conditions of its environment, both physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (novel pathogens, competitors, predators, etc.). This ability, referred to as adaptive capacity, is essential for viability because species need to continually adapt to their continuously changing environment (Nicotra et al. 2015, p. 1269). Species adapt to novel changes in their environment by either 1) moving to new, suitable environments or 2) by altering their physical or behavioral traits (phenotypes) to match the new environmental conditions through either plasticity or genetic change (Beever et al. 2016, p. 132; Nicotra et al. 2015, p. 1270).

Maintaining a species' ability to disperse and colonize new environments fosters adaptive capacity by allowing species to move from areas of unsuitable conditions to regions with more favorable conditions. It also fosters adaptive capacity by increasing genetic diversity via gene flow, which is, as discussed below, important for evolutionary adaptation (Hendry et al. 2011, p. 173; Ofori et al. 2017, p. 1). Thus, maintaining natural levels of connectivity among populations is important for preserving a species' adaptive capacity (Nicotra et al. 2015, p. 1272).

Maintaining a species' ability to adapt to novel conditions also requires preserving the breadth of genetic variation. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either genetic change or plasticity (see Text Box 1.1). For adaptation to occur, whether through plasticity or evolutionary adaptation, there must be genetic variation upon which selection can act (Hendry et al. 2011, pp. 164-165; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 326). Without genetic variation, the species cannot adapt and is more prone to extinction (Spielman et al. 2004, p. 15263; also see Text Box 1.1).

Text Box. 1.1. ***Species Adaptation***. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either *genetic change* or *plasticity* (Chevin et al. 2010, p. 2-3; Hendry et al. 2011, p. 162; Nicotra et al. 2015, p.1270). *Genetic change,* referred to as evolutionary adaptation or potential, involves a change in phenotypes via an underlying genetic change (specifically, a change in allele frequency) in response to novel environmental cues (Nicotra et al. 2015, p. 1271; Ofori et al. 2017, p. 2). *Plasticity*, unlike evolutionary adaptation, involves a change in phenotypes (phenotypic plasticity) without undergoing changes in the genetic makeup (Nicotra et al. 2015, p. 1271-1272). Plasticity is an important mechanism for species to adapt both in immediate and future time frames. In the immediate time frame, plasticity directly acts to allow species to persist despite novel changes in the environment. In the longer time frame, plasticity contributes to a species' adaptive capacity by buying time for adaptive evolution to occur through genetic changes (referred to as genetic assimilation, see Ghalambor et al. 2007, p. 395; Nicotra et al. 2015, p. 1271). Not all genetic and plastic induced changes are adaptive; changes must lead to improved fitness to be adaptive (Nicotra et al. 2015, p. 1271-1272). Importantly, however, adaptive traits can vary over space and time; what is adaptive in one location may not be adaptive in another, and similarly, what is adaptive today may not be under future conditions and vice versa (Nicotra et al. 2015, p. 1271-1272). Thus, maintaining the full breadth of variation in both plastic traits and genetic diversity is important for preserving a species' adaptive capacity.

Genetic variation that is adaptive is difficult to identify for a species and represents a significant challenge even when there is genetic information available. To denote variation as 'adaptive' we need to identify which loci are under selection, which traits those loci control, how those traits relate to fitness, and what the species' evolutionary response to selection on those traits will be over time (Hendry et al. 2011, p. 162-163; Lankau et al. 2011, p. 316; Teplitsky et al. 2014, p. 190). Although new genomic techniques are making it easier to obtain this type of information (see Funk et al. 2019), it is lacking for most species. Fortunately, there are several proxies that collectively can serve as indicators of potentially underlying adaptive genetic variation. One of the easiest proxies to measure is variation in biological traits (also described as phenotypic variation). Phenotypic variation, which on its own can be a mechanism for adapting to novel changes, can be due to underlying adaptive genetic variation (Crandall et al. 2000, p. 291; Forsman 2014, p. 304; Nicotra et al. 2015, p. 3). A second proxy for adaptive genetic variation is neutral genetic variation, which is usually the type of genetic data first reported in species-specific genetic studies (see Text Box 1.2). A third, and more distant, proxy for adaptive genetic variation is disjunct or peripheral populations (Ruckelhaus et al. 2002, p. 322). These populations can be exposed to the extremes in climate tolerances for the species and thus harbor unique and potentially adaptive traits. Similarly, populations that occur across steep environmental gradients can be indicators of underlying adaptive genetic diversity because local adaptation is driven by environmental conditions, which are continually changing at different rates and scales (Sgro et al. 2011, p. 330, 333).

Text Box. 1.2. ***Genetic diversity***. Genetic variation can be partitioned into two types: adaptive and neutral genetic diversity. Both types are important for preserving the adaptive capacity of a species (Moritz 2002, p. 243), but in different ways. Genetic variation under selection underlies traits that are locally adaptive and that determine fitness (Holderegger et al. 2006, pp. 801, 803; Lankau et al. 2011, p. 316); thus, it is the variation that underpins adaptive evolution (Sgro et al. 2011, p. 328). This type of genetic variation is referred to as adaptive genetic diversity and determines the capacity for populations to exhibit an adaptive evolutionary response to changing environmental conditions. Conversely, neutral genetic variation refers to regions of the genome that have no known direct effect on fitness (i.e., selectively neutral) and change over time due to non-deterministic processes like mutation and genetic drift (Sgro et al. 2011, p. 328). Although, by definition, neutral genetic variation is not under selection, it contributes to the adaptive capacity of a species in a couple of ways. First, neutral genetic variation that is statistically neutral in one environment may be under selection--and thus adaptive--in a different environment (Nicotra et al. 2015, p. 1271-1272). Second, neutral markers can allow us to infer evolutionary lineages, which is important because distinct evolutionary lineages may harbor locally adaptive traits (Hendry et al. 2011, p. 167), and hence, serve as an indicator of underlying adaptive genetic variation. Thus, maintaining the full breadth of neutral and adaptive genetic diversity is important for preserving a species' adaptive capacity.

Lastly, preserving a species' adaptive capacity requires maintaining the natural levels of the processes that allow for evolution to occur; namely, natural selection and gene flow (Crandall et al. 2000, p. 290-291; Sgro et al. 2011, p. 327; Zackay 2007, p. 1). Natural selection is the process by which heritable traits can become more (selected for) or less (not selected for) common in a population via differential survival or reproduction (Hendry et al. 2011, p. 169). To preserve natural selection as a functional evolutionary force, it is necessary to maintain populations across an array of environments (Hoffmann and Sgro 2011, p. 484; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 332; Shaffer and Stein 2000, p. 308). Gene flow serves as an evolutionary process by introducing new alleles (variant forms of genes) into a population, thereby, increasing the gene pool size (genetic diversity). Maintaining the natural network of genetic connections between populations will foster and preserve the effectiveness of gene flow as an evolutionary process (Crandall et al. 2000, p. 293). Along with maintaining large effective population sizes, preserving genetic connections among populations also helps minimize the loss of genetic variation due to genetic drift (Crandall et al. 2000, p. 293). Large population numbers also important to adaptive capacity because the level of diversity is influenced by population size and the rate of evolutionary adaptation is faster in populations with high diversity (Ofori et al. 2017, p.2).

Chapter 2: Species Ecology

This chapter describes the ecological requirements for survival and reproduction at the individual, population, and species levels (the first step of our analytical framework).

Species Description

The monarch, *Danaus plexippus* (Linneaus, 1758), is a species of butterfly in the order Lepidoptera (family Nymphalidae) that occurs in North, Central, and South America; Australia; New Zealand; islands of the Pacific and Caribbean, and elsewhere (Malcolm and Zalucki 1993, p. 3-5; Fig. 4.1). Adult monarch butterflies are large and conspicuous, with bright orange wings surrounded by a black border and covered with black veins. The black border has a double row of white spots, present on the upper side and lower side of forewings and hindwings (Bouseman and Sternburg 2001, p. 222). Adult monarchs are sexually dimorphic, with males having narrower wing venation and scent patches (CEC 2008, p.11; Figure 2.0). The bright coloring of a monarch is aposematic, as it serves as a warning to predators that eating them can be toxic.

Figure 2.0. Male monarch on milkweed. Note the arrow pointing to the black dots known as androconial scent patches on the hind wings. These are not present on female monarchs. Photo by Tim Koerner, USFWS.

Taxonomy

In 2014, a petition was received to list the subspecies of the monarch butterfly (*Danaus plexippus plexippus*) under the Endangered Species Act (Center for Biological Diversity et al. 2014). The petition also requested a determination of whether any new North American subspecies of *Danaus plexippus* should be listed. After careful examination of the literature and consultation with experts, there is no clearly agreed upon definition of potential subspecies of *Danaus plexippus* or where the geographic borders between these subspecies might exist. Given these findings, we examined the entire range of *Danaus plexippus* for this assessment. For more information on taxonomy, see Appendix 1.

Individual-Level Ecology and Requirements

Below we describe the ecological needs for monarch individuals to survive and reproduce; these needs are summarized in Table 2.1. During the breeding season, monarchs lay their eggs on their obligate milkweed host plant (primarily *Asclepias* spp.), and larvae emerge after two to five days (Zalucki 1982, p. 242; CEC 2008, p. 12). Larvae develop through five larval instars (intervals between molts) over a period of 9 to 18 days, feeding on milkweed and sequestering toxic cardenolides as a defense against predators (Parsons 1965, p. 299). The larva then pupate into chrysalis before eclosing 6 to 14 days later as an adult butterfly. There are multiple generations of monarchs produced during the breeding season, with most adult butterflies living approximately two to five weeks; overwintering adults enter into reproductive diapause (suspended reproduction) and live six to nine months (Cockrell et al. 1993, pp. 245-246; Herman and Tatar 2001, p. 2509; Figure 2.1).

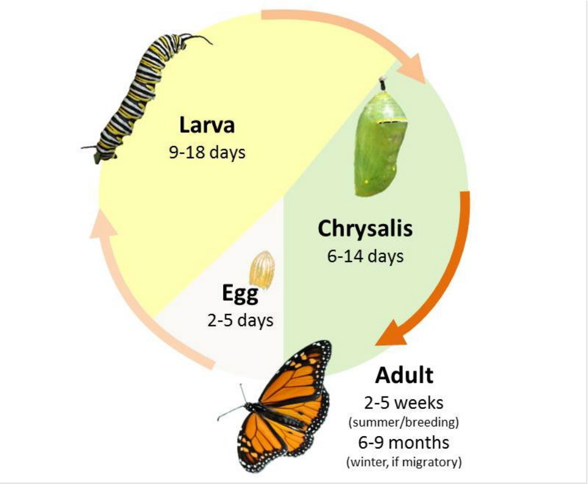
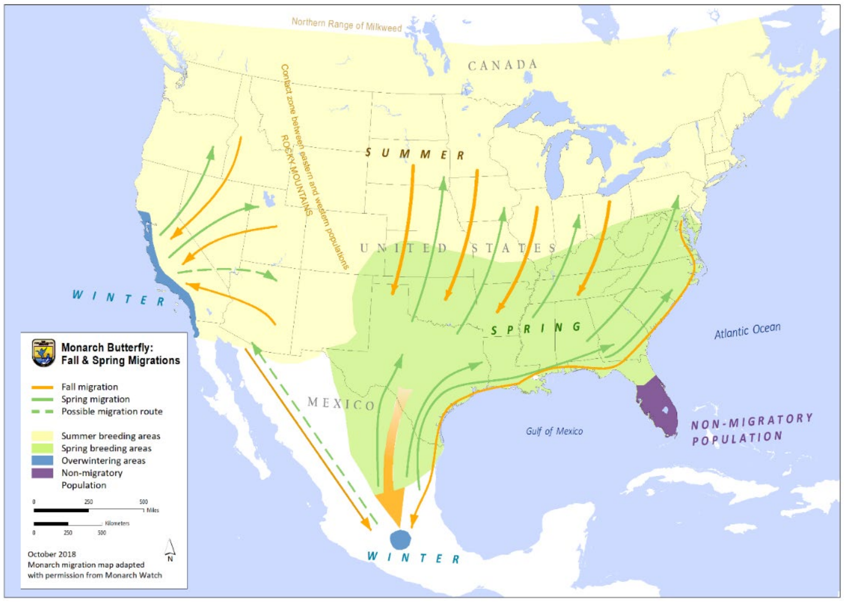


Figure 2.1. Monarch life cycle. Development times calculated from Zalucki (1982) based on temperatures ranging from 22°-32°C. Adult life span based on Herman and Tatar (2001).

The monarch life cycle varies by geographic location. In many regions where monarchs are present, monarchs breed year-round, repeatedly following the above-referenced life cycle throughout the year. Individual monarchs in temperate climates, such as eastern and western North America, undergo long-distance migration, where the migratory generation of adults is in reproductive diapause and lives for an extended period of time (Herman and Tatar 2001, p. 2509). In the fall, in both eastern and western North America, monarchs begin migrating to their respective overwintering sites. This migration can take monarchs distances of over 3,000 km (Urquhart and Urquhart 1978, p. 1760) and last for over two months (Brower 1996, p. 93). Migratory individuals in eastern North America predominantly fly south or southwest to mountainous overwintering grounds in central Mexico, and migratory individuals in western North America generally fly shorter distances south and west to overwintering groves along the California coast into northern Baja California (Solensky 2004, p. 79; see Figure 2.2). Data from monarchs tagged in the southwestern states in the fall suggest that those in Nevada migrate to California, those in New Mexico migrate to Mexico, and those in Arizona migrate to either

Mexico or California (Southwest Monarch Study Inc. 2018). In early spring (February-March), surviving monarchs break diapause and mate at the overwintering sites before dispersing (Leong et al. 1995, p. 46, van Hook 1996, pp. 16-17). The same individuals that undertook the initial southward migration begin flying back through the breeding grounds and their offspring start the cycle of generational migration over again (Malcolm et al. 1993, p. 262).

In eastern North America, monarchs travel north in the spring, from Mexico to Canada, over two to three successive generations, breeding along the way (Flockhart et al. 2013, p. 4-5; Figure 2.2). Individual monarchs disperse as far north as they can physiologically tolerate based on climatic conditions and available vegetation; the most specific predictors of the northern distribution of individual monarchs are monthly mean temperature and precipitation (Flockhart et al. 2013, p. 4; Flockhart et al. 2017, p. 2568). The number of generations of monarchs produced in a given year can vary between three and five and is dependent upon environmental conditions (Brower 1996, p. 100). While a majority of the eastern monarchs shift to the more northern reaches of their range, western monarchs continue to occupy and breed in warmer climates throughout the summer, while also expanding to include the farther reaches of their range. In the spring in western North America, monarchs migrate north and east over multiple generations from coastal California toward the Rockies and to the Pacific Northwest (Urquhart and Urquhart 1977, p. 1585; Nagano et al. 1993, p. 157; Figure 2.2). In the southwestern states, migrating monarchs tend to occur more frequently near water sources such as rivers, creeks, roadside ditches, and irrigated gardens (Morris et al. 2015, p. 100). While the overwintering areas shown in Figure 2.2 represent the sites where most monarchs in North America overwinter in reproductive diapause, there are other sites and overwintering strategies (see *Uncertainties* section in Chapter 8).



October 2018
Monarch migration map adapted with permission from Monarch Watch

Figure 2.2. North American monarch migration map.

Adult monarch butterflies during breeding and migration require a diversity of blooming nectar resources, which they feed on throughout their migration routes and breeding grounds (spring through fall). Monarchs also need milkweed (for both oviposition and larval feeding) embedded within this diverse nectaring habitat. The correct phenology, or timing, of both monarchs and nectar plants and milkweed is important for monarch survival. The position of these resources on the landscape is important as well (see *Population-Level Ecology* section in this chapter). In western North America, nectar and milkweed resources are often associated with riparian corridors, and milkweed may function as the principal nectar source for monarchs in more arid regions (Dingle et al. 2005, p. 494; Pelton et al. 2018, p. 18; Waterbury and Potter 2018, p. 38; Dilts et al. 2018, p. 8). Individuals need nectar and milkweed resources year-round in non-migratory populations. Additionally, many monarchs use a variety of roosting trees along the fall migration route (Table 2.1).

Migratory individuals of eastern and western North America require a very specific microclimate at overwintering sites. The eastern population of monarchs overwinter in Mexico, where this microclimate is provided by forests primarily composed of oyamel fir trees (*Abies religiosa*), on which the monarchs form dense clusters (Williams and Brower 2015, pp. 109-110). The sites used for overwintering occur in mountainous areas west of Mexico City located between elevations of 2,900 and 3,300 m (Slayback and Brower 2007, p. 147). The temperature must remain cool enough to prevent excessive lipid depletion (Alonso-Mejía et al. 1997, p. 935), while at the same time staying warm enough to prevent freezing (Anderson and Brower 1996, pp. 111-113). Exposure to these cooler temperatures also helps orient the monarchs northward in the spring (Guerra and Reppert 2013, pp. 421-422). The oyamel fir forest provides essential protection from the elements, including rain, snow, wind, hail, and excessive solar radiation (Williams and Brower 2015, p. 109). Many sites also provide a source of hydration via nectar plants or a water source (Brower et al. 1977, pp. 237-238). Most of the observed overwintering sites are located within the Monarch Butterfly Biosphere Reserve, which covers over 56,000 ha (Vidal and Rendón-Salinas 2014, p. 169; Ramírez et al. 2015, p. 158).

Migratory monarchs in the western population primarily overwinter in groves along the coast of California and Baja California (Jepsen and Black 2015, p. 149). The location and structure of these sites provide the specific microclimate (although different from the Mexico overwintering microclimate) needed for survival in the western overwintering areas. There are approximately 400 groves that have been occupied, but only a portion of these sites is occupied in any given year. These sites, typically close to the coast, span approximately 1.225 km of coastline (COSEWIC 2010, p. 10). These groves are populated by a variety of tree species, including blue gum eucalyptus (*Eucalyptus globulus*), Monterey pine (*Pinus radiata*), and Monterey cypress (*Hesperocyparis macrocarpa*) (Griffiths and Villablanca 2015, pp. 41, 46-47), all of which act as roost trees. These groves provide indirect sunlight for the overwintering monarchs, sources of moisture for hydration, defense against freezing temperatures, and protection against strong winds (Tuskes and Brower 1978, p.149; Leong 1990, pp. 908-910, Leong 1999, p. 213). The close proximity to the coast (average distance of 2.37 km ± 0.39 SE) also provides a mild winter climate (Leong et al. 2004, p. 180).

Table 2.1. Individual-level requisites for monarch survival and reproduction.

Life Stage	Requirements	Description
Eggs, Larvae, and Adults – breeding	Milkweed resources	Healthy and abundant milkweed is needed for oviposition and larval consumption.
Adult – breeding and migration	Nectar resources	Sufficient quality and quantity of nectar from flowers is needed for adult feeding throughout the breeding and migration seasons.
Adult – overwintering	Suitable habitat for overwintering	Habitat that provides a specific roosting microclimate for overwintering: protection from the elements (e.g., rain, wind, hail, excessive radiation) and moderate temperatures that are warm enough to prevent freezing yet cool enough to prevent lipid depletion. Nectar and clean water sources located near roosting sites.
Adult – migration	Connectivity & Phenology	Nectar and milkweed resources along the migration route when butterflies are present; the size and spatial arrangement of habitat patches are generally thought to be important aspects, but currently unknown. Roosting sites may also be important for monarchs along their fall migration route.

Population-Level Ecology

The ecological requirements of a healthy monarch population are summarized in Table 2.2. To be self-sustaining, a population must be demographically, genetically, and physically healthy (see Redford et al. 2011, entire). Demographically healthy means having robust survival, reproductive, and growth rates. Genetically healthy populations have large effective population sizes (N_e), high heterozygosity, and gene flow between populations. Physically healthy means individuals have good body condition. Monarchs, like many insects, are sensitive to environmental conditions (temperature and precipitation) and can experience large swings in population numbers year-to-year in response to these conditions (Rendón-Salinas et al. 2015, p. 3; Schultz et al. 2017, pp. 3-4). During favorable conditions, monarch survival and reproductive rates are high and population numbers increase; conversely, when environmental conditions are unfavorable, survival and reproductive rates are low and population numbers can plummet. Thus, to successfully recruit over generations and years, they must be capable of withstanding large swings in population sizes (N). Specifically, they need a robust population growth rate (lambda, or λ). Given that environmental fluctuations vary spatially, robust growth rates likely vary across populations.

To support a strong growth rate, monarch populations require large population sizes and sufficient quantity and quality of habitat to accommodate all life stages. Large population sizes also help maintain genetic health (via large N_e) and facilitate thermoregulation during the winter, which is important for good physical health. It may also be important for mate finding and aposematism (S. Malcolm, pers. comm. 2018). The quality of habitat needed to support healthy demographic rates and physical health is described under *Individual-Level Ecology and Requirements*. The quantity of habitat likely varies among populations, and exact requirements may vary (e.g., the type of trees needed for overwintering).

Migratory monarch populations can have individuals that can fly distances of over 3,000 km (Urquhart and Urquhart 1978, p. 1760; see *Individual-Level Ecology and Requirements* earlier in this chapter). During migration to overwintering sites, most monarchs are in reproductive diapause, but continue to need blooming nectar plants throughout the migratory habitat to provide sugar that is eventually stored as lipid reserves (Brower et al. 2015, p. 117). On their return, monarchs are laying eggs, and thus need both nectar sources and milkweed. This habitat needs to be distributed throughout the landscape to ensure connectivity throughout their range and maximize lifetime fecundity (Zalucki and Lammers 2010, p. 84; Miller et al. 2012, p. 2). However, the specific optimal amount of habitat and its spatial distribution are unknown; more research is needed on optimal distances between habitat patches, as well as optimal patch sizes and milkweed density and characteristics of patches selected for female oviposition (Kasten et al. 2016, p. 1055; Stenoien et al. 2016, p. 8; Grant et al. 2018, p. 48; Waterbury and Potter 2018, p. 48).

Table 2.2. The population-level requisites for a healthy population.

Parameter	**Requirements**
Population growth rate, λ	The long-term λ must be sufficiently high to rebound from population lows. On average, λ must be >1; how much greater than 1 is dictated by the degree of environmental variability.
Population size, N	Sufficiently large *N* to withstand periodic population lows; the minimum *N* required is dictated by the degree of environmental variability and varies geographically across populations.
Habitat	Sufficient seasonally and geographically specific quantity and quality of milkweed, breeding season nectar, migration nectar, and overwintering resources to support large population sizes.
Connectivity	A matrix of seasonally specific habitat patches throughout the landscape to support breeding and migrating monarchs and allow migration throughout the population's range each year.

Species-Level Ecology

The ecological requisites at the species level include having a sufficient number and distribution of healthy populations to ensure it can withstand annual variation in its environment (resiliency), catastrophes (redundancy), and novel biological and physical changes in its environment (representation). We describe the monarch's requirements for resiliency, redundancy, and representation below, and summarize the key aspects in Table 2.3.

Resiliency

Monarch resiliency requires maintaining healthy populations across spatially heterogeneous conditions. Healthy monarch populations are better able to withstand and recover from environmental variability and stochastic perturbations (e.g., storms, dry years) than those populations that are less demographically, genetically, or physically healthy. The greater the number of healthy populations, the more likely it is that the monarch will withstand perturbations and natural variation, and hence, have greater resiliency. Additionally, given the monarch's sensitivity to environmental conditions (experiencing large swings in population numbers year-to-year; Rendón-Salinas et al. 2015, p. 3), monarchs occupying a diversity of environmental conditions and being widely distributed helps guard against populations being exposed to adverse conditions concurrently, and thus, fluctuating in synchrony. Asynchronous dynamics within and among populations minimizes the chances of concurrent losses, and thus, provides species' resiliency. Lastly, maintaining the natural patterns and levels of connectivity between populations also contributes to monarch resiliency by facilitating population-level heterozygosity via gene flow and demographic rescue following population decline or extinction due to stochastic events.

Redundancy

Monarch redundancy is best achieved by having multiple, widely distributed populations of monarchs relative to the spatial occurrence of catastrophic events. In addition to guarding against a single or series of catastrophic events that extirpate monarch populations, redundancy is important to protect against reducing the species' adaptive capacity. Having multiple monarch populations, occupying areas of unique diversity will guard against losses of adaptive capacity due to catastrophic events.

Representation

The monarch's ability to withstand novel changes is influenced by its adaptive capacity, which is primarily a function of the species' ability to colonize new areas and its breadth of variation in biological traits and genetic diversity (both neutral and adaptive genetic variation). In addition, and as explained in Chapter 1, maintaining large populations across an array of environments as well as the natural networks of genetic connections among populations are important components of preserving a species' adaptive capacity. Below we describe monarch adaptive capacity by using the best available data to incorporate the multiple proxies for adaptive variation described in Chapter 1. These proxies include genetic, morphological, behavioral, and ecological data drawn from published literature and expert knowledge. Based on these data, we delineated eight geographical units, referred to as adaptive capacity units (ACUs), which are depicted in Figure 2.3 and described below.

Table 2.3. Species-level requisites for species' viability (i.e., ability to sustain populations over time).

3 Rs	Species-Level Requisites	Details
Resiliency	Healthy populations distributed across spatially heterogeneous conditions	Healthy populations distributed across a diversity of temperatures, precipitation levels, elevations, and aspects.
Redundancy	Healthy populations distributed across geographical areas with low risks to catastrophic events	Widely spread, healthy populations to ensure all populations are not exposed to a single or series of catastrophic events.
Representation	Having healthy populations distributed across the breadth of genetic and phenotypic diversity; maintaining evolutionary processes	Breadth of variation in biological traits and genetic diversity via persistent populations within the 8 ACUs. Also, functional evolutionary processes via ensuring populations occupy an array of environments, maintaining genetic connections, and ensuring large N_e.

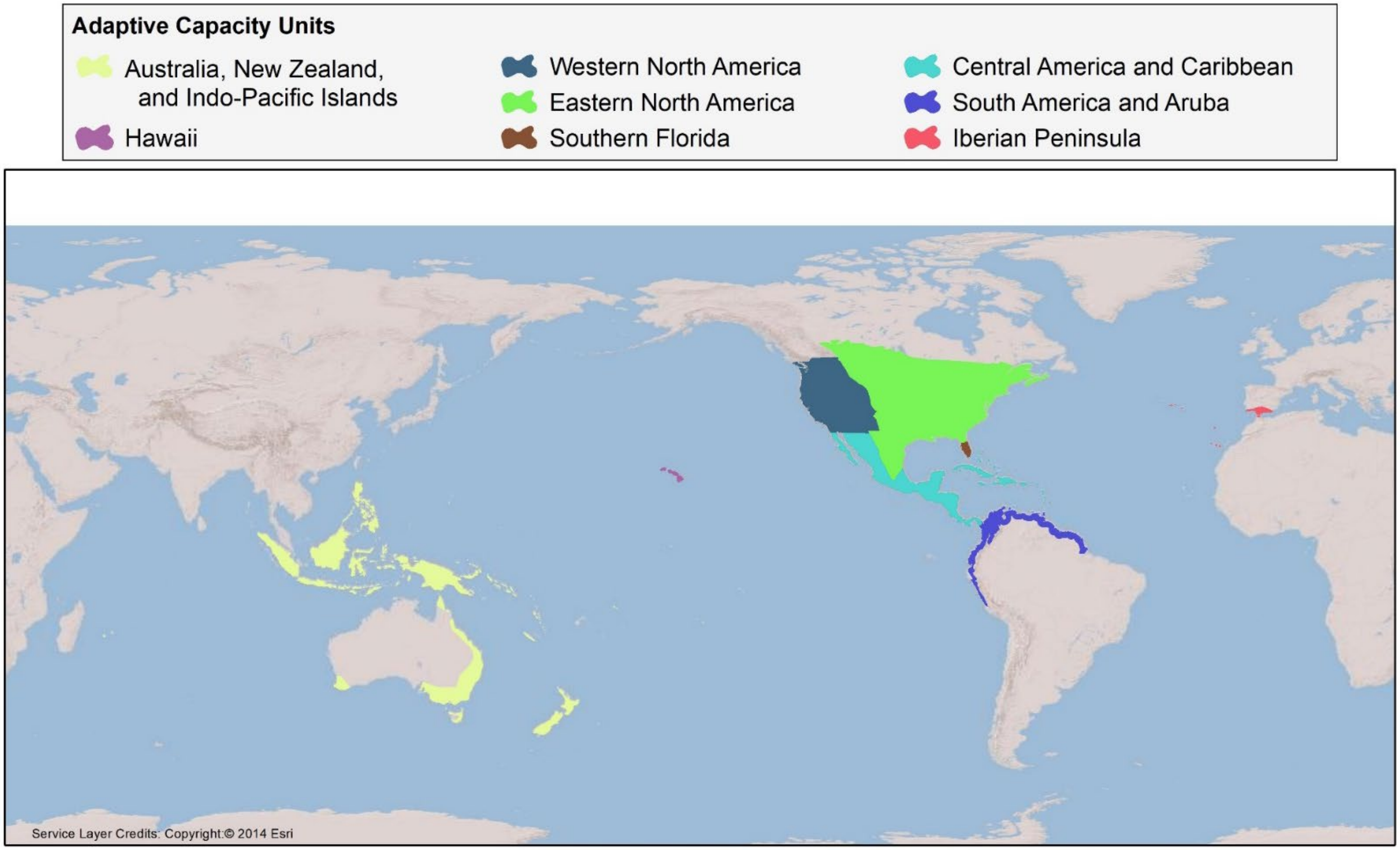


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Figure 2.3. Worldwide range of monarchs organized into eight ACUs.

1. Eastern North America:

   Eastern North American monarchs are identified as an ACU because they contribute unique phenotypic variation in long-distance migratory behavior, wing morphology, and disease/parasite infection resistance, in addition to unique genetic variation. They also occupy unique ecological conditions and serve (along with the western North American ACU) as the ancestral origin for the species worldwide.

   Eastern North American monarchs undergo long-distance migration every fall, a behavior that differentiates this population from other non-migratory populations or from migratory populations that fly shorter distances and to different locations. Further, the migratory phenotype of monarchs in the eastern ACU is distinct from monarchs in other ACUs that may have latent migratory phenotypes (Tenger-Trolander et al. 2019, p. 14673). Experimental comparisons between non-migratory and migratory individuals in the Eastern ACU reveal a unique phenotype present only in the migratory monarchs in the Eastern ACU. This migratory phenotype consists of both reproductive diapause and directional flight orientation to the south, and this migratory behavior of monarchs is remarkably sensitive to genetic and environmental change (Tenger-Trolander et al. 2019, p. 14673). In order to maintain full representation within the eastern North American monarch population, it is crucial to conserve the long-distance migratory phenotype for the unique adaptive capacity this behavior and its associated traits may offer.

