Monarch Butterfly (*Danaus plexippus*)
Species Status Assessment Report, version 2.3
December 2024


Photo: Kelly Nail

Prepared by:

U.S. Fish and Wildlife Service

Acknowledgments

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Suggested Reference

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Note on Document History

In 2020, the U.S. Fish and Wildlife Service (Service) conducted a species status assessment (SSA) for the monarch butterfly. The SSA provides the biological support for the decision on whether the monarch warrants listing under the Endangered Species Act. On December 17, 2020, we announced that listing the monarch butterfly as an endangered or threatened species under the Endangered Species Act was warranted but precluded by higher priority actions. Starting in 2023, we (a team of Service biologists) evaluated information that had become available since 2020, updated the SSA report to reflect new information relevant to the species status, and shared the updated assessment with peer reviewers. Following receipt of the peer reviewer feedback, which included additional literature references as well as questions and suggestions, we further refined the report incorporating peer review suggestions. A description of changes made from one document version to the next is found in Appendix 6.

Executive Summary

The monarch (*Danaus plexippus*) is a species of butterfly well known for its phenomenal long-distance migration in the North American populations. The two North American 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 to list the monarch butterfly for protection under the Endangered Species Act of 1973, as amended.

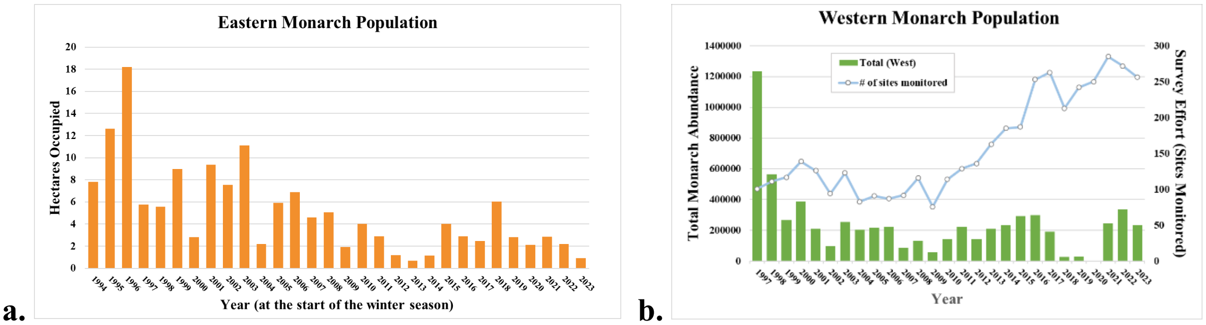

Figure E1. Eastern (a) and western (b) North American monarch population sizes, as measured at overwintering sites in terms of hectares (eastern) and total number (western). The western population count also has a blue line indicating survey effort (number of sites monitored).

Migratory monarchs in North America are the ancestral population for all other monarch populations, and more than 90 percent of monarchs worldwide occur in the North American migratory populations. Historical records suggest that monarchs dispersed from North America in the past two centuries and now occur in non-native or naturalized populations throughout 90 countries, islands, and island groups where milkweed (their larval host plant) was already present or introduced.

Using the best available scientific information about monarchs in and beyond North America, 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).

The primary drivers affecting the health of the two North American migratory populations are 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 (see Figure E1). Using the best available science, we estimated the probability of the population abundance reaching the point at which extinction is likely inevitable (*pE*) for each population given their current abundance and growth rate, as well as under projected future conditions. The *pE* for the eastern population is high (less than 10% chance of extinction within 10 years, 56% to 74% chance within 60 years). The *pE* for the western population is higher (60% to 68% chance of extinction within 10 years, reaching 99% by year 60) under current conditions and increases under projected future conditions. The range in the estimates represents the best and worst plausible 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 fully captured in the *pE* estimates and could increase the probability of extinction over the timeframes modeled. 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 are also not fully captured in the *pE* estimates (see Chapter 8: Viability).

For monarchs outside of the two North American migratory populations, 69 of the 90 countries, islands, and island groups are considered currently extant. Monarch presence within the remaining 21 countries, islands, and island groups has not been confirmed since 2000, but the best available information does not indicate they are extirpated, and thus they are all presumed extant. Of the 29 populations outside of North America, 25 are confirmed extant, and the remaining 4 are presumed extant.

In the future, we found that 15 of the 29 nonmigratory or dispersed populations are at risk due to threats associated with climate change (6 due to sea level rise and 9 due to unsuitably high temperatures). The populations susceptible to sea level rise are at risk of losing at least some of their monarch habitat; thus, redundancy could decrease with the loss of those areas. However, the best scientific and commercial data available do not indicate if populations susceptible to high temperatures are at risk of losing all or just a portion of their monarch habitat; thus, they may continue to contribute to redundancy. Therefore, we conclude that rangewide the species is likely to maintain considerable redundancy through continued presence in an estimated 84 of the 90 countries, islands, and island groups where it occurred historically or to where it has dispersed. We anticipate the species will continue to have adaptive capacity (representation) through its presence over a large geographical range where the climatic conditions and habitat vary widely.

The extinction of either the western or eastern North American migratory 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 populations 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.

Our analysis estimates that the chance of *both* populations persisting above the extinction threshold over the next 10 years is 27% to 33% (under plausible future scenarios) and drops to 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

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
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
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 (Service) 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, the Service 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 Service 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 (Figure 1.1). 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 appendices providing further information on taxonomy, methodology, results, other drivers considered, insecticides and herbicides, and the modeling workflow.

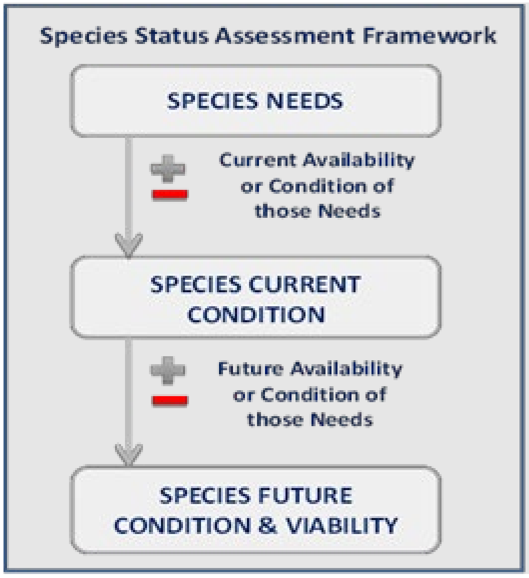


Figure 1.1. Conceptual diagram of SSA analytical framework.

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). 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 a species to withstand environmental stochasticity (normal, year-to-year variations in environmental conditions, such as temperature or rainfall), periodic disturbances within the normal range of variation (fire, floods, storms), and demographic stochasticity (normal variation in demographic rates such as mortality and fecundity) (Redford et al. 2011, p. 40). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions.

Redundancy is the ability of a species to withstand catastrophes. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health and for which adaptation is unlikely (Mangal and Tier 1993, p. 1083). 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.

Representation is the ability of a species to adapt to both near-term and long-term changes in its physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (pathogens, competitors, predators, etc.) environments. This ability to adapt to new environments, referred to as adaptive capacity, is essential for viability 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) 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).

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).

Chapter 2: Species Ecology

This chapter describes the ecological requirements (or needs) for survival and reproduction at the individual, population, and species levels (the first step of our analytical framework; see Figure 1.1).

Species Description

The monarch (*Danaus plexippus* (Linneaus, 1758)) is a species of butterfly in the order Lepidoptera (family Nymphalidae). 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, Service.

Taxonomy and Petition

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 pupates into a 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; in migratory populations the 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). Monarchs cannot develop in temperatures lower than 12°C (53°F) or higher than 33°C (91°F) (Zalucki 1982, pp. 242, 245). Monarchs at all life stages are impacted by diseases and natural enemies (see Appendix 3 for more information). The protozoan parasite *Ophryocystis elektroscirrha* (OE) is especially well-studied and infection rates among migratory and non-migratory monarchs in different areas are further discussed below.

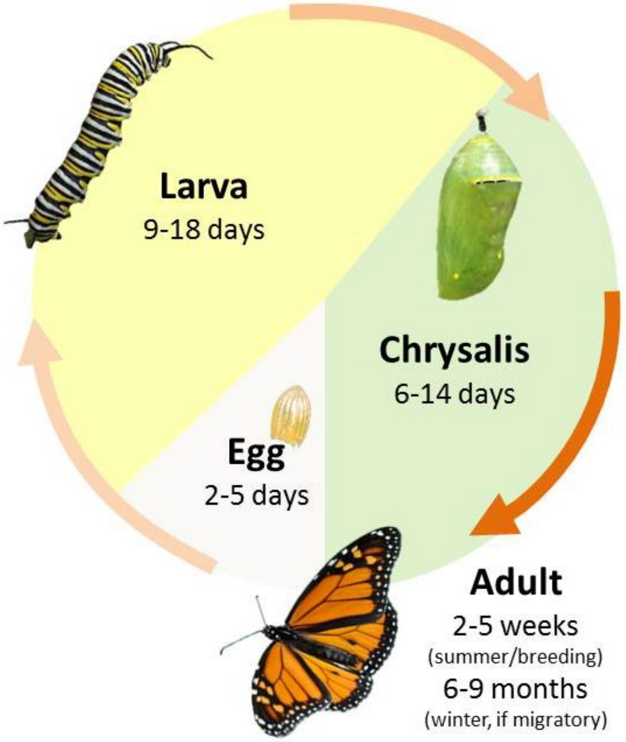


Figure 2.1. Monarch life cycle. Development times calculated from Zalucki (1982) based on temperatures ranging from 22°–32°C (72°–91°F). Adult life span based on Herman and Tatar (2001).

The monarch life cycle varies by geographic location. Migratory monarchs in North America are the ancestral population for all other monarch populations (Pierce et al. 2014a, p. 4; Zhan et al. 2014, p. 318; Hemstrom et al. 2022, pp. 4551–4552), which dispersed from North America via human assistance, potentially aided through wind dispersal events (Brower 1995, p. 354), and now occur in Central and South America; Australia; New Zealand; islands of the Pacific and Caribbean, and elsewhere (Malcolm and Zalucki 1993, pp. 3–5) where milkweed (their larval host plant) was already present or introduced. In a few parts of North America and in most areas outside North America, monarchs breed year-round, repeatedly following the above-referenced life cycle throughout the year. However, monarchs in temperate climates such as in eastern and western North America undergo long-distance migration, where the migratory generation of adults enters 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 and 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 distribution of individual monarchs are monthly mean temperature, precipitation, and land use (Flockhart et al. 2013, p. 4; Flockhart et al. 2017, p. 2570; Kesler and Bunch 2020, p. 60). 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, entire; 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).

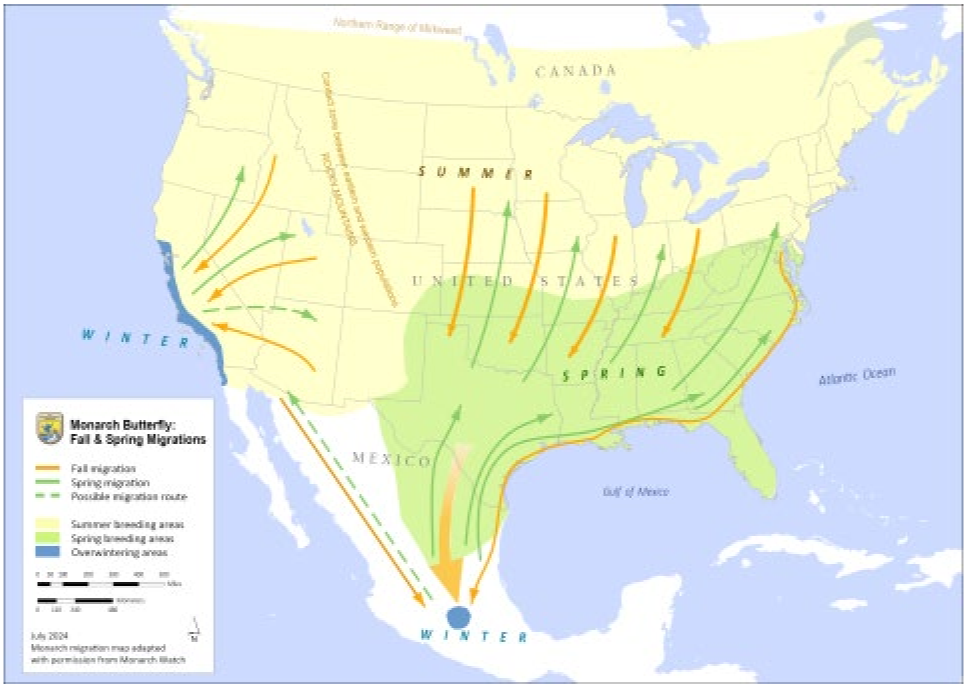


Figure 2.2. North American monarch migration map.

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 overwintering 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). 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 are not fully understood. Roosting sites may also be important for monarchs along their fall migration route.
All	Suitable climatic conditions	Temperatures, precipitation levels, and other climatic conditions supportive of life processes (e.g., breeding, migrating, and overwintering).

Population-Level Ecology

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. The ecological requirements of a healthy monarch population include population growth rate and population size, connectivity, and habitat. These are discussed below and summarized in Table 2.2.

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. 345–346). 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, populations must be healthy enough to withstand large swings in population sizes (N). Specifically, they need a robust population growth rate (lambda, or λ) and sufficient carrying capacity. Population growth rates greater than 1 indicate growth, while values less than 1 indicate decline (see Table 2.2). 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, quality, and distribution 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. Large population sizes may also be important for mate finding and warning coloration effectiveness (Malcolm 2018, pers. comm.). 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). Monarchs need a matrix (distribution) of habitat to support connectivity and timing of their movements during the breeding and migratory seasons (Crone and Schultz 2022, p. 173; Grant et al. 2022, p. 1199; Solis-Sosa et al. 2021, p. 14).

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 range to ensure local and landscape-scale connectivity and maximize lifetime fecundity (Zalucki and Lammers 2010, p. 84; Miller et al. 2012, p. 2; Crone and Schultz 2022, p. 173). However, the specific optimal amount of habitat and its spatial distribution are not fully understood; 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. 535; 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, overwintering resources to support large population sizes, and appropriate climatic conditions.
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 for species viability 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 populations with a robust growth rate, large effective populations size, and suitable quality, quantity, and distribution of habitat conditions 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. Given the monarch's sensitivity to environmental conditions, which can cause 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, enhances species' resiliency.

Redundancy

Monarch redundancy is best achieved by having a broad geographic distribution 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, having monarchs occupying areas of unique diversity will guard against losses of adaptive capacity due to catastrophic events.

Over 90 percent of monarchs conduct the annual North American long-distance fall migration to concentrated overwintering areas in Mexico and California. In the breeding season each year, these monarchs return to vast and diverse habitats across much of North America. Non-migratory monarchs remain year-round at the southern end of their breeding range in North America, including in parts of Florida, the Gulf Coast, and California. A very small proportion of the species additionally occurs in non-native or naturalized populations across even more diverse habitats throughout 90 countries, islands, and island groups.

Representation

The monarch's ability to withstand novel changes in its environment is influenced by its adaptive capacity, which is primarily a function of the species' breadth of variation in biological traits and genetic diversity. 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). Below we describe monarch adaptive capacity by using the best available data.

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 broadly distributed. Also, functional evolutionary processes via ensuring populations occupy an array of environments, maintaining genetic connections, and ensuring large N_e.

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; Hemstrom et al. 2022, pp. 4551–4552). Their unique genetics separate them from non-migratory monarchs (see Freedman et al. 2021, p. 7). Genetic sampling indicates that the monarchs from eastern and western North America form an admixed population (Lyons et al. 2012, p. 3441) with continued interchange between the two populations contributing to low genetic differentiation (Talla et al. 2020, p. 2573; Freedman et al. 2021, pp. 7–8). During the annual breeding season, the North American migratory populations use a vast and diverse array of habitats. Following a long-distance migration through more varied habitats and conditions, the North American migratory monarchs use sites with a diversity of physical structures and climatic conditions for overwintering. Non-migratory monarchs represent a life history strategy that exposes them to selection pressures that differ from the migratory monarchs.

Eastern North American monarchs undergo long-distance migration every fall, a behavior that differentiates this population from non-migratory populations or from migratory populations that fly shorter distances and to different locations. The migratory phenotype of monarchs in the eastern migratory population is distinct from monarchs in other populations that may have latent migratory phenotypes (Tenger-Trolander et al. 2019, p. 14673). 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). Monarchs from the eastern North American migratory population 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).

Western North American monarchs also migrate long distances, although their migration is shorter than monarchs in eastern North America. 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.

Recent genetic fingerprinting and anecdotal evidence suggests that some monarchs from eastern North America migrate through Florida to Cuba and the Yucatán Peninsula (Dockx et al. 2023, pp. 314–317; see also *Alternative Overwintering Strategies* in Chapter 8). Monarch tagging efforts have yet to confirm this migration route (Howard and Davis 2009, p. 284). While we assume most monarchs found in southern Florida are non-migratory, it is possible that some are migratory.

Western North American migratory monarchs' range encompasses as much as 30% of the total North American monarch range (Dilts et al. 2019, p. 11). Western monarchs use ecologically different breeding, migrating, and overwintering habitats (Brower et al. 1995, p. 542) compared to monarchs in eastern North America. 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, entire). 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).

Migratory monarchs in eastern North American have larger wings compared to migratory monarchs in western North America (Altizer and Davis 2010, p. 1025; Talla et al. 2020, p. 2572). Within the eastern North American population, 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).

Eastern monarchs migrating to Mexico have higher lipid reserves than those overwintering in California (Brower et al. 1995, p. 542) and may have a longer diapause compared to western North American monarchs (Herman et al. 1989, pp. 56–57). Monarchs from eastern North America 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 areas (Brower et al. 1995, p. 542).