   Monarchs from the eastern North American migratory population also have unique physical characteristics. They tend to have larger bodies, and larger and elongated wings compared to monarchs from most non-migratory populations (Altizer and Davis 2010, pp. 1023-1025). Relative to monarchs in western North America, eastern monarchs differ at isolated spots in the genome in relation to flight muscles (Kronforst, M. and A. Tenger-Trolander, pers. comm. 2018). Additionally, within the eastern North American ACU, long-distance migrants tend to have redder coloration (Davis 2009, p. 3). Redder coloration is associated with the ability to fly for longer periods of time, although the mechanism for this correlation is unknown (Davis et al. 2012, p. 4). Furthermore, compared to monarchs in the western North American ACU and the southern Florida ACU, eastern North American monarchs have lower rates of infection by the protozoan parasite *Ophryocystis elektroscirrha* (*OE*) (<10%; Altizer et al. 2000, p. 131), which may be due in part to their long-distance migration (Bartel et al. 2011, p. 348). Eastern monarchs migrating to Mexico also have higher lipid reserves than those overwintering in California (Brower et al. 1995, p. 542) and may have a longer diapause compared to monarchs from the western North American ACU (Herman et al. 1989, pp. 56-57).

   Monarchs from the eastern North American ACU also differ from monarchs in other ACUs in their overwintering habitat use and requirements. These monarchs overwinter in the mountainous forests composed primarily of oyamel fir roosting trees (Slayback and Brower 2007, pp. 147-148; Williams and Brower 2015, pp. 109-110), which provide a protective microclimate that is unique relative to those used by overwintering monarchs in other ACUs (Brower et al. 1995, p. 542).

Migratory monarchs in North America are the ancestral population for all other monarch populations around the world (Pierce et al. 2014a, p. 4; Zhan et al. 2014, p. 318). Their unique genetics separate them from non-migratory monarchs within North America (e.g., southern Florida; Pfeiler et al. 2016), as well as populations for the other ACUs described below. While some results show that the monarchs from eastern and western North American ACUs form an admixed population (Lyons et al. 2012, p. 3441), the differences in biological traits and ecological conditions they occupy warrant separating the populations into two ACUs.

2. Western North America:

   Western North American monarchs form a separate ACU because they contribute unique variation in migratory behavior, ecology, reproductive behavior, wing morphology, flight performance, and disease/parasite resistance. In addition, along with the eastern North American ACU, the western North American ACU serves as the ancestral origin for the species worldwide (Pierce et al. 2014a; Zhan et al. 2014).

   Like the monarchs in the eastern North American ACU, monarchs in the western North American ACU possess the unique migratory phenotype that is absent in the other six ACUs (Tenger-Trolander et al. 2019, p. 14673). Western North American monarchs also migrate long distances, although their migration is shorter than monarchs in the eastern North American ACU. Whereas eastern monarchs may fly well over 3,000 km to reach the Mexican overwintering sites, western monarchs reach the California coast by flying ~500 km to 1,600 km (Yang et al. 2016, p. 1002). Western monarchs occupy warmer climates throughout the summer to include the farther reaches of their range while they continue to breed in the hotter regions (expand their range). Eastern monarchs, in contrast, follow more of a stepping-stone path into the northern states, vacating areas as they warm and recolonizing their range.

   Additionally, western monarchs use ecologically different breeding, migrating, and overwintering habitats (Brower et al. 1995, p. 542), and the western North American ACU comprises as much as 30% of the area occupied by monarch butterflies in North America (Dilts et al. 2019, p. 11). Differences in breeding habitat include climate (Zalucki and Rochester 2004, pp. 220-221) and availability and abundance of native nectar and native milkweed plants (Borders and Lee-Mäder 2015, pp. 190-196). It is hotter and drier in the west than the east, and the milkweed and nectar resources used by monarchs in west and east differ (Dilts et al. 2019, entire). In the fall, western monarchs migrate from Canada and states west of the Rockies to overwintering groves located primarily along the California coast south into Baja California, Mexico (Jepsen and Black 2015, pp. 147-156). Roosting tree species used by western monarchs are different than those of the eastern population, and include blue gum eucalyptus, Monterey pine, and Monterey cypress (Griffiths and Villablanca 2015, pp. 43-44). There are fewer monarchs in the western population, spread out among hundreds of overwintering sites compared to fewer than 20 sites in Mexico for the eastern population (Jepsen and Black 2015, pp. 147-156; Vidal and Rendón-Salinas 2014, entire).

   In addition to differences in migratory behavior and habitats occupied, the designation of a separate ACU for western North American monarchs is supported by variation in reproductive behavior, wing morphology, flight performance, and disease/parasite resistance. Western North American overwintering monarchs may have a shorter diapause compared to those in eastern North America (Herman et al. 1989, pp. 52-54), and there may also be differences in mating behavior at the western overwintering grounds compared to the eastern overwintering grounds (Brower et al. 1995, p. 542). Eastern and western North American monarchs have divergent wing morphology (see the Eastern ACU discussion above, Freedman and Dingle, 2018, p. 66) and differences in flight performance resulting from differential gene expression related to non-muscular motor activity (Talla et al. 2020, p. 2572-2573). Monarchs in the west have *OE* infection rates (averaging 5-30%) that are lower than most non-migratory populations but higher than the rates of infection in eastern North America (Altizer and de Roode 2015, p. 91).

Thus, in order to maintain representation within the western North American monarch population, it is crucial to conserve the long-distance migratory phenotype in the west for the unique adaptive capacity this behavior and its associated traits may offer.

3. Southern Florida:

   Southern Florida monarchs form a separate ACU because they contribute unique variation primarily in genetics and phenotypic characteristics of non-migratory behavior, year-round breeding, and resistance to both high OE loads and a different strain of OE.

   Monarchs in southern Florida live in areas where the climate permits year-round breeding, and thus are able to reside continually without migrating. These non-migratory monarchs are genetically distinct from the migratory North American monarchs, although the southern Florida population gets an annual influx of individuals from the eastern monarch population (Knight and Brower 2009, p. 821; Zhan et al. 2014; Pfeiler et al. 2016). Non-migratory Florida monarchs experience some of the highest recorded OE infection rates compared to other monarchs worldwide and particularly high rates compared to eastern and western North America monarch infection rates (75-100% average infection rates in Florida vs. 5-30% infection rates in the western North American population and less than 10% infection rates in the eastern North American population; Altizer and de Roode 2015, p. 91). This may be due both to their inability to escape infected habitat, as well as the non-migratory behavior not leading to any migratory culling (Bartel et al. 2011, entire). Sternberg and colleagues (2013, pp. E239-E241) further determined that in lab settings, monarchs from southern Florida had lower OE spore loads (relative to eastern migratory monarchs) and were less likely to become infected with OE, potentially indicating that non-migratory southern Florida monarchs have increased resistance to OE (however, see also Altizer 2001, p. 622). In cross-population laboratory experiments, the OE parasites from southern Florida caused higher parasite loads than those from the eastern population (Altizer 2001, p. 622). For additional information, see Disease and Natural Enemies in Chapter 6.

4. South America and Aruba:

   Monarchs in South America and Aruba are grouped together to form an ACU due to genetic uniqueness.

   Monarchs in South America, based on samples from Ecuador, are markedly distinct from other populations of monarchs when analyzing microsatellite markers (Pierce et al. 2014a, 2015). They are occasionally classified as a separate subspecies (Danaus plexippus nigrippus).

While there is some indication that monarchs in Aruba are genetically distinct from South American monarchs (Pierce et al. 2014a), there is also evidence to the contrary (Zhan et al. 2014). Thus, based on this and on expert input suggesting that the small Aruba population is probably not genetically or ecologically distinguishable from South American monarchs, we grouped Aruba (and nearby islands) and South American monarch into the same ACU.

5. Central America and the Caribbean:

   Central American and Caribbean monarchs are grouped together to form an ACU based on genetic and behavioral differences relative to monarch elsewhere.

   Microsatellite analyses showed that Caribbean and Central American monarchs are distinct from South American monarchs and other non-migratory monarchs (Pierce et al. 2014a), and single nucleotide polymorphism analysis showed that Caribbean and Central American monarchs are also genetically distinct from the two migratory North American monarch populations (Zhan et al. 2014). Given that monarchs in Central America and the Caribbean are genetically distinct from these other populations and given the uniqueness of the southern Florida population (outlined above), we classified these monarchs as a separate ACU. Mexican non-migratory monarchs were also included in this unit (rather than the eastern North American ACU), based on similar ecological habitat, behavior (lack of migration), and recent genetic work showing genetic differentiation between migratory and non-migratory Mexican monarchs (Pfeiler et al. 2016).

6. Australia, New Zealand, and other Pacific Islands:

   Monarchs across Australia, New Zealand and other Pacific Islands are grouped together to form an ACU based on genetic characteristics and phenotypic characteristics of migration and disease/parasite resistance.

   Monarchs are found on many islands throughout the Pacific Ocean, including larger populations in Australia and New Zealand. Microsatellite analyses of monarchs in several Pacific island locations (Australia, New Zealand, New Caledonia, Fiji, and Samoa) indicate that these monarchs are genetically distinct from other areas and have lower allelic diversity than North American monarchs (Shephard et al. 2002, entire; Pierce et al. 2014a, p. 4). In addition to genetic differences, monarchs in the Pacific Islands show variation in migratory behavior. Monarchs on most of the smaller islands are non-migratory, but some Australian monarchs in New South Wales have been shown to migrate up to 380 km in autumn (James 1993, p. 193). However, there is little evidence for a regular long-distance migration, making it unique from the migration of the western and eastern North American monarchs (James 1993, p. 190).

   Researchers working with non-migratory Australian monarchs also discovered unique phenotypic responses upon exposure to environmental conditions thought to induce migration. Non-migratory monarchs exposed to cooler temperatures and shorter day lengths showed longer larval development periods, greater adult mass (thought to represent greater lipid reserves), and longer forewing development, all characteristics associated with potentially regaining the migratory phenotype (Freedman et al. 2017, p. 7, 10). Additionally, these responses varied significantly between the offspring of different mothers, suggesting that a migratory phenotype is potentially present within that Australian population (Freedman et al. 2017, p. 7, 10). Finally, incidence of OE in Australia is higher than in most other populations (~66% infection rate; Barriga et al. 2016, p. 76).

7. Hawaii:

   Hawaiian Island monarchs form an ACU because of unique genetic variation and increased disease/parasite tolerance.

   Monarchs exist on all major Hawaiian Islands and are non-migratory. Analysis using single nucleotide polymorphisms shows that monarchs in Hawaii are genetically distinct from other worldwide populations (Zhan et al. 2014). Microsatellite analyses also indicate that Hawaiian monarchs are genetically distinct from populations outside of Hawaii and that they have lower allelic diversity than continental North American monarch populations (Pierce et al. 2014b). Additionally, work indicates that monarchs in Hawaii form an admixed population (suggesting movement among islands; Pierce et al. 2014b). Monarchs in Hawaii persist with only moderate fitness reduction under strains of OE that are both more virulent and more prevalent than that of North American monarchs (Sternberg et al. 2013, p. E239). Thus, monarchs in the Hawaiian ACU contribute unique variation to the species in resistance to OE.

8. Iberian Peninsula (including Spain, Portugal, Morocco, and nearby Atlantic islands):

   Monarchs on the Iberian Peninsula (Spain and Portugal), along with monarchs in Northern Morocco and nearby Atlantic Islands, form an ACU because of unique genetic variation and ecological and climatic conditions.

   The non-migratory, introduced monarchs in Spain, Portugal, and Morocco form a genetically distinct, derived population based on a single nucleotide polymorphism analysis of the entire monarch genome (Zhan et al. 2014, p. 2). There may be some genetic variation between the Spanish monarchs and the monarchs in Portugal and Morocco based on microsatellite analyses (Pierce et al. 2014a). However, we did not consider Spanish monarchs as a separate ACU because these monarchs occupy very similar ecological and climatic conditions to the rest of the monarchs in this ACU (Fernández-Haeger et al. 2015, entire) but differ from those of other ACUs.

Chapter 3: Methodology

This chapter describes our methods for assessing viability of the monarch over time. The specific methodology for each step of the framework is described below. Briefly, our approach entailed: 1) gathering occurrence data globally, 2) assessing the number, health, and distribution of populations historically and currently, 3) identifying the substantive factors leading to the species' current condition and predicting the future states of these influences, 4) forecasting the health and distribution of populations given the future states of the influences, and 5) evaluating the resulting change in resiliency, redundancy, and representation over time and the implications for the species' viability (Figure 3.1).

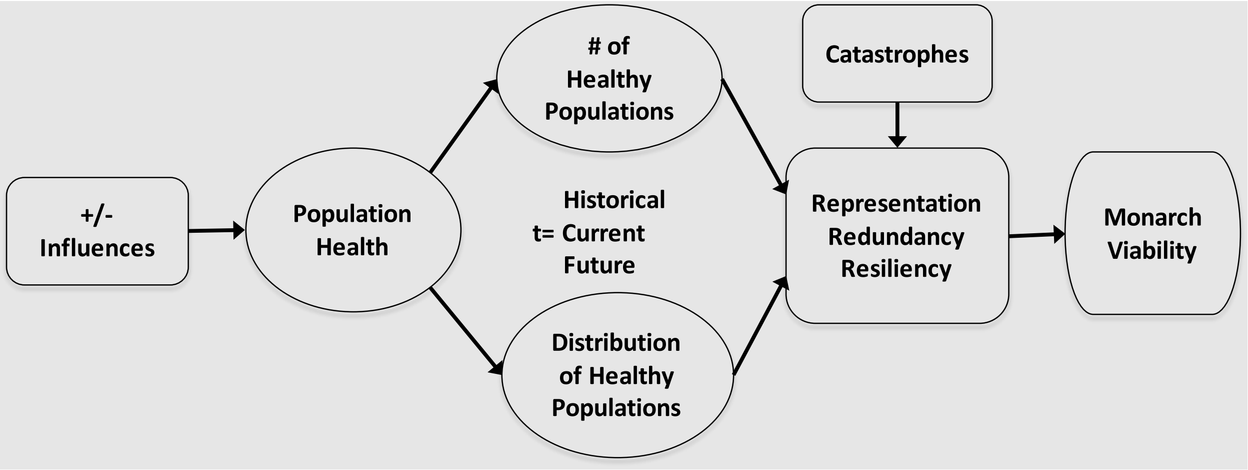

Figure 3.1. Simplified conceptual diagram depicting the analytical framework for assessing monarch viability over time.

Historical Condition: Number, health, and distribution of monarch populations (Ch. 4)

We examined the published literature to determine the historical distribution of the monarch butterfly populations. In order to assess the change in the number, health, and distribution of monarch populations over time, we delineated populations as follows. The monarchs in North America were separated into three populations—eastern, western, and southern Florida—based on distinct behavioral differences and limited movement between these populations. While differences at neutral markers have not been found between the western and eastern populations, a recent genomics analysis indicates low levels of dispersal between eastern and western monarch butterflies suggesting that they are demographically independent (Talla et al., 2020). The monarchs in the Caribbean, Central America, and South America were grouped according to documented genetic differences (Pierce et al. 2015). The remaining monarch locations were delineated based on distance. For monarchs occurring in countries and islands in the Pacific, monarch locations separated by more than 200 miles were considered disjunct populations. Tagging and observational data suggest that monarchs can travel up to approximately 70-75 miles a day during migration, with the longest recorded flight of a tagged eastern North American monarch at 265 miles (Journey North 2018). We thus chose a distance of 200 miles for separating populations because it was at the upper limits of the range of observed distances flown by tagged monarchs, and it is unlikely that monarchs separated by 200 miles or more could successfully move among these locations. If the distance between islands was less than 200 miles, we assumed that movement between islands was plausible and thus did not consider the islands as disjunct populations.

To assess population health, we sought out information on historical population abundance (*N*) and population growth rate (λ). Population size (*N*) estimates were derived from published survey counts; eastern North American monarchs have been surveyed yearly using a standardized protocol at the Mexican overwintering sites since 1994 (Monarch Watch 2020) and the western North American population has been monitored since 1997 at coastal overwintering sites in California (Xerces Society for Invertebrate Conservation 2020). The historical population growth rates (λ) for eastern and western North American populations were available from Semmens et al. (2016) and Schultz et al. (2017) for the eastern and western populations, respectively, and we updated both to reflect changes in growth rates since publication. Prior to 1994, we have limited information on population size (*N*) or growth (λ), but assume both populations were healthy (i.e., λ and N met conditions of Table 2.2) at some point in the historical time period. For all other populations, there are no systematic, multi-year surveys for any time period, so we assume those populations were healthy at some point in the historical time period as well.

Current & Future Conditions: Number, health, and distribution of monarch populations (Ch. 5 and Ch. 6)

To assess the current and future number, health, and distribution of monarch populations, for each population we: 1) determined the current abundance and population trend (λ), 2) identified the current and likely future primary influences, and 3) forecasted the change in health given these influences. We reviewed the available literature and sought out expert input to identify both the negative (threats) and positive (conservation efforts) drivers of monarch population numbers. We identified the following drivers: disease/natural enemies; herbicides; logging/tree loss; habitat degradation (succession, western overwintering site aging of trees); climate change (drought, storm events, temperature extremes); collection/tourism; grazing/incompatible farming; change in nectar and milkweed resources; loss of urban/greenspace; mowing; insecticides; change in western overwintering habitat. Of these, we identified the subset that are the key drivers influencing monarch dynamics (referred to as influences). We carried this subset through the rest of our analyses. For the worldwide populations, we researched potential issues related to land use change, insecticides, and disease.

Population-specific information for monarchs varies from highly detailed data for the eastern and western North American populations to very limited data (occurrence only) for most of the other 29 populations. To fully apply the best available data, we developed a population model for the eastern and western populations while using a coarser-scaled, qualitative approach for the remaining populations. We refer to the non-eastern, non-western populations as "worldwide populations."

Worldwide Populations

To assign status, we categorized populations based on last date observed and survey effort. We assumed that all populations in which at least a single monarch has been reported since the year 2000 are extant today and were assigned 'extant' status. Populations lacking a sighting since the year 2000 and lacking multi-year survey efforts were assigned 'unknown' status (neither extant nor extirpated). Populations lacking sightings with multiple years of surveys were assigned 'extirpated' status. We garnered the available data by: 1) searching for records in Google Scholar using each known country with a historical monarch occurrence and the phrase "*Danaus plexippus*" as search terms; 2) requesting personal knowledge and unpublished information regarding monarch occurrence from international entomologists and species' experts; and 3) searching geotagged photos on Flickr and reports from the citizen science database iNaturalist for monarch records. We did not use these records if we could not verify the species, or if the photo appeared to have been taken in a butterfly exhibit (potentially with non-native butterflies present).

In absence of demographic data, we assessed the current health of each worldwide population by evaluating the past trend in population numbers, the current status of milkweed and nectar resources, the current levels of insecticide exposure, and the current status of overwintering habitat. We compiled these data and assigned a population condition category of 'high,' 'moderate,' 'low,' or 'unknown' for each population. Condition categories were assigned using the descriptions presented in Table 3.1 (for similar condition category table approaches, see NatureServe 2013; IUCN 2018; and Puget Sound Stream Benthos 2018). If the information available was insufficient to assign a condition category, the population was marked as unknown status (Table 3.1).

Table 3.1. Categories used to define the health of the worldwide populations. Unknown indicates insufficient information about habitat quality, quantity, and corresponding monarch population trends.

Condition Rating	Past Trend	Current status of Milkweed and Nectar	Current status of Insecticides	Overwintering Habitat
High	$\lambda > 1$	Milkweed/Nectar not thought to be limiting monarch numbers	Current level of insecticide exposure to and/or toxicity of insecticides not thought to impact population-level	Overwintering habitat quality and quantity not thought to be limiting monarch numbers
Moderate	$\lambda \approx 1$	Milkweed/Nectar resources have been lost and are limiting monarch numbers in some portion of the population	Current level of insecticide exposure to and/or toxicity of insecticides limiting monarch numbers in some portion of the population	Overwintering habitat quality and quantity are limiting monarch numbers in some portion of the population
Low	$\lambda < 1$	Milkweed/Nectar resources have been lost and are limiting monarch numbers throughout the entire population	Level of insecticide exposure to and/or toxicity of insecticides are limiting monarch numbers throughout the entire population	Overwintering habitat quality and quantity are limiting monarch numbers throughout the entire population
Unknown	Unknown	Unknown	Unknown	Unknown

To assess future health of the worldwide populations, we searched the published literature and contacted international lepidopterists to identify the primary influences. For most influences (e.g., insecticides, land cover change, etc.), there was insufficient information to make an assessment.

Eastern & Western North American Populations

Unlike the worldwide populations, there are 20+ years of standardized survey data from which we can derive current abundance and population trend (λ) for eastern and western North American monarch populations. Thus, to assess the current and future health of these populations, we used published stochastic, geometric growth models for eastern (Semmens et al. 2016) and western (Schultz et al. 2017) populations. We updated the models with population data obtained since 2015 and incorporated the future state conditions of the influences (Figure 3.2). We briefly describe our models here; for additional detail see Voorhies et al. (2019) and see Appendix 2 for a list of small improvements made since the publication of Voorhies et al. (2019).

Our models assume that next year's population size in their wintering grounds, N_{t+1}, is a function of the monarch population size in the current time-step, N_t, and their log population growth rate, λ. To incorporate future threats and conservation actions into monarch population projections we added an additional term, δ, which represents a net change in population size (*N*) due to both positive and negative influences. We used published data, expert knowledge, and professional judgment to project the expected future state of each influence. To capture the uncertainty in our future state projections, we identified plausible optimistic and pessimistic changes for each influence. The most optimistic and pessimistic states for each influence were then combined to create composite plausible "best case" and "worst case" scenarios.

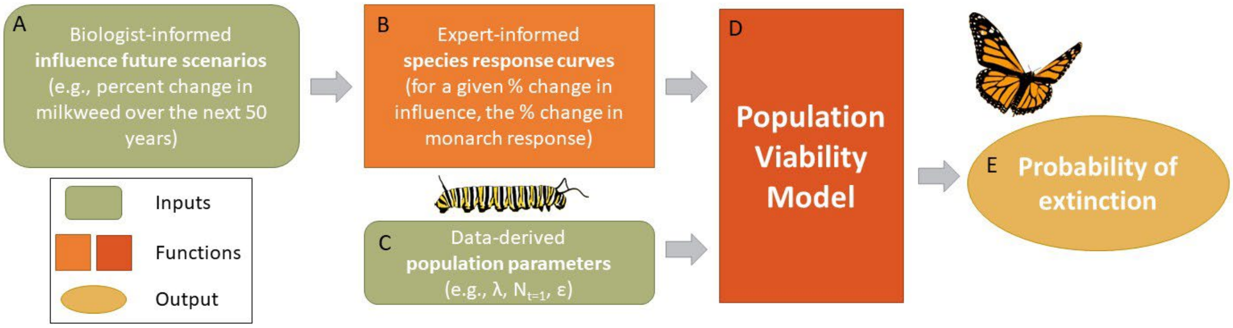

Figure 3.2. An overview of the monarch modeling framework. Biologist-informed scenarios (A) represent expected range in % change in a given influence over time. Expert-elicited population response curves (B) specific to each influence provide the proportional change in monarch response given a proportional change in the influence. Population response curves differ by influence and region (eastern and western populations). Population demographic data (C) were sourced from existing literature and used to initialize the model (D), which also received inputs from (B). Simulation outputs from the population viability analysis were compared against a range of extinction threshold values (E) to estimate the cumulative pE over time.

The health metric, pE, reflects the probability of the population size dropping below a threshold at which extinction would become inevitable (via a mechanism known as an extinction vortex). As others have done (e.g., Flockhart et al. 2015, p. 159; Semmens et al. 2016, p. 2; Schultz et al. 2017, p. 345), the extinction threshold is our primary mechanism for incorporating the consequences of Allee effects and environmental stochasticity at small population sizes. In addition to the extinction threshold, we introduced a population cap to address the limitation of a density-independent growth model which, as noted by Courchamp et al. (1999, p. 408), implicitly assumes populations increase linearly to carrying capacity.

Mechanisms that may trigger an extinction vortex in monarch populations include the following component effects:

- reduced survival on the overwintering grounds (Williams and Brower 2015; Berec et al. 2007, p. 187)
- increased predation on the overwintering grounds (Berec et al. 2007, p. 187; Brower and Calvert 1985, p. 857 and 861; Calvert et al. 1979, p. 849)
- reduced reproduction (e.g., mating depression due to difficulty finding mates [Berec et al. 2007, p. 187] and the subsequent reduction of female overwintering survival due to additional nutrients from multiple matings [Wells et al. 1993, p. 66])
- inability of small population sizes to rebound from sustained threats (Hutchings 2015, p. 6) or natural environmental variation (e.g., poor weather years)

The extinction thresholds for the eastern population were derived from expert-elicited estimates. We defined our lower and upper bounds for the extinction threshold as the median across the experts' "lowest" (0.05 ha) and "highest" (0.61 ha) estimates. For the western population, we used extinction thresholds reported in the literature. Our lower bound was set at 20,000 individuals (Schultz et al. 2017) and the upper bound at 50,000 (Wells et al. 1990). We assumed that all values between the lower and upper bounds were equally probable; thus, we used the upper and lower estimates to set the bounds of a uniform distribution (refer to Voorhies et al. 2019 for further discussion).

We calculated starting population size by taking the average of the last 5 years and calculated population growth rate (λ) and environmental stochasticity value (epsilon; $\mathcal{E}$) by using the Semmens et al. (2016) and Schultz et al. (2017) models, respectively, and updating the population data and time period. All input values are provided in Appendix 2.

Viability (Ch. 8)

To describe monarch viability over time, we evaluated how the change in the number, health, and distribution of monarch populations from historical to present to future influences the resiliency, redundancy, and representation of monarchs.

We used the results from our current and future forecasts--specifically the change in the number, health, and distribution of monarch populations over time--to evaluate the species' resiliency to environmental stochasticity, disturbances, and stressors. To assess monarch's redundancy, we qualitatively assessed how the current and forecasted number and distribution of populations affect the risk of catastrophic losses within each ACU. A catastrophe is an event that is outside the normal range of variation for a stressor and for which adaption is unlikely (Mangal and Tier 1993, p. 1083), and therefore, inevitably leads to population collapse (extinction).

For the eastern North American population, we identified overwintering habitat loss, monarch disease, widespread drought, extreme storm events (both at the Mexican overwintering sites and during migration funnel points), and widespread insecticide spray events as potential catastrophic events. Of these, we found reliable evidence for widespread drought and extreme storm events as sources for causing catastrophic losses, and thus, were carried forward in our analyses. For the western North American population, we identified extreme widespread drought, disease, severe storms events, wildfire, widespread milkweed loss, widespread insecticide spray events, and co-occurrence of a poor environmental conditions and low population abundance as potential catastrophic events. Of these, we found reliable evidence for widespread drought and the co-occurrence of poor environmental conditions and low population abundance as sources for causing catastrophic losses, and thus, were carried forward in our analyses.

For the worldwide populations, we identified climate change induced sea level rise and maximum temperature increases as potentially catastrophic events. We classified risk as either "No Known Risk" or "At Risk" (Table 3.2). Using the Third Assessment Report developed by the International Panel on Climate Change (IPCC), we determined which low-lying islands occupied by monarchs may be at risk of permanent inundation, and used the maximum elevation of those islands to develop thresholds for the risk classifications (IPCC 2001). We also qualitatively assessed where daily maximum surface temperatures exceeding 42°C (a temperature threshold that leads to mortality; Nail et al. 2015b, p. 99) are projected to increase by the year 2069 (~50 years from now) under Representative Concentration Pathways (RCP) scenarios 4.5 and 8.5 using climate projections obtained from the Earth System Grid Federation (Cinquini 2014). Given scale and magnitude of impact (whether population would be exposed to events that would lead to population extinction), this analysis falls under a catastrophic risk.

Table 3.2. Categories used to define the risk of the worldwide populations to predicted climate change impacts.

Future Influence	**Risk Category**	**Definition**
Sea Level Rise	No Known Risk	Not at low elevation (highest point >100m above sea level).
Sea Level Rise	At Risk	Very low elevation (highest point ≤100m above sea level) and single location represents an entire population.
High Temperatures	No Known Risk	Number of days and/or areas with daily maximum surface air temperatures above lethal levels (42˚C) not projected to increase under moderate (RCP 4.5) or severe (RCP 8.5) scenarios.
High Temperatures	At Risk	Number of days and/or areas with daily maximum surface air temperatures above lethal levels (42˚C) are projected to increase under the moderate (RCP 4.5) or severe (RCP 8.5) scenarios.

Lastly, we evaluated the monarch’s ability to adapt to novel changes in its physical and biological environment by assessing the likelihood of monarchs persisting in each of the 8 ACUs given the forecasted influences and catastrophes. Specifically, for the eastern and western North American ACUs, we used the results of our population modeling to predict the likelihood of persistence of monarchs within both ACUs over the next 50 years. For the remaining 6 ACUs, we qualitatively express the likelihood of persistence within each of the 6 ACUs over the next 50 years given the risks of catastrophic sea level rise or high temperature conditions.

Chapter 4: Results – Analysis of Historical Condition

This chapter describes the number, health, and distribution of monarch populations up to the present day. The historical condition provides the baseline condition from which we evaluated changes in monarch viability over time.

Worldwide

There are no reliable records of monarchs outside of continental North America or the Caribbean before 1840 (Vane-Wright 1993, p. 180). However, by 1883 the monarch was reported as one of the most common butterflies in many Pacific Islands (Walker 1914, p. 187). Host plants used by monarchs in these non-North American locations include *Asclepias* spp., *Gomphocarpus* spp., and *Calotropis* spp. (all either milkweed or closely related genera; Blakley and Dingle 1978, p. 134; Buden and Miller 2003, p. 4). It is generally accepted that both monarchs and milkweed dispersed from North America via human assistance, potentially aided through wind dispersal events (Brower 1995, p. 354). For the purposes of our analysis, we assume that monarchs in locations outside of North America have become naturalized, and thus, these records, along with the North American occurrences, comprise the historical range of the species (Figure 4.1).

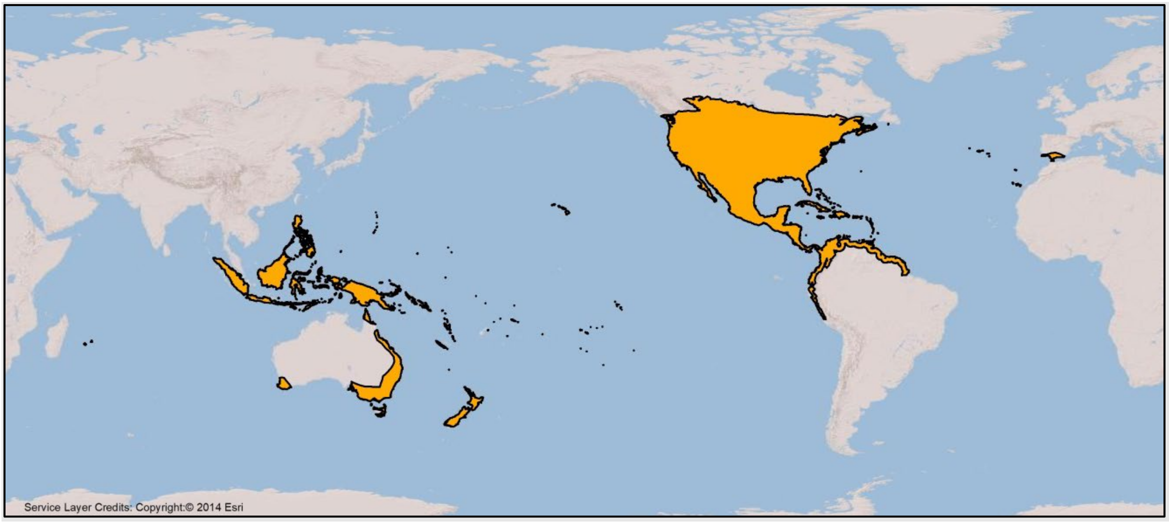

Figure 4.1. Map showing global range of monarchs (orange shows known range).

We found monarch occurrence records in 90 countries, islands, or island groups. We delineated these occurrences into 31 different populations (Table 4.1). We assume that at some point in the past, all populations were healthy. Table 4.1 also shows how these 31 populations are distributed among the eight ACUs (see Chapter 2 for description of the ACUs and how they were delineated). This organization is visually represented in Figure 4.2. While the Australia, New Zealand, and Indo-Pacific Islands ACU appears the largest in spatial extent, the eastern North American population has the most individuals (even accounting for large variation in estimates; Figure 4.3).

Table 4.1. The 31 delineated monarch populations, with their associated ACUs and the countries and islands that comprise each population.

ACU	**Population**	**Countries/Islands within Population**
Australia, New Zealand, and Indo-Pacific Islands	Australia	Commonwealth of Australia
	Cook Island	Cook Islands
	French Polynesia	French Polynesia
	Greater Indonesia	Nation of Brunei, Republic of Indonesia, Malaysia, Democratic Republic of Timor-Leste
	Guam & CNMI	Guam, Commonwealth of Northern Mariana Islands (CNMI)
	Johnston Atoll	Johnston Atoll
	Kiribati	Republic of Kiribati
	Marquesas Islands	Marquesas Islands
	Marshall Islands	Republic of the Marshall Islands
	Mascarene Islands	Republic of Mauritius, Réunion
	Micronesia	Federated States of Micronesia
	Nauru	Republic of Nauru
	New Zealand	New Zealand
	Norfolk Island	Norfolk Island
	Palau	Republic of Palau
	Papua New Guinea	Independent State of Papua New Guinea
	Philippines	Republic of the Philippines
	Samoa	American Samoa, Samoa
	South Pacific Islands	Republic of Fiji, New Caledonia, Society Islands, Solomon Islands, Vanuatu
	Tokelau	Tokelau
	Tonga	Kingdom of Tonga
	Tuvalu	Tuvalu
	Wallis & Futuna	Territory of the Wallis and Futuna Islands
Central America & the Caribbean	Caribbean	Anguilla, Antigua and Barbuda, Bahamas, Barbados, Bermuda, Bonaire, British Virgin Islands, Cayman Islands, Cuba, Dominica, Dominican Republic, Grenada, Guadeloupe, Haiti, Jamaica, Martinique, Montserrat, Puerto Rico, Saba, Saint Barthélemy, Sint Eustatius, Saint Kitts and Nevis, Saint Lucia, Saint Martin, Saint Vincent and the Grenadines, Sint Maarten, Turks and Caicos Islands, U.S. Virgin Islands
	Central America	Belize, Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua, Panama, Mexico
S. Florida	Florida	United States (FL)
Hawaii	Hawaii	United States (HI)
Iberian Peninsula	Iberian Peninsula	Azores, Canary Islands, Gibraltar, Madeira, Morocco, Portugal, Spain
South America & Aruba	South America and Aruba	Aruba, Colombia, Curacao, Ecuador, French Guiana, Guyana, Peru, Suriname, Trinidad and Tobago, Venezuela
E. North America	Eastern North America	Canada, Mexico, Saint Pierre and Miquelon, United States (East)
W. North America	Western North America	Canada, United States (West), Mexico

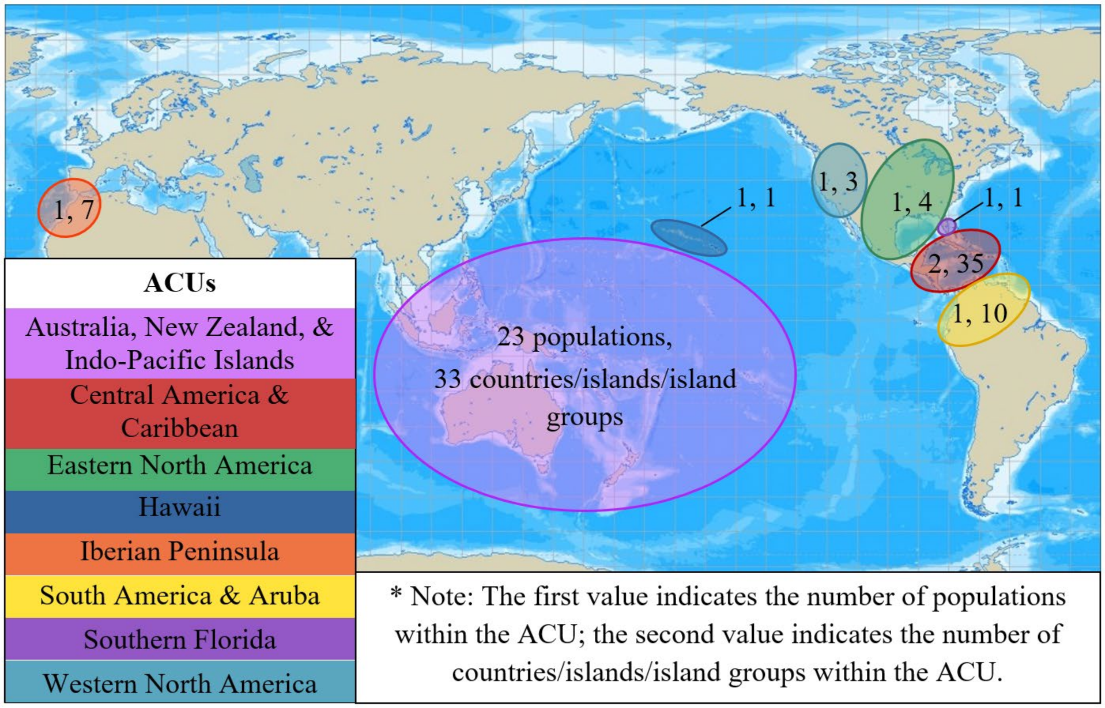

Figure 4.2. Generalized map of the eight ACUs, with the number of populations and countries contained within each ACU provided. Note that the total number of countries/islands/island groups do not add up to 90 because some are present in multiple ACUs.

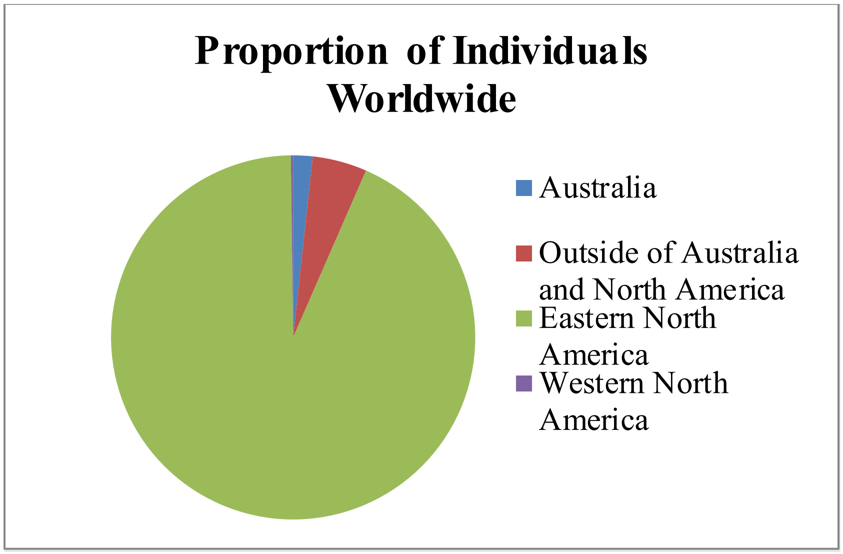

Figure 4.3. Estimated relative proportion of individual monarchs by geographical area. The numbers are based on the following: eastern North America (77,141,600; based on average of last 5 years overwintering estimates, assuming a 21.1 million monarch/ha density), western North America (168,365; based on average of past five years of overwintering counts); Australia (1,424,790; based on estimates from M. Zalucki, pers. comm.); and outside of Australia and North America (4,000,000; based on 3-5 million monarch estimate; M. Zalucki, pers. comm.).

Note that throughout the rest of the document, when the term ‘worldwide’ is used in relation to monarchs, we are referring to 29 monarch populations excluding the eastern and western North American populations.

Eastern North American Population

The eastern North American monarch population has been systematically censused annually since 1994 (Figure 4.4; Vidal and Rendón-Salinas 2014, pp. 167-168). Although varying year-to-year, monarchs consistently numbered in the hundreds of millions throughout the 1990s and early 2000s (assuming a 21.1 million monarch/hectare density; Thogmartin et al. 2017a, p. 1). There are additional survey data suggesting that monarch populations were as high or higher in the two decades prior to standardized monarch monitoring at the Mexican overwintering sites (Vidal and Rendón-Salinas 2014, p. 172, Calvert and Brower 1986, pp. 167-169).

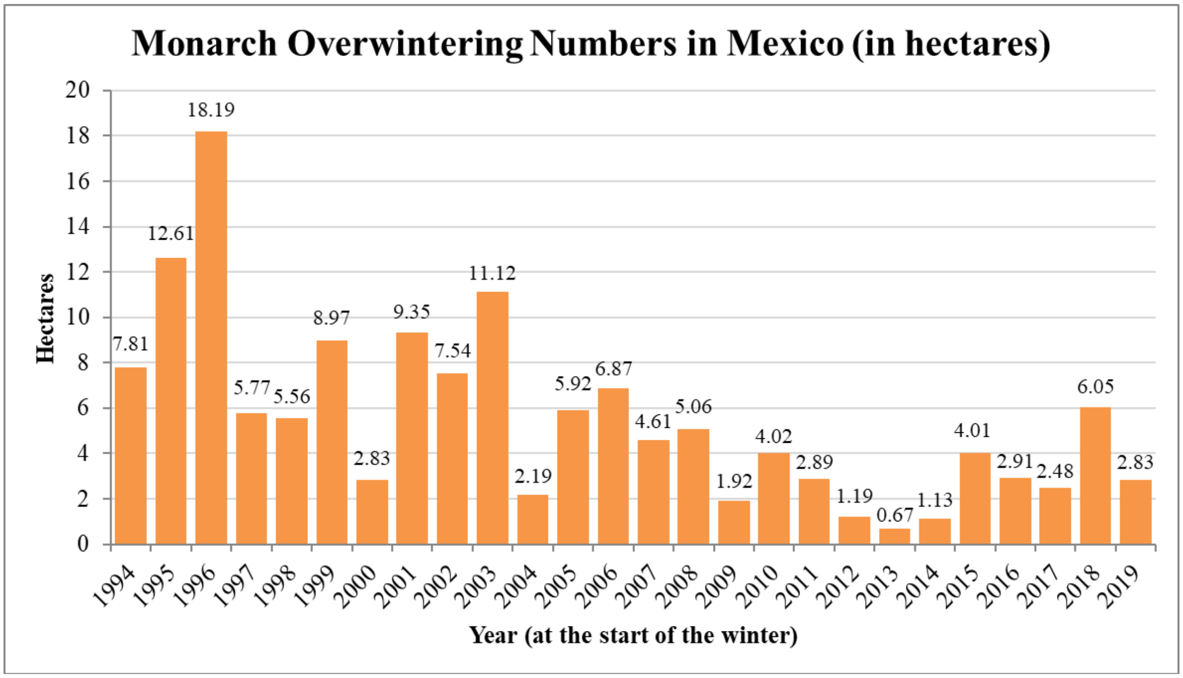

Figure 4.4. Area occupied (in hectares) by eastern North American monarch butterflies at overwintering sites in Mexico (actual hectare measurement displayed above each bar). Year displayed is the beginning year for the winter (e.g., 2017 represents the number for the winter of 2017-2018). Data from Monarch Watch (2020).

Western North American Population

The western North American population has been censused annually since 1997, providing an estimate of annual population size (Figure 4.5). Recent work, using past survey data, gives estimates of millions of butterflies in the mid-1980s (Schultz et al. 2017, p. 3).

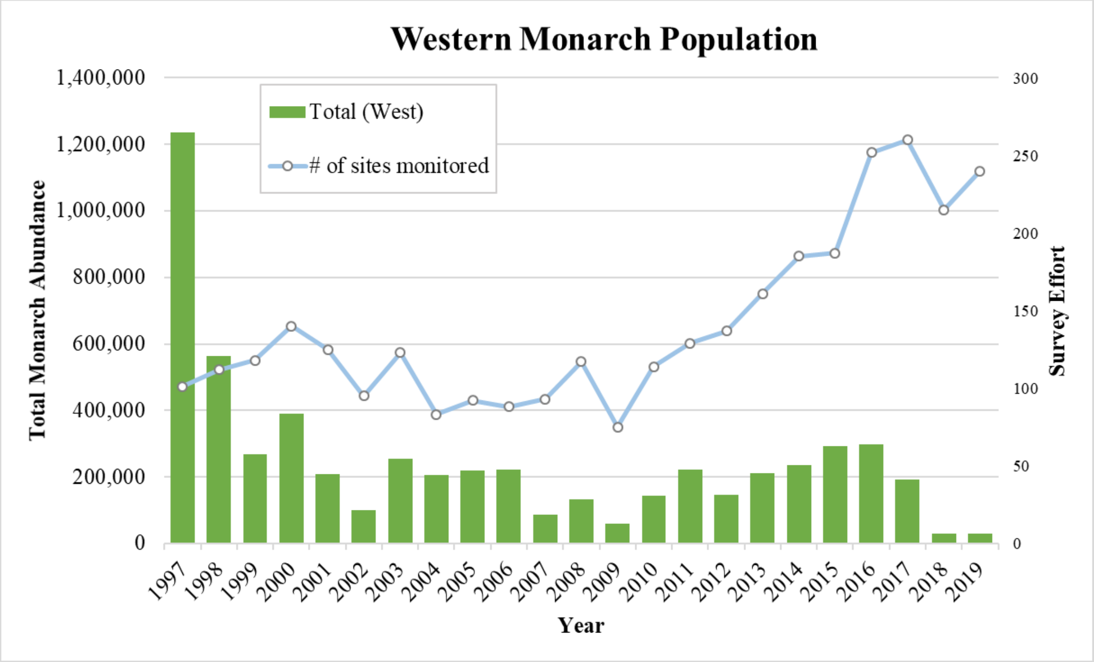

Figure 4.5. Thanksgiving counts showing the number of western North American monarch butterflies observed at overwintering sites (green bars). Blue line shows the number of sites monitored (survey effort) for a given year. Data from The Xerces Society for Invertebrate Conservation 2020, entire.

Chapter 5: Results – Analysis of Current Condition & Current Influences

This chapter describes the number, health, and distribution of monarch populations given current state conditions and describes the influences that have led to this current condition. We present the current condition and influences that led to the condition for the eastern and western North American populations first, followed by the current conditions and influences for the worldwide populations.

Eastern North American Population – Current Condition

Based on the past annual censuses, the eastern North American population has been generally declining over the last 26 years (Figure 4.4). Although the numbers at the overwintering sites have declined, we did not find a corresponding change in the spatial extent of the population during the breeding season. Given its current population size and population growth rate, the pE over the next 60 years is 61% (48%-69%; CI 50%) (Figure 5.1).

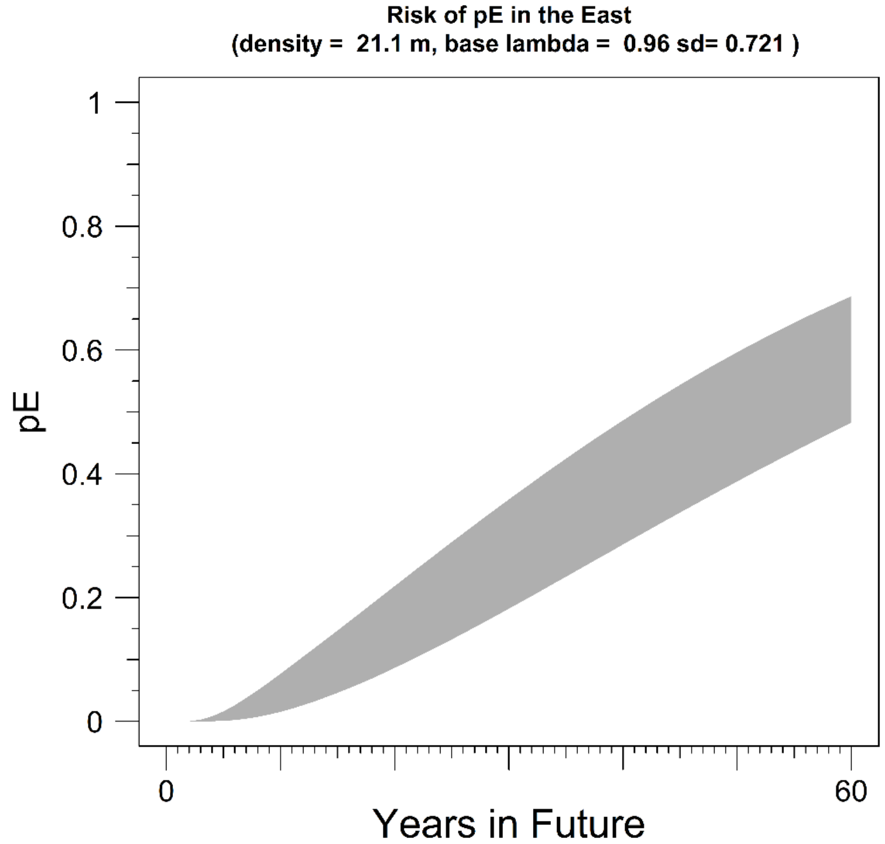

Figure 5.1. pE for the eastern North American monarch population over time, represented by 50% confidence interval (gray space). Probability based on current trend in growth.

Western North American Population – Current Condition

Based on the past annual censuses, the western North American population has been generally declining over the last 23 years, despite an increasing number of sites being counted (Figure 4.5). Under current conditions, the risk of extinction over time is predicted to increase sharply, with the pE over 60 years reaching 99% (98%-99%, CI 50%) (Figure 5.2).

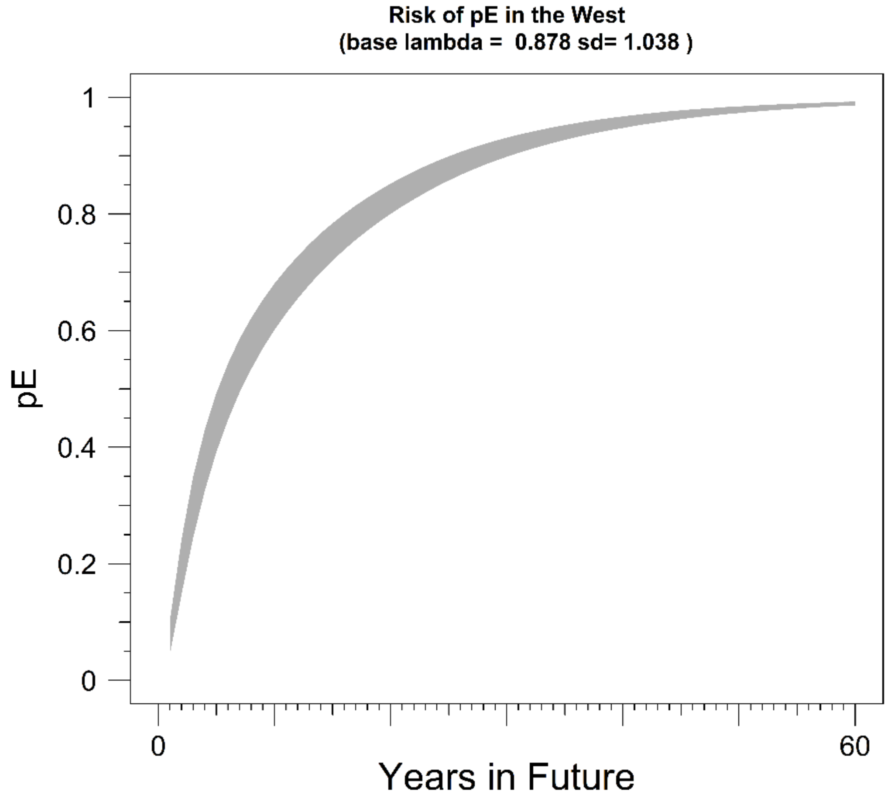

Figure 5.2. pE for the western North American monarch population over time, represented by 50% confidence interval (gray space). Probability based on current trend in growth.

North American Populations – Current Influences

There are a myriad of influences operating on the North American populations. With the assistance of monarch experts, we identified the important factors driving monarch population dynamics for the eastern and western North American populations (Tables 5.1 and 5.2). The primary drivers affecting the health of the two North American migratory populations are changes in breeding, migratory, and overwintering habitat (due to conversion of grasslands to agriculture, urban development, widespread use of herbicides, logging/thinning at overwintering sites, unsuitable management of overwintering groves , and drought), continued exposure to insecticides, and effects of climate change (Figure 5.3). Below, we discuss the key influences on monarch populations—the aforementioned stressors and monarch conservation efforts.

Table 5.1. Expert-elicited rank and extent of impact (% contribution to the decline from the historical period) of the influences on the eastern North American population. % Contribution = median value across experts; the lowest and highest expert judgment among the experts provided in parentheses (see Voorhies et al. 2019, Suppl.2).

Influence	**Rank**	**% Contribution**
Availability, spatial distribution, and quality of milkweed	1	25 (10-60)
Availability and quality of overwintering habitat	2	20 (10-30)
Climate (storms, drought, temperatures)	3	12.5 (6-23)
Availability, quality, and spatial distribution of migration resources	4	12 (2-20)
Disease and natural enemies	5	9.5 (1-15)
Insecticides	6	8 (1-10)
Availability, spatial distribution, and quality of nectar resources (breeding)	7	5 (1-10)
Road mortality and pollutants	8	3 (1-5)
Biogeographical scrambling of milkweed spp. (includes non-native spp.)	9	2 (0-4)
Other	10	2 (0-8)
Monarch releases, captive breeding, and translocation	11	1.5 (0-3)

Table 5.2. Expert-elicited rank and extent of impact (% contribution to the decline from the historical period) of the influences on the western North American population. % Contribution = median value across experts; the lowest and highest expert judgment among the experts provided in parentheses (see Voorhies et al. 2019, Suppl.2).

Influence	**Rank**	**% Contribution**
Availability, spatial distribution, and quality of milkweed	1	22 (15-25)
Availability, spatial distribution, and quality of nectar resources (breeding)	2	18 (13-20)
Insecticides	3	18 (15-22)
Climate change effects via impacts to habitat	4	17 (10-19)
Availability and quality of overwintering habitat	5	16 (12-18)
Climate change via non-habitat mediated effects	6	8 (3-14)
All others	7	4 (0-7)

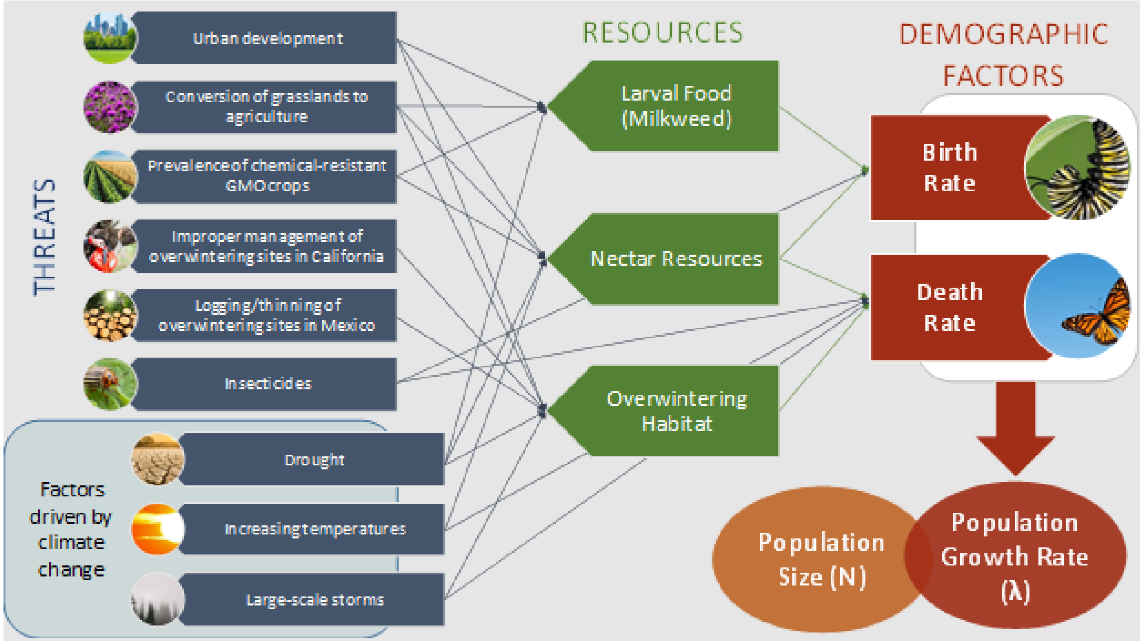

Figure 5.3. Influence diagram showing the key Influences and how they drive monarch population abundance (N) and growth rate (lambda, λ). Note, conservation efforts can decrease all the listed threats and improve all resources for monarchs.

Availability, Distribution, and Quality of Milkweed

The availability of milkweed is essential to monarch reproduction and survival. Reductions in milkweed is cited as a key driver in monarch declines (Brower et al. 2012, p. 97; Pleasants and Oberhauser 2013, p.7; Inamine et al. 2016, p. 1081; Thogmartin et al. 2017b, p.12; Waterbury and Potter 2018, pp. 42-44; Saunders et al. 2019, p. 8612).

A majority of the milkweed loss has occurred in agricultural lands, where intensive herbicide usage for weed control has resulted in widespread milkweed eradication. Pleasants (2017, p. 7), for example, estimated that over 860 million milkweed stems were lost in the Midwest between 1999 and 2014, a decline of almost 40%. Currently, approximately 89% and 94% of corn and soybean crop acreage, respectively, are planted as glyphosate (herbicide)-tolerant crops (USDA 2018). Glyphosate use in western agricultural lands has also increased dramatically since the 1990s, especially within the Central Valley of California, Snake River Plain of Idaho, and the Columbia River Basin, which spans the border between Washington and Oregon (USGS NAWQA 2017; Waterbury and Potter 2018, p. 42). As weed species develop increasing resistance to glyphosate, other herbicide (e.g., dicamba) tolerant crops are developed, which can lead to a corresponding increase in herbicide use. Accordingly, herbicide impacts to milkweed and nectar plants will continue to impact monarch resources.

Milkweed is also lost on the landscape through development and conversion of grasslands (Lark et al. 2015, pp. 3-4). Between 2008 and 2012, a total of 5.7 million acres of grassland were converted to new cropland, including up to 3 million acres of Conservation Reserve Program (CRP) land (Lark et al. 2015, p. 5). Pleasants and Oberhauser (2013, pp. 5-6) estimate that the loss of agricultural milkweeds in the Midwest has resulted in an 81% decline in monarch production, in part because monarch egg densities were higher on milkweed in agricultural fields (3.89 times more eggs than on non-agricultural milkweed). This particularly impacts the eastern monarch population because more Mexico overwintering monarchs originate from the Midwest crop belt region than any other region (with estimates ranging from 38% to over 85% of all overwintering monarchs originating from the Midwest; Wassenaar and Hobson 1998, pp. 15438-15439; Flockhart et al. 2017, p. 4). Accordingly, herbicide impacts to milkweed and nectar plants will continue to impact monarch resources available in agricultural lands.

Availability, Distribution, and Quality of Breeding Range Nectar Resources

Reductions in nectar resources are also cited as a potential key driver in monarch declines (Thogmartin et al. 2017b, p.12). Losses of nectar resources are due to same stressors identified above for milkweed resources.

Availability, Distribution, and Quality of Migration Nectar Resources

Losses of nectar sources during migration have also been particularly implicated as a potential key driver in monarch declines (Inamine et al. 2016, p. 1081; Thogmartin et al. 2017b, p.12; Saunders et al. 2019, p. 8612). Losses of nectar resources are due to same stressors identified above for milkweed resources. Additionally, with a warming climate, drought impacts may become more important, especially in the western population and in the migratory bottleneck for the eastern population (see *Climate Change* in Current Influences section within this chapter for more details).

Availability and Quality of Overwintering Habitat

Both western and eastern monarchs rely on the microclimate provided by the trees at their overwintering sites (Leong et al. 2004, entire; Williams and Brower 2015, entire). Loss of trees occurs at overwintering sites in Mexico primarily through small- and large-scale logging, storms, and an increasingly unsuitable climate (see *Climate Change* section below for more details). Most overwintering sites used by eastern monarchs occur within the Monarch Butterfly Biosphere Reserve (Reserve), a 56,259-ha protected area. Within this area, there is a logging ban within the 13,551-ha core zone (Ramírez et al. 2015, p. 158). However, recent logging has occurred both legally (including salvage logging allowed after storms) and illegally at multiple colonies (Vidal et al. 2014, pp. 180-185; Brower et al. 2016, entire).

Logging was estimated by Vidal and colleagues (2014, p. 180) in the core zone of the Reserve from 2002 through 2012. Within this period, 2,179 ha of core zone were deforested (<10% canopy cover remained; 1,254 ha) or degraded (a decrease in canopy cover; 925 ha). Most of these losses were attributed to illegal logging (2,057 ha), with the remaining 122 ha lost due to floods, drought, strong winds, and fire. Current estimates of forest loss throughout the Reserve vary from 0-2.4% per year (Ramírez et al. 2015, p. 163). While anti-logging and reforestation efforts are underway (López García 2011, p. 631), logging is still ongoing within the Reserve (Brower et al. 2016, entire). Although clearcutting of forests destroys habitat directly, thinning of the forest also changes the microclimate needed by overwintering monarchs, making them more susceptible to winter mortality (Brower et al. 2011, p. 43).

Western monarch overwintering habitat along the Pacific Coast has been subject to loss through various forms of development, particularly urban development (Sakai and Calvert 1991, p. 149; Frey and Schaffner 2004, p. 172). Habitat alteration, both natural and anthropogenic, can also alter the microclimate of the western overwintering sites, leading to less suitable habitat conditions (Jepsen et al. 2015, p. 17). There are many other stressors that can work alone or in tandem on the western overwintering sites, including disease and pests that impact the trees used for overwintering, as well as senescence and improper grove management. Fire is also a threat, both indirectly through habitat loss and directly to overwintering monarchs (Pelton et al. 2016, pp. 28, 32). Drought in the West can further exacerbate the stressors on the western overwintering sites (see *Climate Change* section below).

Insecticide Exposure

Insecticides are pesticides with chemical properties that are designed to kill insects, and most are non-specific and broad-spectrum in nature. That is, insects exposed to these insecticides are susceptible to mortality and/or sub-lethal effects. Furthermore, the larvae of many Lepidopterans are considered major pest species and insecticides are tested specifically on this taxon to ensure that they will effectively kill individuals at labeled application rates. Monarchs may also be exposed to insecticides in areas beyond the insecticide application points due to drift (Olaya-Arenas and Kaplan 2019, p. 1; Halsch et al. 2020, p. 3).