In addition to differences in migratory behavior and habitats occupied, western migratory North American monarchs also differ from eastern migratory North American monarchs 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 (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).

Compared to monarchs in western North America and to non-migratory monarchs in southern Florida, eastern North American monarchs have lower rates of infection by the protozoan parasite *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). 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).

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 migratory monarch population (Knight and Brower 2009, p. 821; Zhan et al. 2014, p. 322). 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 migratory population (Altizer 2001, p. 622). For additional information, see Appendix 3 ("*Other Stressors*" section).

In the West, the population of migratory monarchs has dropped from several million butterflies in the 1980s (Schultz et al. 2017, p. 345). Concurrently in the West, a portion of non-migratory monarch butterflies in urban gardens has been growing (Crone and Schultz 2021, entire). The increase in numbers of these non-migratory monarchs do not seem to make up for the decline of the migratory population (Crone and Schultz 2021, entire). Additionally, the non-migratory portion also probably lacks the demographic capacity to expand its range inland during summer months. Non-migratory monarch butterflies build up high levels of a protozoan parasite *OE*, at least in part because of a lack of migratory culling and migratory escape (Satterfield et al. 2015, pp. 4–5). In California, about 8% of migratory monarch butterflies are infected with *OE*, compared to about 75% of non-migratory individuals (Satterfield et al. 2016, p. 346). *OE*-infected monarch butterflies have both lower survival and lower egg-laying rates, and only produce about 0.8 adult daughters per female on average (Crone and Schultz 2021, Supplement S4). While this rate of increase is enough for non-migratory monarch butterflies to persist in urban areas, it does not allow them to rapidly expand to other western states (Crone and Schultz 2021, p. 1536).

For information about the genetic structuring of the dispersed monarch populations outside North America, please see Appendix 2.

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 rangewide, 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) modeling to forecast 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).

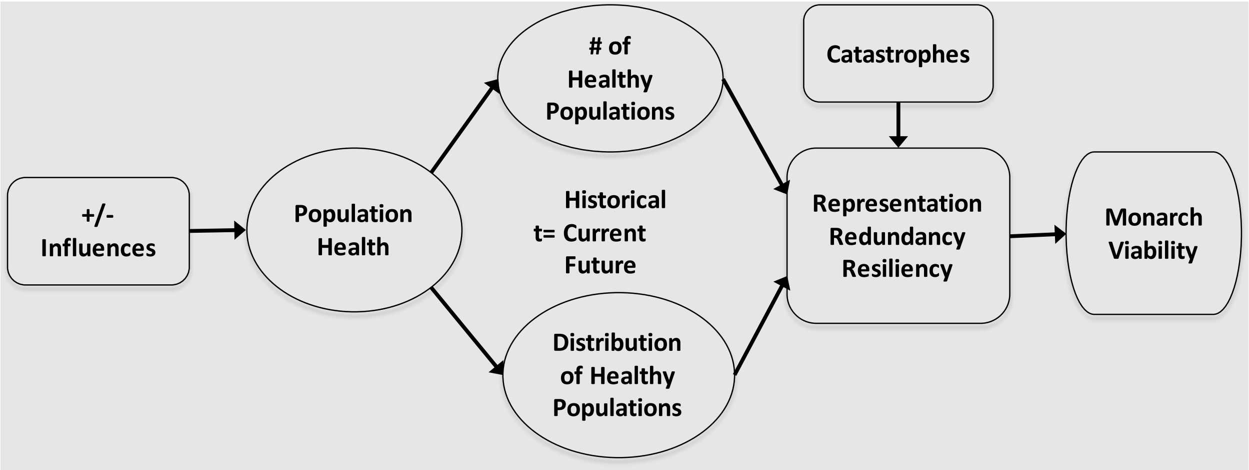


Figure 3.1. Simplified conceptual diagram depicting the analytical framework for assessing monarch viability over time.

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 (See Appendix 2).

To assess population health, we sought out information on historical population abundance (*N*) and population growth rate (λ). Over 90 percent of monarchs worldwide occur in the North American migratory populations. Population size (*N*) estimates for these populations 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 (Rendón-Salinas et al. 2023) and the western North American population has been monitored since 1997 at coastal overwintering sites in California (Xerces Society for Invertebrate Conservation 2024). The historical population growth rates (λ) for eastern and western North American migratory 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 monarchs populations other than the North American migratory populations, there are no systematic, multi-year surveys for any time period. We examined the published literature to determine the historical distribution of the monarch butterfly populations.

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 and to the extent that available information allowed 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 dispersed populations outside of North America, 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 migratory populations to very limited data (occurrence only) for the other populations. Using 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.

Eastern & Western North American Populations

Over 90 percent of monarchs worldwide occur in the North American migratory populations. There is 20+ years of standardized survey data from which we can derive current abundance and population trend (λ) for eastern and western North American migratory monarch populations based on annual estimates of overwintering population levels. Stochastic geometric growth models have been published for both eastern (Semmens et al. 2016) and western (Schultz et al. 2017) migratory populations. These statistical models produce estimates of extinction risk based on population size, trajectory, and year-to-year variability. However, published attempts at quantifying extinction risk (e.g., Semmens et al. 2016, Schultz et al. 2017, and Thogmartin 2024) derive their estimates from past and current conditions and thus do not incorporate the influence of changing future conditions on extinction risks. Changing drivers, such as habitat loss, climate change, or conservation efforts, can have a major influence on future vulnerabilities (e.g., Zylstra et al. 2022) and thus are a necessary aspect of a species status assessment. Therefore, in our analysis we used a similar statistical modeling framework that we modified to allow for the projection of a range of changing future conditions (i.e., threats and conservation efforts). We briefly describe our modeling framework here (Fig 3.2); for additional detail see Voorhies et al.

(2019) and see Appendix 3 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.

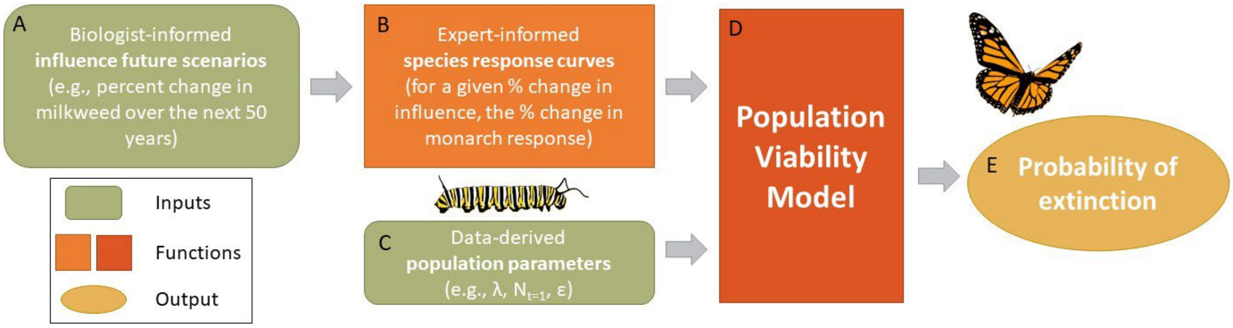


Figure 3.2. An overview of the monarch modeling framework. Biologist-informed scenarios (A) represent expected range in % change in a given influence over time. Expert-elicited population response curves (B) specific to each influence provide the proportional change in monarch response given a proportional 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 likely 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 serves as the primary mechanism for incorporating the consequences of Allee effects and environmental stochasticity at small population sizes. In addition to the extinction threshold, we applied 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, pp. 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 lack of additional nutrients from multiple matings [Wells et al. 1993, p. 66]) and
- Inability of small population sizes to rebound from sustained threats (Hutchings 2015, p. 6) or natural environmental variation (e.g., poor weather years) (Thogmartin 2024, entire)

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, reported as area occupied by overwintering monarchs in Mexico. 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). Given the western population rebound observed from below 2,000 individuals in 2019–2020 to nearly 250,000 individuals the following winter, these estimates may be conservative. Much uncertainty remains about the size of extinction thresholds and the time lag necessary to confirm assumptions about population stability or extinction trajectories. 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).

Monarch butterfly populations, like other insects, naturally fluctuate from year to year in response to the temperature, rainfall, the availability of food, and other factors. However, the western migratory monarch population has undergone a sustained and significant decline and is now fluctuating to a greater degree than previously observed (Pelton et al. 2021, unpaginated; see figure 4.5). It is plausible that migratory monarchs experienced good weather that resulted in abundant and well-timed milkweed and nectar resources across their breeding habitat in the West in 2021, which provided for a significant increase in the overwintering count from below 2,000 individuals in 2019–2020. Smaller populations can also experience reduced competition for resources which can help them bounce back to higher numbers under favorable conditions. Other hypotheses include that western monarchs could be overwintering across the broader landscape in lower densities that were not detected, or monarchs in northern parts of the overwintering range could instead now be breeding in the winter instead of aggregating at overwintering sites where they would be counted (Taylor 2023, pers. comm.). With only a few years of improved population numbers, it is impossible to say whether this is the western migratory population's status improving or more data supporting the continuation of its fluctuation and decline (Howard and Pelton 2022, unpaginated).

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 3.

Dispersed and Non-Migratory Populations

To assign persistence status to the dispersed and non-migratory monarchs, 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 dispersed 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 area. 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 dispersed and non-migratory populations. Unknown indicates insufficient information about habitat quality, quantity, and 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 dispersed and non-migratory 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.

Viability (Ch. 8)

We used the results from our current and future forecasts 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. 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 dispersed 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, entire). Given scale and magnitude of impact (whether monarchs 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 dispersed 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 assessed the likelihood of monarchs persisting across the species' current range (and the corresponding probability of extinction) given the forecasted influences and catastrophes. Specifically, for the eastern and western North American migratory monarchs, we used the results of our population modeling to predict the likelihood of persistence or extinction of eastern and western migratory monarch populations over the next 60 years (see Appendix 3 for details on modeling methodology). For the dispersed populations, we qualitatively express the likelihood of persistence or extinction 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.

Among monarch populations distributed worldwide, the ancestral eastern North American monarch population represents by far the largest proportion of individuals (even accounting for large variation in estimates; Figure 4.1).

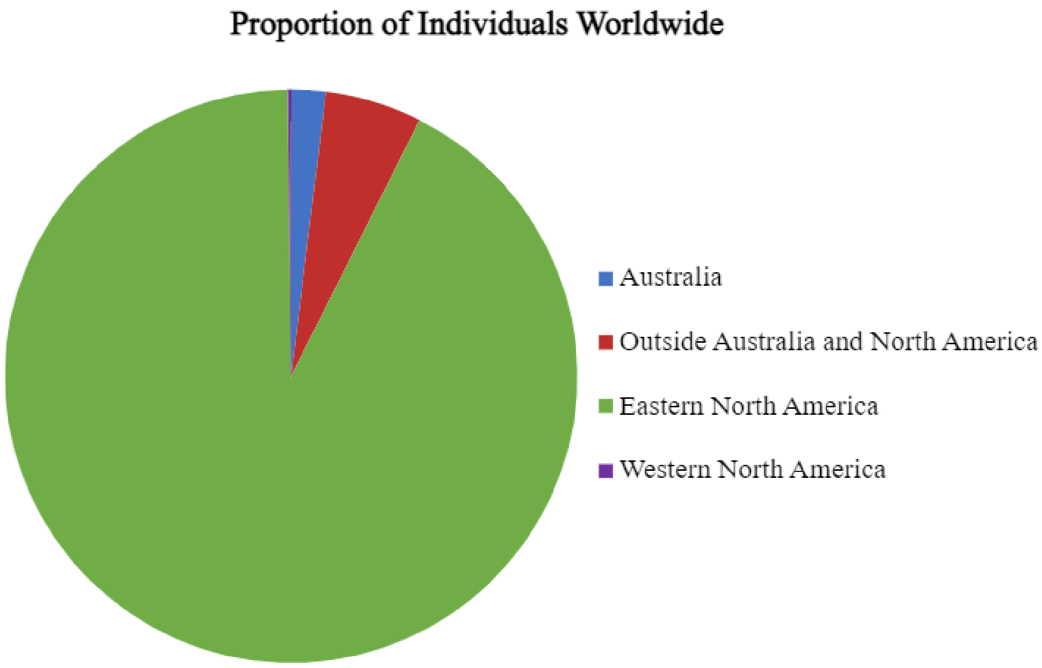


Figure 4.1. Estimated relative proportion of individual monarchs by geographical area. The numbers are based on the following: eastern North America (67,646,600; based on average of last five years overwintering estimates, assuming a 21.1 million monarch/ha density), western North America (128,357; based on average of past five years of overwintering counts); Australia (1,424,790; based on estimates from M. Zalucki 2017, pers. comm.); and outside of Australia and North America (4,000,000; based on 3–5 million monarch estimate; M. Zalucki 2017, pers. comm.).

Eastern North American Population

The eastern North American migratory monarch population has been systematically censused annually since 1994 (Figure 4.2; 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 likely generally comparable in size and may have been higher, though early estimates are difficult to compare due to different methods and incomplete coverage across overwintering sites (Vidal and Rendón-Salinas 2014, p. 172).

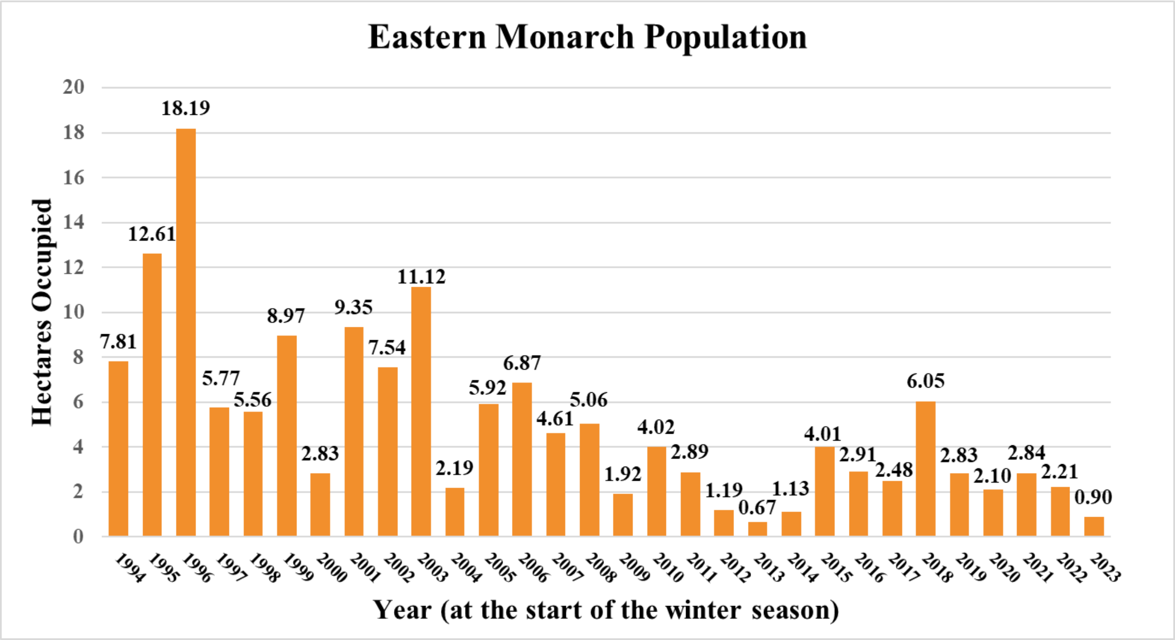


Figure 4.2. Area occupied (in hectares) by eastern North American monarch butterflies at overwintering sites in Mexico (actual hectare measurement displayed above each bar). Year displayed is the beginning year for the winter (e.g., 2017 represents the number for the winter of 2017–2018). Data from Rendón-Salinas et al. 2023.

Based on genomic data to reconstruct demographic histories for monarch and common milkweed (*A. syriaca*), monarchs and milkweed experienced significant population expansion roughly 10,000–20,000 years ago, aligning with the most recent period of North American glacial recession, then increased again roughly 200 years ago, aligning with extensive human-caused landcover change predominated by deforestation and agricultural expansion (Boyle et al. 2023, entire). However, genomic analyses did not detect effective population declines over the past 40 years.

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). These estimates were for overwintering abundance in coastal California from 1981–2015. The researchers used state-space models to estimate the western monarch population growth rate from spatially and temporally erratic sampling data going back to the early 1980s and found annual abundance estimates were high in the 1980s, fluctuating in the 1990s, and low in the 2000s.