The monarch butterfly is widely distributed across the United States, occurring in a variety of urban and rural habitat types that include milkweed plants and other flowering forbs. Insecticide impacts to monarchs are primarily influenced by the extent to which monarchs are exposed to insecticides throughout their range. Although insecticide use is most often associated with agricultural production (for example, between 2005 and 2012, 60% of insecticide applied occurred on agricultural lands, USEPA 2017, p.11), any habitat where monarchs are found may be subject to insecticide use. Insecticides can be used for insect pest control anywhere there is a pest outbreak or for general pest prevention. Homeowners may treat yards and gardens to protect plants from pests or purchase plants from nurseries that sell neonicotinoid-treated plants as ornamentals. Natural areas, such as forests and parks, may be treated to control for insects that defoliate, bore into wood, or otherwise damage trees. Outbreaks of pests such as gypsy moths, Mormon crickets, or grasshoppers may trigger insecticide treatments over larger areas to control populations. Use of insecticides in vector control, especially pyrethroids and organophosphates, may be significant in areas of the country where mosquitoes pose a public health threat or reach nuisance levels. The use of insecticides in the U.S. is ubiquitous; in 2012 for example, expenditures on insecticides topped $5 billion in the United States, with 64 million pounds used for agriculture, home and garden, and other purposes (USEPA 2017, see Tables 2.2 & 3.1).

The most widely used classes of insecticide include organophosphates, pyrethroids, and neonicotinoids. Neonicotinoids entered the market in the mid- to late-1990s (Figure 5.4), and because of their high insecticidal activity at low application rates, they are now the most used class of insecticides in the world (Braak et al. 2018, p. 507). By 2008, for example, neonicotinoid insecticides accounted for 80% of global seed treatment sales (Jeschke et al. 2011, p. 2898), and by 2011, >79% of the corn hectares and 34% to 44% of soybean acreage in the U.S. were planted with neonicotinoid-treated seeds (Douglas and Tooker 2015, p. 5092). Neonicotinoid insecticides are absorbed into plants and distributed throughout their tissues to their stems, leaves, roots, fruits, and flowers. They kill and injure insects by attacking their central nervous system.

Studies looking specifically at dose-response of monarchs to neonicotinoids, organophosphates and pyrethroids have demonstrated monarch toxicity (e.g., Krischik et al. 2015, entire; James 2019, entire; Krishnan et al. 2020, entire; Bargar et al. 2020, entire). Moreover, the magnitude of risk posed by insecticides may be underestimated, as research usually examines the effects of the active ingredient alone, while many of the formulated products contain more than one active insecticide (e.g., Swagger contains bifenthrin and imidacloprid, Krishnan et al. 2020, p. 17, but see Oberhauser et al. 2009, entire). The additional risk posed from compounds added to improve the kill rate (referred to as synergists) are often not assessed. The use of synergists is not uncommon. Olaya-Arenas and Kaplan (2019, p. 13), for example, reported that fungicides (often used as a synergist) were most commonly detected on milkweed samples (e.g., 98% of the milkweed sample in one year contained the fungicide, Propiconazole) and, in many of these cases, co-occurred with insecticides like deltamethrin and thiamethoxam. See *Insecticide Supplemental* for further discussion of the risk of pesticides to the monarch, including data, references, and supporting information.

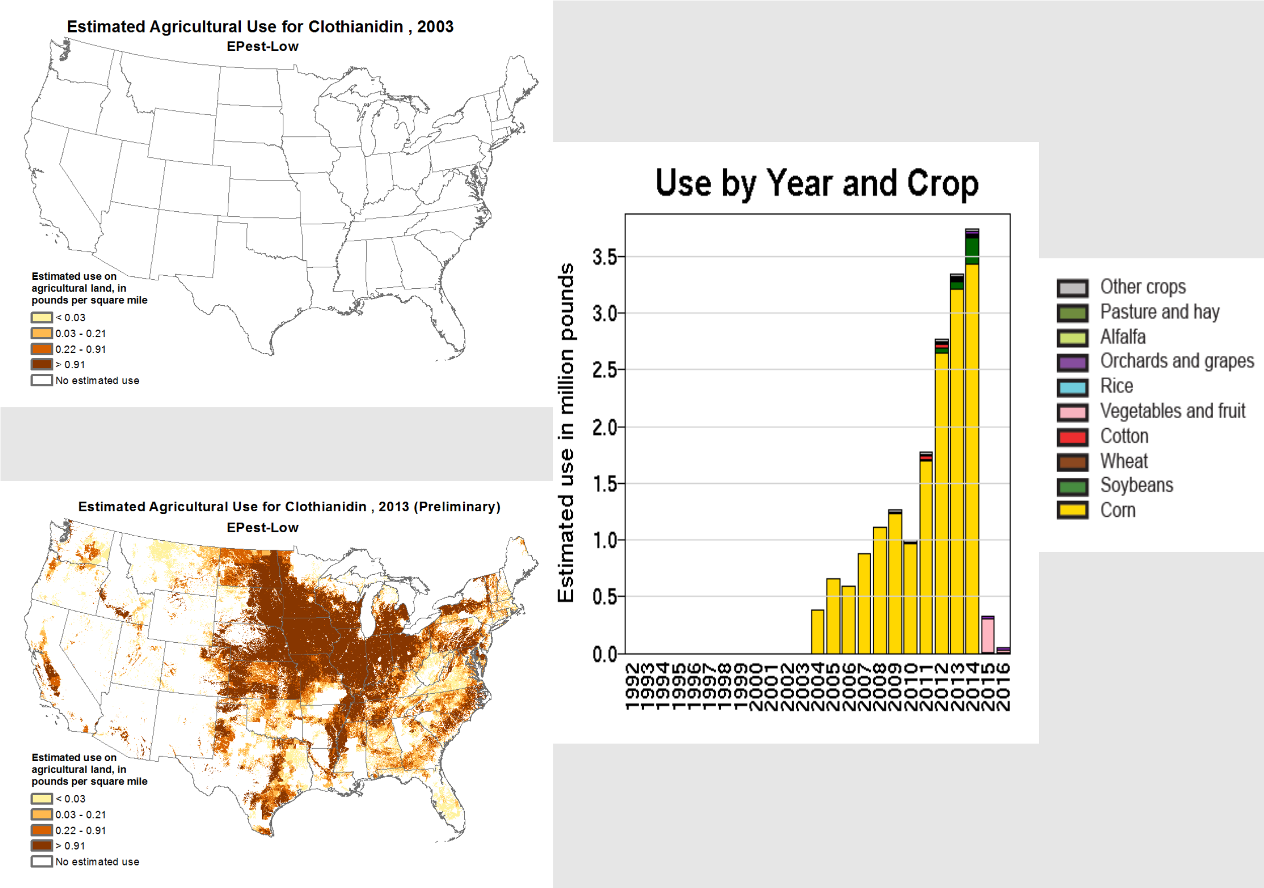

Figure 5.4. Estimated use of Clothianidin (a neonicotinoid) by location in 2003 (top) and 2013 (bottom) and by year (right). USGS National Pesticide Synthesis Project, accessed 2020; https://water.usgs.gov/nawqa/pnsp/usage/maps/

<u>Climate Change Effects</u>

Climate change can affect monarchs both directly and indirectly (Nail and Oberhauser 2015, entire) on both the overwintering and breeding grounds. Increasing storm frequency in the Mexican overwintering colonies can lead to catastrophic (up to 80%) mortality through the freezing temperatures that accompany these storms (Anderson and Brower 1996, p. 112; Brower et al. 2004, entire). Severe storms may become more frequent with precipitation predicted to increase during the winter when monarchs are present in Mexico (Oberhauser and Peterson 2003, p. 14067).

Monarchs need a very specific microclimate at their overwintering sites not just to avoid storm mortality, but also to avoid early lipid depletion (see Chapter 2, *Monarch Life History* section). Additionally, changing precipitation patterns and temperatures may influence the microclimate needed by overwintering monarchs (Williams and Brower 2015, p. 116). Current modeling of the monarch's fundamental niche predicts the loss of 38.6% to 69.8% of current suitable habitat within the Monarch Butterfly Biosphere Reserve (Zagorski 2016, p. 17). In western North America, climate change is predicted to cause a significant change in the distribution of overwintering monarchs in coastal California. Results from climatic niche modeling by Fisher et al. (2018, p. 10) suggest that climate change will result in an inland and upslope displacement of suitable overwintering conditions. The probability of occurrence of suitable overwintering conditions becomes roughly proportional to elevation.

Climate change impacts, particularly increasing temperatures, may impact monarch fecundity (Oberhauser 1997, pp. 168-169), mating success (Solensky and Oberhauser 2009, p.6), and survival during migration and while overwintering (Masters et al. 1988, entire; Alonso-Mejía et al. 1997, entire). Laboratory studies indicate optimal temperatures for monarch range from 27–29°C with sublethal effects beginning around 30–36°C range and an upper lethal thermal limit of 42°C (Zalucki 1982, p. 243; York and Oberhauser 2002, p. 294; Zalucki and Rochester 2004, p. 225; Nail et al. 2015b, p. 101). Nail and colleagues (2015b) also found nighttime temperatures of 34°C during periods with daytime temperatures of 38°C resulted in lower survival, showing that respites from elevated temperatures are important in allowing monarchs to survive temperature stress (Nail et al. 2015b, p. 104). Temperatures consistently above 33°C to 35°C are unsuitable for monarchs and may account for their general absence from southern U.S. states after spring (Malcolm et al. 1987, p. 78; Zalucki and Rochester 1999, pp. 155- 157). High temperatures and drought conditions may be particularly impactful during the crucial spring migration (Chip Taylor, pers. comm. 2020).

In addition to the impact of climate change on overwintering monarchs directly, the Mexico overwintering sites are predicted to be less suitable for oyamel fir trees, the predominant monarch roosting tree. The overwintering sites are predicted to become increasingly warm throughout the year, potentially making 50% or more of the sites unsuitable for oyamel fir trees in 2030, and completely unsuitable for the oyamel fir trees by 2090 (Sáenz-Romero et al. 2012, p. 102; Ramírez et al. 2015, p. 167). Widespread drought is similarly likely to impact trees in the western overwintering areas both directly and indirectly due to increased susceptibility to pests (Paine and Millar 2002, p. 148).

A warming climate may influence breeding habitat by altering suitable locations for both monarchs (Batalden et al. 2007, pp 1369-1370) and their milkweed host plant (Lemoine 2015, entire). Saunders et al. (2019, p. 8612) suggested that nectar resources during migration may be reduced under climate conditions (decreased precipitation) projected for south-central Texas. Drought may also influence the amount and availability of nectar needed for migrating butterflies (Brower et al. 2015, entire; Stevens and Frey 2010, p. 740; Espeset et al. 2016; p. 826; see *Widespread Drought* section). The coastal non-migratory population may also be impacted by loss of habitat through rising sea levels due to climate change (Tampa Bay Climate Science Advisory Panel 2015, entire). While drought and increased temperatures may reduce monarch habitat in some areas, the climatically suitable niche for monarchs may increase, potentially increasing their summer breeding grounds if both monarchs and milkweed are able to adapt (Lemoine 2015, pp. 10-17).

Climate change may additionally impact monarchs in ways that are more difficult to measure. This may include phenological mismatch (e.g., timing of milkweed and nectar sources not aligning with monarch migration; Thogmartin et al. 2017b, p. 13) or range mismatch with associated species (e.g., changed environmental suitability of monarch natural enemies; McCoshum et al. 2016, p. 229-233). Furthermore, recent research suggests that carbon dioxide may impact the medicinal properties of some milkweed species, potentially leading to increased *OE* parasite virulence and decreased monarch tolerance of *OE* infections (Decker et al. 2018, p. 7; see Appendix 2 for more information on *OE*).

Conservation Efforts

While many factors have been implicated in the decline in monarch populations, the loss of milkweed and nectar resources (i.e., breeding and migratory habitat) has been targeted as the threat that can be most easily addressed through conservation efforts. Protection, restoration, enhancement and creation of habitat is a central aspect of recent monarch conservation strategies, thus highlighting the importance of restoring and enhancing milkweed and nectar resources (Oberhauser et al. 2017a, p. 6-8; Pleasants 2017, p. 43; Thogmartin et al. 2017b, p. 2-3; MAFWA 2018, p. 52; Pelton et al. 2019, p. 4-5, WAFWA 2019). Improved management at overwintering sites in California has also been targeted to improve the status of western North American monarch butterflies (Pelton et al. 2019, p. 4; WAFWA 2019).

Major conservation plans and efforts include the Mid-America Monarch Conservation Strategy developed by the Midwest Association of Fish and Wildlife Agencies (MAFWA), the Western Monarch Butterfly Conservation Plan developed by the Western Association of Fish and Wildlife Agencies (WAFWA), and the Nationwide Candidate Conservation Agreement for Monarch Butterfly on Energy and Transportation Lands (CCAA/CCA) developed by entities from the energy and transportation sectors and the Energy Resources Center at the University of Illinois – Chicago. The Mid-America Monarch Conservation Strategy established a goal of adding 1.3 billion stems of milkweed on the landscape by 2038 (MAFWA 2018). The 1.3 billion stem goal is an estimated goal for adding enough habitat to support 6 hectares of overwintering population for the eastern North American population, per Pleasants and Thogmartin et al. (2017; 2017c). Twenty states—including Arkansas, Illinois, Indiana, Iowa, Kansas, Kentucky, Maryland, Michigan, Minnesota, Missouri, Nebraska, New York, North Dakota, Ohio, Oklahoma, Pennsylvania, South Dakota, Texas, West Virginia, and Wisconsin—have agreed to participate in the effort to reach the 1.3 billion stem goal, which will require contributions from multiple sectors of society, including private land owners, agricultural and non-governmental organizations, rights-of-way organizations, and federal, state and local governments. The Western Monarch Butterfly Conservation Plan currently encompasses the states of Arizona, California, Idaho, Nevada, Oregon, Utah, and Washington, which comprise the core of the western monarch range (WAFWA 2019). The plan includes short-term goals of: 1) protecting and managing 50% of all currently known and active monarch overwintering sites, including 90% of the most important overwintering sites by 2029; and 2) providing a minimum of 50,000 additional acres of monarch-friendly habitat in California's Central Valley and adjacent foothills by 2029. It also includes overwintering and breeding habitat conservation strategies, education and outreach strategies, and research and monitoring needs. The monarch CCAA will also contribute to the goals of these plans by coordinating and providing guidance to businesses and organizations in the energy and transportation sectors seeking to implement conservation efforts for monarchs. In exchange for implementing voluntary conservation efforts and meeting specific requirements and criteria, those businesses and organizations enrolled in the CCAA will receive assurance from the USFWS that they will not have to implement additional conservation measures should the species be listed. The goal of the CCAA is enrollment of up to 26 million acres of land in the agreement, providing over 300 million additional stems of milkweed (Cardno, Inc. 2020).

There are many other conservation efforts implemented under agreements, such as the Farm Service Agency's Conservation Reserve Program and the Natural Resource Conservation Service's Environmental Quality Incentives Program, Wetland Reserves Program, and Conservation Stewardship Program, which will be critical for meeting MAFWA and WAFWA's stated goals. Additionally, multiple federal, state and local governments, non-governmental organizations, and private businesses and individuals have provided information about regional and local monarch conservation plans and efforts. Although not associated with any formal plans or agreements, we have also obtained information on thousands of small and backyard pollinator gardens through organizations such as Monarch Watch.

Several land managers who oversee overwintering sites in California have developed and implemented grove management strategies (e.g., Ardenwood Historical Farm, Lighthouse Field) or have added monarch groves in their general management plans (e.g., Vandenberg Air Force Base). Others are in the process of developing grove management plans for which funding has already been established (e.g., Ellwood Mesa Complex). At this time, grove management plans have been implemented by at least three overwintering sites and are currently being developed for at least seven more. An additional 37 overwintering sites are on public land that has a general management plan that specifically includes protections for monarch groves (IELP and Xerces Society 2012, entire). Management and restoration of these sites may include activities such as replacing dead trees, modifying canopy structure, planting fall- and winter-blooming shrubs as nectar sources, and addressing monarch predation issues (Jepsen et al. 2017, entire).

The USFWS developed the Monarch Conservation Database (MCD) to capture information about monarch conservation plans and efforts to inform the listing decision. As of June 1, 2020, there are 48,812 complete monarch conservation effort records in the MCD that have a status of completed, implemented, or planned since 2014, and 113 monarch conservation plans. These efforts constitute a total of 5,635,992 acres of land area in the continental United States and Hawaii (5,534,451 acres and 97,949 acres in the eastern and western populations, respectively) enhanced or created for monarchs, with the most common conservation effort being direct planting of milkweed and other nectar resources [note that these values include all completed, implemented, and not yet completed efforts; completed and implemented efforts to-date total 4,542,323 acres nationally].

Worldwide – Current Condition

Today, there are 30 extant populations and 1 presumed extant (Table 5.3, 5.4). The current health of these populations, however, is unknown, as there is insufficient information available (with the exception of the eastern and western North American populations, described above; Table 5.5).

Table 5.3. The current status (extant; unknown or presumed extant; or extirpated) of ACUs, populations, and countries/islands.

Status	# ACUs	# Pops	# Countries/ Islands	Definition
Extant	8	27	69	Observed since 2000
Unknown or Presumed Extant	0	4	21	Not observed since 2000, but lacking multi-year survey efforts
Extirpated	0	0	0	No observations despite multi-year survey efforts

Table 5.4. Current status of monarchs in 90 known countries, islands, or island groups occurrences and 31 populations worldwide. Status = presumed extant (P), known extant (E).

Population	Country/Island	Status	Population	Country/Island	Status
Australia (E)	Australia	E	Guam & Commonwealth of the Northern Mariana Islands [CNMI] (E)	CNMI	E
Caribbean (E)	Anguilla	E		Guam	E
	Antigua and Barbuda	P	Hawaii (E)	Hawaii	E
	Bahamas	E	Iberian Peninsula (E)	Azores	P
	Barbados	E		Canary Islands	E
	Bermuda	E		Gibraltar	E
	Bonaire	E		Madeira	E
	British Virgin Islands	P		Morocco	E
	Cayman Islands	P		Portugal	E
	Cuba	E		Spain	E
	Dominica	E	Johnston Atoll (E)	Johnston Atoll	E
	Dominican Republic	E	Kiribati (E)	Kiribati	E
	Grenada	E	Marquesas Islands (E)	Marquesas Islands	E
	Guadeloupe	E	Marshall Islands (E)	Marshall Islands	E
	Haiti	E	Mascarene Islands (E)	Mauritius	P
	Jamaica	E		Réunion	E
	Martinique	E	Micronesia (E)	Federated States of Micronesia	E
	Montserrat	P	Nauru (E)	Nauru	E
	Puerto Rico	E	New Zealand (E)	New Zealand	E
	Saba	E	Norfolk Island (E)	Norfolk Island	E
	Saint Barthélemy	P	Palau (E)	Palau	E
	Saint Kitts and Nevis	P	Papua New Guinea (E)	Papua New Guinea	E
	Saint Lucia	P	Philippines (P)	Philippines	P

Population	Country/Island	Status	Population	Country/Island	Status
	Saint Martin	E	Samoa (E)	American Samoa	P
	Saint Vincent & Grenadines	P		Samoa	E
	Sint Eustatius	E	South America and Aruba (E)	Aruba	E
	Sint Maarten	E		Colombia	E
	Turks and Caicos Islands	P		Curaçao	E
	US Virgin Islands	E		Ecuador	E
Central America (E)	Belize	E		French Guiana	P
	Costa Rica	E		Guyana	E
	El Salvador	E		Peru	E
	Guatemala	E		Suriname	P
	Honduras	E		Trinidad and Tobago	E
	Nicaragua	E		Venezuela	E
	Panama	E	South Florida (E)	South Florida*	E
Cook Islands (E)	Cook Islands	E	South Pacific (E)	Fiji	E
E. North America (E)	Canada (also part of the W. N. America population)	E		New Caledonia	E
	Mexico (also part of W. N. America and Central American populations)	E		Society Islands	E
	Saint Pierre & Miquelon	P		Solomon Islands	E
	E. United States	E		Vanuatu	E
Austral Islands (E)	Austral Islands	E	Tokelau (P)	Tokelau	P
Greater Indonesia (P)	Brunei	P	Tonga (E)	Tonga	E
	Indonesia	P	Tuvalu (E)	Tuvalu	E
	Malaysia	P	Wallis & Futuna (P)	Wallis and Futuna	P
	Timor-Leste	P	W. North America (E)	W. United States*	E

*Country that is listed multiple times, but not counted again (note that countries may be counted multiple times if they have distant islands; e.g., Hawaii is counted separately from the contiguous United States.)

Table 5.5. Population health: current status, past trend in population size (N), current status of milkweed & nectar resources, current status of insecticides, and overwintering habitat

ACU	Population	Status	Trend in N	MW/Nectar	Insecticides	OW Habitat	Overall Condition
Australia, New Zealand, and Indo-Pacific Islands	Australia	Extant	Unknown	Unknown	Unknown	Unknown	Unknown
	Cook Island	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	French Polynesia	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Greater Indonesia	Unknown	Unknown	Unknown	Unknown	N/A	Unknown
	Guam and CNMI	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Johnston Atoll	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Kiribati	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Marquesas Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Marshall Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Mascarene Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Micronesia	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Nauru	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	New Zealand	Extant	Unknown	Unknown	Unknown	Unknown	Unknown
	Norfolk Island	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Palau	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Papua New Guinea	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Philippines	Unknown	Unknown	Unknown	Unknown	N/A	Unknown
	Samoa	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	South Pacific Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Tokelau	Unknown	Unknown	Unknown	Unknown	N/A	Unknown
	Tonga	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Tuvalu	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Wallis & Futuna	Unknown	Unknown	Unknown	Unknown	N/A	Unknown
Central America, Caribbean	Caribbean	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Central America	Extant	Unknown	Unknown	Unknown	N/A	Unknown
Southern Florida	Florida	Extant	Unknown	Unknown	Unknown	N/A	Unknown

<table>
<tr><th>ACU</th><th>Population</th><th>Status</th><th>Trend in N</th><th>MW/Nectar</th><th>Insecticides</th><th>OW Habitat</th><th>Overall Condition</th></tr>
<tr><td>Hawaii</td><td>Hawaii</td><td>Extant</td><td>Unknown</td><td>Unknown</td><td>Unknown</td><td>N/A</td><td>Unknown</td></tr>
<tr><td>Iberian Peninsula</td><td>Iberian Peninsula</td><td>Extant</td><td>Unknown</td><td>Unknown</td><td>Unknown</td><td>N/A</td><td>Unknown</td></tr>
<tr><td>South America, Aruba</td><td>South America and Aruba</td><td>Extant</td><td>Unknown</td><td>Unknown</td><td>Unknown</td><td>N/A</td><td>Unknown</td></tr>
<tr><td>Eastern North America</td><td colspan="2">Eastern North America</td><td>Extant</td><td colspan="4"><i>See Eastern North American Population section below</i></td></tr>
<tr><td>Western North America</td><td colspan="2">Western North America</td><td>Extant</td><td colspan="4"><i>See Western North American Population section below</i></td></tr>
</table>

Worldwide – Current Influences

There is little to no information on the status and health for most of these populations, as well as information regarding positive or negative influences acting upon these populations. Below we discuss what little information is known or can be assumed.

There is limited information on predation, parasitism, and disease outside of eastern and western North American populations. Parasitism rates from Tachinid flies have been documented in Australia, Hawaii, throughout Central America, and Brazil. In Australia, the rates fluctuate throughout the year, ranging from very low to up to 100% of sampled monarchs in February (Smithers 1973, p. 38). Another parasitoid, the wasp *Pteromalus puparum*, is also known to attack monarch pupae in other locations (Ramsay 1964, p. 15). The protozoan parasite, *OE*, infects monarchs throughout Australia, Central and South America (Altizer et al. 2000, p. 135), and Hawaii (Pierce et al. 2014b, p. 1). Thus, given this limited information, we are unable to ascertain to what extent predation, parasitism, and disease impact worldwide monarch populations. Similarly, while data suggest global use of insecticides is increasing, we are unable to estimate the degree of overlap with monarch populations and thus derive a credible projection of impact on the worldwide monarch populations.

Chapter 6: Results –Future Influences and Catastrophic Events

This chapter describes our projections for the future states of the influences. To capture the uncertainty in our future projections, we identified both plausible optimistic and pessimistic changes for each influence. These optimistic and pessimistic states for each influence were then combined to create composite plausible "best case" and "worst case" scenarios. Additionally, we describe the events that are likely to be catastrophic should they occur.

North American Populations – Future Scenarios

To assess the future condition of monarch populations, we organized the key factors driving monarch population dynamics into 5 categories: 1) milkweed availability, 2) nectar availability, 3) migration nectar availability, 4) climate change effects, and 5) insecticide exposure. We then forecasted how each of these five influences is expected to change (i.e., its expected future state condition). We described the expected changes as the percent change from current state conditions (Figure 6.1 & 6.2). Lastly, we combined the most optimistic and pessimistic expected state conditions of each influence to form composite plausible best and plausible worst scenarios, respectively. The range of plausible future state conditions for each influence is described below and summarized in Table 6.1 (eastern population) and Table 6.2 (western population).

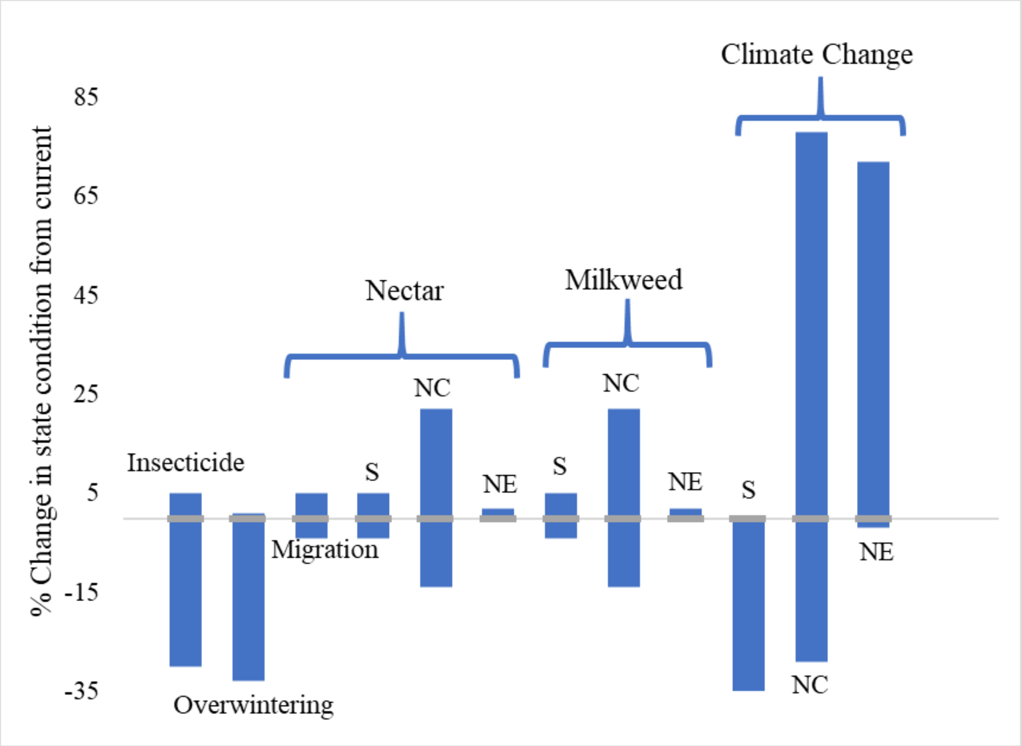

Figure 6.1. Range of forecasted % change from current state conditions for eastern population influences. Bars above and below the x-axis represent positive and negative changes, respectively, relative to monarch numbers. S, NC, and NE represent the Southern, Northcentral, and Northeastern subregions of the breeding range, respectively.

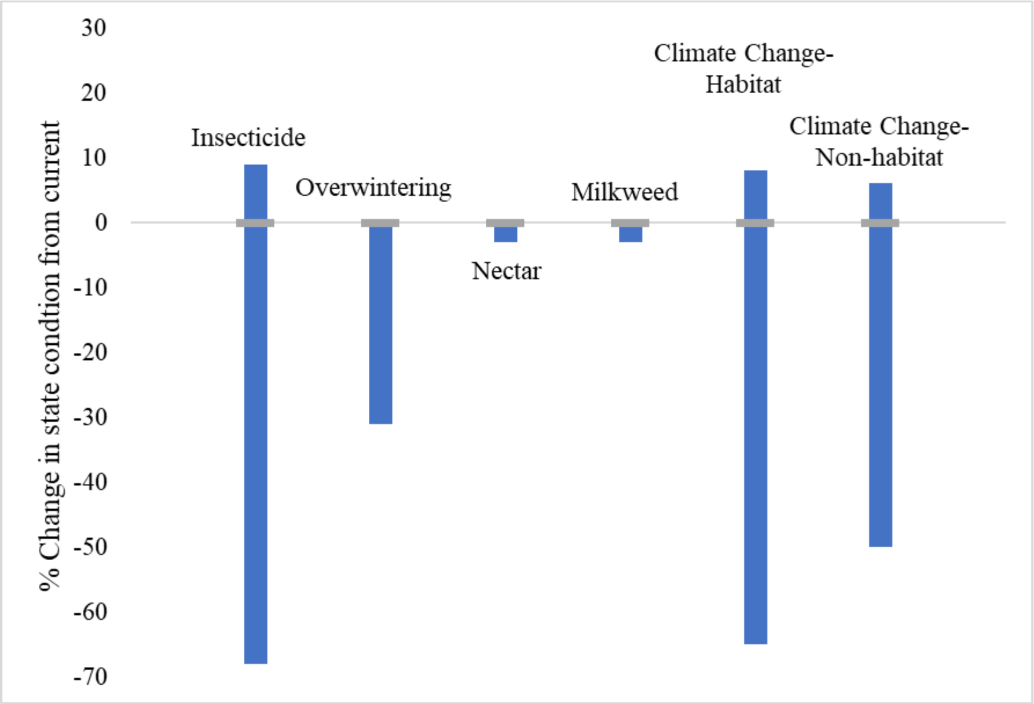

Figure 6.2. Range of forecasted % change from current state conditions for western population influences. Bars above and below the x-axis represent positive and negative changes, respectively, relative to monarch numbers.

Table 6.1. Description of the future state conditions for the influences for the eastern population. Time = the time period over which the change will occur. % Change = estimated % change in influence. NC = northcentral, NE = northeast, S = south.

Influence	**Time/% Change**	**Description**
Milkweed	18 years	
Best	NC: 22% increase NE: 3% increase S: 5% increase	Successful implementation of the Mid-America Monarch Conservation Strategy and other planned efforts, alongside gains of CRP habitat (22% increase in CRP acreage relative to 2018 levels), and a 2% milkweed stem gain driven by future land cover change, results in widespread habitat gains, primarily occurring in the North Core geography.
Worst	NC: 11% increase NE: 1% loss S: 6% loss	Successful implementation of the Mid-America Monarch Conservation Strategy and other planned efforts, occurring alongside losses of CRP habitat (35% decline compared to 2018 CRP levels), with no impact from future land cover change, results in modest habitat gains overall but variable by geography.
Nectar Resources	Same as Milkweed	*Same as Milkweed conditions*
Migration Nectar	18 years	
Best	S: 5% increase	*(Same as Milkweed "Best")*
Worst	S: 6% loss	*(Same as for Milkweed "Worst")*

Influence	Time/% Change	Description
Overwintering habitat	25 years	
Best	1% gain	Natural forest regeneration caused by reduced illegal logging and grazing pressures is projected to result in very slight gains of habitat over time.
Worst	33% loss	Losses of trees due to large-scale illegal logging and climatic factors are projected to continue at rates that have been observed in the recent past. This assumes that funding and programs implemented very recently are not sustained.
Climate change - Habitat	60 years	
Best	NC: 78% increase NE: 72% increase S: no change	Climate change drives increased habitat suitability and northward range expansion, up to a boundary of approximately 50°N latitude, resulting in widespread habitat increases throughout the eastern geography, particularly in Canada.
Worst	NC: 29% loss NE: 2% loss S: 83% loss	Climate change reduces overall habitat suitability across the current range; monarchs and milkweed do not effectively shift their range northward to track changing climatic conditions, resulting in habitat losses occur across the range, most notably in the southern geography.
Insecticides	25 years	
Best	5% decrease	Increasing attention for monarch conservation via MAFWA, CCAA, and MP3 plans, as well as increasing opportunities for VRT & newer equipment with the shift to larger farming operations.
Worst	30% increase	Increasing demand for food production leading to increases pest management; increasing trend in crop and disease-vector pests leading to aggressive insecticide response to prevent crop damage (e.g., soybean aphid) and disease outbreaks (e.g. Zika, West Nile).

Table 6.2. Description of the future state conditions for the influences on the western population. Time = the time period over which the change will occur. % Change = the % change estimated.

Influence	Time/% Change	Description
Milkweed	50 years	
Best	2% loss	Incorporates a low human growth scenario and conservation efforts implanted via the WAFWA plan and nonprofit groups.
Worst	3% loss	Incorporates a high human growth scenario and conservation efforts implanted via the WAFWA plan and nonprofit groups.
Nectar Resources		*Same as Milkweed conditions*
Overwintering habitat	50 years	

Influence	Time/% Change	Description
Best	18% loss	Projected losses of overwintering habitat are decreased from of those losses observed caused by urban development between 1990-1998 due to increased coastal development regulations and recent decreasing population growth rate in California. Conservation efforts implanted via the WAFWA plan are also included.
Worst	31% loss	Projected losses of overwintering habitat are consistent with those losses observed caused by urban development between 1990-1998 due to continued increasing population in California. Conservation efforts implanted via the WAFWA plan are also included.
Climate change – habitat	20 years	
Best	8% decrease	Increases in suitable climate niche due to projected increases temperatures.
Worst	65% increase	Losses of breeding and overwintering habitat due to projected increases in drought intensities & frequencies; the combined effect of dry spring conditions and warmer summer temperatures.
Climate change – non-habitat	20 years	
Best	6% decrease	Projected increases in *minimum* temperatures may expand the amount of time available for western monarch reproduction, thereby allowing for more generations per year to be produced and boosting monarch numbers.
Worst	50% increase	Reductions in reproduction and survival due to projected increases maximum daily temperatures, and hence, the number of days where temperatures exceed critical monarch thresholds.
Insecticides	20 years	
Best	9% decrease	Increasing attention for monarch conservation via WAFWA, CCAA, and MP-3 plans, as well as increased awareness of pollinator declines could lead to reduced and more targeted insecticide use.
Worst	68% increase	Increasing demand for food and projected land conversion from rangeland to agriculture; significant overlap of agricultural lands and the areas of most important to monarch production--CA Central Valley; and lack of standardize, broad-scale efforts and difficulty regulating use needed to reduce exposure

Availability, Distribution, and Quality of Milkweed

Eastern Population

Future scenarios for milkweed and nectar resources for the eastern population include a combination of 1) projected conservation effort, 2) projected changes in CRP acreage, and 3) other habitat change driven by projected land cover change. Scenarios are described in terms of percent change in "habitat" as indicated by milkweed stem estimates (with habitat assumed to consist of both milkweed and nectar resources, effectively co-occurring in a 1:1 ratio on average at broad scale), where percent change is reported relative to 2020 milkweed estimates, respectively for each subregion (Northcentral, Northeast, and South). "Baseline" (2020) habitat estimates were derived from the USGS "seamless" land cover spatial data (Rohweder and Thogmartin 2016; see Appendix 2 for additional methodological details), also including all completed and implemented efforts reported since 2014 via the national MCD.

For the eastern population, our future milkweed scenarios incorporated all not yet implemented (i.e., future) formalized conservation efforts reported to the MCD. For each subregion, the same level of formalized future conservation effort was projected for both the upper and lower bounds. For the Northcentral subregion, projected future formalized conservation effort associated with the Mid-America Monarch Conservation Strategy results in an additional 1.3 billion milkweed stems. We assumed conservation efforts occurring since 2014 effectively contribute to that goal. For our upper bound, we assumed achieving that goal would also include projected gains in CRP, meaning that any increase in CRP acres (in this case a 22% gain relative to 2018 levels; 156,485,213 stems) are not additive (beyond the 1.3 billion stem target) but rather are a contribution toward the overall target in the Northcentral subregion. For the lower bound in the Northcentral subregion, we assumed a similar level of effort would occur compared to the upper bound but with 35% less CRP contributions. Lacking any comparable overarching multi-state plan for much of the South and Northeast, we assumed CRP changes would be additive to future formalized conservation efforts in those subregions. For the conservation effort component of the eastern population future scenario, relative to 2020 levels, we projected an estimated 17% increase for the Northcentral, a 0.28% increase for the South, and a 0.03% increase in in milkweed/nectar in the Northeast subregions.

For CRP, we relied on USDA agricultural projections (USDA 2020), along with national CRP trend data and expert input from USDA-Farm Service Agency (Skip Hyberg, retired Senior Economist; personal communications). US Farm Bill programs are inherently difficult to predict, occurring at roughly 5-year legislative cycles and reflecting national and global economic and policy drivers that influence commodity prices and agricultural land values. We used current USDA projections (USDA 2020) for CRP to inform our upper bound, assuming that CRP increase under their stated assumptions could occur linearly over the next 18 years. Relative to 2018 CRP acreage, our upper bound scenario projected a 22% increase in CRP habitat and our lower bound scenario projected a 35% decline in CRP acres, respectively for each subregion. The lower bound CRP scenario was based on 10-year national CRP acreage declines (2008-2018). For purposes of milkweed stem estimates, future CRP losses/gains were assumed to change to/from cropland land cover.

For broader land cover change, we used the USGS FORE-SCE (Sohl et al. 2018) spatial data projections, which are informed by International Panel on Climate Change Special Report on Emissions Scenarios (IPCC 2000), to evaluate predicted milkweed stem change, respective to each subregion. Our scenarios account for land cover change occurring independent of conservation effort and CRP changes. Milkweed stem estimates, by land cover type, were based on a modified interpretation of Thogmartin et al. (2017c) where a subset of land cover types were lumped or split when necessary to align with the land cover classification scheme available in the FORE-SCE spatial data. We assumed land cover change would occur roughly linearly; therefore, we annualized the projected rate of change relative to the 2050 model output provided by FORE-SCE. For the land cover change component of our future scenarios, we estimated a 4% increase in milkweed stems in the Northcentral subregion, a 5% increase in the Northeast subregion, and a 4% increase in the South subregion over 40 years for the upper bound (primarily driven by urbanization trends). For the lower bound, we assume no habitat change due to projected land cover change.

When conservation effort, CRP, and land cover were considered holistically, overall projected changes in milkweed and nectar habitat range from a 11-22% increase in the Northcentral subregion, a 1% decrease to 3% increase in the Northeast subregion, and a 6% decrease to 5% increase in the South subregion (Table 6.1).

Western Population

The western population future state conditions are predicated upon projections of 1) human population growth rate in California and corresponding changes in landuse/cover and 2) conservation efforts throughout the West. California's Central Valley is an important production area for western monarchs (Crone et al. 2019, p. 10) and important migration pathway. Thus, the availability of milkweed or nectar resources in this area greatly influences the western population dynamics. Hence, we primarily relied on trends in California—and the Central Valley, in particular—to project the future state condition of milkweed and nectar availability. Loss of rangelands (an important land cover for monarchs) represented the largest land cover change in California's Central Valley, with a loss of approximately 1,054 km^2 (~260,450 acres) between 1980 and 2000 (Sleeter 2016). To project future trends, we used the results from Sleeter et al. (2017) analyses. They projected future land use change in California under three human population growth projections, and we chose the low and high human population growth scenarios to bound the range of plausible human population growth and the associated land use projections to estimate the change in monarch breeding habitat. The human growth projections were developed by the California Department of Finance (2019), which monitors human population growth trends at state and county scales. We believe that the methods used to develop these projections were scientifically rigorous, and thus, the scenarios represent the best available data and realistic projections of human population growth in California. In the low human population growth scenario, by the year 2070 approximately 2,600 km^2 will be converted from grassland or shrubland habitat to land use types that do not support monarchs. This represents a loss of 1.7% from the current amount of grassland and shrubland habitat currently available in California. In the high human population growth scenario, by the year 2070 approximately 5,300 km^2 will be converted from grassland or shrubland habitat to land use types that do not support monarchs. This represents a loss of 3.4% from the current amount of grassland and shrubland habitat currently available in California.

To forecast plausible future conservation efforts, we relied upon the WAFWA plan (2019, p. 39) and ongoing projects by nonprofit groups. Under the WAFWA plan, a minimum target of 202 km^2 of breeding habitat and adjacent foothills will be restored by 2029. The key drivers in realizing the plan's restoration goals are adequate funding and partner willingness. These issues are discussed within the plan and we agree with the rationale given for why these targets are plausible (WAFWA 2019, pp. 86-87). We also believe that additional conservation will be achieved by nonprofit groups and use information from the Xerces Society as proxy for estimating the quantity of habitat restored to project habitat restoration into the future. The Xerces Society has received funding to restore 2.65 km^2 of breeding habitat over the next five years and we use this value to project restoration by nonprofit groups over the next 50 years (an estimated total of 26.5 km^2). It is reasonable to expect similar levels of effort and funding for nonprofit groups to continue because supporting organizations such as the Monarch Joint Venture have shown that they are committed to furthering the conservation of the species in the West by funding these projects into the future. Thus, under both scenarios, we assumed 228 km^2 of habitat will be restored, yielding 2,384 km^2 (-2%) and 5,116 km^2 (-3%) for the best (low population growth) and worst (high population growth) case scenarios, respectively (Table 6.2).

Availability, Distribution, and Quality of Breeding Range Nectar Resources

Milkweed stem density is assumed to be a reasonable proxy for the availability, abundance, and phenological diversity of nectar resources. Monarch conservation best management practices generally tend to focus on producing more milkweed alongside diversified vegetation composition and structure, leading to more abundant and more diverse nectar resources that may be available for extended periods of the growing season (additionally, milkweed itself serves as a nectar source throughout a portion of the year). The ratio of milkweed:nectar outputs is largely unknown, is difficult to quantify, and likely varies by land cover, sector, conservation practice, geography, and climatic conditions. While some efforts may produce disproportionate changes in milkweed or nectar resources, 1:1 the relationship between nectar and milkweed is generally assumed to be correlated on average over broad spatial scales. As the mechanisms affecting the availability of nectar and milkweed are assumed to be the same, our future projections for them are proportionally the same as well.

Availability, Distribution, and Quality of Migration Nectar Resources

See the previous section "Nectar Resource Availability" for our rationale on why our southern milkweed scenario is a suitable proxy for nectar. As the mechanisms affecting the availability of nectar and milkweed are generally assumed to be the same, our future projections for them are proportionally the same as well.

Availability and Quality of Overwintering Habitat

Eastern Population

The future projections of the availability of overwintering habitat are largely predicated upon the analyses within Honey-Rosés et al. (2018), Vidal et al. (2014), and Flores-Martínez et al. (2019), the key findings of which are described below.

Under the best case scenario (1% increase, Table 6.1), we assumed that: 1) forest regeneration within the Monarch Butterfly Biosphere Reserve continues at the current rate (0.04% annually), and 2) the negative effects from illegal logging and climate change will lessen over time. Honey-Rosés and colleagues (2018) estimated 0.04% gains in reforestation annually due to natural forest regrowth and concerted replanting efforts. The current regeneration rate is driven largely by reduced logging and grazing pressures, a trend we can plausibly foresee continuing over two or more decades. We also assumed that this rate captures any loss of overwintering habitat (and regeneration outweighs these negative stressors, assuming that illegal logging will continue to decline as well and both oyamel fir trees and monarchs will adapt to the projected environmental conditions under climate change; see Sáenz-Romero et al. 2012).

Under the worst case scenario (33% decrease, Table 6.1), we assumed that: 1) illegal logging returns to rates observed prior to involvement and funding by stakeholders, and 2) the recent loss of habitat due to climatic factors continues. Vidal and colleagues (2014) observed a high percentage of loss due to illegal logging between 2001 and 2012 (2,179 hectares of core zone were impacted due to illegal logging over 11 years; Vidal et al. 2014). Flores-Martínez and colleagues (2019) observed the highest recently recorded rate of habitat loss due to climatic factors between 2012 and 2018 (125 hectares impacted due to climatic factors over 6 years) and we can foresee this trend continuing over two or more decades. Combined, these factors result in an annual loss of approximately 219 ha of overwintering habitat per year (5,473 ha by the year 2045). We assumed that the recent reductions in illegal logging (Flores-Martínez et al. 2019) do not continue or are no longer effective going forward, and thus, rates of illegal logging revert to levels previously observed (since 2000). This is plausible because many of these improvements rely on funding and programs offered by the government and outside entities; if they can no longer be funded, then both large- and small-scale logging operations are expected to resume (Flores-Martínez et al. 2019, p. 7).

Western Population

The future projections of the availability of overwintering habitat are predicated upon: 1) forecasts of urban development and associated monarch habitat loss along coastal California and 2) conservation efforts under full implementation of the WAFWA plan. There is a strong interest by the State and conservation groups to protect and manage key monarch overwintering sites, and thus, under both scenarios, we assume that the actions proposed by WAFWA and conservation groups will be fully implemented. Under the WAFWA plan, 50% of all known overwintering sites will be protected and managed for monarchs by 2029 (WAFWA 2019, p. 35). It is reasonable to expect the WAFWA plan to be fully implemented because the plan outlines the steps required and identifies the key players (WAFWA 2019, pp. 87-88) and the State of California continues to further legislation designed to support implementation of the plan (State of California 2018).

Although the current rate of monarch overwintering habitat loss is unknown, rate of loss from 1990 to 1998 (due primarily to urban development) was 12% (Griffths and Villablanca 2015, entire). The threat of urban development in coastal California remains. Given continued increases in the human population (California Department of Finance 2019), we expect loss of overwintering sites due to urban development to continue. However, we can foresee the rate of habitat loss decreasing because California's population growth rate has been below 1.0 percent since 2005, with the 2019 growth rate being the lowest since 1900 (California Department of Finance 2019). Given this, we can foresee a reduction from the rate of overwintering habitat reported by Griffiths and Villablanca (2015) to 6% loss every 9 years, which is half of the rate observed in the 1990s. Under this foreseeable best case scenario, considering protection and maintenance of 50% of the overwintering sites starting in 2029 and a decreased rate of habitat loss at the remaining 50% of sites, we estimate a total loss of 18% of habitat over the next 50 years (Table 6.2). Under the foreseeable worst case, considering protection and maintenance of 50% of the overwintering sites starting in 2029 and continued loss of habitat at the observed rate (Griffiths and Villablanca 2015), we estimate a total loss of 31% over the next 50 years (Table 6.2).

Climate Change Effects

Eastern Population

Our future scenarios for habitat related climate changes were derived primarily from the model results of Lemoine (2015, entire). For the best case climate change scenario, suitable habitat increases by 78% in the Northcentral subregion, increases by 72% in the northeast subregion, and has no gain or loss in the southern subregion. This was based on the slightly modified monarch and milkweed ecological niches as modeled by Lemoine (2015), using the moderate B2 emission scenario. While Lemoine (2015) found an overall increase in suitable breeding habitat for eastern monarchs, we assume that this increase will ultimately be constrained by the current northern extent of the monarch's range (approximately 50°N). This is reasonable to expect because while there could be some northward expansion in suitable habitat driven by climate change, there are simultaneous factors that limit the degree to which milkweed and monarchs will be able to fully realize a northward range expansion (particularly in terms of population-level outcomes). First, northern expansion of milkweed is expected to lag behind changing climatic conditions, both because of the time it takes the species to colonize large, new areas and because of other potential differences in suitable habitat (e.g., different soil types or competing vegetation). Second, monarchs are mobile, but northward expansion might also be limited for physiological reasons (e.g., lack of directional flight after certain dates, insufficient energetic resources, etc.; Taylor, pers. comm. 2020). Third, even if monarchs and milkweed were able to effectively colonize beyond their current northern limit (~50°N), these monarchs would not be able to successfully migrate such a long distance to Mexico, as evidenced by the limited tag returns from similarly far away areas in the north and northeast (Taylor, pers. comm. 2020). Furthermore, those monarchs that did successfully make the extended journey to the overwintering grounds might subsequently have lowered fecundity due to the increased energetic constraints relative to monarchs that migrated from more optimal core breeding grounds. Thus, we assumed future range expansion will be limited to 50°N latitude.

Under the same moderate emissions scenario, Lemoine (2015) estimates that the southern subregion of the current eastern population breeding range will have a loss of the southernmost portion of the range but backfilling in the more northern part of the southern subregion. Overall, there was more backfilling than loss of southern habitat (for a potential 34% increase); however, this does not account for the importance of the southern portion of the breeding range, particularly for migratory demographic connectivity (Flockhart et al. 2015, p. 5). Thus, for this likely best case scenario, we took a moderate approach and assumed neither an increase nor decrease in the suitable habitat in the southern subregion (Table 6.1).

For the worst case scenario, we used Lemoine's more severe modeled climate change scenario (A2 emissions scenario), but again we constrained monarch expansion to 50°N latitude. Under this climate change scenario, habitat losses will occur in all 3 subregions: 29% loss in the Northcentral subregion, 2% loss in the northeast sub-region, and 83% loss in the southern subregion (Table 6.1).

Western Population

We relied upon expert predictions and other information to quantify the change in impacts from climate change to western monarchs over the next 20 years. We elicited the current and predicted future influence of non-habitat and habitat mediated effects of climate change on monarch numbers (Voorhies et al., 2019, Suppl. 2).

Non-habitat mediated climate change effects

The median (across experts) predicted percent change in influence from the current condition ranged from a 6% decrease in impact to a 50% increase in impact over the next 20 years (Table 6.2). The key underlying premise for the experts' predicted lessening impact from climate change effects is predicated upon recent findings suggesting increases in temperatures could improve reproduction. Svancara et al. (2019), for example, found that the projected increases in minimum temperatures in Idaho will expand the amount of time available for western monarch reproduction (by a half to a full month), thereby allowing for more generations per year to be produced and boosting monarch numbers.

The key underlying premises for predicting increasing impact from climate change include increasing maximum daily temperatures and severe precipitation events. Increasing temperatures–extremes and nighttime temperatures–can hinder reproduction and lead to increased mortality when temperatures exceed critical thresholds (38ºC and 42ºC, respectively; see the climate change section under Influences above). Projected changes in climate show continued and accelerated increases in temperature across the western U.S. through the twenty-first century (Sillmann et al. 2013, entire). In California, for example, statewide warming of 2-4°C (RCP 4.5) to 4-7°C (RCP 8.5) is projected by the end of the century (Pierce et al. 2018, pp. iv, 17-18); extreme temperature events are predicted to increase as well (Pierce et al. 2018, p. 22-28; see also Climate Change discussion under the Current Influences section above).

The experts also forecasted increased mortality from increasing intensity of strong precipitation events at overwintering sites. Unlike the temperature projections, regional changes in precipitation are more variable among global climate models (Kharin et al. 2013, entire). However, climate models generally project an increase in extreme precipitation events in California, including the overwintering coastal areas for monarch (Pierce et al. 2018, p. 26; Swain et al. 2018, entire).

We believe the experts' projections are supportable given the climate change projections available and the knowledge on monarch critical temperature thresholds. Under the best case scenario, the experts assumed that with projected increases in temperature, the number of generations and thus number of monarchs will increase and the number of days where the maximum temperatures exceeds critical thresholds will not increase. Under the worst case scenario, the experts forecasted increased mortality and reductions in reproduction given projected increases in maximum temperatures and the intensity of "most intense" precipitation events at overwintering sites.

Habitat-mediated climate change effects

The median (across experts) predicted percent change in influence from current condition ranged from an 8% decrease to a 65% increase over the next 20 years (Table 6.2). The experts' predictions are predicated upon anticipated changes in: 1) drought frequencies and severities, 2) the suitability of monarch overwintering habitats along coastal California, and 3) the suitability of monarch breeding habitat throughout the West.

The experts' prediction for a reduction in impact is predicated upon recent analyses that show monarch distribution being largely a function of milkweed occurrence (Dilts et al. 2019, p. 6; Lemoine 2015, p. 11; Svancara et al. 2019, p. 14), and with increasing temperatures, the area of suitable climate niche may expand (Svancara et al. 2019, p. 15).

The experts' prediction of an increasing impact is predicated on increasing drought intensities and or frequencies, which will reduce milkweed and nectar plant availability throughout the West. Stevens and Frey (2010, entire) found moisture regime acts as a strong bottom-up driver of monarch abundance patterns via resource availability in the West. Drought indices for California, Idaho, Nevada, and Oregon (but not Arizona, Utah, or Washington) were each significantly associated with monarch wintering abundance patterns, with California exhibiting the strongest relationship. Variation in moisture availability within a block of three contiguous central California climate divisions (Sacramento Drainage, San Joaquin Drainage, and Southeast Desert Basin) significantly predicted inter-annual abundance of migrant generation monarchs. Similarly, Espeset et al. (2016, p. 824, 826) found a positive effect of precipitation and western monarch numbers at focal sites. These findings suggest that precipitation may be a limiting factor and thus increased drought—frequency or intensity—will negatively affect western monarchs.

Even though annual precipitation changes due to climate change are predicted to be modest, year-to-year variability is predicted to increase due to the wetter winter conditions and drier spring conditions in California (Pierce et al. 2018, p. 27). The overall result is an increase in the frequency of dry years due to fewer wet days, but more precipitation on wet days (Pierce et al. 2018, p. 27). In addition, maximum July temperatures are expected to increase and heat waves may span longer durations (Pierce et al. 2013, entire). This could lead to increased evapotranspiration (Diffenbaugh et al. 2015, p. 3994) and a greater likelihood of monarch habitats drying, both inland breeding and coastal overwintering (Pierce et al. 2018, p. 25). The combined effect of dry spring conditions and warmer summer temperatures would reduce the amount of milkweed and nectar resources across the landscape available for nectaring and egg-laying, particularly in the early part of the year when western monarchs are migrating away from the overwintering sites to produce the first generation. These overwintering monarchs have low energy reserves and lack the flexibility to continue moving if resources are not immediately available. Thus, they may die before finding suitable breeding habitat.

The experts indicated that severe drought can cause overwintering tree loss and degradation, decreasing the availability and quality of roosting habitat for monarch butterflies in the West (Pelton et al. 2016, p. 29). Many groves are dominated by one or a few tree species, especially blue gum eucalyptus, which are not native to California and are considered drought sensitive (Marcar et al. 1995, p. 46). Drought-stressed eucalyptus trees are vulnerable to infestation by insect borers, exacerbating tree loss in these groves (Paine and Millar 2002, p. 148), thereby reducing roosting habitat and wind protection. Stressed blue gum eucalyptus may also cease flowering, eliminating the main source of nectar available to monarchs during the overwintering season at some sites. Other dominant trees, such as Monterey pines and Monterey cypress, are more resistant to drought, but these species are the primary species in fewer than 25% of groves.

Furthermore, Fisher et al. (2018, entire) modeled the future location of western monarch overwintering habitat under climate change scenarios in Santa Barbara County, California. They found a substantial shift in predicted overwintering habitat distribution. Monarchs currently overwinter along the coast to take advantage of the mild winter temperatures (Leong 1990, p. 906; Weiss et al. 1991, p. 173), and if temperatures in California are predicted to rise through the year 2100, then similarly cool temperatures, and overwintering monarchs, should be found at higher elevations later this century. Under a plausible scenario (RCP 4.5), the probability of occurrence of overwintering habitat directly reflects elevation, with coastal regions having a reduced probability relative to today, and higher elevation sites increasing in probability. Under a more extreme scenario (RCP 6.0), high probability sites are located only along ridgelines and in mountaintop regions of the county.

We believe the experts' projections are reasonable given (1) there may be small increases in milkweed availability in some portions of the range, and (2) greater losses of monarch habitat from increased temperatures and drought.

Insecticide Exposure

We relied upon expert judgments to quantify the change in insecticide impact, i.e., the expected change in the insecticides state conditions and monarch response for the eastern and western populations (see Voorhies et al. 2019, Supplemental 2). Using the experts' estimates and other information, we devised future projections for the percent change in impact to monarchs. We briefly describe key underlying premises and supporting evidence here; see Insecticide Supplemental for further detail.

Eastern Population

The expert-elicited projected future percent change in the magnitude of impact (monarch population-level response) is a 5% decrease to 30% increase over the next 25 years (Table 6.1). The expert's range is predicated upon the three key premises: 1) there will be no change due to changes in the insecticide doses applied to kill insect pests that reduce crop yields, land use patterns, residential practices, or monarch use of milkweed across the various land uses, 2) there will be a small decrease due to changes in farming practices, and 3) there will be small to high increases in impacts due to additional applications of insecticides because of new agricultural pests that threaten crop yields, new human health threats, and increased vigor of insect pests.

Insecticides are used across a diversity of sectors, with agriculture being the largest source of insecticide exposure for the eastern monarch population (the agriculture comprises 30% of land use within eastern monarch population range and 60% of insecticide use nationwide). The Food and Agriculture Organization of the United Nations (FAO) estimates that a 50% increase in food production by 2050 is needed to meet the demand of the growing human population (FAO 2017, p. 46). In response, corn and soybean production is projected to increase by 16% and 33%, respectively, over the next 10 years (USDA 2020, p. 30, Table 5; p. 35, Table 10). Because only nominal increases in agricultural land expansion is expected in the eastern U.S. (USDA 2020, p. 29; see Milkweed & Nectar Resources section above), this demand will be met primarily through increased yields. Crop production can be greatly diminished by pests. Crop and forest production losses from invasive insects and pathogens in the U.S., for example, have been estimated at $40 billion/year (Paini et al. 2016, p. 7575); similarly, corn and soybean yield losses from pests are estimated to be 54% and 46%, respectively (USDA 2014, p.7). Thus, it is reasonable to foresee efforts to control insect pests intensifying over the next 30 years to meet the increasing demand for food. Additionally, increasing insecticide use among other sectors (e.g., homeowners, forestry, vector control districts) beyond agriculture is expected as well. The number of insect-borne diseases in the U.S., for example, tripled from 2004 to 2016 (CDC 2018), and the causes (e.g., land use changes, increasing transcontinental movements, warming climate) underlying these trends are accelerating (Bradshaw et al. 2016, p. 4-5, FAO 2017, p. 56, 58; Petersen et al. 2016, p. 280).

Moreover, a warming climate is expected to exacerbate insect-borne diseases and pest burden via: 1) improved overwintering survival and faster development and hence increased pest population growth, 2) increased number of generations per season, 3) earlier arrival of migratory pests, and 4) expanding suitable climate envelopes leading to novel pest outbreaks (Caminade et al. 2019, p. 158; Sangle et al. 2015, p.3581; Sharma and Prabhakar 2014, p. 25). Deutsch et al. (2018, p. 918, figure 3) projected, for example, 18% and 32% increase in wheat and corn losses due to insect pests, respectively, with 2°C rise in global temperatures. Although the response of insect pests to climate change will vary, the preponderance of evidence suggests that warmer temperatures in temperate climates will yield more types and higher populations of insect pests and pathogens (Sangle et al. 2015, p. 3580, Wolfe et al. 2008, p. 568). These data indicate an increasing impact from escalating insecticide use into the future.

Some of this increased impact will be mitigated through efforts (e.g., MAFWA, MP3, Rights-of-ways CCAA) to reduce monarch exposure by promoting monarch-specific conservation efforts and increased awareness of the potential harm of insecticides to pollinators, in general. Additionally, the trend towards larger farming operations—which have the capital and capacity to more fully integrate newer technology such as variable rate technology (VRT) and upgrade to newest equipment—may also reduce the monarch's exposure to insecticides. This reduction, however, is likely to be modest as small and mid-size farms still represent a large fraction of acres farmed (e.g., based on a nationwide sample [n=19,600] in 2015, 71% of land was operated by small and mid-size farms; USDA 2016, p. 4).

Given the demand for increasing crop yields and the continued increasing trend in insect pests and insect-borne diseases, increases in insecticide use is foreseeable. Conservation efforts, via reduced exposure potential, are likely to prevent the full impact of these increases from occurring. Thus, we believe the expert's 5% decrease to 30% increase represents a plausible projection of insecticide impacts on the eastern population over the next 25 years.

Western Population

The expert-elicited projected future percent change in the magnitude of impact (monarch population-level response) is a 9% decrease to 68% increase over the next 20 years (Table 6.2). The experts' range is predicated upon the three key premises: 1) areas with high insecticide use overlap significantly with areas most important to monarch production—California's Central Valley, eastern Washington, southern Idaho; 2) the trend in land conversion from rangeland to agriculture will lead to an increasing demand for insecticides by the agricultural sector, and 3) despite California having the strongest pesticide registration in the country, ability to regulate exposure is difficult.

Insecticide use is widespread across the most important breeding areas (Figure 6.4) for the western monarch, and it has been implicated as one of the key drivers in the decline of the western monarch population (Crone et al. 2019, p. 10; Forister et al. 2016, entire; Halsch et al. 2020, entire). Based on volume alone, exposure to insecticides is greatest on or near agricultural lands. Between 2005 and 2012, the agricultural sector, for example, accounted for 60% of insecticide use (USEPA 2017, p. 11). Given the overlap of agriculture and monarch breeding areas, the trend in insecticide use on agriculture greatly influences monarch exposure to insecticides. The increasing demand for food production is expected to expand trade for all the projected agricultural commodities (USDA 2020, p. 55). California is the leading U.S. state in cash farm receipts, and its agricultural production includes more than 400 commodities representing over a third of the United States' vegetables and two-thirds of the country's fruits and nuts (California's Managed Pollinator Protection Plan (MP3); CDPR 2018, p. 1) and ~15% of U.S. agricultural exports for 2017. In the western U.S., this demand for food will be met by expanding agricultural lands (Sleeter et al. 2017) and through increased yields (Popp et al. 2013, p.253), both of which will increase insecticide use in the western U.S.