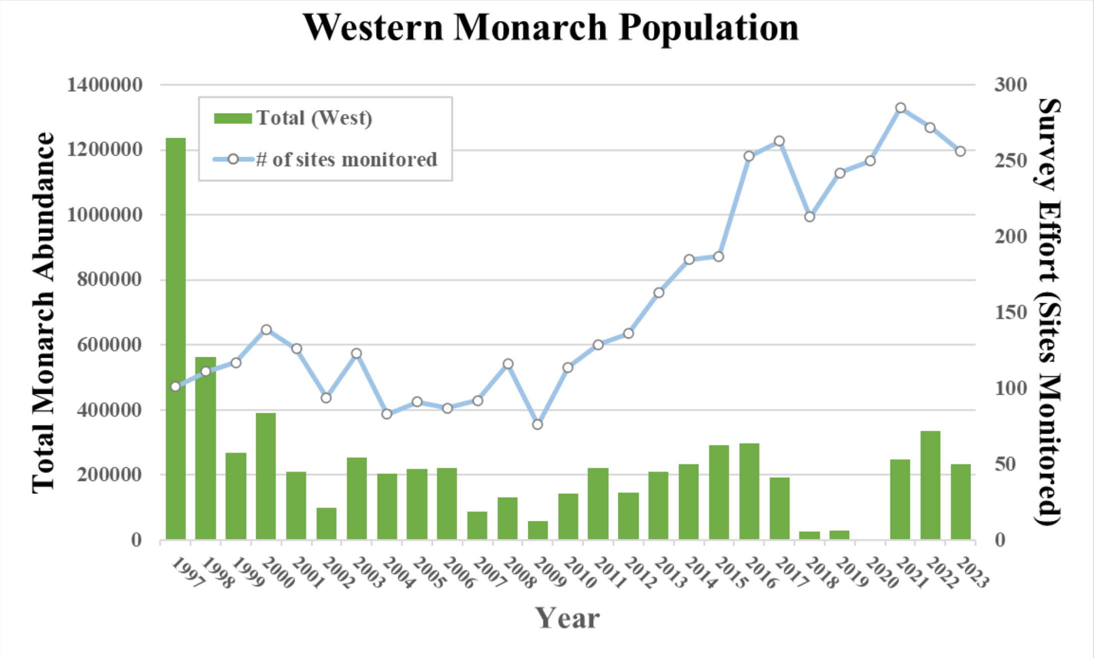


Figure 4.3. Thanksgiving counts showing the number of western North American monarch butterflies observed at overwintering sites (green bars). Blue line shows the number of sites monitored (survey effort) for a given year. Data from The Xerces Society for Invertebrate Conservation (2024, entire).

Dispersed and Non-Migratory Populations

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 4.4). We found occurrence records of monarchs in 90 countries, islands, or island groups. For our analyses, we grouped these occurrences into populations (See Appendix 2, Table 2A.2).

We lack abundance estimates for the non-migratory monarchs that reside year-round in southern Florida and along the California coast.

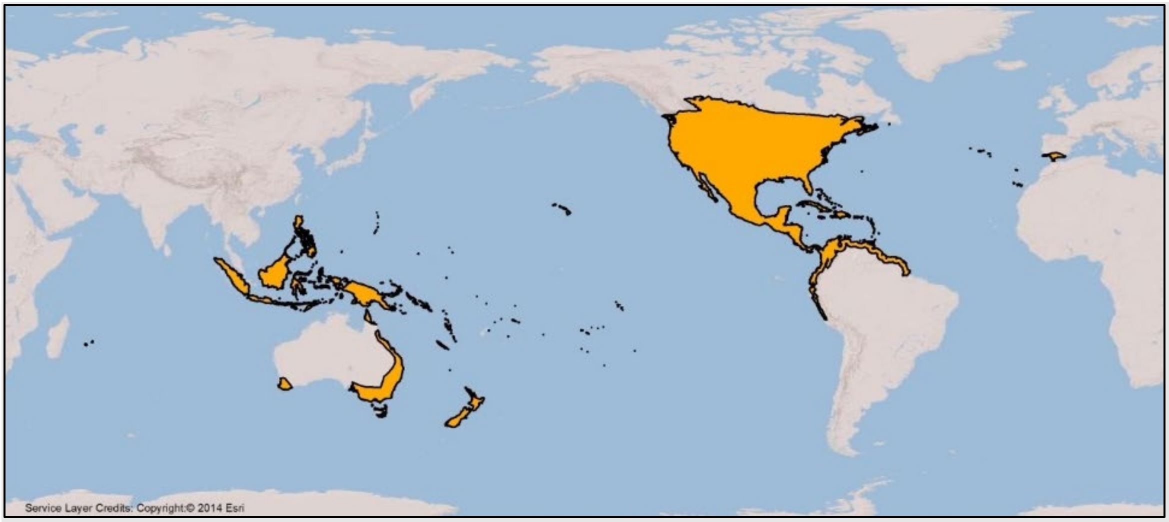


Figure 4.4. Map showing global range of monarchs (orange shows known range).

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

This chapter describes the current condition of the species, including the status and health of monarch populations and forecasted probability of extinction based on those current conditions. The chapter further describes the influences, including threats and conservation efforts, that have led to this current condition and extinction risk under current conditions.

Eastern North American Population – Current Condition

Based on the past annual censuses, the eastern North American migratory monarch population has been generally declining over the last 29 years (Figure 4.2). 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 estimated probability of extinction (*pE)* over the next 60 years is 61% (48%–69%; CI 50%) (Figure 5.1).

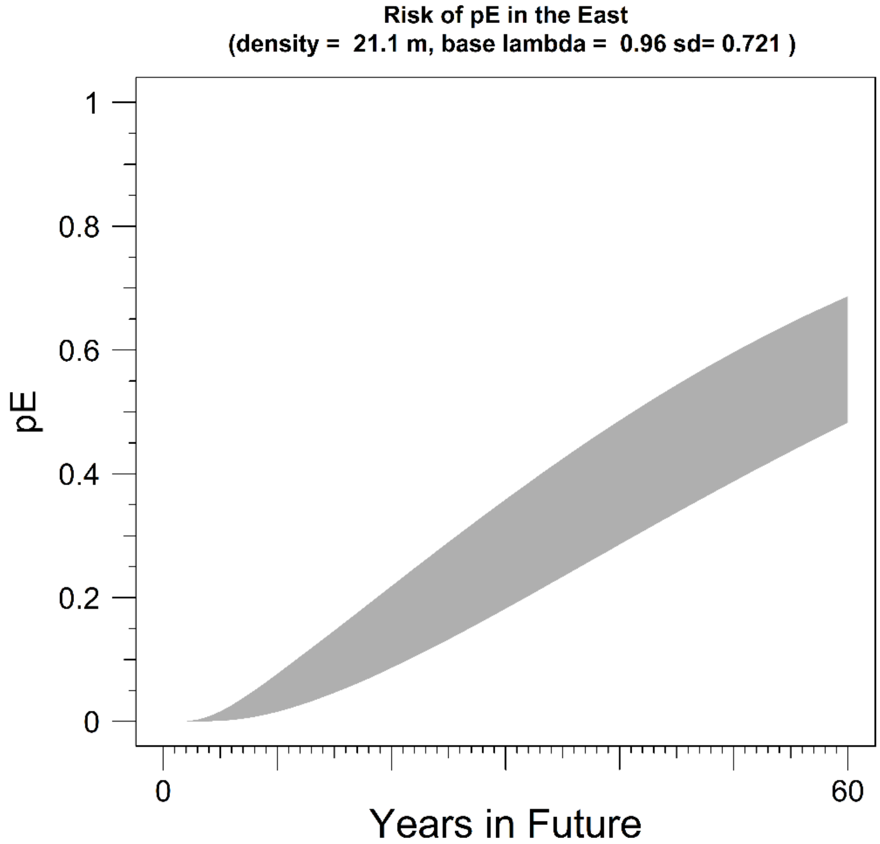


Figure 5.1. pE *for the eastern North American monarch population over time, represented by 50% confidence interval (gray space). Probability based on current trend in growth.*

Western North American Population – Current Condition

Based on the past annual censuses, the western North American migratory population has been generally declining over the last 26 years, despite an increasing number of sites being counted (Figure 4.3). Our model indicates that under current conditions the risk of extinction over time is predicted to increase sharply, with the estimated *pE* over 60 years reaching 99% (98%–99%, CI 50%) (Figure 5.2).

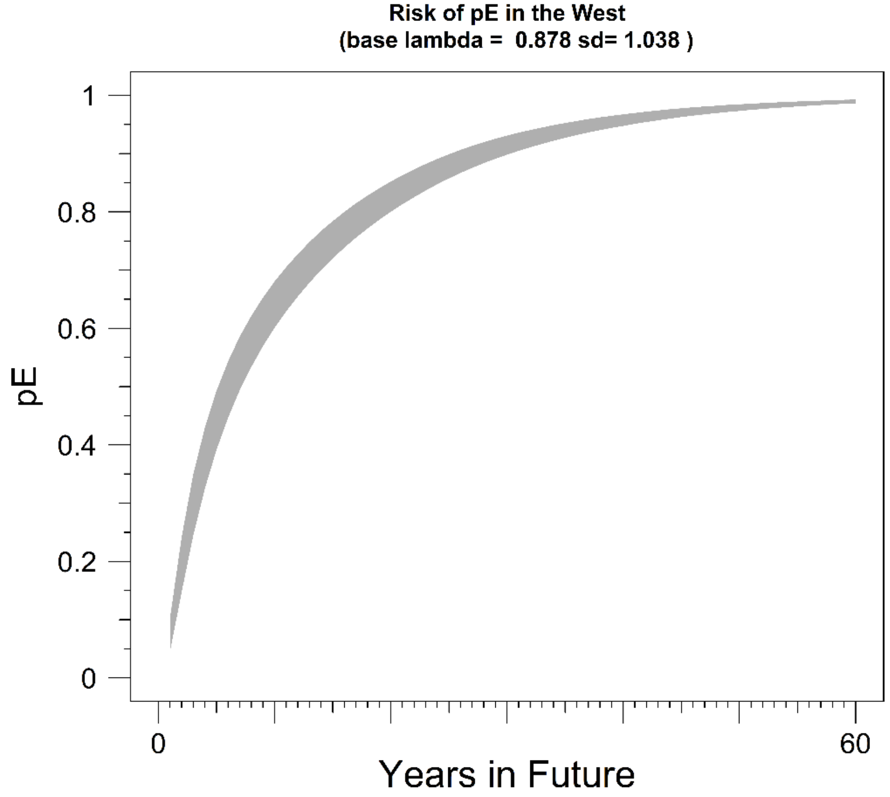


Figure 5.2. pE *for the western North American monarch population over time, represented by 50% confidence interval (gray space). Probability based on current trend in growth.*

North American Populations – Current Influences

To identify and evaluate the factors that are influencing the modeled *pE* predictions under current conditions for the eastern and western North American populations (Figures 5.1 and 5.2), we conducted expert elicitation with monarch experts (see Voorhies et al. 2019, p. 4). This elicitation informed our identification of the important factors driving population dynamics for each population (Tables 5.1 and 5.2). We then summarized the factors that are most highly influential on the two populations in an influence diagram (Figure 5.3). 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. Below, we discuss the key influences on monarch populations—the aforementioned stressors as well as the beneficial contributions of 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)

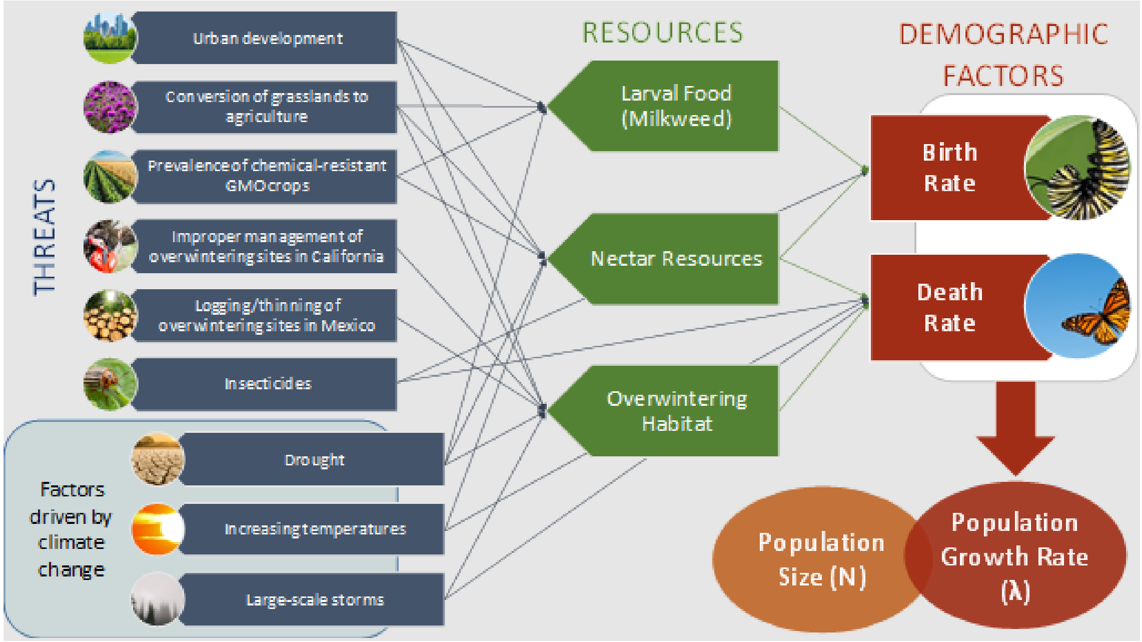


Figure 5.3. Influence diagram showing the key Influences and how they drive monarch population abundance (N) and growth rate (lambda, λ). Note, conservation efforts can decrease all the listed threats and improve all resources for monarchs.

<u>Availability, Distribution, and Quality of Milkweed</u>

The availability of milkweed is essential to monarch reproduction and survival. Reduction in milkweed is cited as a key driver in monarch declines (Brower et al. 2012, p. 97; Pleasants and Oberhauser 2013, p. 141; 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; Solis-Sosa 2021, p. 16).

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. 48), 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, unpaginated). 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, unpaginated; 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 are expected to 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 2016, a total of 4.9 million acres of grassland were converted to new cropland, including up to 3 million acres of Conservation Reserve Program (CRP) land between 2008 and 2012 (Zhang et. al. 2021, p. 11; Lark et al. 2015, p. 5). Pleasants and Oberhauser (2013, pp. 139–140) 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. 2568). Development and conversion of grasslands will continue to impact monarch resources 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 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). Some conversion of forest to avocado plantations has also occurred in the Reserve buffer zone (see *Availability and Quality of Overwintering Habitat* section in Chapter 6).

Logging was estimated in the core zone of the Reserve from 2002 through 2012 (Vidal et al. 2014, p. 180). 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. Estimates of forest loss throughout the Reserve were approximately 0–2.4% per year between 1986 and 2012 (Ramírez et al. 2015, p. 163). While anti-logging, reforestation, and other conservation efforts are underway (López-García 2011, p. 631; López-García et al. 2022, p. 245), logging is still ongoing within the Reserve (Brower et al. 2016, entire). Flores-Martínez et al. (2019, entire) found that forest cover loss in the core zones of the Reserve showed a decreasing trend from 2012 to 2018, with total illegal logging estimated at <50 ha during this period, suggesting that other factors have likely driven monarch population changes since 2010. 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).

Risk from Insecticides

Insecticides are pesticides with chemical properties that are designed to kill insect pests. Most insecticide active ingredients are non-specific or broad-spectrum in nature. 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 insects at application rates specified on product labels. Although monarchs are not the target species of these insecticides, individuals may be exposed to insecticides on use sites and in areas beyond application sites due to spray 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, including milkweeds (Halsch et al. 2022, entire). 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 spongy moths, mosquitoes, 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.

Insecticides are a threat to monarchs based on their mode of action to target insects and their potential exposure to monarchs. Lab studies determining a concentration dose and behavioral or physiological response of monarchs to neonicotinoids, organophosphates and pyrethroids have demonstrated toxicity at product label application rates and at field concentration levels (e.g., Krischik et al. 2015, entire; James 2019, entire; Krishnan et al. 2020, entire; Bargar et al. 2020, entire). The primary insecticide threat to monarchs is from application of liquid conventional insecticides or Lepidoptera-active biopesticides (i.e., pesticides derived from certain natural materials). Monarchs can be exposed to liquid insecticides from direct spray at the time of application, contact with vegetation contaminated with an insecticide, or ingestion of leaves or nectar contaminated with insecticide following a spray. Insecticides applied by broadcast spraying generally involve applications to large areas and require the use of greater volumes of product compared to more targeted and small-scale treatments. These applications may result in a greater likelihood for insecticides to drift off-field under certain conditions and come in contact with monarchs. Biopesticides, another type of liquid insecticides, generally affect only the target pest and closely related organisms. This contrasts with broad spectrum and conventional insecticides that target all insects. The primary identified biopesticide exposure threat to monarchs is limited to the liquid application of certain types of *Bacillus thuringiensis* that are active against Lepidopterans and often used to control caterpillar pests (such as spongy moth). Considering application method and pesticide type, liquid conventional insecticides and Lepidoptera-active biopesticides account for most of the threat to monarchs from insecticide use.

Monarchs may also be exposed to insecticides in forms other than liquids, such as those that have been systemically incorporated into plant tissues on which monarchs feed (e.g., milkweed leaves, flowers, pollen, and nectar) or dust that has drifted off treated seeds at the time of planting. Numerous types of insecticides may be incorporated into plants systemically or used to treat seeds, including neonicotinoids. However, there are limited data to suggest that insecticides used in this manner achieve concentrations likely to result in negative effects to monarchs. Thus, while monarchs may be exposed to insecticide residues from these application methods, the available information indicates negative effects for insecticide residues are likely minimal and needs further study. For insecticides applied in solid forms such as granules, no exposure is expected to monarchs; thus, negative effects from application of pesticides in solid forms are unlikely.

Studies on the exposure and effects of fungicides to the monarch are more limited than insecticides; however, there are two studies that indicate further investigations on the effects of fungicides is warranted. For example, research for two fungicides have been shown to have direct sub-lethal effects on monarchs. In one study, exposure to fungicide active ingredients (azoxystrobin and trifloxystrobin) resulted in a 12.5% reduction in monarch wing length (Olaya-Arenas et al. 2020, p. 5), which could affect the success of fall migrants to overwintering grounds (Inamine et al. 2016, p. 1089; Olaya-Arenas et al. 2020, p. 8). A second study found the same two fungicides were present on 75% of store-bought milkweed plants at concentrations sufficient to affect wing size (Halsch et al. 2022, p. 2). Additional risk from added compounds to the formulated product that increase the effectiveness of controlling pests (referred to as synergists) are often not assessed. Olaya-Arenas and Kaplan (2019, p. 13) reported that fungicides (which can be used as a synergist with certain insecticides) 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. Other pesticide classes (e.g., herbicides, fungicides) are used within or near areas where monarchs may be present; however, only a subset of pesticides within these classes have data on direct negative effects to insects (i.e., as opposed to insecticides, where all pesticides within this class will cause adverse effects to insects). See Appendix 5 for further discussion on the risk of insecticides and herbicides, including data, references, and supporting information.

Climate Change Effects

Climate change can affect monarchs both directly and indirectly (Nail and Oberhauser 2015, entire) on both the overwintering, breeding, and migratory 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 by 2050 (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. However, we do not have evidence that monarchs would move to or use these upslope areas.

In addition to the impact climate change may have on overwintering monarchs directly, the Mexico overwintering sites are predicted to be less suitable for oyamel fir trees, the predominant monarch roosting tree, as climate change progresses. Flores-Martínez et al. (2019, p. 5) found that climate-related factors have become the predominant driver of forest cover loss since around 2010, whereas past deforestation was largely attributed to illegal logging. 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).

Direct effects of climate change, particularly increasing temperatures, may impact monarch fecundity (Oberhauser 1997, pp. 168–169), mating success (Solensky and Oberhauser 2009, p. 333), 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). 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 (Taylor 2020, pers. comm.). Spring temperatures and precipitation in the Southern U.S. portion of the migratory range have been modeled to have a high relative importance affecting summer population size of eastern monarchs (a larger impact than compared to summer weather, summer herbicide use in cropland, and late-winter population size) (Zylstra et al. 2021, p. 1443). However, considerable uncertainty remains regarding the extent, intensity, and biological impacts of climate change during spring migration (Neupane et al. 2022, pp. 10–14).

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). Widespread drought caused by climate change is expected to increase into the future (IPCC 2023, p. 69) negatively impacting monarchs and their habitat.

Sea level rise is increasing due to climate change. These rising sea levels are likely to impact monarch populations in coastal areas (e.g., along the Gulf Coast) and low-lying islands through loss of habitat (TBCSAP 2015, entire; IPCC 2023, p. 69). 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. 1359; see Appendix 3 ("*Other Stressors*" section) 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, pp. 56–58; Pleasants 2017, p. 43; Thogmartin et al. 2017b, pp. 2–3; MAFWA 2018, p. 52; Pelton et al. 2019, pp. 4–5; WAFWA 2019, p. 41). 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, pp. 37–40).

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 2023, p. 16). 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. As of April 2023, State agencies had implemented milkweed restoration efforts on over 3,896,773 acres, adding an estimated 264 million milkweed stems to the landscape nationwide. 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 Service 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 (ROWHWG 2020, p. 3). By April 9, 2023, CCAA enrollment included approximately 850,000 “adopted” acres (acres with habitat conservation measures for monarch).

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.

The Service developed the Monarch Conservation Database (MCD) in 2018 to capture information about monarch conservation plans and efforts to inform the listing decision. As of June 27, 2023, there are 145,455 complete monarch conservation effort records in the MCD that have a status of completed, implemented, or planned since 2014, and 126 monarch conservation plans (Table 5.3). These efforts constitute a total of 10,457,316 acres of land area in the continental United States and Hawaii (10,246,867 acres and 178,920 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. Conservation efforts implemented or entered into the MCD by State agencies accounted for nearly 50% of the efforts completed and implemented as of April 2023 (3,896,773 acres). The MCD allowed users to provide information about the density and diversity of milkweed and blooming nectar plants present before and after the conservation effort was implemented, but few MCD users had monitoring data at this detail. With respect to the Mid-America Monarch Conservation Strategy goal of 1.3 billion additional stems of milkweed, completed and implemented conservation efforts have contributed an estimated 269 million milkweed stems. Conservation efforts in California’s Central Valley currently amount to nearly 9,000 acres.

Table 5.3. Total conservation efforts entered into the MCD as of June 27, 2023, and total conservation efforts from the MCD combined with changes in CRP between 2014 and 2022.

	Eastern U.S.	**Western U.S.**	**Entire U.S.**
Total number of conservation efforts	138,747	5,820	145,455
Total size of conservation efforts (includes both efforts that have been completed and those planned/not implemented)	10,246,876 ac	178,920 ac	10,457,316 ac
Size of completed or implemented conservation efforts	7,227,362 ac	161,583 ac	7,415,731 ac
Estimated number of milkweed stems from completed or implemented conservation efforts	433,193,531	3,208,508	438,027,995
Net size of conservation efforts from MCD and CRP	5,596,782 ac	-840,069 ac	4,788,233 ac
Estimated net number of milkweed stems from MCD and CRP	257,705,816	-90,543,227	169,226,741

Many land managers who oversee overwintering sites in California have developed and implemented grove management strategies or have included monarch groves in their general management plans. Many others are in the process of developing grove management plans. As of January 2024, grove management plans are being implemented at no fewer than 24 overwintering sites and are currently being developed for at least a dozen more. 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).

Dispersed and Non-Migratory Monarch Populations – Current Condition

In addition to North America, we have records of monarchs occurring in 90 countries, islands, or island groups (Tables 5.4 and 5.5). Of the 90 countries, islands, and island groups, 69 are considered currently extant. Monarch presence within the remaining 21 countries, islands, and island groups has not been confirmed since 2000, but the best available information does not indicate they are extirpated, and thus they are all presumed extant. Of the 29 populations outside of North America, 25 are confirmed extant, and the remaining 4 are presumed extant.

Table 5.4. The current status (extant; unknown or presumed extant; or extirpated) of populations, and countries/islands.

Status	# Countries/ Islands	Definition
Extant	69	At least a single monarch observed since 2000
Unknown or Presumed Extant	21	Not observed since 2000, but lacking multi-year survey efforts
Extirpated	0	No observations despite multi-year survey efforts

Table 5.5. Status of monarchs in 90 known countries, islands, or island groups.

Population	Country/Island	Status
Austral Islands (Extant)	Austral Islands	Extant
Australia (Extant)	Australia	Extant
Caribbean (Extant)	Anguilla	Extant
	Antigua and Barbuda	Presumed extant
	Bahamas	Extant
	Barbados	Extant
	Bermuda	Extant
	Bonaire	Extant
	British Virgin Islands	Presumed extant
	Cayman Islands	Presumed extant
	Cuba	Extant
	Dominica	Extant
	Dominican Republic	Extant
	Grenada	Extant
	Guadeloupe	Extant
	Haiti	Extant
	Jamaica	Extant
	Martinique	Extant
	Montserrat	Presumed extant
	Puerto Rico	Extant
	Saba	Extant
	Saint Barthélemy	Presumed extant
	Saint Kitts and Nevis	Presumed extant
	Saint Lucia	Presumed extant
	Saint Martin	Extant
	Saint Vincent & Grenadines	Presumed extant
	Sint Eustatius	Extant
	Sint Maarten	Extant
	Turks and Caicos Islands	Presumed extant
	U.S. Virgin Islands	Extant

Population	Country/Island	Status
Central America (Extant)	Belize	Extant
	Costa Rica	Extant
	El Salvador	Extant
	Guatemala	Extant
	Honduras	Extant
	Nicaragua	Extant
	Panama	Extant
Cook Islands (Extant)	Cook Islands	Extant
Migratory Eastern North America (Extant)	Canada* (also part of the Western North America population)	Extant
	Mexico* (also part of Western North America and Central American populations)	Extant
	Saint Pierre & Miquelon	Presumed extant
	Eastern United States*	Extant
Greater Indonesia (Presumed extant)	Brunei	Presumed extant
	Indonesia	Presumed extant
	Malaysia	Presumed extant
	Timor-Leste	Presumed extant
Guam & Commonwealth of the Northern Mariana Islands [CNMI] (Extant)	CNMI	Extant
	Guam	Extant
Hawaii (Extant)	Hawaii	Extant
Iberian Peninsula (Extant)	Azores	Presumed extant
	Canary Islands	Extant
	Gibraltar	Extant
	Madeira	Extant
	Morocco	Extant
	Portugal	Extant
	Spain	Extant
Johnston Atoll (Extant)	Johnston Atoll	Extant
Kiribati (Extant)	Kiribati	Extant
Marquesas Islands (Extant)	Marquesas Islands	Extant
Marshall Islands (Extant)	Marshall Islands	Extant
Mascarene Islands (Extant)	Mauritius	Presumed extant
	Réunion	Extant
Micronesia (Extant)	Federated States of Micronesia	Extant
Nauru (Extant)	Nauru	Extant
New Zealand (Extant)	New Zealand	Extant
Norfolk Island (Extant)	Norfolk Island	Extant
Palau (Extant)	Palau	Extant
Papua New Guinea (Extant)	Papua New Guinea	Extant
Philippines (Presumed extant)	Philippines	Presumed extant
Samoa (Extant)	American Samoa	Presumed extant
	Samoa	Extant

Population	Country/Island	Status
South America and Aruba (Extant)	Aruba	Extant
	Colombia	Extant
	Curaçao	Extant
	Ecuador	Extant
	French Guiana	Presumed extant
	Guyana	Extant
	Peru	Extant
	Suriname	Presumed extant
	Trinidad and Tobago	Extant
	Venezuela	Extant
South Florida (Extant)	South Florida*	Extant
South Pacific (Extant)	Fiji	Extant
	New Caledonia	Extant
	Society Islands	Extant
	Solomon Islands	Extant
	Vanuatu	Extant
Tokelau (Presumed extant)	Tokelau	Presumed extant
Tonga (Extant)	Tonga	Extant
Tuvalu (Extant)	Tuvalu	Extant
Wallis & Futuna (Presumed extant)	Wallis and Futuna	Presumed extant
Migratory Western North America (Extant)	Western United States*	Extant

*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.)

Dispersed and Non-Migratory Populations – Current Influences

There is little to no information on the status and health of monarchs outside of the North American migratory populations, nor information regarding the positive or negative influences acting upon these populations (Appendix 2, Table 2A.2). 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 dispersed 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 dispersed 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 North American migratory monarch populations, we organized the key factors driving monarch population dynamics into 6 categories: 1) milkweed availability, 2) breeding nectar availability, 3) migration nectar availability, 4) overwintering habitat availability, 5) climate change effects, and 6) insecticide exposure. We then forecasted how each of these six influences is expected to change (i.e., its expected future state condition). 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). We then used these changed parameters to re-run the population model and present the resulting probability of extinction predictions in Chapter 7.

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	**Scenario (time frame)**	**% Change**	**Description**
Milkweed	Best (18 years)	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.
Milkweed	Worst (18 years)	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*	*Same as Milkweed conditions*
Migration nectar	Best (18 years)	S: 5% increase	*(Same as Milkweed "Best")*
Migration nectar	Worst (18 years)	S: 6% loss	*(Same as for Milkweed "Worst")*

Overwintering habitat	Best (25 years)	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.
Overwintering habitat	Worst (25 years)	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	Best (60 years)	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.
Climate change - habitat	Worst (60 years)	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	Best (25 years)	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.
Insecticides	Worst (25 years)	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	**Scenario (time frame)**	**% Change**	**Description**
Milkweed	Best (50 years)	2% loss	Incorporates a low human growth scenario and conservation efforts implanted via the WAFWA plan and nonprofit groups.
Milkweed	Worst (50 years)	3% loss	Incorporates a high human growth scenario and conservation efforts implanted via the WAFWA plan and nonprofit groups.
Nectar resources	*Same as Milkweed*	*Same as Milkweed*	*Same as Milkweed conditions*
Overwintering habitat	Best (50 years)	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.
Overwintering habitat	Worst (50 years)	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	Best (20 years)	8% decrease	Increases in suitable climate niche due to projected increases temperatures.
Climate change - habitat	Worst (20 years)	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	Best (20 years)	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.
Climate change – non-habitat	Worst (20 years)	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	Best (20 years)	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.
Insecticides	Worst (20 years)	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

The future scenarios that we developed 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). We divided the breeding range into subregions based on Oberhauser et al., 2017a, which reflect differences in the relative contribution of individuals to the population (Voorhies et al., 2019). “Baseline”

(2020) habitat estimates were derived from the USGS "seamless" land cover spatial data (Rohweder and Thogmartin 2016; see Appendix 3 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. We projected scenarios for milkweed and nectar from conservation efforts, CRP, and land cover change for the eastern migratory population range out 18 years, in line with the Mid-America Monarch Strategy. We believe there is additional uncertainty regarding conservation efforts following implementation of the MAFWA Strategy. Although land cover models project change well beyond 18 years, applying our estimates of milkweed density based on land cover type would result in overestimates of increased milkweed and nectar due to land cover change (see Appendix 3, part 4). 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 (Hyberg 2018; pers. comm.). U.S. 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, entire) 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. Although the FORE-SCE model predicts a loss in grassland type habitat and an increase in developed land cover types in our future timeframe, once estimated milkweed stem densities were factored in, increases in milkweed resulted under all FORE-SCE scenarios considered (A1, A2, B1, and B2).

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 that we developed 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 105,400 ha (~260,450 acres) between 1980 and 2000 (Sleeter 2016, unpaginated). To project future trends, we used the results from Sleeter et al. (2017, 1074–1081) 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, unpaginated), 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 260,000 additional ha (640,000 ac) 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 530,000 additional ha (1.3 million ac) 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 20,000 ha (50,000 ac) 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 265 ha (655 ac) 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 2,650 ha (6,550 ac)). 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 22,800 ha (56,300 ac) of habitat will be restored, yielding 238,400 ha (590,000 ac; -2%) and 511,600 ha (1.3 million ac, -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). Tools to quantify past, present, and future nectar resources at broad scales are not available. We assume data on milkweed densities associated with landcover classes and conservation actions serve as a proxy of habitat quality for monarchs. While the relationships between milkweed density and nectar may vary significantly site-by-site, landscape-scale patterns of milkweed densities and the availability of flowering plants will be generally correlated at broader scales due to management practices resulting in the embedded vegetation communities and vegetation structures within which milkweed occur. The degree to which nectar may be a limiting factor under various conditions, or in particular geographies or habitat types, remains uncertain. 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

Our future projections of the availability of overwintering habitat are largely predicated upon the analyses within Honey-Rosés et al. (2018, pp. 6–10), Vidal et al. (2014, p. 180), and Flores-Martínez et al. (2019, pp. 2–4), 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, pp. 101–104).

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. There was 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, p. 180). Flores-Martínez and colleagues (2019, p. 4) 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, p. 5) 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). We also note that while some conversion of land to avocado plantations within the Monarch Butterfly Biosphere Reserve buffer zone has occurred, the amount is relatively small (1.7% of the total Reserve, with only 0.07% of the Reserve where oyamel fir forests were converted) and projections show that conversion of oyamel fir forest is a low threat through 2050 (Sáenz-Ceja and Pérez-Salicrup 2021, pp. 304–305; Denvir 2023, pp. 5–7).

Western Population

Our 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, unpaginated).

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% (Griffiths 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, unpaginated), 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, unpaginated). 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, entire), using the moderate B2 greenhouse gas emission scenario. While Lemoine (2015, entire) 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, spring and summer weather conditions that limit breeding, etc.; Taylor 2020, pers. comm.; Zylstra et al. 2021, p. 1443). 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 2020, pers. comm.). 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, pp. 7–20) estimates that the southern subregion of the current eastern population breeding range will have a loss of the southernmost portion of the range but will add habitat in the more northern part of the southern subregion. Overall, there was more addition in the northern habitat 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, entire). 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 greenhouse gas 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, p. 10), 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. 3934) 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 Appendix 5 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, unpaginated), 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.

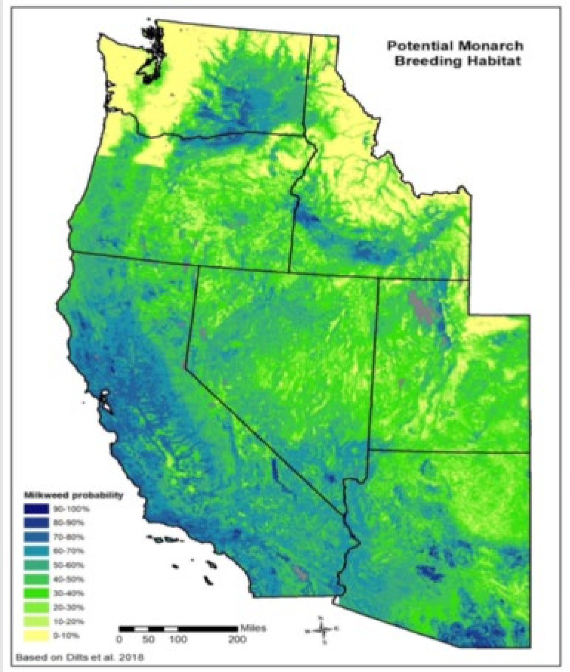
Insecticide use is widespread across the most important breeding areas (Figure 6.1) for the western monarch, and it has been implicated as one of the key drivers in the decline of the western monarch population (Crone et al. 2019, p. 10; Forister et al. 2016, entire; Halsch et al. 2020, entire). Based 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, entire) 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.