In addition, insecticide exposure is occurring across a wide variety of land use sectors. A study in the central valley of California, for example, detected pesticides in all land use types (Halsch et al. 2020, p. 13). Insecticides are used by: homeowners to control pests in yards and gardens or planting neonicotinoid-treated ornamentals from garden centers; municipalities to control mosquito populations (WAFWA 2019, p. 16) to prevent the spread of infectious diseases (i.e., West Nile virus, Zika virus); and federal, state, and private entities to control pest irruptions on rangelands (WAFWA 2019, p. 16). These data indicate an increasing impact in the future due to increasing use of and exposure to insecticides.

We expect that some of this impact will be mitigated through efforts (e.g., WAFWA, MP3, Rights-of-ways CCAA) to reduce monarch exposure by promoting monarch-specific conservation efforts and increasing awareness of the potential harm of insecticides to pollinators, in general. The WAFWA plan, for example, points to monarch-specific BMPs and training for all sectors. Additionally, the states of California and Washington have MP3 plans in place and Idaho has a similar plan. The purpose of these plans is to mitigate the pesticide risk to bees, but in doing so, can also lead to reduced monarch insecticide exposure.

Given the increasing demand for agricultural products and the substantial overlap of agricultural lands with important monarch breeding areas, increases in insecticide use or toxicity are foreseeable. Conservation efforts, via reduced exposure potential, are likely to prevent the full impact of these increases from occurring. Thus, we believe the experts' 9% decrease to 68% increase represents a plausible projection of insecticide impacts on the western population over the next 20 years.

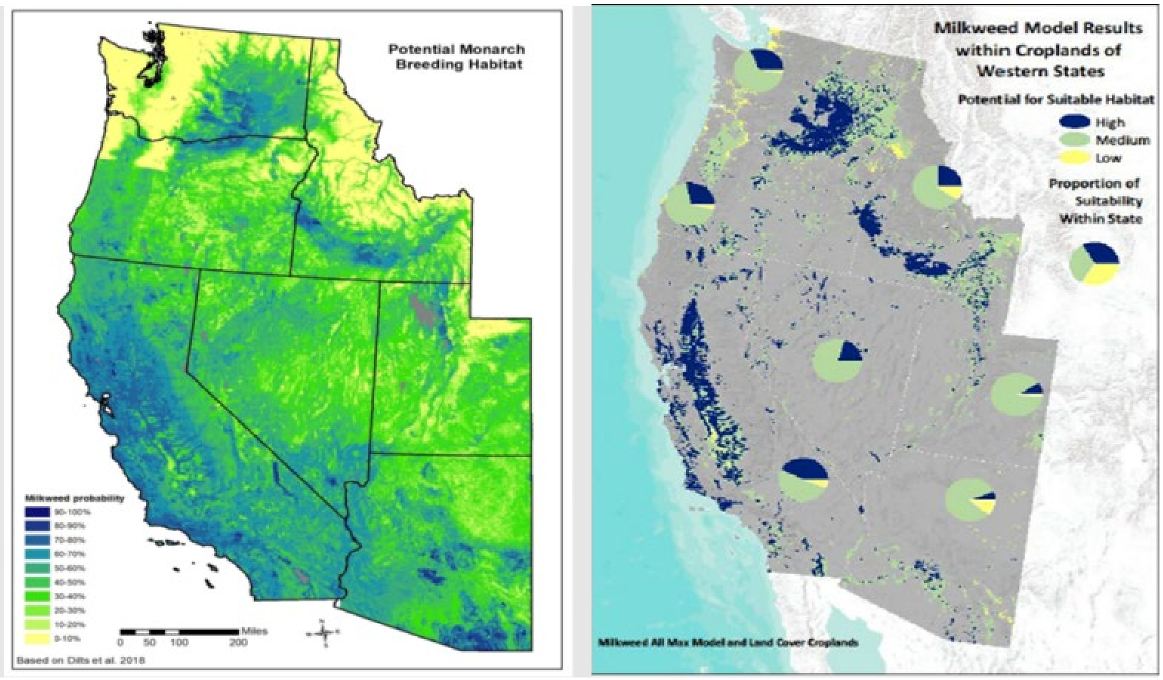

Figure 6.4. A Predicted distribution of milkweed and thus extent of potential monarch breeding areas—derived from a habitat suitability model (Dilts et al. 2018). B. Suitable habitat—milkweed potential—overlaid with croplands in western U.S. (WAFWA 2019, Fig. 6, p. 15). Dark blue spots correlate with the important for breeding areas -- the Central Valley, Columbia River, and Snake River Plain.

North American Populations – Catastrophic Events

We defined catastrophic event as an event that is expected to extirpate the population should the event occur. We evaluated several potential events to determine if they were of sufficient magnitude and severity to cause a population collapse. Below, we describe the events that are likely to be catastrophic should they occur.

Eastern North American Population

We assessed the following events for their potential to cause catastrophic losses: overwintering storms, widespread drought, fire, habitat loss, broad-scale insecticide spray events, and monarch disease and predation. Of these, we determined that two—extreme storm events and widespread drought— have sufficient magnitude (scope) and severity (causing population collapse) potential to pose a catastrophic risk to the eastern population.

Extreme Storm Mortality

Storms during the annual cycle can cause high levels of mortality when monarchs are congregated (during migration and at the overwintering grounds). During migration, storms could be catastrophic if they occurred in areas where monarchs are funneled together (e.g., Texas, where the eastern migratory population funnels through in the spring and fall). However, after an extensive literature search, we found only a few documented incidences of storm mortality during migration (but see Howard and Davis 2012, entire). Moreover, although large numbers of monarchs funnel through at the same time, it is unlikely that storms will cover the relatively large area occupied at any time during migration and thus, not likely to rise to the level of causing population-level losses. Given this, we have insufficient information that the magnitude and severity of storms during migration pose a catastrophic risk.

There is, however, well-documented mortality events at the Mexican overwintering sites from storms (e.g., mortality upwards of 80% has been documented [Brower et al. 2004, p. 158]). Monarchs are particularly sensitive to storms in Mexico because once wetted, monarchs freeze at a warmer temperature (approximately -4°C for wet butterflies, compared to -8°C for dry; Larsen and Lee 1994). Monarch freezing mortality from storms at overwintering sites has been documented during the winters of 1980-1981, 1995-1996, 1999-2000, 2001-2002, 2003-2004, 2009-2010, and 2015-2016 (Oberhauser and Peterson 2003, p. 14063, Brower et al. 2005, p. 970, Fink et al. in prep). Given the potential severity and the high magnitude across the relatively small overwintering sites, extreme storm events pose a catastrophic risk for the eastern monarch population. A previous model shows a potential increase in precipitation events in the winter (Oberhauser and Peterson 2003, p. 14066-14067). However, other modeling efforts show a potential decline in freezing storm events due to warming temperatures (Flockhart et al. 2015, p. 160). Additionally, with logging and climate change negatively impacting the oyamel overwintering forests, freezing events may be more likely and more severe because of the loss of the protective effects of an intact forest (Williams and Brower 2015, entire). When combined with a decreasing population size, there is a higher risk that extreme storms of magnitudes similar to previously documented storms would now be catastrophic.

Widespread Drought

Monarchs can be affected by drought at multiple points during their migratory cycle, including during the breeding season as both larvae and adults, and as adults nectaring along their migratory route (nectar can be converted to stored lipids for use while overwintering; Brower et al. 2015). Water availability can affect both milkweed quality and milkweed and nectar availability (Brower et al. 2015, Couture et al. 2015; see also Widespread Drought section under the Western North American discussion below). Given the expansive breeding ground, drought events are unlikely to affect a large enough area to evoke a population level response, and hence not likely to pose a catastrophic risk to the eastern breeding population.

Eastern migratory monarchs funnel through Texas and Mexico in the fall, where it is imperative that they consume enough nectar to be converted to lipids and used as needed throughout their overwintering period (when nectar resources are scarce; Brower et al. 2015). Brower and colleagues (2015) found that monarchs in Texas nectaring on wildflowers during a drought had lowered lipids (compared to monarchs nectaring on flowers from an irrigated garden at the same time). However, they also found that monarchs arriving at Mexican overwintering sites that same year had higher lipid reserves, suggesting that non-drought areas in Mexico may provide sufficient nectar even when Texas is in a drought. This area is also important in the spring, as monarchs funnel through this same area and rely on milkweed and nectar sources as they lay the first generation of the new year. Thus, monarchs in the spring could be similarly impacted by drought. Given the above, it is possible that drought conditions in Texas or Mexico pose a catastrophic risk for the eastern monarch population.

<u>Western North American Population</u>

We assessed the following events for their potential to catastrophic losses: widespread drought, wildfire, extreme overwintering storm events, and co-occurrence of poor environmental conditions and low population numbers. Of these, we determined that two—widespread drought and co-occurrence of poor environmental conditions and low population abundance—have sufficient magnitude (scope) and severity (causing population collapse) potential to pose a catastrophic risk to the western population.

Widespread Drought

Severity and intensity of drought have been suggested as a major driver of monarch populations in the West (Stevens and Frey 2010, p. 740). Severe drought affects both milkweed and nectar resources, and overwintering habitat resources. The frequency of years with precipitation "much below normal" in California and Nevada has increased from 1910 to current (Figure 6.6) and are predicted to increase with climate change (Diffenbaugh et al. 2015, p. 3934; Williams et al. 2015, p. 6826; Cook et al. 2015, p. 6). Under climate change projections, wetter winter conditions and drier spring conditions will lead to greater year-to-year precipitation variability and an overall increase in the frequency of dry years due to fewer wet days (Pierce et al. 2018, p. 27). Additionally, the forecasted higher maximum July temperatures and increased duration of heat waves (Pierce et al. 2013, entire) is likely to increase evapotranspiration (Diffenbaugh et al. 2015, p. 3994) and drying of monarch habitats, especially along the central and southern California coast (Pierce et al. 2013, p. 843).

If the tolerance threshold of milkweed and nectar resources to consecutive years of drought is reached, this could result in catastrophic breeding and migratory habitat degradation and loss. A decrease in nectar resources could result in starvation and reduced reproductive output of adults. Milkweed with limited water availability can have more viscous latex, which has been shown to negatively influence larval performance (Bell 1998, p. 133). A decrease in milkweed resources may leave monarchs with fewer resources on which to feed and lay their eggs, resulting in decreased recruitment for the population. However, the majority of milkweeds are deciduous perennials that have adapted to seasonal dry conditions (Borders et al. 2013, p. 7). A mild drought or one that was limited in extent or duration would likely reduce the availability of milkweed to breeding individuals, but the effects to the overall distribution of milkweeds would be short-term. Though a single year of drought could cause fecundity to decline sharply, only a drought that was severe, widespread, and sustained would be catastrophic for a population of monarch butterflies. The breeding ground is widespread for the western population, but large-scale drought could be as equally as widespread (Williams et al. 2020, entire), such that it could occur throughout most of the breeding grounds. Given the above, extreme drought affecting milkweed and nectar resources poses a catastrophic risk for the western monarch population. When combined with a decreasing population abundance, there is a higher risk that drought would be catastrophic.

Severe drought can also cause tree loss and degradation, decreasing the availability and quality of overwintering roosting habitat (Pelton et al. 2016, p. 29). Many groves are dominated by one or a few tree species; one of the most prevalent—blue gum eucalyptus—is drought sensitive (Marcar et al. 1995, p. 46). Drought-stressed eucalyptus trees are vulnerable to infestation by insect borers, which can exacerbate tree loss in these groves (Paine and Millar 2002, p. 148). Eucalyptus loss and degradation reduces availability of roosting habitat, lessens wind protection, and eliminates the primary overwintering source of nectar at many sites. Other dominant trees, such as Monterey pines and Monterey cypress, are more resistant to drought, but are the primary species in fewer than 25% of overwintering sites. Although overwintering grounds are widespread, drought could be equally as widespread, such that it could occur throughout many or most of the overwintering sites simultaneously. Given the above, extreme drought at overwintering sites poses a catastrophic risk for the western monarch population.

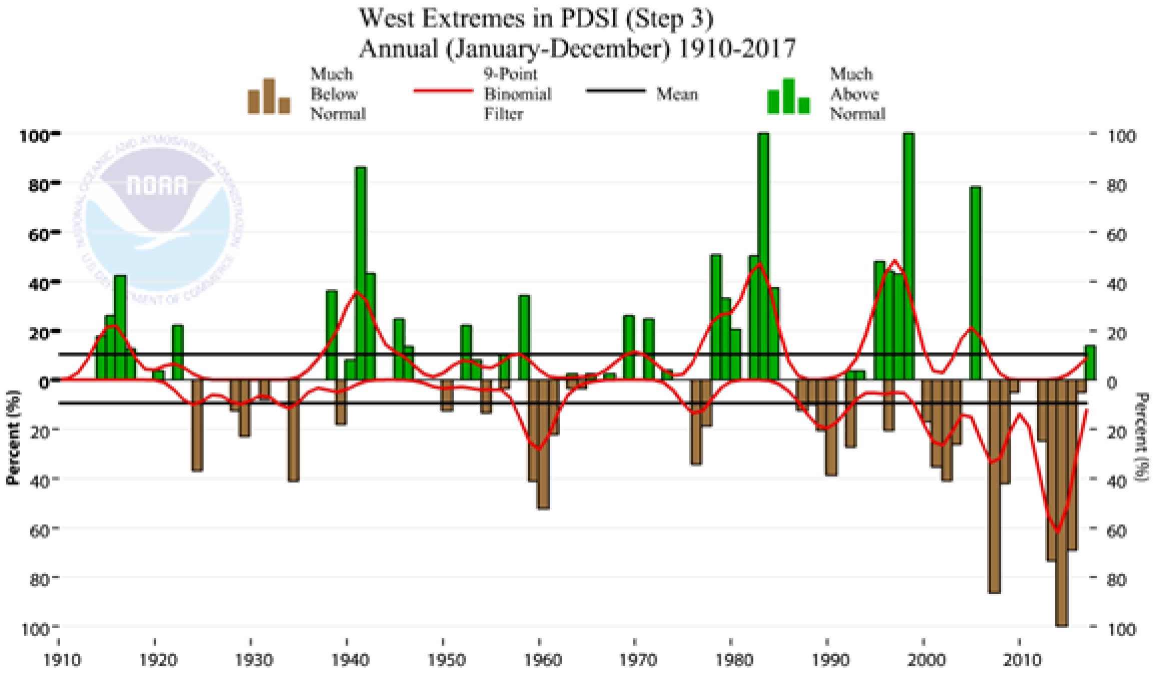

Figure 6.6. Extremes in the Palmer Severity Drought Index (PDSI) for the western U.S. (i.e., California and Nevada). Figure from the National Oceanic and Atmospheric Administration (NOAA 2018).

Co-occurrence of Poor Environmental Conditions with Low Abundance

If the large population fluctuations that were observed in the 1990s (presumably due to poor environmental conditions) were to occur when the population abundance is low (as it has been in recent years), extinction of the western North American population is likely. Given that environmental variability, and thus large swings in abundance, will increase with a changing climate (Pierce et al. 2018, entire) and given that the population has remained at lowest ever abundances for the last 2 years, co-occurrence of poor environmental conditions and low population abundances numbers poses a catastrophic risk for the western monarch population.

Worldwide – Future Scenarios & Catastrophic Events

Due to a lack of information on current influences, we were unable to forecast future scenarios for these populations.

We identified, however, two potential catastrophic events—both of which are climate change effects: sea level rise and lethal high temperatures. To forecast future changes in temperature and sea levels, we relied upon the Third Assessment Report developed by the International Panel on Climate Change (IPCC) to identify the low-lying islands that are at risk of permanent inundation and used the maximum elevation of those islands to develop thresholds for the risk classifications. To forecast changes in daily temperatures, we used downscaled General Circulation Model under RCP scenarios 4.5 and 8.5 obtained from the Earth System Grid Federation (CORDEX 2018; Cinquini 2014). Using these data, we assessed where daily maximum surface temperatures would exceed 42°C (a temperature threshold that leads to significantly reduced monarch larvae survival; Nail et al. 2015b, p. 99) by the year 2069 (see Appendix 2 Methods – Climate change projections for further details).

Sea Level Rise

Several low-lying islands in the pacific region are at risk of permanent inundation according to the Third Assessment Report from the IPCC (IPCC 2001). Many of these low-lying islands are inhabited by monarch butterflies. Additionally, many of these islands are remote and represent an entire population of monarchs. A mix of elevations occurs on these islands. We assumed that monarch populations on islands with higher elevations are at a lower risk level. However, we do not have any data on the population size or extent of habitat on these islands.

Unsuitably High Temperatures

In addition to sea level rise, temperatures are expected to increase throughout parts of the monarch's range (IPCC 2001). While monarchs can tolerate a range of thermal conditions, there are known upper limits (Nail et al. 2015b). Therefore, we also examined future predicted temperatures throughout the global range of monarchs, presuming that areas exceeding these lethally high thermal thresholds would have catastrophic losses of monarchs.

Chapter 7: Results – Analysis of Future Condition

This chapter describes the forecasted health of monarch populations over time. We first describe the results from our analysis of direct effects from high temperatures due to climate change. Next, we provide the forecasted health of the North American populations given the best and worst case scenarios. Lastly, we provide the results of the catastrophic events analysis for the worldwide populations.

Eastern North American Population – Future Condition

Under both best and worst case scenarios, the population continues to decline ($\lambda < 1$, Figure 7.1). The greatest impact on the population occurs during the first 20 years for both scenarios; lambda increases by 1.5% from 0.960 to 0.975 under the best case scenario and declines by -4.5% from 0.960 to 0.917 under the worst case scenario (Figure 7.1). As expected under a declining trajectory, the pE increases over time (Figure 7.2). By year 60, pE ranges from 56% to 74% (see Appendix 3, Table 3A3 for decadal projections).

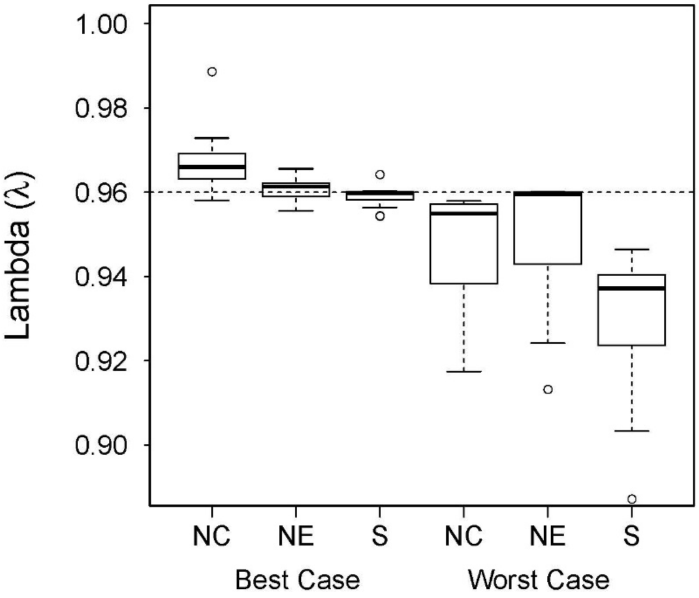

Figure 7.1. Box plot for population growth rate (lambda, λ) under the best and worst case scenarios for each of the subregions of the eastern population (NC=Northcentral, NE= Northeast, S=South). The dashed line represents the current population growth rate (λ=0.96).

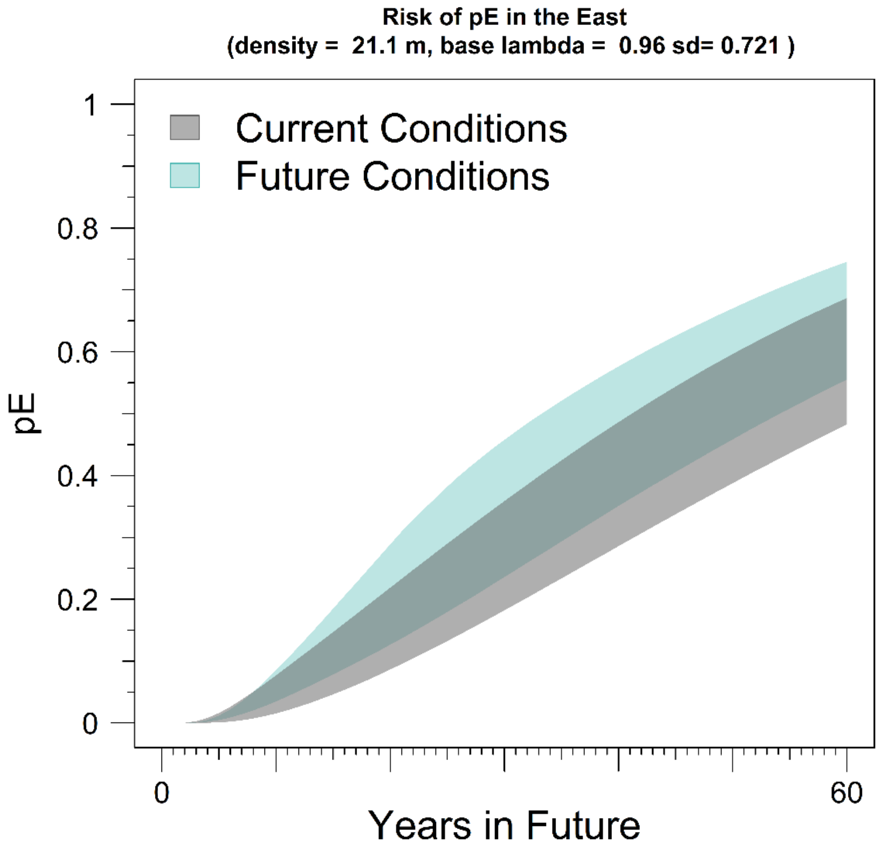

Figure 7.2. pE, for the eastern North American monarch population over time, given both current (gray band) and projected changes in state conditions (blue band). By year 60, pE ranges from 56% to 74% under the best and worst case future scenarios, respectively.

Direct Effects from High Temperatures & Catastrophic Events

We were unable to incorporate direct effects from increasing temperatures and catastrophic risks into the population models, so we qualitatively discuss the implications of these factors on the future condition of the population. We evaluated the change in the spatial extent and number of "cell days" (i.e., raster grid cells) with projected temperatures above thermal thresholds during critical time periods in monarch migration (see Appendix 2 - Climate change projections for further details). Under the RCP 4.5 scenario, both the spatial extent and the average number of >38°C days (sublethal and moderate survival reductions) are projected to decrease in the northcentral subregion but markedly increase in the south (94% and 331%, for area and number of days, respectively) and northeast subregions in April and May (Figure 7.3, see Appendix 3 for values for all subregions). The spatial extent and average number of cell days above the lethal threshold (42°C) are projected to increase dramatically for the south (6,630% and 8,147%, respectively) during the same period (Figure 7.3). Given these results, monarch reproductive success and survival rates of the first generation of monarchs are likely to decline, although the extent of which these rates will decline is unknown.

Similarly, given the projections of monarch health described above, the eastern population will be increasingly vulnerable to catastrophic losses due to both extreme storm and widespread drought events. Although we cannot quantify this increased risk, the longer the eastern population remains unhealthy, the more likely it is that catastrophic losses will occur and the greater the extinction risk for the eastern population.

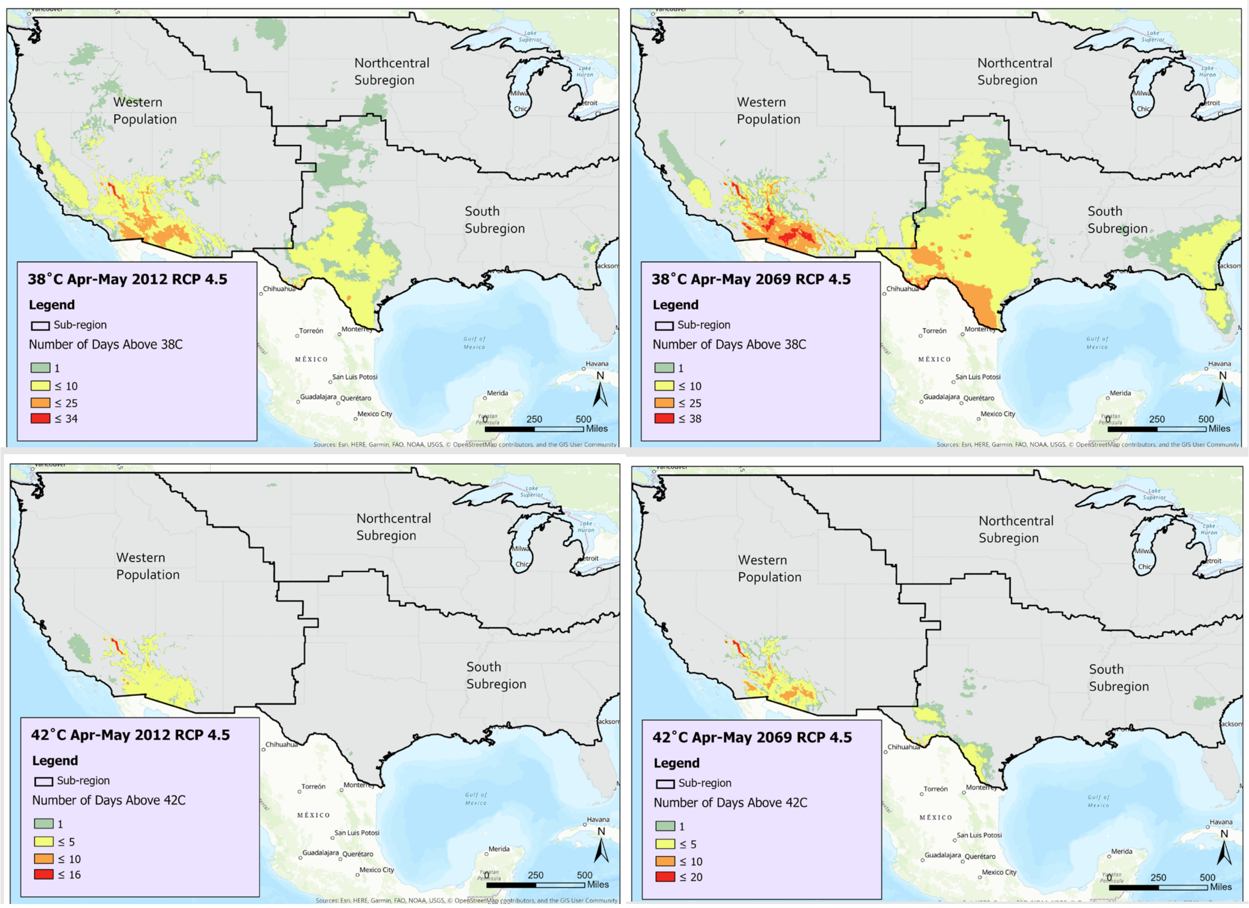

Figure 7.3. The projected spatial extent and average number of cell days between April and May where temperatures >38˚C (top) and 42˚C (bottom) in 2012 (left) and 2069 (right) under RCP 4.5. Colors represent number of cell days above >38˚C and 42˚C.

Western North American Population – Future Condition

Under both scenarios, the population continues to decline ($\lambda < 1$, Figure 7.4). Under the best case scenario, greatest positive effect occurs in years 21-50 when lambda slightly increases by 0.3% from 0.878 to 0.881; under the worst case scenario, the population is most affected during the first 20 years when lambda decreases -5.8% from 0.878 to 0.828. As would be expected with a declining growth, the pE increases over time (Figure 7.5). At year 10, pE ranges from 66 to 71% and reaches 99% by year 60 (see Appendix 3, Table 3A3 for decadal projections).

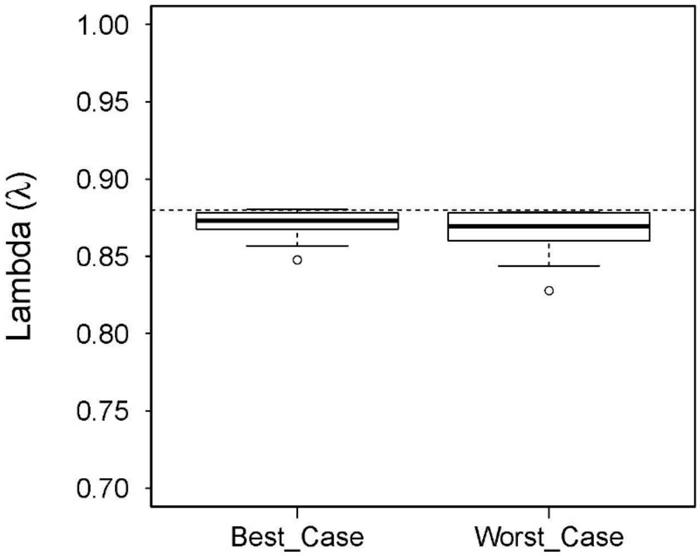

Figure 7.4. Box plot for population growth rate (lambda, λ) under the best and worst case scenarios for the western population. The dashed line represents the current population growth rate (λ=0.878).

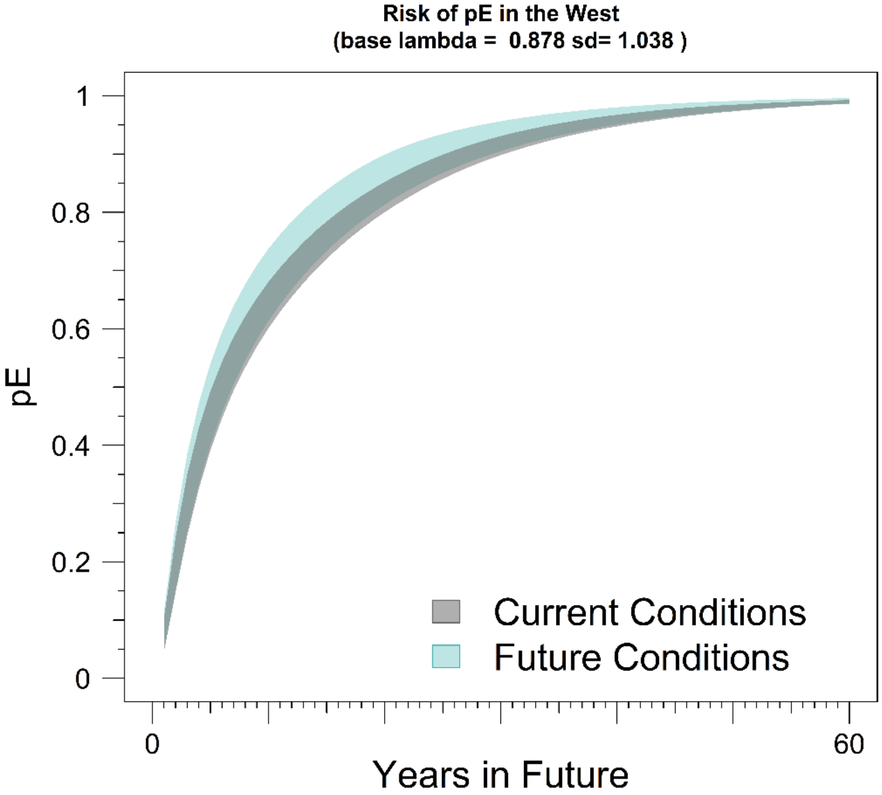

Figure 7.5. pE, for the western North American monarch population over time, given both current (gray band) and projected changes in state conditions (blue band). By year 60, pE reaches 99% under the best and worst case future scenarios.