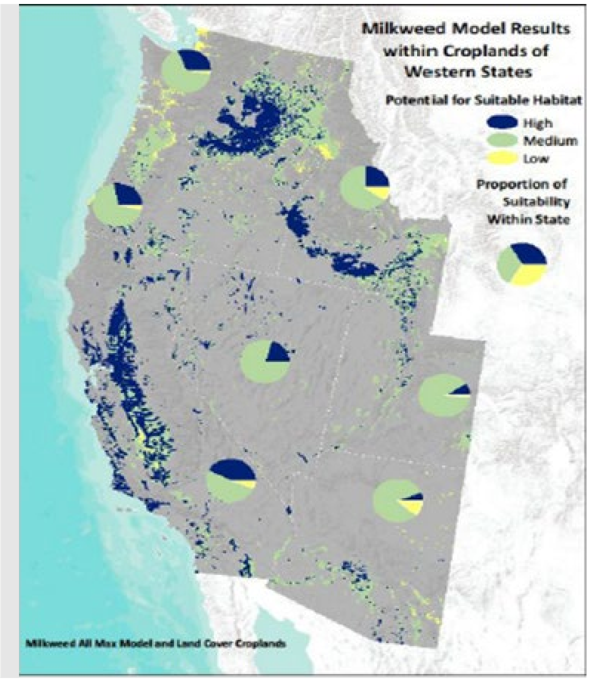


Figure 6.1. A. Predicted distribution of milkweed and thus extent of potential monarch breeding areas—derived from a habitat suitability model (Dilts et al. 2018). B. Suitable habitat—milkweed potential—overlaid with croplands in western U.S. (WAFWA 2019, Fig. 6, p. 15). Dark blue areas correlate with 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, entire). Water availability can affect both milkweed quality and milkweed and nectar availability (Brower et al. 2015, pp. 124–126; Couture et al. 2015, pp. 8–10; 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, p. 117). Brower and colleagues (2015, pp. 127–129) 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.

Western North American Population

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 6.2) 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. 3934) 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.

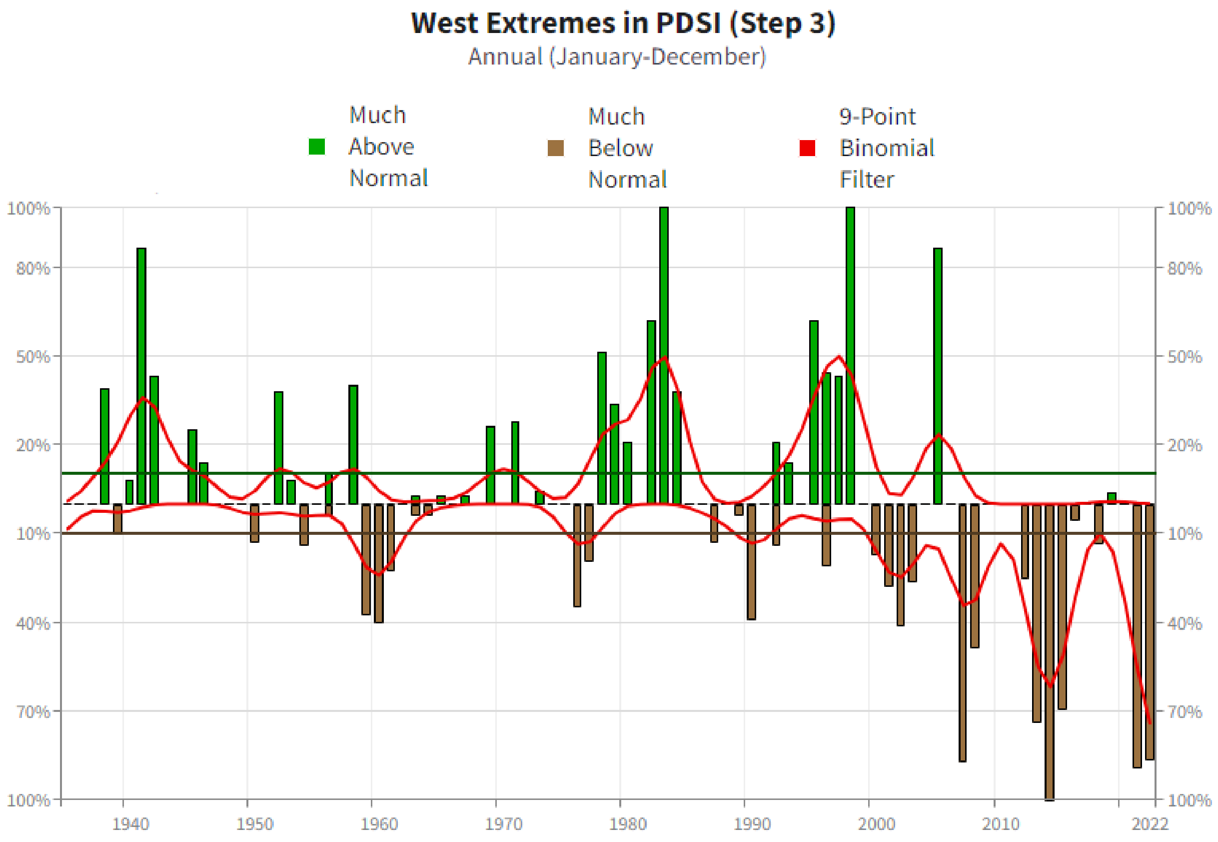


Figure 6.2. Extremes in the Palmer Severity Drought Index for the western U.S. (i.e., California and Nevada). Figure from the National Oceanic and Atmospheric Administration (NOAA 2023).

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.

Dispersed and Non-Migratory Populations – 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, unpaginated; Cinquini 2014, entire). 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 3 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, and forecasted probability of extinction, given scenarios of expected future changes to current influences (threats and conservation efforts). 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 migratory populations given the best and worst case scenarios presented in Chapter 6. Lastly, we provide the results of the catastrophic events analysis for the dispersed and non-migratory populations.

Eastern North American Population – Future Condition

Under both best and worst case future scenarios, the population is likely to continue 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 estimated *pE* increases over time (Figure 7.2). By year 60, estimated *pE* ranges from 56% to 74% (see Appendix 4, Table 4A4 for decadal projections).

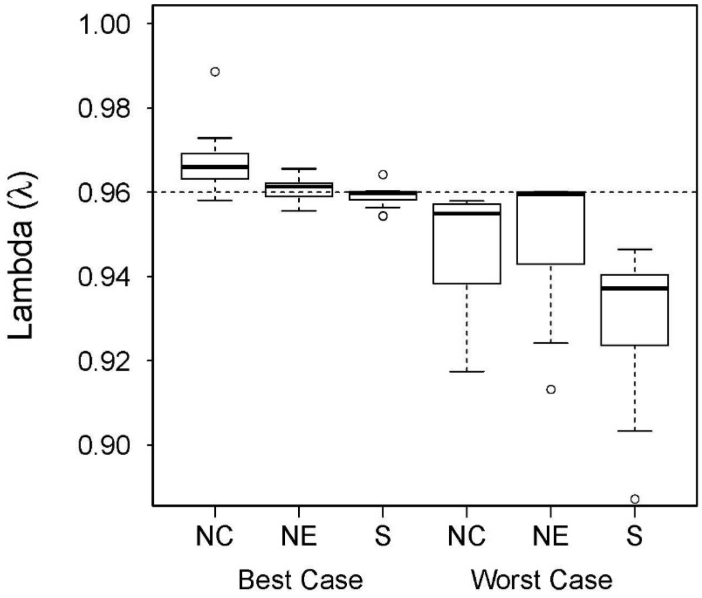


Figure 7.1. Box plot for population growth rate (lambda, λ) under the best and worst case scenarios for each of the subregions of the eastern population (NC=Northcentral, NE= Northeast, S=South). The dashed line represents the current population growth rate (λ=0.96).

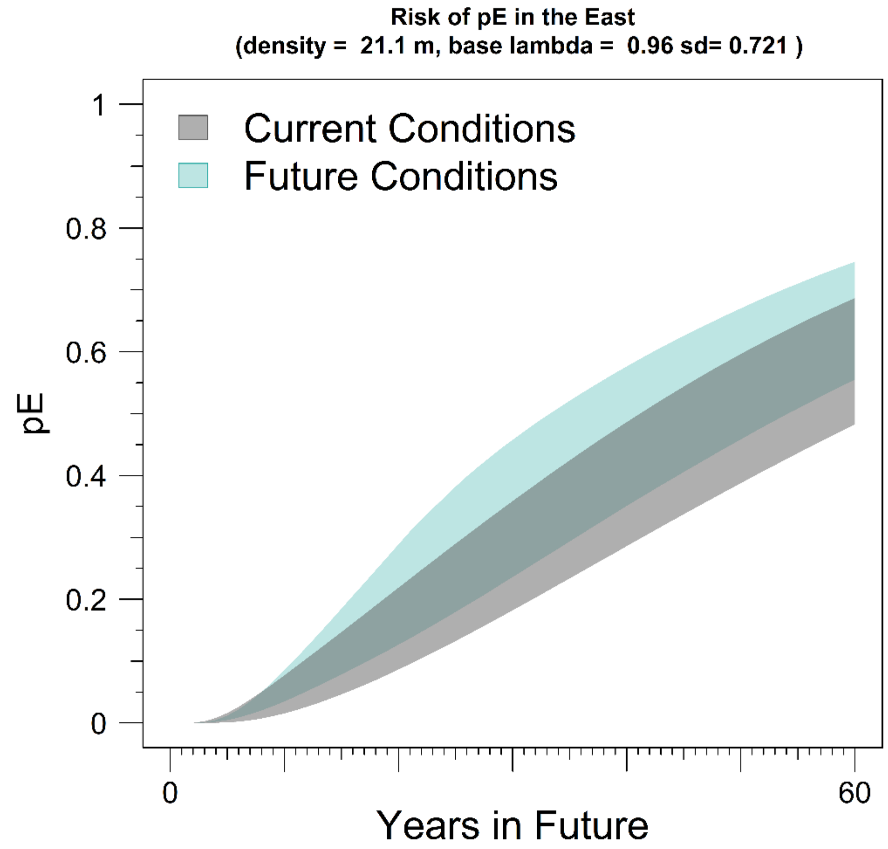


Figure 7.2. pE, *for the eastern North American monarch population over time, given both current (gray band) and projected changes in state conditions (blue band). By year 10,* pE *is under 10% and by year 60,* pE *ranges from 56% to 74% under the best and worst case future scenarios, respectively.*

Direct Effects from High Temperatures & Catastrophic Events

Due to modeling framework limitations, 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 3 - 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 4 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.

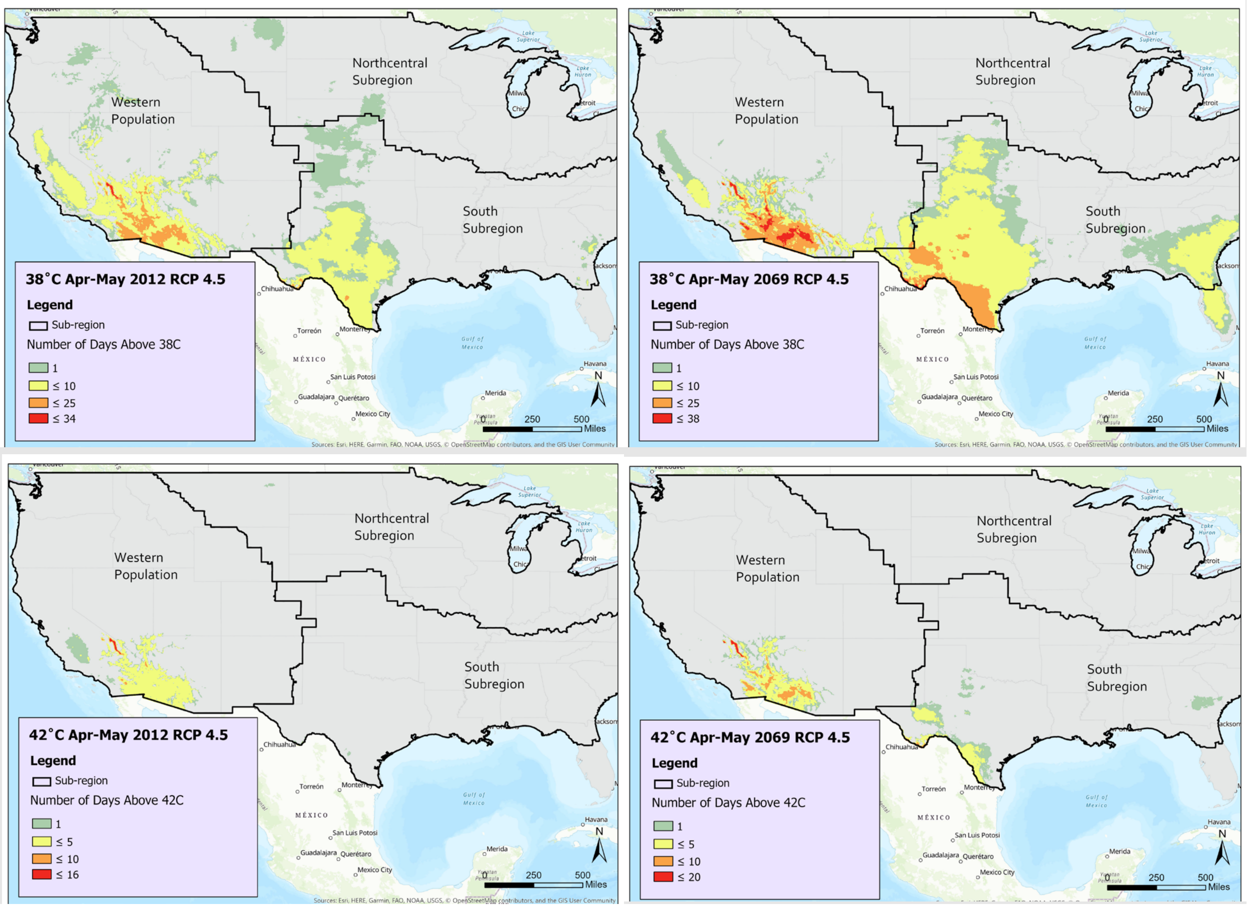


Figure 7.3. The projected spatial extent and average number of cell days between April and May where temperatures >38˚C (top) and 42˚C (bottom) in 2012 (left) and 2069 (right) under RCP 4.5. Colors represent number of cell days above 38 and 42˚C.

Western North American Population – Future Condition

Under both best and worst case future scenarios, the population is likely to continue to decline ($\lambda < 1$, Figure 7.4). As would be expected with a declining growth, the estimated *pE* increases over time (Figure 7.5). At year 10, estimated *pE* ranges from 66 to 71% and reaches 99% by year 60 (see Appendix 4, Table 4A3 for decadal projections).

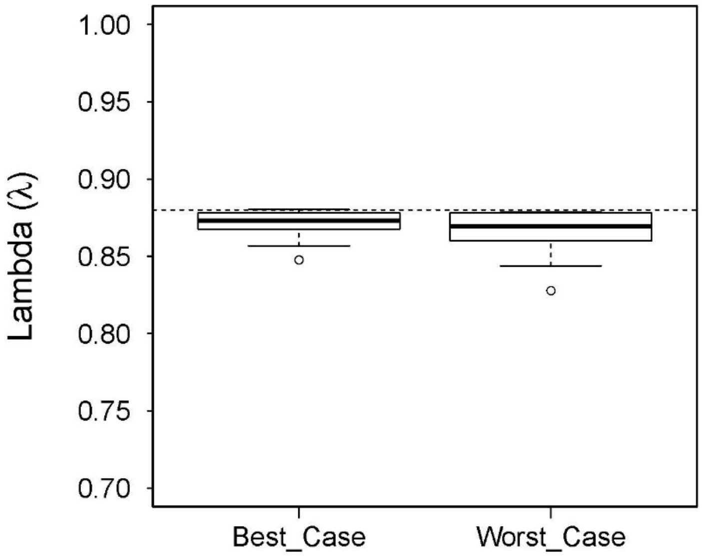


Figure 7.4. Box plot for population growth rate (lambda, λ) under the best and worst case scenarios for the western population. The dashed line represents the current population growth rate (λ=0.878).

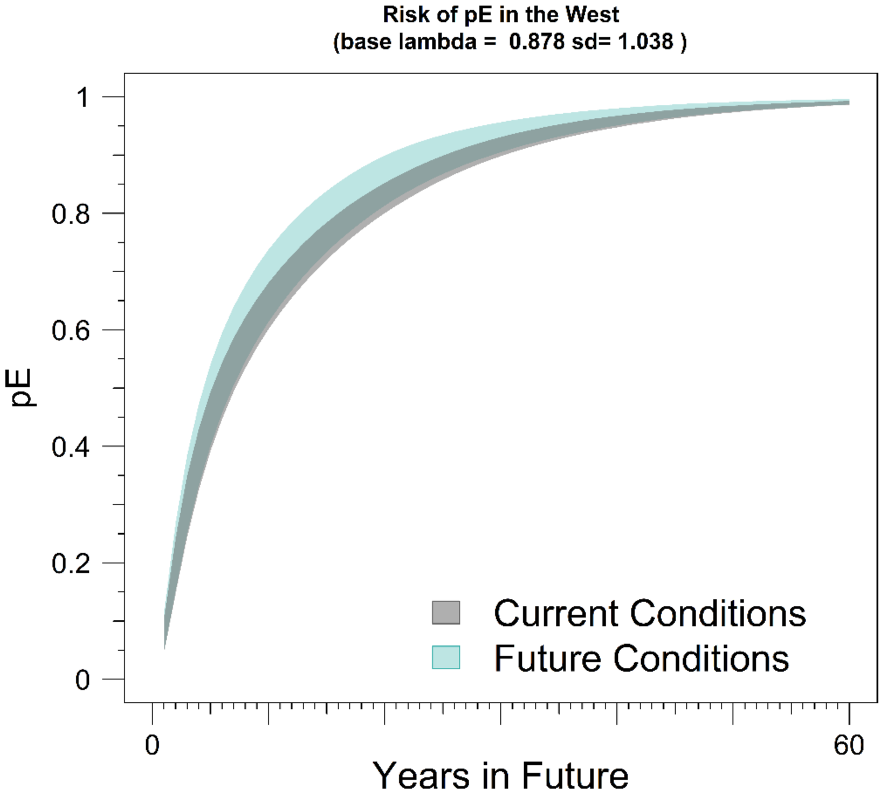


Figure 7.5. pE, *for the western North American monarch population over time, given both current (gray band) and projected changes in state conditions (blue band). By year 10,* pE *is 60%–68% and by year 60,* pE *reaches 99% under the best and worst case future scenarios.*

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 greater than 38°C and 42°C is projected to decrease (-23% and -11%, respectively), while increases in the average number of days greater than 38°C and 42°C are projected to increase (38% and 11%, respectively) (see Figure 7.1 and Appendix 4, Tables 4A1–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.

Dispersed and Non-Migratory Populations – Risks due to Catastrophic Events

We qualitatively assessed the impact due to predicted climate change effects. Fifteen of 29 population groupings are classified as being “at risk” to extinction due to threats associated with climate change (6 due to sea level rise and 9 due to unsuitably high temperatures; Table 7.1). The populations susceptible to sea level rise (Johnston Atoll, Kiribati, Marshall Islands, Nauru, Tokelau, and Tuvalu) are at risk of losing at least some of their monarch habitat; thus, redundancy could decrease with the loss of those areas. However, the best available information does not indicate if populations at risk to high temperatures will lose all or just a portion of their monarch habitat; thus, they may continue to contribute to redundancy. Each of the populations at risk due to sea level rise contains a single country, island, or island group. Therefore, in the dispersed nonmigratory populations the species is likely to have continued presence in an estimated 84 of the 90 countries, islands, and island groups where it occurred historically or to where it has dispersed.

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

Population	Status	High Temps	Sea-Level Rise
Austral Islands	Extant	No Known Risk	No Known Risk
Australia	Extant	At Risk	No Known Risk
Cook Islands	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
Caribbean	Extant	At Risk	No Known Risk
Central America	Extant	At Risk	No Known Risk
Florida	Extant	At Risk	No Known Risk
Hawaii	Extant	No Known Risk	No Known Risk
Iberian Peninsula	Extant	At Risk	No Known Risk
South America and Aruba	Extant	At Risk	No Known Risk
Eastern North America	Extant	See E. North American pop above	See E. North American pop above
Western North America	Extant	See W. North American pop above	See W. North American pop above

Chapter 8: Synthesis – Implications for Viability

This chapter synthesizes the results from our historical, current, and future analyses and discusses the consequences of the forecasted change in the health and probability of persistence 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 the species' ability to sustain populations in the face of normal environmental stochasticity, catastrophes, and novel changes in its environment. 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. Populations retaining genetic and phenotypic diversity help to preserve species' adaptive capacity, which is essential for adapting to their continuously changing environment.

Migratory monarchs in North America are the ancestral population for all other monarch populations, and more than 90 percent of monarchs worldwide occur in the North American migratory populations.

While the North American migratory populations naturally fluctuate year-to-year with environmental conditions, they have declined over the last 20 years (Figures 4.2 & 4.3). 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. The magnitude or frequency (or both) of threats are expected to increase.

Because monarch population sizes fluctuate with environmental conditions, populations must be large and have strong population growth potential to withstand natural environmental variation and disturbances. The eastern and western North American migratory populations comprising the core of the species have declining growth rates and increasing extinction risks (Figures 7.2 & 7.5). The estimated probability of extinction for the eastern population is high (less than 10% chance of extinction within 10 years, 56% to 74% chance within 60 years). The estimated probability of extinction for the western population is higher (60% to 68% chance within 10 years, reaching 99% by year 60). The ranges in the estimates represent mean *pE*s under the best and worst case plausible future scenarios, respectively.

The annual long-distance migration of most monarchs to a few concentrated overwintering areas in Mexico and California results in costs and benefits to the species' viability. The high percentage of the world's monarch butterflies that concentrate at the migratory monarch overwintering sites makes the species vulnerable to catastrophic events during the overwintering period, while the expansive areas of North America occupied by migratory monarchs during the annual breeding season provides protection against the impacts of catastrophic events during the breeding season.

The probability of extinction estimates presented in this report 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. 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.

Historical records suggest that monarchs dispersed from North America in the past two centuries and now also occur in non-native or naturalized populations throughout 90 countries, islands, and island groups where milkweed (their larval host plant) was already present or introduced. Though widespread across a diversity of habitats, environmental gradients, and climates, little information is available about the abundance and health of the monarchs in these locations, and 15 of the 29 dispersed monarch populations may be at risk of extinction. Rangewide, we conclude that the species is likely to maintain considerable redundancy through continued presence in an estimated 84 of the 90 countries, islands, and island groups where it occurred historically or to where it has dispersed. We anticipate the species will continue to have adaptive capacity (representation) through its presence over a large geographical range where the climatic conditions and habitat vary widely.

The North American migratory monarch populations comprise over 90 percent of the species' abundance, possess distinct and greater allelic diversity than other populations, exhibit the long-distance migratory phenotype, occupy different climates and habitat niches, differ in reproductive behavior, and possibly differ in disease resistance as a result of their migratory behavior. Therefore, loss of these populations would impair the overall ability of the species to adapt in the future. In the scenarios that we modeled, the chance of *both* populations persisting above the extinction threshold over the next 10 years is 27% to 33% (that is, the probability of at least one of the populations going extinct is 67% to 73% over the next 10 years). Within 30 years, the chance of both populations persisting above the extinction threshold drops to under 10% (that is, probability of extinction for at least one population is greater than 90%). Much of this risk is due to the poor condition of the western population.

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.

The key uncertainties that impact our assessment of current and future monarch viability include: (1) estimates for the eastern migratory population size, (2) population trend and growth rate estimation, (3) modeling the multi-generational growth of the migratory eastern North American monarch population, (4) extinction thresholds for the migratory eastern and western North American populations, (5) current, independent population growth of North American populations (the lack of links between their population numbers), (6) the relationship between influences and population responses, (7) influences from alternative overwintering strategies, (8) current status and future health and persistence of dispersed and non-migratory populations, and (9) additional influences.

Estimates for the Eastern North American Migratory 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. Ground-based LiDAR techniques were tested at multiple sites in a single season and derived density estimates similar to the median density estimated by Thogmartin et al. 2017a (Hristov 2020, pers. comm.). 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.

Monitoring at overwintering sites allows for the most practical and direct assessment of annual population levels, where the near-entirety of the migratory population can be evaluated consistently and comprehensively, reflecting the full range of threats and population dynamics encountered over the prior annual breeding and migration cycle. One paper (Crossley et al. 2022) raised questions about the optimal approach for annual census of the populations (e.g., overwintering data v. non-overwintering data), however subsequent papers (Pleasants et al.

(2024, entire) review that methodology and conclude “the size of the population as measured by overwintering area is an accurate representation of the size of the population that is produced every summer” (p. 62).

Population Trend and Growth Rate Estimation

Estimates of population viability (and extinction risk) are based on the overall population trajectory (growth rate), along with the current population size and the bounciness exhibited year-to-year. Growth rate and variability are derived using time series data, which may be influenced by the number of years being analyzed.

Different analyses have explored the possibility that past declines in milkweed, or habitat more generally, alongside the expansion of genetically-modified crops and associated herbicides and pesticides, may have effectively plateaued in recent years. Thogmartin et al. (2020, entire) documented some evidence of a change in trajectory for the eastern population since 2014, but determined there was insufficient statistical support for this, as of 2020. More recently, Thogmartin (2024, entire) analyzed overwintering data for the eastern population and estimated a median annual rate of growth of 0.93 (0.67–1.30, 95-percent CI) determining likelihood of there being a declining population, even when assuming relatively stable land use and landcover change relative to the conditions that had driven steeper population declines in the previous decades.

Other published analyses estimate monarch population growth rates and persistence/extinction risks using alternative models and different data sets. These include non-overwintering population data, different time-series of annual population estimates (i.e., ten years v. twenty+ years), and different modeling assumptions about density-dependence, the relationship between population size and growth (i.e., increased density of monarchs will result in decreased population growth and decreased density of monarchs will result in increased population growth). Meehan and Crossley (2023, entire) used a variable change model to characterize the trend in eastern North American monarch abundance suggesting that the monarch population has rebounded after a period of decline, thereby leading to a markedly lower risk of extinction relative to other assessments (e.g., Semmens et al. 2016, entire; Schultz et al. 2017, entire; Thogmartin 2024, entire), including our SSA. Meehan and Crossley (2023, entire) assume density dependence, which constrains the uncertainty associated with the predicted abundance estimates (small populations recover faster under density dependent assumptions than density independent assumptions). Though their density dependent model provides a good fit for the 10-year dataset they analyzed, the models cannot identify the mechanism behind the apparent decline in growth rate as populations increase. This assessment did not incorporate the impacts of changing future conditions. In response to Meehan and Crosley (2023), Crone et al. (2024, entire) subsequently challenged the conclusions published by Meehan and Crossley (2023) as being “much less robust than implied” and lacking “convincing evidence that density dependence reduces extinction risk in monarch butterflies”.

Another analysis that also did not incorporate the impacts of changing future conditions assumes the variability in the population numbers is now driven by environmental and demographic stochasticity (Thogmartin 2024; entire). This approach is more conservative because it results in increased variability at small populations versus the assumption of density dependence (which assumes increased per capita growth when populations are small). In this modeling approach, when the population abundance has been reduced to the small level, the demographic and environmental stochasticity alone (and not necessarily a declining trend) are now the driving factors, resulting in an increased risk of extinction.

Multi-Generational Growth of the Migratory Monarch Populations

Published models of monarch population growth vary in accounting for the multi-generational migration and growth of the eastern monarch population. Some models only estimate growth of the overwintering population (Semmens et al. 2016, entire) while others model the growth of subregions within the eastern monarch population (Flockhart et al. 2015, entire; Oberhauser et al. 2017a, entire). 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.

The variability of recent years in western monarch overwintering numbers is consistent with a broader, long-term declining trend in overwintering western monarchs (Espeset et al. 2016, entire; Schultz et al. 2017, entire). Sequences of favorable or unfavorable seasonal conditions can strongly affect monarch recruitment (Yang et al. 2022, entire). Unfavorable conditions in either the early or late season can constrain recruitment, whereas years with favorable conditions in both the early and late season have the potential to support rapid population increases (Yang et al. 2022, p. 20).

Extinction Threshold

Another key uncertainty is the population size at 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 population model that we used 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, p. 3) and Wells et al. (1990, p. 124) for the western population. Given the western population rebound observed from below 2,000 individuals in 2019–2020 to nearly 250,000 individuals the following winter, these estimates may be conservative. Much uncertainty remains about the size of extinction thresholds and the time lag necessary to confirm assumptions about population stability or extinction trajectories Therefore, we could be either overestimating or underestimating extinction risk (*pE*) depending on the accuracy of the thresholds.

Exchange of Individuals among the North American Populations

Marking data (Morris et al. 2015, pp. 100, 102) and genetic analyses (Talla et al. 2020, p. 2577) 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 (see Freedman et al. 2021, pp. 7–8). 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, entire; Schultz et al. 2017, entire), our model does not include immigration and emigration parameters 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.

The Relationship between Influences and Population Response

The SSA must evaluate and incorporate information on current and future threats to accurately estimate future monarch population resiliency. 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. Additionally, in our model, we do not account for synergistic interactions and/or heterogeneity of influences, spatial or temporal patterns that carry additional uncertainties.

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. However, climate change may shift monarch distribution across the expansive summer breeding grounds in other directions, with potentially higher projected abundances in areas that become wetter but not appreciably hotter (e.g., northern Ohio) and declines in abundance where summer temperatures are projected to increase well above those observed in the recent past (e.g., northern Minnesota) (Zylstra et al. 2022, p. 6147). In addition, recent modeling efforts of the eastern population suggest that the amount of milkweed in the central and southern parts of the range has a greater influence on the population size than the northern portion (Solis-Sosa et al. 2021, pp. 12–13). 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.

Alternative 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, p. 347; Satterfield et al. 2018, p. 1676), 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 South Carolina (McCord and Davis 2010, p. 413; Kendrick and McCord 2023, p. 2). 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 migratory monarch populations, respectively. Monarchs exhibiting non-migratory or alternative migratory behaviors are not well studied. While the total number of monarchs falling under these categories appears to be quite small, relative to the North American migratory populations that overwinter in Mexico and California, and some experts consider them effectively acting as population sinks (see Crone and Schultz 2021, p. 1536), it is uncertain how important they may be in terms of overall population dynamics and providing adaptive capacity into the future.

Recent work has elucidated potential alternative migratory routes, including that of a relatively small number of individuals from North America through Florida, Cuba, the Yucatán Peninsula, and then possibly overwintering at yet undiscovered sites in Guatemala (Dockx et al. 2023, p. 314). We make the same assumption as above: because of the relatively small number of monarchs potentially taking alternative migratory routes, we assume that Mexico annual counts continue to represent the large majority of the eastern migratory population (there have been reports of transient migratory monarchs in the Yucatán Peninsula as recently as 2007 with up to thousands of monarchs, compared to the tens of millions of monarchs consistently migrating and overwintering in Mexico) (Dockx et al. 2023, pp. 296–297).

Status and Health of Dispersed and Non-Migratory Populations

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 for the purpose of our analysis that monarchs that are present outside of North America have become naturalized. This assumption may overestimate the historical importance and/or viability of monarchs worldwide.

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 dispersed populations are extant will overestimate the current representation and resiliency of monarchs globally and, subsequently, overestimate the viability of the species.

We are unable to evaluate the impact of threats like habitat loss (land-use change) or pesticide use on these dispersed populations because we lack information on the specific locations of monarchs within these geographies. The populations susceptible to sea level rise are at risk of losing at least some of their monarch habitat; thus, redundancy could decrease with the loss of those areas. However, the best available information does not indicate if populations at risk to high temperatures will lose all or just a portion of their monarch habitat; thus, they may continue to contribute to redundancy. In all other cases (i.e., populations not found to be at risk of sea level rise or high temperatures), we assume that monarch populations will persist into the future, and this may lead to overestimating the viability of the species.

Additional Influences

We considered the primary influences on eastern and western North American monarch populations, which were based the best available information. However, it is possible that other influences may impact monarchs, or may later emerge as primary influences. Drivers of past trends do not necessarily determine future trends. Many of these potential negative influences are discussed in more detail in Appendix 4 under the section “Other threats and catastrophic events considered.”

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Appendices

Appendix 1. Taxonomy

The monarch butterfly is a milkweed butterfly (subfamily Danainae, family Nymphalidae) in the genus *Danaus*. 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" (Pelham 2017, pers. comm.). 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. Dispersed and Non-Migratory Monarch Populations: Additional Description and Results

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; Hemstrom et al. 2022, pp. 4551–4552). We found occurrence records of monarchs in 90 countries, islands, or island groups. For our analyses, we grouped these occurrences into populations.

To assess the change in the number, health, and distribution of monarch populations over time, we grouped populations as follows. Based on distinct behavioral differences and limited movement between these populations (see Chapter 2), we grouped North American monarchs into eastern and western migratory populations and a non-migratory Florida population. The monarchs in Central America, the Caribbean, and South America were grouped into populations if there were documented genetic differences (Pierce et al. 2015). Other monarch populations were delineated based on distance. Monarch locations separated by more than 200 miles (such as islands in the Pacific Ocean) 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 regularly. 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.

Table 2A1. Dispersed monarch population groupings, with the countries and islands that comprise each population.

Population	**Countries/Islands within Population**
Austral Islands	Austral Islands
Australia	Commonwealth of Australia
Cook Islands	Cook Islands
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
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
Hawaii	United States (HI)
Iberian Peninsula	Azores, Canary Islands, Gibraltar, Madeira, Morocco, Portugal, Spain
South America and Aruba	Aruba, Colombia, Curacao, Ecuador, French Guiana, Guyana, Peru, Suriname, Trinidad and Tobago, Venezuela

Genetic diversity in dispersed populations

1. South America and Aruba:

   Monarchs in South America and Aruba are grouped together due to genetic similarity.

   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 monarchs together.

2. Central America and the Caribbean:

   Central American and Caribbean monarchs are grouped together 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).

   Mexican non-migratory monarchs were also included in this group (rather than grouped with the eastern North American monarchs), 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).

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

   Monarchs across Australia, New Zealand and other Pacific Islands are grouped together 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. Analyses of monarchs in several Pacific island locations (mainland Australia, New Zealand, Norfolk Island, 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; Hemstrom et al 2022, p. 4550). 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).

4. Hawaii:

   Hawaiian Island monarchs have 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 dispersed populations (Zhan et al. 2014, p. 317). 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, p. 6; Hemstrom et al. 2022, pp. 4552). Additionally, work indicates that monarchs in Hawaii form an admixed population, suggesting movement among islands (Pierce et al. 2014b, p. 1; Hemstrom et al. 2022, p. 4552). 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 Islands may contribute unique variation to the species in resistance to *OE*.

5. 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, are grouped because of unique genetic variation and ecological and climatic conditions.

The non-migratory 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, Spanish monarchs occupy very similar ecological and climatic conditions to the rest of the monarchs in in Portugal and Morocco (Fernández-Haeger et al. 2015, entire).

Status, Health, and Influences of Dispersed Populations

Table 2A.2. Population health: current status, past trend in population size (N), current status of milkweed & nectar resources, current status of insecticides, and overwintering habitat

Population	**Status**	**Trend in N**	**MW/Nectar**	**Insecticides**	**OW Habitat**	**Overall Condition**
Austral Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
Australia	Extant	Unknown	Unknown	Unknown	Unknown	Unknown
Cook Islands	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

Population	**Status**	**Trend in N**	**MW/Nectar**	**Insecticides**	**OW Habitat**	**Overall Condition**
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
Caribbean	Extant	Unknown	Unknown	Unknown	N/A	Unknown
Central America	Extant	Unknown	Unknown	Unknown	N/A	Unknown
Hawaii	Extant	Unknown	Unknown	Unknown	N/A	Unknown
Iberian Peninsula	Extant	Unknown	Unknown	Unknown	N/A	Unknown
South America and Aruba	Extant	Unknown	Unknown	Unknown	N/A	Unknown

Appendix 3. 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 Voorhies et al. 2019, 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 timeframe (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 3A1. Initial starting values for the population model. Inds= individuals

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

*Parameter values differ slightly from Schultz et al. 2017 and Semmens et al. 2016 because we added values through winter 2019–2020 to the population datasets. See the Summary of Changes (pg. ii) for discussion of why subsequent years (2021 to present) were not incorporated.