Direct Effects from High Temperatures & Catastrophic Events

Under the RCP 4.5 scenario, the spatial extent of the area over which the average number of degree days >38°C and >42°C is projected to decrease (-23% and -11%, respectively), while increases in the average number of days >38°C (38%) and >42°C (11%) are projected (see Figure 7.1 and Appendix 3, Tables 3A1-A3 for further results). Given these results, monarch reproductive success and survival rates are likely to decline, although the extent of which these rates will decline is unknown.

Similarly, given the projections of monarch health described above, the western population is vulnerable to catastrophic losses due to both widespread drought events and the co-occurrence of poor environmental conditions and low population abundance. The risk of extinction due to these events increases the longer the population remains at the current low abundances.

Worldwide Populations – Risks due to Catastrophic Events

We qualitatively assessed the impact due to predicted climate change effects. Fifteen of the 29 populations are classified as being "at risk" to extinction due to sea level rise or due to increasing temperatures (Table 7.1).

Table 7.1. Qualitative expression of risk due to predicted sea level rise and high temperatures. See definitions of terms in Table 3.2.

ACU	Population	Status	High Temps	Sea Level Rise
Australia, New Zealand, and Indo-Pacific Islands	Australia	Extant	At Risk	No Known Risk
	Cook Island	Extant	No Known Risk	No Known Risk
	French Polynesia	Extant	No Known Risk	No Known Risk
	Greater Indonesia	Unknown	At Risk	No Known Risk
	Guam & CNMI	Extant	No Known Risk	No Known Risk
	Johnston Atoll	Extant	No Known Risk	At Risk
	Kiribati	Extant	No Known Risk	At Risk
	Marquesas Islands	Extant	No Known Risk	No Known Risk
	Marshall Islands	Extant	No Known Risk	At Risk
	Mascarene Islands	Extant	No Known Risk	No Known Risk
	Micronesia	Extant	No Known Risk	No Known Risk
	Nauru	Extant	No Known Risk	At Risk
	New Zealand	Extant	No Known Risk	No Known Risk
	Norfolk Island	Extant	No Known Risk	No Known Risk
	Palau	Extant	No Known Risk	No Known Risk
	Papua New Guinea	Extant	At Risk	No Known Risk
	Philippines	Unknown	At Risk	No Known Risk
	Samoa	Extant	No Known Risk	No Known Risk
	South Pacific Islands	Extant	No Known Risk	No Known Risk
	Tokelau	Unknown	No Known Risk	At Risk
	Tonga	Extant	No Known Risk	No Known Risk
	Tuvalu	Extant	No Known Risk	At Risk
	Wallis & Futuna	Unknown	No Known Risk	No Known Risk
Central America & the Caribbean	Caribbean	Extant	At Risk	No Known Risk
	Central America	Extant	At Risk	No Known Risk
Southern Florida	Florida	Extant	At Risk	No Known Risk
Hawaii	Hawaii	Extant	No Known Risk	No Known Risk
Iberian Peninsula	Iberian Peninsula	Extant	At Risk	No Known Risk
South America & Aruba	South America and Aruba	Extant	At Risk	No Known Risk
Eastern North America	Eastern North America	Extant	See E. North American pop below	
Western North America	Western North America	Extant	See W. North American pop below	

Chapter 8: Synthesis – Implications for Viability

This chapter synthesizes the results from our historical, current, and future analyses and discusses the consequences of the change in the number, health, and distribution of populations over time for the viability of the monarch. We assessed monarch viability by evaluating the species' ability to withstand environmental stochasticity (resiliency), catastrophes (redundancy), and changes in its environment (representation). We also discuss the key uncertainties and their implications for the analyses.

Viability

Monarch viability depends upon its ability to sustain populations in the face of normal environmental stochasticity, catastrophes, and novel changes in its environment. The species' ability to do so is influenced by the health and distribution of its populations. Demographically and physically healthy populations are better able to withstand and recover from environmental variability and disturbances and are more likely to withstand and recover from events that would otherwise be catastrophic. Populations spread across heterogeneous conditions are unlikely to be exposed at the same time to poor environmental conditions, thereby guarding against synchronous population losses. Lastly, populations spread across the breadth of genetic and phenotypic diversity help to preserve species' adaptive capacity, which is essential for adapting to their continuously changing environment (Nicotra et al. 2015, p. 1269). Without such variation, species are less responsive to change and more prone to extinction (Spielman et al. 2004, p. 15263). Additionally, as populations with higher genetic diversity can more quickly adapt to novel changes, species with genetically healthy populations (large N_e, which begets genetic diversity) are better able to adapt (Ofori et al. 2017, p.2).

Historically, monarchs were widely distributed across 90 countries, islands, and island groups. Currently, monarchs remain widespread with 27 extant populations and 4 with unknown status. Despite being widespread across a diversity of habitats, environmental gradients, and climates, we found 15 of the worldwide populations are 'at risk' of extinction, and the populations comprising the core of the species—eastern and western North American populations—have declining growth rates and increasing extinction risks. While the North American migratory populations naturally fluctuate year-to-year with environmental conditions, they have declined over the last 20 years (Figure 8.1). These declines are due primarily to: (1) loss and degradation of habitat [from conversion of grasslands to agriculture, widespread use of herbicides, logging/thinning at overwintering sites in Mexico, senescence and incompatible management of overwintering sites in California, urban development, and drought]; (2) continued exposure to insecticides; and (3) effects of climate change. Because monarch populations fluctuate with environmental conditions, populations must be large and have strong population growth potential to withstand natural environmental variation and disturbances. Given their current low population sizes and declining growth rates, these populations will likely continue to decline without threat abatement. The magnitude or frequency (or both) of these threats, are expected to increase (Figures 6.1 & 6.2) further exacerbating declines (in abundance and growth rates) and increasing extinction risks (Figures 7.3 & 7.5). The recent steep decline of the western population may be a consequence of small population effects (i.e., an extinction vortex due to Allee effects and increased sensitivity to environmental stochasticity); in which case, amelioration of threats may not be enough to stall extinction. The western population trajectory may portend the future for the eastern population if its declining population trend is not reversed (i.e., insufficient resiliency to rebound from poor years resulting in steep and rapid declines). The health of the North American populations is declining, rendering both less able to withstand and recover from poor environmental conditions and withstand stressors. Under future state conditions, the resiliency of these populations will continue to decline as reflected in their increasing pE (the probability of the population abundance reaching the point at which extinction is inevitable) estimates over time.

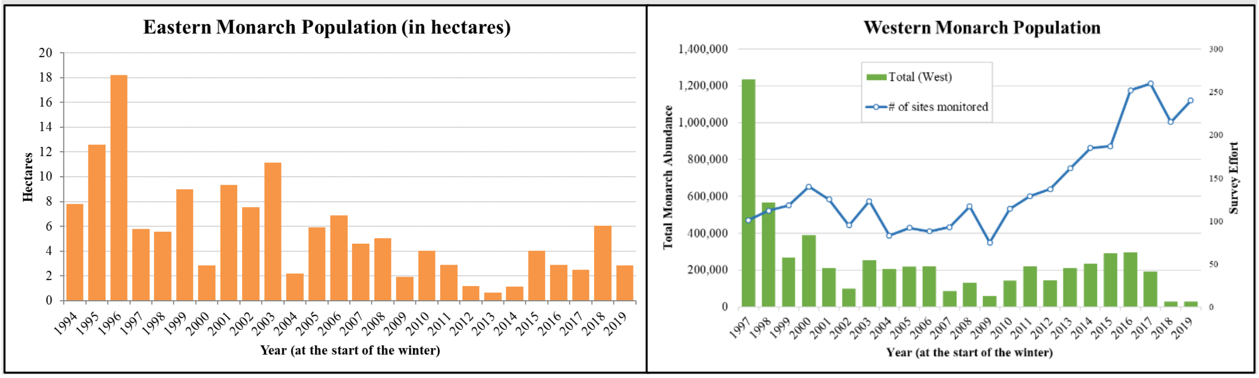

Figure 8.1. Eastern (a) and western (b) North American monarch population sizes, as measured at overwintering sites in terms of hectares (eastern) and total number (western). The western population counts also has a blue line indicating survey effort (number of sites monitored).

Moreover, the estimates of pE do not include risks from large, consequential stochastic events and direct effects of high temperatures due to climate change. At their current low abundances, these populations are more vulnerable to events that would otherwise be non-catastrophic. For example, had either of the two potentially catastrophic storms (where estimated mortality exceeded 70%) on the Mexico overwintering sites occurred during a low abundance year, the eastern North American population may have been extirpated. The longer these populations remain unhealthy (i.e., impaired growth potential and low abundance), the greater their risk to extinction due to stochastic events alone. Additionally, under climate change projections (both RCP 4.5 and RCP 8.5), the number of days with—as well as the spatial extent where monarchs will be exposed to—lethally high temperatures is projected to increase markedly and thus reduce monarch survival and reproductive rates in the affected subregions. Neither the risks from catastrophic events nor high temperature effects are fully captured in our pE estimates.

The extinction of either the western or eastern North American migratory population would increase the risk of losing the North American migratory phenomenon, as its persistence would depend solely upon the continued survival of a single population. Moreover, loss of either population would impair the overall ability of the species to adapt in the future. Although each of the 8 delineated ACUs represent unique sources of adaptive diversity, and therefore individually contribute to the monarch's adaptive capacity, the eastern and western ACUs are especially important. In addition to being genetically distinct and possessing greater allelic diversity than all other ACUs, monarchs in the eastern and western North American ACUs exhibit the long-distance migratory phenotype, occupy different climates and habitat niches, and differ in reproductive behavior and possibly disease resistance. Further, these North American populations represent the historical and current core of the species and the ancestral lineage of the species. Accordingly, loss of these two ACUs would reduce monarch diversity, rendering the species less able to adapt to novel changes in its environment now and in the future and thereby increasing the extinction risk of the species. The chance of *both* populations persisting above the extinction threshold over the next 10 years is 27% to 33% (under future conditions) and drops under 10% within 30 years.

Much of this risk is due to the poor condition of the western population. The western North American population comprises approximately 30% of the area occupied by monarch butterflies in North America and contributes unique variation in migratory, overwintering, and reproductive behavior; ecology; wing morphology; and flight power. Western monarchs expand outward from their overwintering sites, while monarchs in the eastern population shift the range northward. Western overwintering monarchs may have a shorter diapause and may also differ in mating behavior. Western monarchs differ in their ecology from eastern monarchs in their use of different species of nectar and milkweed plants and different roosting tree species. Lastly, differences are seen in divergent wing morphology and flight power between eastern and western monarchs. Additionally, a recent genomics analysis indicates low levels of dispersal between eastern and western monarch butterflies, suggesting that they are demographically independent. So, although unquantifiable, the loss of the western population would reduce the monarch's diversity and likely its ability to adapt to changes in its environment, thereby increasing the extinction risk of the North American monarchs.

Based on this information and other analyses included in this SSA, monarch viability is declining and projected to continue declining over the next 60 years.

Uncertainties

Our analysis includes both aleatory (i.e., inherent, irreducible) and epistemic (i.e., ignorance, reducible) uncertainty that we address by developing a range of future scenarios, adding environmental stochasticity to our model, applying stochastic extinction thresholds, and making reasonable assumptions. These assumptions, albeit necessary, impact the results of our analyses. Here, we highlight the key uncertainties, our accompanying assumptions, and our assessment of the relative influence they impose on the results. When we say that these key assumptions impact the analysis of monarch viability, we mean they may directly impact estimates of the monarch's (a) ability to withstand environmental stochasticity (resiliency), (b) ability to withstand catastrophic events (redundancy), (c) ability to adapt to novel changes in their environment (representation), and (d) vulnerability to extinction.

Historical Conditions

The historical range of monarch includes sites outside of North America, with monarchs documented throughout this range from the mid- to late-1800s. We know monarchs were present in North America prior to the 1800s, but we do not know the full extent of their range. We assume that monarchs that are present outside of North America have become naturalized. This assumption may overestimate the historical viability of monarchs worldwide.

Current Conditions

The key uncertainties that impact our ability to interpret current monarch viability include: (1) current status and health of worldwide populations, (2) current, independent population growth of North American populations (the lack of links between their population numbers), (3) extinction thresholds for both eastern and western populations, and (4) density estimates for the eastern population.

Worldwide Populations Status and Health

There is a paucity of data on monarch occurrence over time, distribution, and habitat use. We assumed that all populations in which at least a single monarch has been documented since the year 2000 are currently extant (either known or presumed). To assume these worldwide populations are extant will overestimate the current representation and resiliency of monarchs globally and, subsequently, overestimate the viability of the species.

Exchange of Individuals among the North American Populations

Marking data from Morris et al. (2015, pp. 100, 102) indicate that at least some individuals migrate from the western United States to overwintering grounds in Mexico and that monarchs can return from Mexico to the western United States to breed (Brower and Pyle 2004, p. 155; Dingle et al. 2005, p. 498), but we do not know at what rate. We also know that some monarchs that migrate south through the eastern United States to overwinter in Mexico break diapause to breed in the Gulf region (Howard et al. 2010, p. 2) and likely supplement non-migratory populations that breed year-round in southern Florida (Knight and Brower 2009, p. 819). Similar to other models (Semmens et al. 2016, Schultz et al. 2017), our model does not include immigration and emigration parameters from our population models for the eastern and western North American populations. This assumption of lack of connectivity could underestimate the current resiliency of each population and thus underestimate monarch viability. This uncertainty and its corresponding assumption also apply for future conditions and again likely underestimate monarch viability into the future.

Alternate Overwintering Strategies

It is believed that a majority of eastern and western North American monarchs overwinter in reproductive diapause in Mexico and along the California and Mexican coast, respectively (see *Individual-Level Ecology and Requirements* in Chapter 2). However, there are known exceptions to this overwintering pattern. There are monarchs that remain or become reproductively active and breed throughout the winter along the Gulf Coast, the southern Atlantic Coast, and the southern Pacific Coast (Howard et al. 2010, p. 3; Satterfield et al. 2016, p. 346). These monarchs are more likely to be infected with *OE* (Satterfield et al. 2016, 2018, p. 347, p. 1676, respectively), and there is some question of whether some of the offspring of these individuals might emerge in diapause and continue to Mexico or California overwintering sites later in the season (Batalden and Oberhauser 2015, p. 223).

Additionally, there are other, smaller overwintering areas for the eastern and western North American population that exist with monarchs overwintering in diapause. For the eastern population, these include small colonies east of Mexico City (e.g., a site with small aggregations along western slopes of the Popocatépetl volcano; Calvert and Brower 1986, p. 171), and along the coast of North Carolina (where 94 monarchs were captured during overwintering dates over the course of 13 years; McCord and Davis 2010, p. 413). For the western population, these include several small inland California and Arizona overwintering sites (Morris et al. 2015, p. 98; Pelton et al. 2016 p. 10). Because of the relatively small number of monarchs at these sites and their transient nature, we have assumed that Mexico and California annual counts represent the large majority of the eastern and western monarch populations, respectively.

Density Estimates for the Eastern North American Population

The density (# of overwintering monarchs/ha) at the overwintering grounds in Mexico is uncertain and fluctuates within and among years. Because monarch overwintering population size in Mexico is measured in hectares, the assumed density value determines the initial population size estimate, *N(t)OW*, which can influence model results. Published estimates of these densities range from 6.9-60.9 million monarchs per hectare (Calvert 2004, p. 125); Thogmartin et al. (2017a) estimated that the 95% credible interval ranges from 2.4 - 80.7 million monarchs per hectare. We used the median density estimate of 21.1 million (Thogmartin et al. 2017a, p. 10) for our initial population size estimates, and we assumed that density, as reported by annual monitoring efforts, has remained consistent year to year. The chosen density greatly influences the probability of persistence estimates, and thus, likely monarch viability. Monarch viability could be over or underestimated due to our choice in density estimate.

Extinction Threshold

Another key uncertainty is the population size in which environmental stochasticity and Allee effects begin to override the population dynamics (i.e., reinforcing processes drive the population downward towards extinction, extinction vortex). The model samples extinction thresholds from a uniform distribution defined by two sources: expert elicitation for the eastern population (Voorhies et al., 2019, Suppl. 2) and Schultz et al. 2017 and Wells et al. 1990 for the western population. Therefore, we could be either overestimating or underestimating extinction risk under current conditions depending on the accuracy of the thresholds. This uncertainty and its assumptions also apply to future conditions.

Future Conditions

Most of our uncertainty related to monarch viability rests with our analyses of future conditions. These key uncertainties include (1) the future health and persistence of global populations, (2) the relationship between threats and population responses, (3) extinction thresholds for the migratory eastern and western North American populations, and (4) the correct way to account for the multi-generational growth of the migratory eastern North American monarch population.

Worldwide Populations Status and Health

Similar to current conditions, there is a lack of monitoring or survey data necessary to predict future population growth trends for worldwide populations. We are unable to evaluate the impact of threats like habitat loss (land-use change) or pesticide use because we lack information on the specific locations of monarchs within these worldwide geographies. We do assume that monarchs will be extirpated from islands that are completely drowned due to sea level rise. In all other cases, we assume that monarch populations will persist into the future and this may lead to overestimating the viability of the species.

The Relationship between Influences and Population Response

Outside of milkweed and breeding, we lack direct and causal relationships between monarch population size and threats. We assume that our expert-elicited response curves and scenarios accurately represent these unknown relationships. Additionally, we assume that influences are additive and that their rates remain constant over time, an assumption mirrored in a retrospective threats analysis done by Thogmartin et al. 2017b (threats analysis). To assume influences can be simply added and remain constant over time (rather than including interactions or rate changes), likely leads to an underestimate of the vulnerability of extinction of both eastern and western populations. These assumptions in our eastern and western population models likely lead to an overestimate of monarch viability by increasing the resiliency of eastern and western populations.

Furthermore, we overestimate the resiliency of eastern and western populations through our assumptions addressing uncertainties in climate and insecticide influences on these populations. For climate change, we assume that the newly available monarch habitat will be in the northern portion of its current breeding range and beyond and that the migration success rates will be unchanged. We assume that they will be able to take advantage of this habitat and successfully migrate, and we also assume that the large scale modeled niche is indicative of suitable microclimate for monarchs. For insecticide use, we lack information on changes to effectiveness of insecticides or societal pressure to reduce insecticide use. Therefore, we assume very little change in the influence of insecticides on monarch populations into the future.

Extinction Thresholds

Just as in current conditions, the extinction thresholds for both eastern and western populations are a source of uncertainty. This uncertainty follows the same discussion and rationale as described in the current conditions section. Therefore, we could be either overestimating or underestimating extinction risk under future conditions depending on the accuracy of our expert-informed thresholds. The uncertainty of extinction thresholds will impact our estimate of monarch resiliency and possibly overestimate or underestimate the viability of the monarch as a species.

Multi-Generational Growth of the Eastern Monarch Population

Published models of monarch population growth vary in accounting for the multi-generational migration and growth of the eastern monarch population. Some only estimate growth of the overwintering population (Semmens et al. 2016) while others model the growth of subregions within the eastern monarch population (Flockhart et al. 2015, Oberhauser et al. 2017a). Here we assume that modeling population growth at the sub-regional level (Northcentral, Northeast, and South regions) is appropriate (as done in Oberhauser et al. 2017a and published in Voorhies et al. 2019). Experts who participated in our expert elicitation provided estimates of the relative importance of each of these regions to the Mexico overwintering population used in our modeling. This assumption leads to redundancy in influences (both negative and positive) in the different subregions. This in turn, can lead to either an under- or overestimation of the vulnerability to extinction of the eastern population. This redundancy occurs because the population can respond differently to these influences in different regions (because of differing population response curves). As an example, if one region is critically impacted by a negative influence, there are still other regions to contribute to the overall population size. Furthermore, because the eastern monarch population is such a large component of the monarch species, the robustness of this population could lead to over- or underestimating the viability of monarch butterfly.

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Appendix 1. Taxonomy

At the time that the monarch butterfly (*Danaus plexippus plexippus*) was petitioned to be listed under the Endangered Species Act of 1973 (Center for Biological Diversity et al. 2014), the petition noted that there were six recognized subspecies of *Danaus plexippus* (*plexippus*, *megalippe*, *nigrippus*, *tobagi*, *portoricensis*, and *leucogyne*; Warren et al. 2013). However, examination of the literature and contact with a butterfly taxonomist, suggest there are only 2 or 3 subspecies, and that the subspecies concept for monarch butterflies is not currently rigorously defined.

In 2005, Smith and colleagues published their findings on *Danaus* taxonomy. They classified *Danaus plexippus* as having only two subspecies: *plexippus* and *megalippe*. *Danaus plexippus plexippus* is the subspecies that resides throughout most of North America, and throughout islands in the Pacific Ocean. *Danaus plexippus megalippe* is non-migratory and resides in parts of the southern U.S., the Caribbean, and Central and South America. They suggest that *tobagi*, *portoricensis*, and *leucogyne* may be color variants of *Danaus plexippus megalippe*, rather than separate subspecies. However, they do not comment on *Danaus plexippus nigrippus* (a potential subspecies that is non-migratory and found in parts of South America). In communications with butterfly taxonomy expert, Jonathan Pelham (Curatorial Associate [Lepidoptera] at the University of Washington Burke Museum), he agrees with the Smith et al. (2005) findings.

The potential third subspecies, *nigrippus*, was mentioned in a study where it was shown to be a different species than the South American-residing southern monarch (*Danaus erippus*; Hay-Roe et al. 2007). However, it is unclear whether any work has defined *nigrippus* as separate from either *megalippe* or *plexippus* subspecies. It is also uncertain whether monarchs in the northern and northwest portions of South America are subspecies *plexippus*, *megalippe*, or *nigrippus*.

J. Pelham stated that "*plexippus* represents the 'Monarch' as we have known it, *megalippe* represents the Caribbean fraction, which is typical of many widespread Neotropical butterfly species and *nigrippus* represents the southernmost entity" (J. Pelham, pers. comm. 2017). This classification depends on *Danaus plexippus plexippus* being migratory, and the other subspecies being non-migratory. However, non-migratory *Danaus plexippus plexippus* exist throughout the range (both within North America and throughout the Pacific). There are many unknowns about the precise borders of the monarch range, and there is even more difficulty in precisely determining where potential subspecies might interface. Most of scientific papers on *D. plexippus* examined do not specify subspecies, further complicating any determination of where potential subspecies might exist.

Given the complexity and uncertainty of monarch subspecies, as well as the petitioners' request to determine "whether any newly identified North American subspecies may warrant federal protection" (Center for Biological Diversity et al. 2014, p. 16), we are considering monarchs (*Danaus plexippus plexippus*) throughout the known range of the species.

Appendix 2. Methods

[1] Updates to Voorhies et al. 2019 model

Since the publication of the Voorhies et al. model, we made several changes to the model:

1) The input values have been updated: lambda values, epsilon values, and starting population sizes.
2) The time-frames for the influences are now "influence-specific." We allow each influence to reach its full magnitude of impact within the time-frame specific to that influence; in the published paper, the magnitude of change was incrementally distributed over 50 years (see paper methods section or p. 4).
3) The influence of climate is modeled differently in a couple ways. [1] The effects of climate change continue to be incorporated via availability of milkweed. In this version, climate change effects are combined with milkweed over the milkweed specific time-frame (20 years) and on its own for an additional 40 years (to reach the full duration of the climate change effect). [2] In this version, climate change is also combined with migration nectar influence in the south subregion. It is combined in the same way it is combined with milkweed and is used as an input to the migration nectar population response curve for the southern sub-region of the eastern population).
4) Future scenarios for milkweed and nectar in breeding habitat in the eastern population now include subregion specific values to be fed to subregion specific population response curves. Previously, we had one future scenario for milkweed and nectar in the breeding range and it was applied to all three subregions using their subregion specific population response curves. Now both inputs and response curves are subregion specific).

[2] Inputs to model

Table 2A1. Initial starting values for the population model. Inds= individuals

Population	Model Parameter	Value	Source
Both	Years	60	SSA Team
	Simulations	1,000,000	SSA Team
Eastern	Ninit	3.656 ha	5-year average
	λ	-0.0408	Semmens et al. 2016*
	ε	0.721	Semmens et al. 2016*
	Extinction threshold low	0.05 ha	Expert-elicited, Voorhies et al. 2019
	Extinction threshold high	0.61 ha	Expert-elicited, Voorhies et al. 2019
	Density/ha	2.11E+07 inds/ha	Thogmartin et al. 2017a
	Cap	36 ha	SSA Team
	Regional Importance_NC	0.68	Expert-elicited, Voorhies et al. 2019
	Regional Importance_NE	0.20	Expert-elicited, Voorhies et al. 2019
	Regional Importance_S	0.12	Expert-elicited, Voorhies et al. 2019
Western	Ninit	168,365 inds	5-year average
	λ	-0.13	Schultz et al. 2017
	ε	0.99	Schultz et al. 2017*
	Extinction threshold low	20,000 inds	Schultz et al. 2017*
	Extinction threshold high	50,000 inds	Wells et al. 1990
	Cap	2,400,000 inds	SSA Team

*Parameter values differ slightly from Schultz et al. 2017 and Semmens et al. 2016 because the population datasets have been updated with values through winter 2019-2020.

[3] Other threats and catastrophic events considered

In addition to the primary influences considered above, we also looked at many other factors that may be impacting monarchs. These included but were not limited to natural enemies (disease/parasitism), captive rearing, collection, impacts of tourism at overwintering sites, invasive swallow-wort plants, vehicle mortality, and natural catastrophes. We also considered other potential positive impacts, such as positive impacts of research and monitoring.

Other Stressors

Monarchs are impacted by a number of diseases and natural enemies. One of the most well-known and well-studied natural enemies of monarchs, *OE* (a monarch parasite), impacts worldwide populations at different rates (see *Representation* section in Chapter 2; Altizer and de Roode 2015, p. 84), with non-migratory populations typically having higher rates of infection (Bartel et al. 2011. p. 348). This protozoan parasite impacts monarchs (*OE*'s only known host), leading to decreased survival and fitness in the monarch (Altizer and Oberhauser 1999, p. 85). While infection rates can be high, we have not seen a large and continuous increase in proportion of monarchs that are heavily infected over time in eastern North America (Project Monarch Health 2016, p. 1). Other diseases can infect monarchs, including nuclear polyhedrosis virus, but most reports of these are anecdotal and no reports to our knowledge indicate increasing rates of disease (Arnott et al. 1968).

In addition to disease and parasites, immature monarchs are heavily preyed upon by natural enemies (upwards of 90% of monarchs are killed in immature stages; Nail et al. 2015a), but there is not any conclusive evidence available that suggests predation rates are currently increasing. These immature monarch predators range from ants, tachinid fly parasitoids, and various other insects for eggs and larvae, and wasps (*Pteromalus cassotis* and *Polites dominulus*) for pupae (Oberhauser et al. 2015, p. 72). The most studied larval natural enemy, the tachinid fly parasitoid, does not show a significant trend in proportion of monarchs parasitized over the years studied (Oberhauser et al. 2017b, p. 6). Adult monarchs also have predators, many of which have been documented at the Mexican overwintering sites (including birds, mice, and wasps; Oberhauser et al. 2015, p. 72). There is thought to be an approximate bird predation rate of 9% (Brower and Calvert 1985, p. 864), with potentially higher rates at smaller sites (Calvert et al. 1979, p. 850). However, these higher rates of predation have not been measured since the recorded decline in the eastern North American population began.

Captive rearing of monarchs was considered, as there are potential negative impacts of this practice on a large scale (Altizer et al. 2015, pp. 1-3). However, the number of monarchs being raised in mass-rearing operations is unclear (Villareal 2015, p. 9-10), and the impacts were difficult to quantify; thus, we did not consider this a primary influence. There is some information on vehicle mortality on insects (Baxter-Gilbert et al. 2015, Keilsohn et al. 2018), and some research on monarch vehicle mortality specifically (McKenna et al. 2001, Mora Alvarez et al. 2019, Kantola et al. 2019), and while this warrants future attention, we did not feel we had enough information to show that this was increasing or one of the current primary drivers of changes in monarch populations, nor was it identified as a primary driver in our expert elicitation. We did not find strong evidence of tourism at overwintering sites or insect collection impacting monarchs at the population level; hence, we did not currently consider them as primary influences. This is not to say that these or other threats could not become primary influences going forward, and thus should continue to be evaluated in the future.

The impact of invasive swallow-wort plants on monarchs was another influence that was considered. Black swallow-wort (*Cynanchum louiseae*) and pale swallow-wort (*C. rossicum*) are two European plants that are invasive in North America. They are in the milkweed family, but monarch caterpillars are unable to feed on these plants. However, there has been observed oviposition on these plants by adult monarchs, leading to speculation that these plants could serve as ecological sinks. However, the evidence for this is limited, with one study showing no oviposition on these species in the laboratory (DiTommaso and Losey 2003, p. 207) and another study showing limited oviposition in the field when common milkweed is scarce (Casagrande and Dacey 2007, p. 633). Given this evidence, we did not think invasive swallow-wort plants were a primary influence for driving the monarch decline.

We also considered the direct impacts of herbicides to monarchs. Results of herbicide toxicity studies suggest that various types of herbicides may result in direct effects to lepidopterans if exposed at recommended field application rates for the labeled land use/cover type. However, the direct effects of most herbicides to monarchs are unknown, and likely to be highly variable. In several studies, the simulated application site was some type of conservation area where chemical control of invasive plants was presumed, resulting in maximum exposure of herbicide to lepidopteran. It is important to note that we found no studies evaluating the effects of herbicides to lepidopterans at concentrations representative of exposure due to drift from an application site to nearby habitat (i.e., exposure concentrations at less than a labeled rate) for this risk assessment. While we acknowledge the potential for toxic effects of herbicides to monarchs under certain exposure conditions, we consider the effects of insecticides to be the primary driver in monarch population impacts due to pesticides (insecticides, herbicides, fungicides, rodenticides, etc.). See our *Supplemental Materials 1b* for a detailed description of the direct impact of herbicides on monarchs, including data, references, and supporting information.

We also considered positive influences, such as research and monitoring (e.g., the information that might be gained from the national integrated monitoring strategy). While these future impacts are difficult to determine or quantify, we note the importance of these efforts and their potential future influence on monarch populations.

Other Catastrophic Events

<u>Fire</u>

The frequency, size, and intensity of wildfire in the western U.S. has increased over time (Littel et al. 2009, p. 1003; Waterbury and Potter 2018, p. 43). The three largest fires in California history occurred in 2017 and 2018. Wildfire pose risks to both breeding and overwintering habitat as well as causing direct mortality of butterflies. Given the broad distribution of breeding habitat throughout the West, it is unlikely, however, that any single fire or series of fires would destroy a sufficient amount of habitat such that catastrophic losses occur. Additionally, monarchs are highly mobile and may be able to escape slow-moving fires and thus, direct mortality is unlikely. Similarly, during the winter, monarchs occupy numerous sites along broad areas of coastal California. Coupled with the close proximity of many of these sites to residential areas (where fire is more likely to be quickly contained), the likelihood of a catastrophic fire is low (Pelton et al. 2016, p. 28). However, if population numbers continue to decline, the impact of losing some portion of breeding habitat or one or two of the largest overwintering sites will increase the risk of extinction for the migratory population. Thus, there is insufficient information indicating that the magnitude and severity of fire poses a catastrophic risk to the western monarch population.