[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 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 dispersed 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). Within the North American migratory populations, the prevalence of heavily *OE* infected monarchs in the west (19.8%) is almost four times that of the east (5.5%), possibly due to a reduced migratory culling phenomenon with the shorter distance western migration (Majewska et al. 2022, p. 786). This protozoan parasite impacts monarchs (the only known host for *OE*), leading to decreased survival and fitness in the monarch (Altizer and Oberhauser 1999, p. 85). Infection rate trends have varied, with the migratory eastern North

American population seeing an increase in infections since 2002. Migratory western North American monarchs primarily declined through 2012, with a more recent increase (Majewska et al. 2022, p. 786). 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 overwintering sites (including birds, mice, and wasps; Oberhauser et al. 2015, p. 72; Ruiz 2018, p. 7). 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 have 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 Appendix 5 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

Fire

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**.**

Hurricanes

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 3A2). Milkweed density values in Table 3A2 for each land cover type are generally based on Thogmartin et al. (2017c), 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 3A2. Baseline and potential milkweed densities for land cover types. Values from Thogmartin et al. 2017c.

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

Classification	Estimated Baseline Milkweed Density	Potential Density
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
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

Classification	**Estimated Baseline Milkweed Density**	**Potential Density**
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; Hyberg 2018, 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 3A3). 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 3A3. 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

Classification (FORE-SCE Model)	Classification (Rohweder and Thogmartin 2016)
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 et al. 2012, entire). For simplicity, we use the period between April and May (i.e., when heat is likely to have the biggest impact on migration of monarchs in the continental United States) 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 3A3). 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 A4). 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 3A4 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 3A4 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).

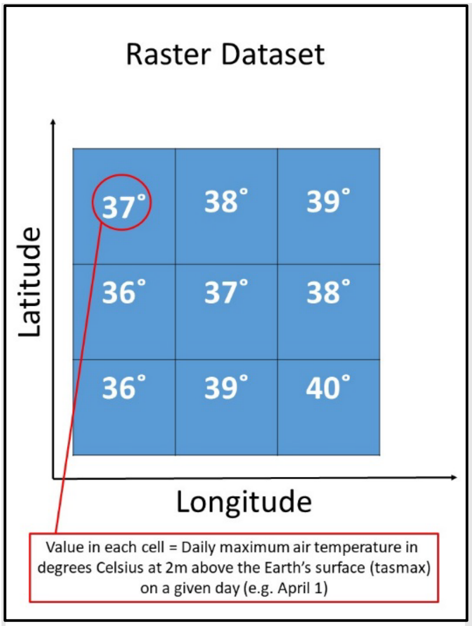


Figure 3A1. A raster dataset is composed of a spatially referenced grid with each grid cell containing data. For this analysis, the data in each cell represents the daily maximum air temperature 2 meters above the surface of the earth.

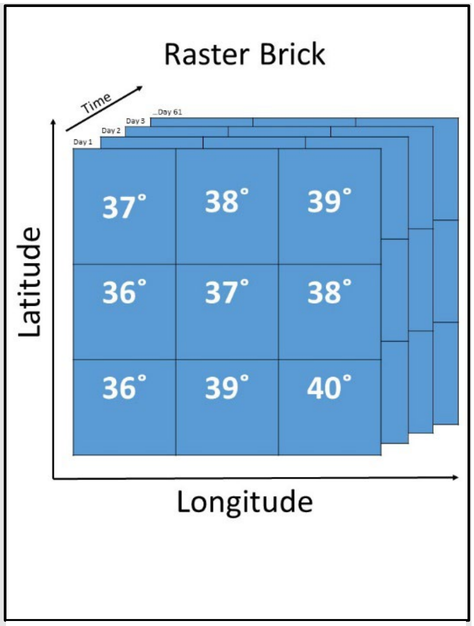


Figure 3A2. A raster brick consists of stacked individual raster datasets. For this analysis, each raster represents a single day between April 1st and May 31st.

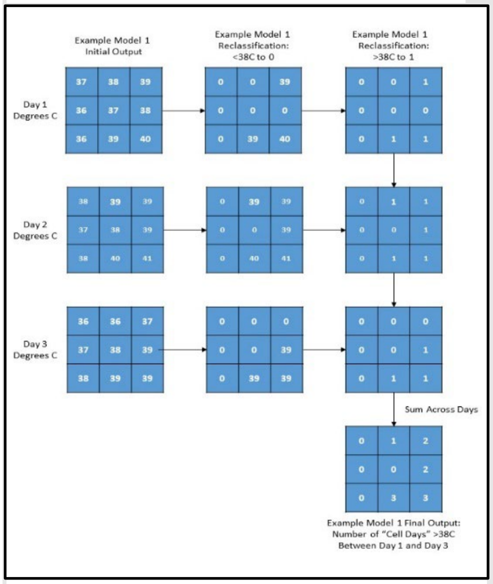


Figure 3A3. Reclassifying the raster dataset and assigning a value of "0" to all cells with tasmax value of 38˚C or lower and a value of "1" to all cells that have a tasmax value of above 38˚C. This resulting value is referred to in this analysis as a "cell day". Summing through the days of a raster brick provides the total number of cell days above 38˚C.

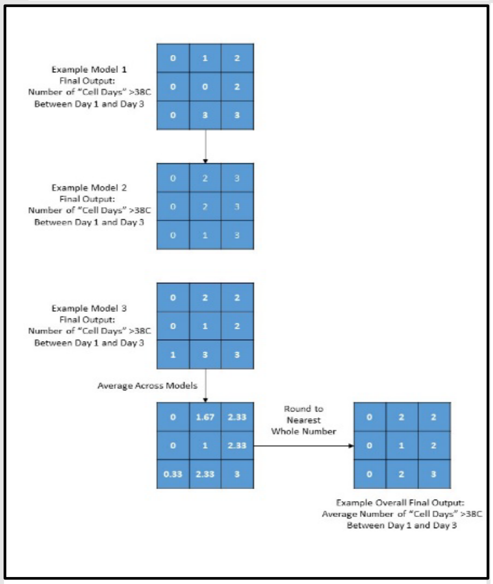


Figure 3A4. Averaging combined model outputs to get the average number of cell days above 38˚C.

Outside North America

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, unpaginated; Cinquini 2014, entire). 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 et al. 2012, entire). 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 4. Additional Results

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

Table 4A1. Comparison between simulated 2012 (April and May) baseline and 2069 (April and May) total number of 4.6km x 4.6km grid cells, including average number of cells and percent change in the area with at least one day above 38˚C and 42˚C under RCP 4.5. Also, the percent change in the area with at least one day above 42˚C under RCP 4.5 in July and August (2012–2069).

Conservation Unit	Total Number of Cells	Number of Cells with at least 1 Day >38°C under RCP 4.5 (April and May 2012)	Number of Cells with at least 1 Day >38°C under RCP 4.5 (April and May 2069)	% Change in Area of >38°C Days under RCP 4.5 (April and May 2012 - 2069)	Number of Cells with at least 1 Day >42°C under RCP 4.5 (April and May 2012)	Number of Cells with at least 1 Day >42°C under RCP 4.5 (April and May 2069)	% Change in Area of >42°C Days under RCP 4.5 (July and August 2012 - 2069)
North Central	134,563	3,845	38	-99	67	0	-100
North East	23,445	0	285	--	0	0	--
South	147,796	32,573	63,306	94	100	6,730	6,630
West	161,501	29,085	22,284	-23	10,452	9,283	-11

Table 4A2. Comparison between simulated 2012 (April and May) baseline and 2069 (April and May) total number of 4.6km x 4.6km grid cells, including average number of cells and percent change in the area with at least one day above 38˚C and 42˚C under RCP 8.5. Also, the percent change in the area with at least one day above 42˚C under RCP 8.5 in July and August (2012–2069).

Conservation Unit	Total Number of Cells	Number of Cells with at least 1 Day >38°C under RCP 8.5 (April and May 2012)	Number of Cells with at least 1 Day >38°C under RCP 8.5 (April and May 2069)	% Change in Area of >38°C Days under RCP 8.5 (April and May 2012 - 2069)	Number of Cells with at least 1 Day >42°C under RCP 8.5 (April and May 2012)	Number of Cells with at least 1 Day >42°C under RCP 8.5 (April and May 2069)	% Change in Area of >42°C Days under RCP 8.5 (July and August 2012 - 2069)
North Central	134,563	0	10,081	--	0	301	--
North East	23,445	0	170	--	0	0	--
South	147,796	35,446	106,223	200	2,777	48,250	1,637
West	161,501	24,983	52,323	109	7,403	18,388	148

Table 4A3. Comparison between simulated 2012 (April and May) baseline and 2069 (April and May) total number of cell days, including average number of cell days and percent change in the number of cells days with at least one day above 38˚C and 42˚C under RCP 4.5. A cell day is a unit of space and time, representing the number of raster cells in a geographic region within a period of time (April–May or July–August in this case) above a temperature threshold.

Conservation Unit	**Total Number of "Cell Days"**	**Number of Cell Days >38°C Under RCP 4.5 (April and May 2012)**	**Number of Cell Days >38°C Under RCP 4.5 (April and May 2069)**	**% Change in Cell Days >38°C Under RCP 4.5 (April and May 2012-2069**	**Number of Cell Days >42°C Under RCP 4.5 (April and May 2012)**	**Number of Cell Days >42°C Under RCP 4.5 (April and May 2069)**	**% Change in Cell Days >42°C Under RCP 4.5 (April and May 2012-2069**
North Central	8,208,343	3,845	38	-99	67	0	-100
North East	1,430,145	0	285	--	0	0	--
South	9,015,556	59,026	254,346	331	114	9,402	8,147
West	9,851,561	123,495	169,865	38	23,542	26,185	11

Table 4A4. Comparison between simulated 2012 (April and May) baseline and 2069 (April and May) total number of cell days, including average number of cell days and percent change in the number of cells days with at least one day above 38˚C and 42˚C under RCP 8.5. A cell day is a unit of space and time, representing the number of raster cells in a geographic region within a period of time (April–May in this case) above a temperature threshold.

Conservation Unit	**Total Number of "Cell Days"**	**Number of Cell Days >38°C under RCP 8.5 (April and May 2012)**	**Number of Cell Days >38°C under RCP 8.5 (April and May 2069)**	**% Change in Cell Days >38°C under RCP 8.5 (April and May 2012-2069**	**Number of Cell Days >42°C under RCP 8.5 (April and May 2012)**	**Number of Cell Days >42°C under RCP 8.5 (April and May 2069)**	**% Change in Cell Days >42°C under RCP 8.5 (April and May 2012-2069**
North Central	8,208,343	0	10,081	--	0	301	--
North East	1,430,145	0	170	--	0	0	--
South	9,015,556	35,446	106,223	200	3,621	133,055	3,575
West	9,851,561	24,983	52,323	109	20,104	56,704	182

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

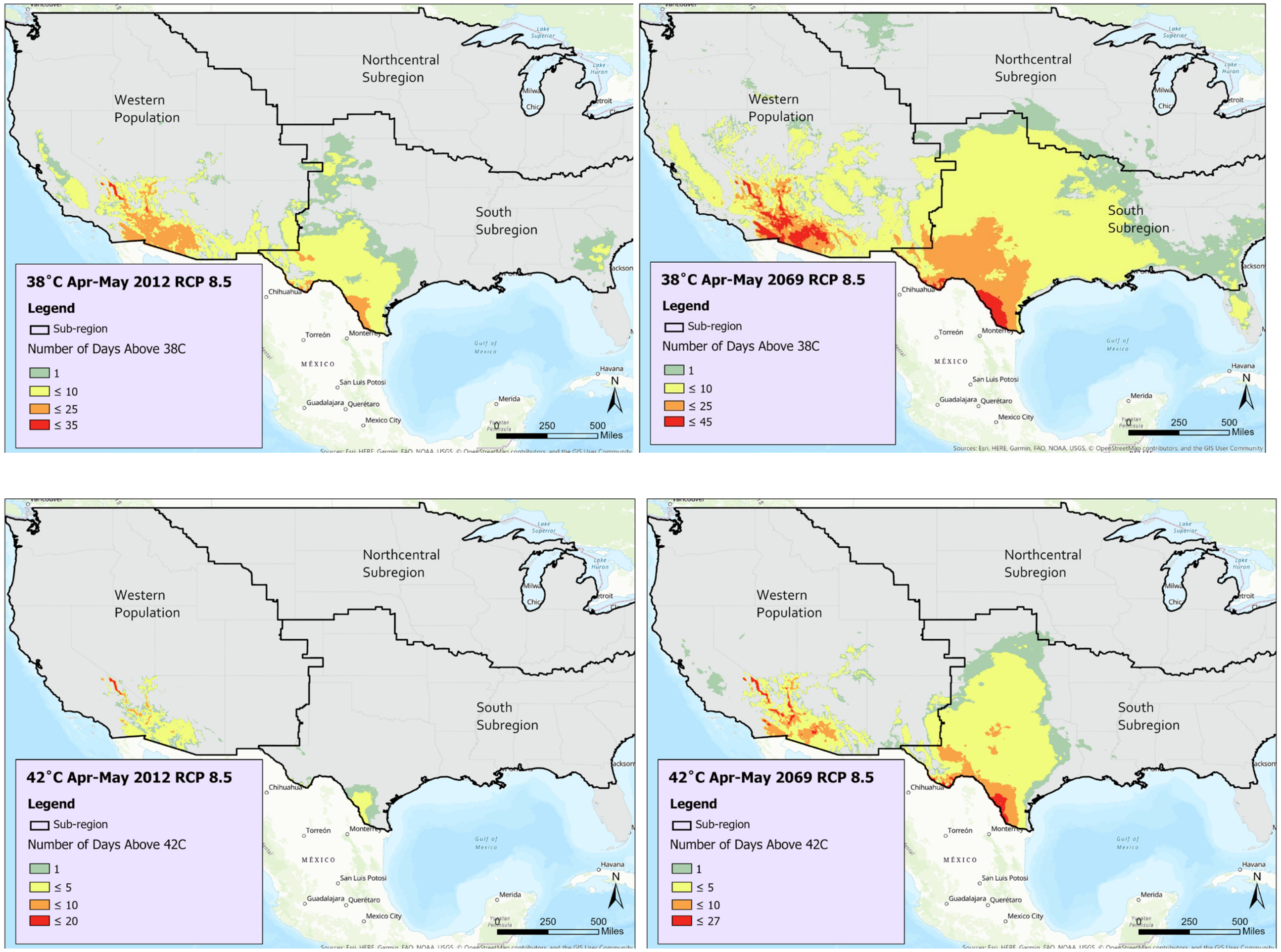


Figure 4A1. The spatial extent and average number of days >38˚C (top) and 42°C (bottom) in April and May 2012 (left) and 2069 (right) under RCP 8.5.

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

Table 4A4. pE *values for the western and eastern North American populations.* pE *predictions under current state conditions represent the 50% confidence interval.*

Population	State Condition	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
Western Pop	Current – 75%	0.68	0.85	0.93	0.97	0.98	0.99
Western Pop	Future – Worst case	0.71	0.88	0.95	0.97	0.99	0.99
Western Pop	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
Eastern Pop	Current – 75%	0.08	0.22	0.36	0.49	0.60	0.69
Eastern Pop	Future – Worst case	0.09	0.29	0.46	0.58	0.67	0.75
Eastern Pop	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
p(both pops persist)	Current – 75%	0.29	0.12	0.04	0.02	0.01	0.00
p(both pops persist)	Future – Worst case	0.27	0.08	0.03	0.01	0.00	0.00
p(both pops persist)	Future – Best case	0.33	0.14	0.06	0.02	0.01	0.00

Appendix 5. Supplementary Information About Pesticides

Authors: Sarah Warner, Nancy Golden, Dave Warburton, Lisa Williams
Last updated February 2, 2024

[1] The Risk of Insecticides to the Monarch Butterfly

The risk of insecticide impacts to monarchs is primarily influenced by the extent to which monarchs are exposed to insecticides throughout their range. This assessment presents an overview of: (1) the use of insecticides within monarch habitat, (2) pathways of monarch exposure to insecticides, (3) toxicity of insecticides to monarchs, and (4) a summary evaluation of insecticide risk. Factors influencing insecticide exposure and the uncertainties inherent in these factors are also presented to guide future research/monitoring and monarch conservation strategies.

Insecticides in Monarch Habitat

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. Monarch habitat includes gardens and yards, urban parks, farmlands and other agricultural production areas, rights of way, and protected natural areas. Though pesticide use is most often associated with agricultural production, any habitat where monarchs are found may be subject to insecticide use or exposure. 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 insecticide-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.