<u>Hurricanes</u>

Much of the coastline of the eastern U.S. has sustained impact by multiple hurricanes in recorded history (NOAA 2010). The states hit hardest by hurricanes are occupied by the eastern migratory population throughout much of the year. Hurricanes have the potential to kill some individual monarchs but only a hurricane in Texas or Mexico during peak migration to Mexico could have catastrophic effects on the eastern population. In an analysis by Ries et al. (2018, pp. 98-101), the authors determined that hurricanes and large masses of migrating monarchs are unlikely to cross paths in time and space because most major hurricanes happen in September or earlier and migrating monarchs funnel through Texas in October and November. Although hurricanes also have the potential to indirectly affect monarchs (Ries et al 2018, pp. 99-101), there is no evidence indicating that indirect effects (e.g., increased fall plant growth) would be catastrophic to the eastern migratory population. Currently, there is no evidence that major storms have directly killed masses of individual monarchs, and there are anecdotal accounts of monarchs surviving or flying in the opposite direction of severe storms (Journey North 2008; Moskowitz et al. 2001, p. 488). Should the timing and duration of hurricane season change in the future, as has been suggested by news outlets but not supported by research (see Karloski and Evan 2016, p. 273), migrating monarchs could be at an increased risk. Thus, there is currently insufficient information indicating that hurricanes pose a catastrophic risk to the eastern monarch population.

[4] Future scenarios

Eastern North American Population – Milkweed and nectar projections for Eastern North America were driven by milkweed stem changes from conservation efforts, Conservation Reserve Program acres, and land cover change.

Conservation Efforts

To calculate milkweed stem estimates, we began by establishing a baseline for the year 2014 using a "seamless" land cover dataset developed by Rohweder and Thogmartin (2016) that combined data from the National Land Cover Dataset (NLCD), Cropland Data Layer (CDL), Topologically Integrated Geographic Encoding and Referencing, and Homeland Security Infrastructure Program. We used the seamless dataset to calculate the number of acres of each land cover type in eastern subregions. We then multiplied the acres of each land cover type by the corresponding milkweed stem density in stems per acre from Thogmartin et al. (2017c), which were derived from literature and expert input. The result was an estimate of the total number of milkweed stems on the landscape in the Northcentral, Northeast, and South subregions. We assumed milkweed density is a reliable proxy for habitat quality, including nectar resources. Further, we assumed that the milkweed density estimates in the upper Midwest can be reasonably applied to Northeast and South subregions.

Using land cover type and acreage information in the Monarch Conservation Database (MCD), we calculated the current amount of habitat due to conservation efforts by adding milkweed from completed and implemented conservation efforts to the 2014 baseline number of milkweed. We calculated the number of milkweed from conservation efforts by tallying the number of acres of each land cover type that have been improved due to completed and implemented conservation efforts, and multiplying those acres by the net change in milkweed. We calculated the net change in milkweed by subtracting baseline milkweed stem density from the user provided data or "potential" milkweed density for the land cover type in question when user provided data was not available (Table 2A2). Milkweed density values in Table 2A2 for each land cover type are generally based on Thogmartin et al. (2017c; further clarified via pers. comms with Thogmartin), and represents the average estimate of biologically reasonable milkweed density for a given land cover type (derived from a combination of literature review and expert input). Potential milkweed density was not available for all land cover types due to discrepancies between land cover types used in Thogmartin et al. (2017c) and the seamless dataset (Rohweder and Thogmartin 2016). The estimated baseline and potential milkweed densities represent the current state of knowledge and can be updated when additional information becomes available.

We then derived a level of future conservation effort, relative to the current amount of habitat with upper and lower bound projections of Conservation Reserve Program acreage and land cover change. Our future scenarios (upper and lower bounds) included formalized, but not yet implemented (i.e. planned) conservation efforts submitted to the MCD. We assume the conservation efforts completed to-date will be maintained and continue to provide monarch milkweed and nectar resources for both scenarios.

For the Northcentral subregion, we assumed implementation of the Mid-America Conservation Strategy, which will result in an estimated 1.3 billion additional milkweed stems by 2038 from monarch conservation efforts. To account for net change since 2014, we calculated the gain in milkweed from completed and implemented efforts in the MCD as described above and subtracted this figure from the 1.3 billion stem goal. The result is the remaining total number of additional milkweed stems needed to meet the 1.3 billion stem goal from all potential sources and sectors. Next, we subtracted the projected gains under the upper bound scenario from Conservation Reserve Program and land cover projections (see below) to calculate the number of additional milkweed stems specifically from non-CRP conservation efforts needed to achieve the 1.3 billion stem goal relative to 2014 levels. For the lower bound in the Northcentral subregion, we assumed that additional conservation effort would occur to offset a portion of projected CRP losses; in this case, conservation effort equated to the same level of effort associated with the upper bound scenarios plus the equivalent gains that we had projected due to CRP increases under the upper bound scenario. In essence, the same level of habitat would be added to the landscape under the lower bound scenario as was assumed under the upper bound scenario (minus the additional benefits that were attributed to projected land cover change); however, additional losses would simultaneously occur due to broader CRP declines at that resulted in losses greater than the CRP gains under the upper bound scenario (also see *Conservation Reserve Program*). For the Northeast and South subregions, given the lack of an overarching monarch conservation strategy analogous the Mid-America Monarch Conservation Strategy, we simply calculated the change in milkweed from future formalized conservation efforts in the MCD using the methodology described above and similarly added the upper and lower projections CRP and land cover.

Table 2A2. Baseline and potential milkweed densities for land cover types. Values from Thogmartin et al. 2017.

Classification	**Estimated Baseline Milkweed Density**	**Potential Density**
22 - Developed Low Intensity (NLCD) (Inside Urban Areas)	1.00	50.00
23 - Developed Med Intensity (NLCD)	0.50	25.00
24 - Developed High Intensity (NLCD)	0.10	10.00
26 - Developed Low Intensity (NLCD) (Outside Urban Areas)	19.74	84.50
21 - Developed Open Space (NLCD) Linear	0.00	16.31
25 - Developed Open Space (NLCD) Core	0.00	3.09
120 - TIGER Secondary Roads	57.15	175.00
110 - TIGER Primary Roads and Ramps	57.15	150.00
140 - TIGER Local Roads	57.15	100.00

Classification	**Estimated Baseline Milkweed Density**	**Potential Density**
174 - TIGER Private Roads	3.09	3.09
180 - All TIGER Roads (Inside Urban Areas)	0.00	0.00
31 - Barren (NLCD)	0.00	0.00
41 - Deciduous Forest (NLCD)	0.00	0.00
42 - Evergreen Forest (NLCD)	0.00	0.00
43 - Mixed Forest (NLCD)	0.00	0.00
76 - Grassland (NLCD)	3.09	40.00
77 - Grassland (NLCD) PADUS Protected	3.09	250.00
100 - HSIP Transmission Line (Outside Urban Areas)	3.09	150.00
101 - HSIP Transmission Line (Inside Urban Areas)	0.00	0.00
200 - TIGER Rails (Outside Urban Areas)	3.09	200.00
201 - TIGER Rails (Inside Urban Areas)	0.00	0.00
52 - Shrubland (NLCD)	3.09	3.09
1 - Corn LOW	0.05	4.04
14 - Soybeans LOW	0.05	4.04
3 - Other Crops (CDL) LOW	3.09	5.56
4 - Other Crops (CDL) MEDIUM	5.30	7.74
5 - Other Crops (CDL) HIGH	7.50	9.93
6 - Fallow Idle (CDL) HIGH	3.09	4.05
7 - Fruit Xmas Trees Vines (CDL) LOW	3.09	5.56
8 - Fruit Xmas Trees Vines (CDL) MEDIUM	5.30	7.74
9 - Fruit Xmas Trees Vines (CDL) HIGH	7.50	9.93
2 - Corn LOW (Marginal)	0.05	200.00
15 - Soybeans LOW (Marginal)	0.05	200.00
10 - Hay Alfalfa (CDL) LOW	3.09	40.00
78 - Pasture (NLCD)	3.09	40.00
79 - Pasture (NLCD) PADUS Protected	3.09	126.55
95 - Herbaceous Wetlands (NLCD)	61.37	68.16
90 - Woody Wetlands (NLCD)	61.37	68.16
Unclassified (Weighted average of all land cover types)	7.03	28.63

Conservation Reserve Program

To calculate the net change in Conservation Reserve Program acres from 2014 and 2018 and current amount of CRP acreage, we began by requesting county-level information from the Farm

Service Agency (FSA) for acres of CRP conservation practices that Thogmartin et al. (2017c) determined to be beneficial for monarchs. We shared with an FSA economist a "non-sensitive" version of the seamless dataset for consistency and the economist was able to extract from their system and the seamless dataset a breakdown of CRP acres for conservation practices benefitting monarchs by land cover type in each county for 2014 and 2018. We then applied the baseline and potential milkweed stem density for each land cover type per Thogmartin et al. (2017c) (see *Conservation Efforts*) to calculate the total number of milkweed from CRP acres and subtracted 2014 county totals from 2018 county totals to get the net change. We added the net change in CRP milkweed to milkweed from completed and implemented conservation efforts to calculate the current habitat due to CRP. For the milkweed and nectar future scenarios with respect to CRP, we assumed a 22% increase relative to 2018 CRP milkweed in the upper bound, and a 35% loss in the lower bound, respective to each subregion, based on USDA projections, recent trends in CRP acreage, and expert opinion (USDA 2020; Skip Hyberg, retired Senior Economist, pers. comm.).

Land Cover Change

We used the FORE-SCE (FOREcasting SCEnarios) land cover change model developed by the USGS Earth Resourses Observation Science (EROS) Center to develop future scenarios with respect to background changes in land cover under a range of emissions scenarios between 2010 and 2050 (Sohl et al. 2018). Unfortunately, the land cover types used in the FORE-SCE model did not all match the land cover types from Rohweder and Thogmartin (2016) or Thogmartin et al. (2017c) despite being based largely upon the same underlying dataset (the 2011 National Land Cover Dataset, NLCD). We matched any mismatched land cover types used in the FORE-SCE model with seamless dataset land cover types using overarching themes (e.g. developed, agriculture, grassland, wetland, etc.; Table 2A3). Additionally, there were land cover sub-types for which the FORE-SCE model did not predict future change but were crucial components of the seamless rater dataset, such as roads and rail lines. For roads and rail lines, we estimated the change based on mile statistics over the past decade from the U.S. Department of Transportation (USDOT 2020a, 2020b). Due to a lack of available data, we assumed no change in acreages of transmission lines. For Conservation Reserve Program, see methodology described above. For seamless dataset land cover types grouped into a single FORE-SCE land cover type (e.g. cropland), we assumed the percent change projected in the FORE-SCE model or other datasets applied evenly to all grouped land cover types. Projected changes in the "Mechanically Disturbed" and "Mining" land cover types used in the FORE-SCE model were not accounted for, as there is no analogous land cover type defined in the seamless dataset. While the projected percent change in some conservation units are significant, they generally accounted for a relatively small proportion of the landscape.

Once we calculated the percent change for each land cover type using the FORE-SCE model, we applied that percent change to the seamless dataset using the Table 1 to calculate projected acres of seamless dataset land cover types and applied the milkweed stem densities per Thogmartin et al. (2017c) to calculate future milkweed. We assumed linear change from 2010 and 2050 and divided the change over the 40-year period to calculate annual change and projected acres of each land cover type in 2018. We subtracted the 2014 baseline milkweed from projected milkweed due to land cover change in 2018 to calculate the net change in milkweed due to background land cover change. Under all scenarios, we project an increase in milkweed due to background land cover change. This projected increase in milkweed stems initially seems counterintuitive given that the FORE-SCE model and other sources of information (i.e., USDOT road mile statistics) generally predict an increase in more "developed" land cover types and a slight decrease in more land cover types such as grassland and shrubland. The numerical increase in milkweed due to land use change is largely a factor of differences in the estimated milkweed stem density for each land cover type. For example, certain types of roadway corridors are estimated to have much higher baseline milkweed stem densities than grassland or shrubland. While land use change appears to result in an increase in milkweed stems numerically, what is not factored in is the overall quality of habitat. As such, we used the projected increase in milkweed stems from the FORE-SCE for the upper bound scenario with respect to milkweed and nectar from land cover change. For the lower bound, we assumed no net change due to land cover change.

Table 2A3. Groupings of land cover type from the USGS EROS FORE-SCE model and Rohweder and Thogmartin 2016.

Classification (FORE-SCE Model)	**Classification (Rohweder and Thogmartin 2016)**
Developed	Developed – Low/ Medium/High Intensity, Exurban, Open Space
NA	Roads – Secondary, Primary & Ramps, Small, Private, Inside Urban Areas
Mechanically Disturbed National Forest, Other Public Lands, Disturbed Private	NA
Mining	NA
Barren	Barren
Deciduous Forest	Deciduous Forest
Evergreen Forest	Evergreen Forest
Mixed Forest	Mixed Forest
NA	CRP - Non-wet, Wet
Grassland	Grassland, Protected Grassland
NA	Transmission Line
NA	Rails
Shrubland	Shrubland
Cropland	Corn, Soy, Other Crops, Fallow Idle, Fruit/Christmas Trees & Vines
Hay/Pasture	Hay Alfalfa, Pasture, Protected Pasture
Herbaceous Wetland	Herbaceous Wetland
Woody Wetland	Woody Wetland

[5] Climate change projections

To calculate the percent change from 2012 to 2069 in the average number of days and spatial extent of which temperatures are above 38˚C between April and May and 42˚C between April and May of 2012 and 2069 in the continental United States, we downloaded climate projections from the Multivariate Adaptive Constructed Analogs (MACA) Climatology Lab (Abatzoglou and Brown 2012). The MACA Climatology Lab provides downscaled climate data from a number of Coupled Model Intercomparison Project 5 (CMIP5) climate models (Taylor 2012). For simplicity, we use the period between April and May in 2012 and 2069 and a threshold of 38˚C to describe our methodology. We downloaded projected daily maximum surface temperature for the continental United States for the 2006-2025 and 2066-2070 timeframes, and to account for variation between models and uncertainty, we downloaded projections under Representative Concentration Pathway (RCP) scenarios 4.5 and 8.5 and averaged outputs from 5 models. Each dataset came in the form of a NetCDF file, which consists of "stacked" raster datasets (Figures 2A1 and 2A2). Each approximately 4.6km x 4.6km grid cell of the dataset contains the daily "tasmax", or maximum air temperature in degrees Celsius 2 meters above the surface of the Earth for one day (Figure 1). We used the raster package in RStudio to import the data as a raster brick, or a stack of the 61 rasters, with each raster representing one day between April 1st and May 31st (Figure 2; Hijmans 2017; RStudio Team 2015). To calculate the total number of cells in a raster with tasmax values above 38˚C between April 1st and May 31st, we reclassified each raster, assigning all cells with tasmax values 38˚C or below a value of 0, and all remaining cells (i.e. cells with tasmax values above 38˚C) a value of 1. The result was a stack of 61 reclassified rasters, each containing cells with values of 0 or 1 indicating whether the tasmax was above 38˚C at that location. We refer to the reclassified value of each cell as a "cell day". Since each individual raster represents a single day, the maximum cell day value for any given cell is 1. We summed the rasters together to get the total number of cell days above 38˚C between April 1 and May 31 (Figure 2A3). The final combined raster is the sum of all 61 individual rasters and therefore, the maximum cell day value for any given cell in the resulting raster is 61, which would mean that every day between April 1st and May 31st has a tasmax above 38˚C at that cell location. We then plotted the final combined raster to get a map that indicates the number of cell days for each cell between April 1st and May 31st that had a tasmax of 38˚C degrees or above for a single model.

We then averaged the number of cell days above 38˚C at each cell across 5 models to capture the range of projections and plotted the average combined raster (Figure 2A4). We followed this process for the year 2012 and 2069. From these data, we were able to calculate the change in the spatial extent of temperatures above 38˚C spatially by calculating the change in the percent of land area occupied by cells with cell day values of at least one, and tabularly by calculating the change in the percent of total number of cell days with tasmax values above 38˚C.

We calculated the number of cells with at least one cell day above 38˚C by summarizing the raster table and summing the number of cells with tasmax values greater than one. The result was a binary dataset with either cells with no days with tasmax values above 38˚C or cells with one or more days with tasmax values above 38˚C. By dividing the number of cells with cell day values greater than one by the total number of cells in the raster, we calculated the change in the spatial extent of cell days with tasmax values above 38˚C. Using Figure 2A4 as an example, there are 6 cells with at least one cell day above 38˚C and thus the spatial extent of temperatures above 38˚C is 67% (6÷9 total cells).

To calculate the change in the percentage of total cell days with values above 38˚C, we first calculated the total number of cell days for each final combined raster by multiplying the total number of cells in the raster by 61, or the total number of days between April 1st and May 31st. Since the final combined rasters represent averaged days above 38˚C across 5 models, we rounded the day values to the nearest whole number to avoid having fractions of days (Figure 5). Next, we used the raster table to calculate the number of cell days with tasmax values above 38˚C. Raster tables generally consist of a “value” column and a “count” column. The “value” in the tasmax rasters used in this analysis represents the number of days above 38˚C and the count is the number of cells in the dataset with that number of days above 38˚C. We multiplied each value by the corresponding count and summed the products to get total number of cell days above 38˚C. Using Figure 2A4 as an example, there are 3 cells with no days above 38˚C, 1 cell with 1 day above 38˚C, 4 cells with 2 cell days above 38˚C, and 1 cell with 3 days above 38˚C. Multiplying each value with its count (0x3, 1x1, 2x4, and 3x1) and summing gives 12 total number of cell days above 38˚C. The total number of cell days in the example is 27 (9 total cells in each raster multiplied by 3 days), and thus the percent of cell days with tasmax values of 38˚C is 44% (12÷27).

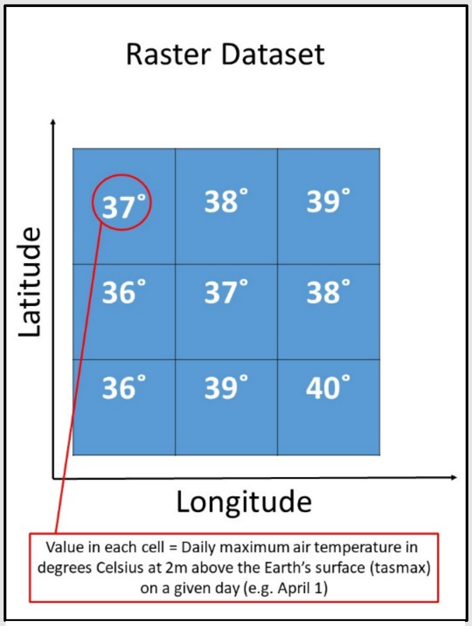

Figure 2A1. A raster dataset is composed of a spatially referenced grid with each grid cell containing data. For this analysis, the data in each cell represents the daily maximum air temperature 2 meters above the surface of the earth.

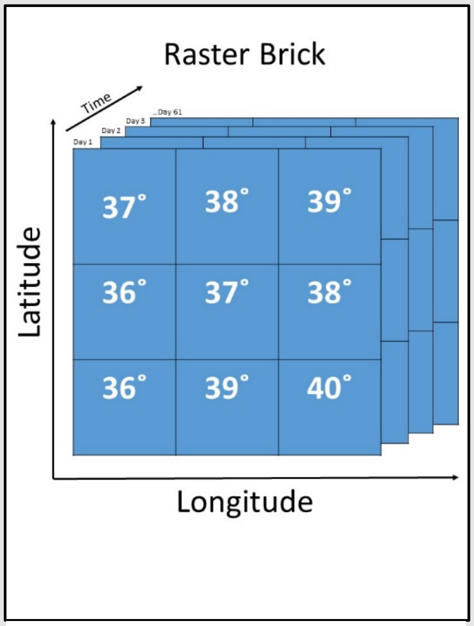

Figure 2A2. A raster brick consists of stacked individual raster datasets. For this analysis, each raster represents a single day between April 1st and May 31st.

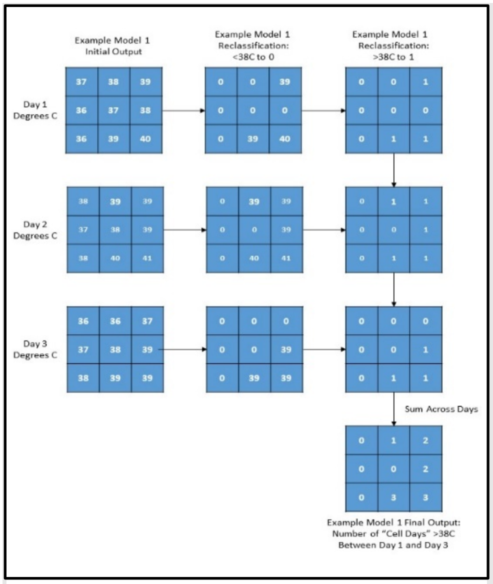

Figure 2A3. Reclassifying the raster dataset and assigning a value of "0" to all cells with tasmax value of 38˚C or lower and a value of "1" to all cells that have a tasmax value of above 38˚C. This resulting value is referred to in this analysis as a "cell day". Summing through the days of a raster brick provides the total number of cell days above 38˚C.

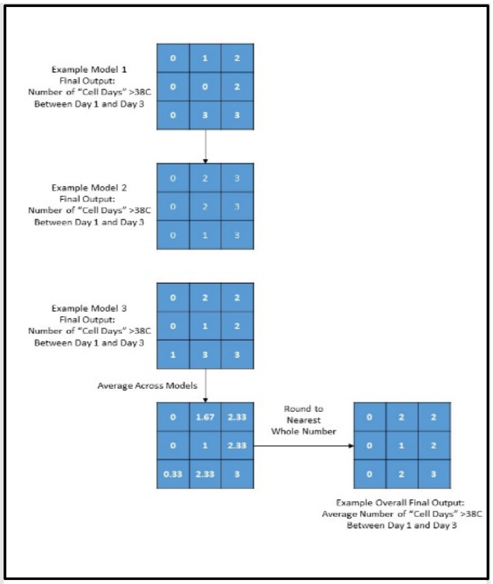

Figure 2A4. Averaging combined model outputs to get the average number of cell days above 38˚C.

Worldwide

We evaluated projections from downscaled General Circulation Models produced by the Coordinated Regional Climate Downscaling Experiment under Representative Concentration Pathways RCP scenarios 4.5 and 8.5 from the Earth System Grid Federation to visually determine if the spatial extent and number of days above the lethal threshold (42˚C) is projected to increase (CORDEX 2018; Cinquini 2014). Where possible, we used bias-adjusted outputs averaged across at least one iteration of each model available to account for variation across models and scenarios. To capture the warmest period for each population, we focused on the July and August timeframe in the northern hemisphere and January and February for Australia and Central America and April and May for Southeast Asia in the southern hemisphere. For Australia and Central America, we were able to average the results over three models; however, downscaled data was only available for scenario RCP 8.5. For the populations in Southeast Asia, we averaged over three models, but only one model output was available for RCP 4.5. We obtained five downscaled and bias-corrected datasets for both RCP 4.5 and 8.5 scenarios for Europe. We also obtained global climate projections from General Circulation Models developed under the Climate Model Intercomparison Project 5 (CMIP5) so we could evaluate projections for all populations more consistently (Taylor 2012). We note that because a population is in the "No Known Risk" risk category does not necessarily mean it has no risk overall (it could be at risk due to one of the influences we were unable to evaluate); rather, it is at no known risk for the two influences that were evaluated.

We acknowledge the World Climate Research Programme's Working Group on Coupled Modelling, which is responsible for CMIP, and we thank the climate modeling groups (Met Office Hadley Centre, Max Planck Institute *for Meteorology, Norwegian climate Centre, Centre National de Recherches Meteorologiques / Centre Europeen de Recherche et Formation Avancees en Calcul Scientifique, European EC-EARTH Consortium, Institut Pierre-Simon Laplace, Canadian Centre for Climate Modelling and Analysis, Centro Euro-Mediterraneo per I Cambiamenti Climatici, Atmosphere and Ocean Research Institute (The University of Tokyo), National Institute for Environmental Studies, and Japan Agency for Marine-Earth Science and Technology, National Center for Atmospheric Research) for producing and making available their model output. For CMIP the U.S. Department of Energy's Program for Climate Model Diagnosis and Intercomparison provides coordinating support and led development of software infrastructure in partnership with the Global Organization for Earth System Science Portals.*

Appendix 3. Additional Results

[1] Percent change in area and average number of days above 38°C and 42°C

Table 3A1. Projected 2012 (May and April) baseline total number of 4.6km x 4.6km grid cells and average number of cells with at least one day above 38˚C and 42˚C under RCP 4.5 and 8.5.

Population Unit	Total Number of 4.6km^2 Raster Cells	Average Number of Cells w/at Least 1 Day			
		>38˚C Under RCP 4.5 (Apr-May 2012)	>38˚C Under RCP 8.5 (Apr-May 2012)	>42˚C Under RCP 4.5 (Apr-May 2012)	>42˚C Under RCP 8.5 (Apr-May 2012)
Eastern					
Northcentral	134,563	3,845	0	67	0
Northeast	23,445	0	0	0	0
South	147,796	32,573	35,446	100	2,777
West	161,501	29,085	24,983	10,452	7,403

Table 3A2. Percent change in the area and average cell days above 38° C for each conservation unit under RCP 4.5 and 8.5 from April and May of 2012 to 2069.

Population Unit	% Change in Area RCP 4.5	% Change in Area RCP 8.5	% Change in Cell Days RCP 4.5	% Change in Cell Days RCP 8.5
Eastern				
Northcentral	-99	1,008,000	-99	1,008,800
Northeast	28,400	16,900	28,400	16,900
South	94	200	331	438
Western	-23	109	38	114

Table 3A3. Percent change in the area and average cell days above 42° C for each conservation unit under RCP 4.5 and 8.5 from April and May of 2012 to April and May of 2069.

Population Unit	% Change in Area RCP 4.5	% Change in Area RCP 8.5	% Change in Cell Days RCP 4.5	% Change in Cell Days RCP 8.5
Eastern				
Northcentral	-99	30,000	-99	30,000
Northeast	0	0	0	0
South	6,630	1,637	8,147	3,575
Western	-11	148	11	182

[2] Projected area and average number of days >38˚C and 42°C under RCP 8.5

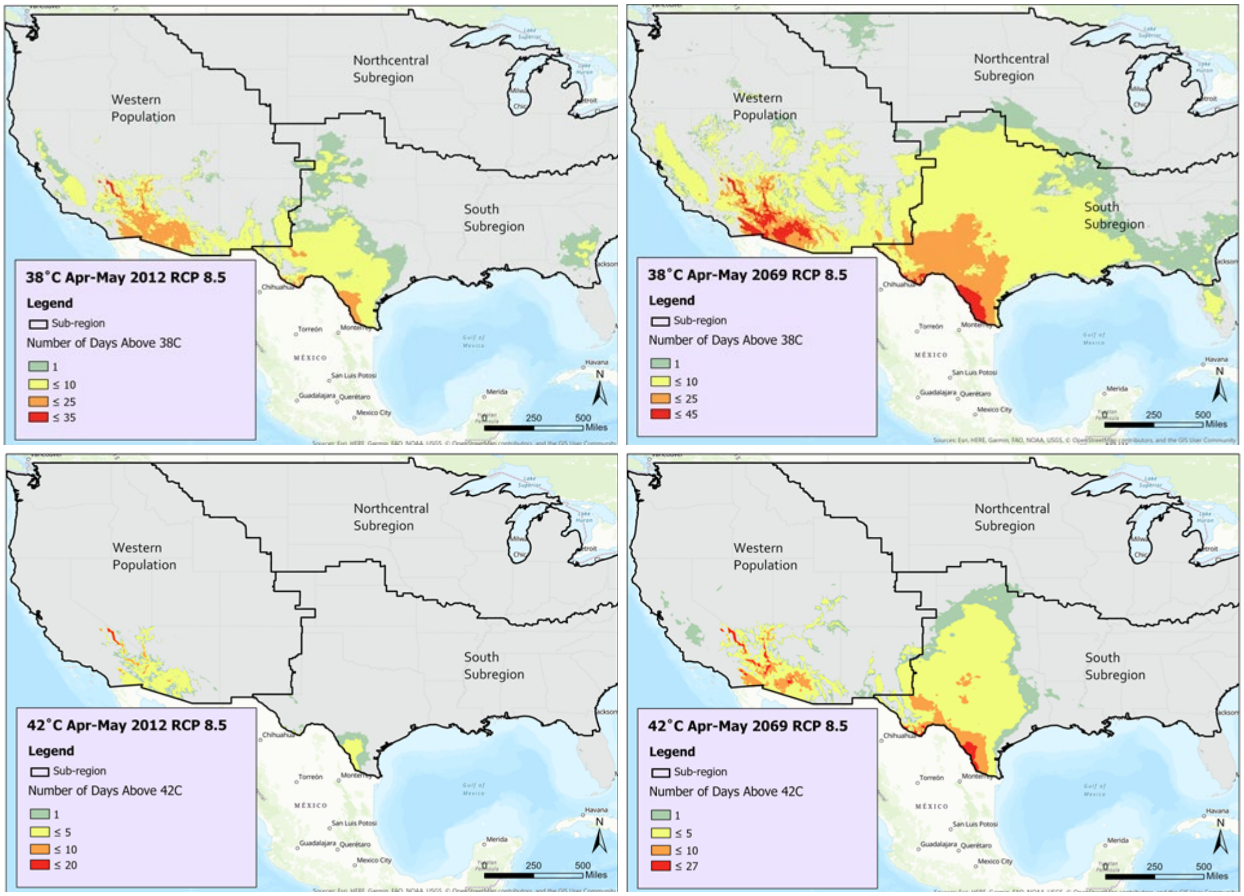

Figure 3A1. The spatial extent and average number of days >38˚C (top) and 42°C (bottom) in April and May 2012 (left) and 2069 (right) under RCP 8.5.

[3] pE over time under current and future state conditions

Table 3A4. pE values for the western and eastern North American populations. pE predictions under current state conditions represent the 50% confidence interval.

	10 Year	20 Year	30 Year	40 Year	50 Year	60 Year
Western Pop						
Current - 25%	0.60	0.80	0.90	0.95	0.97	0.99
Current - 75%	0.68	0.85	0.93	0.97	0.98	0.99
Future - Worst case	0.71	0.88	0.95	0.97	0.99	0.99
Future - Best case	0.66	0.84	0.92	0.96	0.98	0.99
Eastern Pop						
Current - 25%	0.02	0.09	0.18	0.29	0.39	0.48
Current - 75%	0.08	0.22	0.36	0.49	0.60	0.69
Future - Worst case	0.09	0.29	0.46	0.58	0.67	0.75
Future - Best case	0.04	0.13	0.24	0.35	0.46	0.56
p(both pops persist)						
Current - 25%	0.39	0.18	0.08	0.04	0.02	0.01
Current - 75%	0.29	0.12	0.04	0.02	0.01	0.00
Future - Worst case	0.27	0.08	0.03	0.01	0.00	0.00
Future - Best case	0.33	0.14	0.06	0.02	0.01	0.00