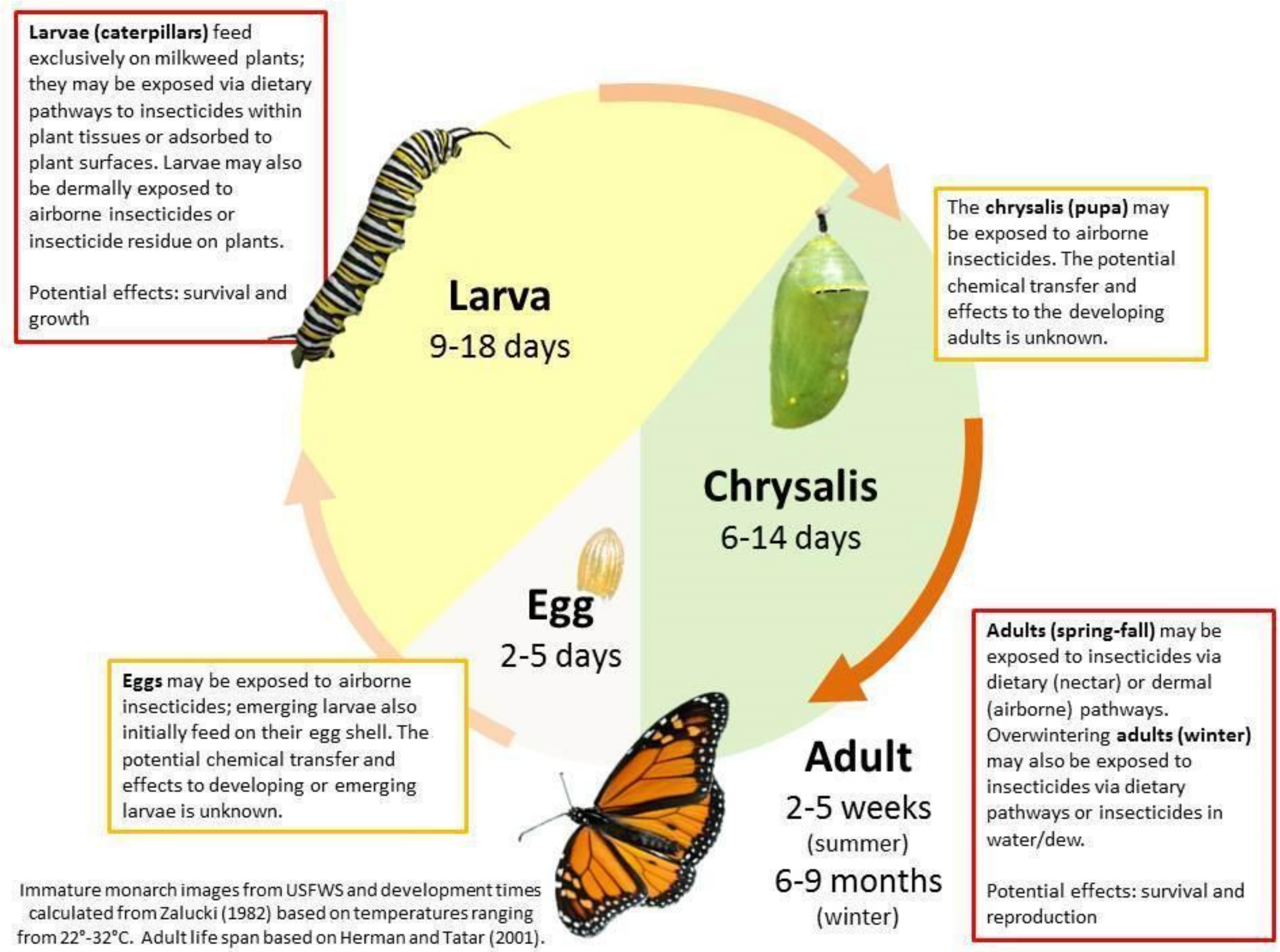
Expenditures on insecticides in 2012 topped $5 billion in the United States, with an estimated 60 million pounds being used for agriculture (57%), home and garden (23%), and in the industrial/commercial/governmental sector (20%; EPA 2017, pp. 6–7). Chemical classes of the most commonly used insecticides during the time of the report (2008 - 2012) were organophosphates and carbamates, and pyrethroids (EPA 2017, pp. 13–19). In addition, neonicotinoid insecticides (a class of insecticides first registered in the 1990s) accounted for 80% of global seed treatment sales by 2008 (Jeschke et al. 2011, p. 2898). Treated seeds are used for nearly all of the corn and soybean crop acreage in the U.S. (Douglas and Tooker 2015, pp. 5088–5097), and neonicotinoid-treated plants are commonly sold as ornamentals for yards and gardens.

Given this extent of insecticide use over the wide distribution of monarch habitat across a variety of land use sectors, there is significant potential for monarchs to be exposed to insecticides in the United States.

Monarch Insecticide Exposure Pathways

Insecticide exposure pathways to both adults and larvae of the monarch include: (1) *dietary exposure* (ingestion of an insecticide on or within plant tissue that the monarch is feeding upon), and/or (2) *contact exposure* (direct contact with airborne insecticides that land on the monarch or are deposited on plants that the monarch comes in contact with). Figure 1 illustrates these potential insecticide exposure pathways to each life stage of the monarch. While the monarch may be exposed to insecticides throughout all life stages, this evaluation is limited to larval and adult stages, as these are considered to be the most significant from a biological perspective, and the most likely in actual environmental settings. Further, there are insufficient data to evaluate exposure and effects to the other life stages beyond a conceptual analysis. Due to overlapping generations of monarchs through the spring-fall months, both larvae and adults may be exposed to insecticides in any given geographic location the species may occur outside of its overwintering areas.

Figure 5A1. Insecticide exposure pathways to monarch life stages.



Larvae (caterpillars) feed exclusively on milkweed plants; they may be exposed via dietary pathways to insecticides within plant tissues or adsorbed to plant surfaces. Larvae may also be dermally exposed to airborne insecticides or insecticide residue on plants.

Potential effects: survival and growth

The **chrysalis (pupa)** may be exposed to airborne insecticides. The potential chemical transfer and effects to the developing adults is unknown.

Eggs may be exposed to airborne insecticides; emerging larvae also initially feed on their egg shell. The potential chemical transfer and effects to developing or emerging larvae is unknown.

Adults (spring-fall) may be exposed to insecticides via dietary (nectar) or dermal (airborne) pathways. Overwintering **adults (winter)** may also be exposed to insecticides via dietary pathways or insecticides in water/dew.

Potential effects: survival and reproduction

Immature monarch images from USFWS and development times calculated from Zalucki (1982) based on temperatures ranging from 22°-32°C. Adult life span based on Herman and Tatar (2001).

Figure produced by Kelly Nail and Dave Warburton, USFWS.

Insecticides can move through the environment and expose monarchs by the following routes:

1) **Direct Spray**: Monarchs that inhabit the same area as insect pests are susceptible to insecticide exposure (through either dietary or contact pathways) via direct spray of insecticides. One significant scenario for this occurrence is in areas subject to mosquito control with pyrethroid and organophosphate insecticides (used as mosquito adulticides).
2) **Pesticide Drift**: Monarchs may be exposed to pesticides via dietary or contact pathways in any area adjacent to a treatment location where the pesticide leaves the site of application ("drifts") via droplets, vapor, or dust. Whether a pesticide will drift, and how far from the treatment area that drift occurs, are influenced by numerous factors including method of application, height of spraying equipment, wind speed, weather conditions, nozzle size, terrain, and the use of best management practices by applicators to control for these factors and limit drift occurrences.
3) **Systemic**: Monarchs may be exposed via dietary pathways to insecticides that become incorporated into plant tissues (e.g., leaves, pollen, nectar). Although numerous insecticides may be systemic to some degree, neonicotinoids in particular are known for this characteristic, and are expressed throughout the plant including nectar and pollen of treated crops and plants (Goulson 2013, p. 977).

The degree to which an insecticide persists and moves through the environment can influence its availability, and thus exposure to monarchs. Pesticides can differ widely in these characteristics, even within the same class of chemicals; those which persist longer or are more mobile can result in greater exposure to monarchs.

For example, chemical characteristics of many neonicotinoids include high water solubility and relatively long persistence in the environment. These characteristics contribute to the propensity of neonicotinoid insecticides to transport long distances beyond use areas. Neonicotinoids have been found in well-water (Starner and Goh 2012, pp. 317–320; Huseth and Groves 2014, pp. 5–9), and can also drift off-site when incorporated into pollen (Bonmatin et al. 2015, pp. 52–60), suggesting far-reaching effects and potential landscape-scale mobility. When used as seed treatments, over 90% of the active ingredient can enter the soil and remain available (reported half-lives range from 200 to over 1000 days; Goulsen 2013, p. 979). During seed sowing, less than 2% is lost in dust-off; more can be lost and deposited in the field margin areas if talcum powder or graphite is added to the seeds (Krupke et al. 2012, p. 2).

For a monarch to be exposed to an insecticide through its diet, residues must be deposited on or incorporated within the dietary item associated with the relevant life stage, specifically milkweed leaves for larvae and nectar from flowers for adults. How the plant metabolizes or stores insecticides in its tissues and how it is expressed in leaves or nectar can influence exposure potential and the degree of risk to monarchs and needs to be studied. While insecticide residues have been documented in both of these media, few studies exist to help estimate concentrations (i.e., the magnitude of exposure) in the variety of areas where monarchs may be exposed, including agricultural and adjacent lands, residential areas, and parks or other presumed natural areas.

Exposure to pesticides in pollen and nectar

While monarchs are not expected to feed on pollen, reports of its widespread contamination in crop areas illustrates the ability of flowering plants to serve as sources of exposure, at least in areas in and around crops. Presence in pollen is likely indicative of presence in nectar and with further investigation into the relative accumulation of residues, concentrations measured in pollen may be used to estimate concentrations in nectar. There is some evidence that residues in nectar may be lower than those in pollen, though factors such as application method, application timing, and environmental conditions are likely to affect concentrations available to monarchs from this source. There are few North American studies measuring concentrations occurring in plants following exposure based on typical or labeled application methods, and a lack of field sampling from active crops and non-crop areas.

Investigations of contaminants in honeybee colonies illustrate that insecticides used in crops are available to pollinating insects. In a large-scale study of colonies in 23 states and one Canadian province, representing several agricultural cropping systems, concentrations of 98 different pesticides were detected in collected bee pollen (Mullin et al. 2010, p. 3). Bee pollen, which aggregates pollen collected from different individuals and flowers, contained an average of 7 pesticides per sample. Chlorpyrifos was the most frequently detected insecticide in 44% of samples.

Residues of insecticides were regularly detected in pollen and nectar following two studies of experimental pesticide applications in field conditions, though concentrations varied. Average concentrations of neonicotinoids in pollen from pumpkins following various methods of application ranged up to 80.2 ng/g imidacloprid (plus an additional 19.1 ng/g metabolites), 88.3 (10.3) ng/g dinotefuran, and 95.2 (26.8) ng/g thiamethoxam (Dively and Kamel 2012, pp. 4449–4456). Concentrations were lower in the second year of the study, presumably due to extreme environmental conditions resulting in heat and moisture stress. Neonicotinoid metabolites accounted for 15 - 27% of total residues across years. Residues in nectar were consistently 74 - 88% lower than pollen residues, and residues in leaves were generally higher, though only correlated with values in pollen and nectar for imidacloprid. At-planting applications resulted in the lowest concentrations, and those applications occurring closer to flowering resulted in higher residues. In another study, concentrations of imidacloprid or thiamethoxam in nectar and pollen of squash treated via soil application or drip irrigation (a subset of the application methods tested in the above study) resulted in similar concentrations in pollen (5–35 ng/g) and nectar (5–20 ng/g) regardless of application method, insecticide, or study year (Stoner and Eitzer 2012, pp. 3–4). Average concentrations were 14 ng/g imidacloprid and 12 ng/g thiamethoxam in pollen, and 10 ng/g imidacloprid and 11 ng/g thiamethoxam in nectar. Residues were similar across two study years despite rainfall totals in the second year about half of those in the first. Data for metabolites were not presented.

In a study simulating greenhouse application, residues of imidacloprid and its metabolites (hydroxy and olefin), were measured in Mexican milkweed (*Asclepias curassavica*) flowers following soil applications at labeled rates for greenhouse use (Krischik et al. 2015, p. 3). Whole flowers contained a mean of 6,030 ng/g midacloprid and 980 ng/g metabolites 21–51 days post-application. A second soil application 7 months after the first resulted in mean concentrations of 21,670 ng/g imidacloprid and 6,440 ng/g metabolites in whole flowers. The authors speculated that the higher residues from this application may be due to concentration in flowers during a time of slower vegetative growth. Metabolites accounted for 14% and 23% of total residues for each year, respectively, similar to the percentages measured in nectar and pollen described above. The authors acknowledge that residues in pollen and nectar may be different than residues in whole flowers and that the correlation needs to be scientifically determined.

Exposure to insecticides in milkweed leaves

Larval monarchs can be exposed to insecticides by ingesting residues that are expressed in the leaf tissue of milkweeds. Insecticides have been detected in milkweed leaves near agricultural fields in at three two studies. Variation in frequency of detection and concentration levels across years or seasons was common to both studies. While the two studies below measure concentrations in common milkweed, it is worthwhile to note that in the toxicity studies reviewed below, monarchs are exposed using four different species of milkweed plants. At present, it is not known whether the pharmacokinetics (i.e., how the plant metabolizes, stores, and expresses systemic insecticides in its tissues) is comparable across milkweed species and how this may affect the exposure and bioavailability to monarchs using these plants.

Clothianidin was measured in common milkweed (*Asclepias syriaca*) leaves that were adjacent to fields (mean distance of 1.47 m) at eight sites in South Dakota shortly after maize planting in 2014 using an ELISA method[1] (Pecenka and Lundgren 2015, pp. 2–4). Mean clothianidin concentration per plant was reported as 0.58 ppb overall and 1.14 ppb in plants with detectable residues, with a maximum 4.02 ppb in one plant. Clothianidin was detected in about half of the samples, with twice the proportion having detectable residues in July (65%) compared to June (37%). Monitoring of plants during sampling revealed that monarchs were actively using these sites, with an average of 1.3 eggs and 0.6 larvae per plant in June, and 1.4 eggs and 0.3 larvae in July.

Olaya-Arenas and Kaplan (2019, pp. 3–10) analyzed pesticides in soil and leaves of common milkweed (*A. syriac*a) within 100 m of crop fields in northwest Indiana to determine if areas adjacent to fields provide greater exposure to monarchs. Three neonicotinoids were detected in leaves with variation in percent detection and concentrations by year. Clothianidin was detected in 15–25% of samples in June, but rarely detected in July or August. Concentrations varied between 2015 (0.71 ng/g mean, 56.5 ng/g maximum) and 2016 (0.48 ng/g mean, 28.5 ng/g max). Thiamethoxam was detected in just 2% of samples in 2015 (0.19 ng/g mean, 94.8 ng/g max), yet found in 75–99% in 2016 (1.87 ng/g mean, 151.3 ng/g max). Imidacloprid was only detected in 0.2% of samples in 2015 (up to 3.7 ng/g) and was not detected in 2016. The pyrethroid deltamethrin was detected in 98.9% of samples in 2016 (37.0 ng/g mean, 1,352.9 ng/g max). Distance from the edge of a crop field or the amount of crop was generally a poor predictor of pesticide detection, with only thiamethoxam demonstrating this relationship. Clothianidin was the only insecticide detected in soil, with concentrations consistent throughout the summer and

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[1] In reviewing the methods as described in this paper and correspondence with one of the authors who stated that he did not think that leaf disks were weighed, it is not clear whether the reported concentrations in ppb are on a ng/g basis in the leaves or a ug/L basis in the leaf extracts, so these concentrations should be considered to be less certain than those from other publications cited in this document.

correlated with those in milkweed leaves. In general, higher concentrations of insecticides were found earlier in the season with year to year variation.

Halsch et al. (2020, pp. 3–7) investigated insecticide exposure to milkweed plants across three land-use sectors that included agriculture, wildlife refuges, urban parks and gardens in northern California. The field study determined what pesticides are available to monarch during a one-time sampling event in late June - when monarch larvae are likely to be present. In this field study, 227 leaf samples of narrowleaf milkweed (*Asclepias fascicularis,*161 samples), common milkweed (*A. speciosa*, 50), woolly pod milkweed (*A. eriocarpa*, 4) and tropical milkweed (*A. curassavica*, 12) were collected from 19 sites across the Central Valley. The sites were located in conventional farms, an organic farm, a milkweed restoration site, a roadside location adjacent to an agriculture field, five in wildlife refuges, four in urban areas, and two from retail nurseries. In addition to the milkweed samples that were collected in the field, milkweed plants were purchased from home and garden stores and leaves were analyzed for pesticides. A total of 64 pesticides were detected across samples: 25 insecticides, 27 fungicides, 11 herbicides, and 1 adjuvant. A greater number of pesticides were detected in plants sampled from agricultural and retail locations compared to samples from refuge and urban sites. Chlorantraniliprole (registered for use in urban areas) was detected in 91% of the samples and methoxyfenozide (registered for use on a variety of crops) was measured in 96% of samples. The authors compared the concentrations detected in milkweed leaves to honeybee and monarch toxicity levels. Sixteen percent (36 out of 227) of the milkweed leaves sampled had concentrations over an LD50 value for honeybee toxicity with exceedances from 7 of the 19 sampled sites. Three other pesticides (cyantraniliprole, fipronil, and methoxyfenozide) exceeded a honeybee LD50 and these were sampled from retail and urban sites. In 25% of the samples, chlorantraniliprole concentrations exceed a tested LD50 for monarchs. Clothianidin was detected above a monarch LD50 from one agriculture site. Authors indicate that for the vast majority of the pesticides detected in the milkweed leaves it is unknown what the biological effects are on monarch caterpillars.

To explore pesticide residues and effects to monarchs within agriculture landscapes, Hall et al. (2022, pp. 160–167) investigated insecticide concentrations in soils and milkweed plants in pollinator plantings (i.e. prairie strips) within maize and soybean fields that use neonicotinoids to control for crop pests. Clothianidin was found in all soil samples (85) within the prairie strips. Thiamethoxam and imidacloprid were detected in >90% of the samples. In milkweed, neonicotinoids were found in 80% of leaves sampled (358), both clothianidin (58%) and imidacloprid (67%) were detected. The results indicate that plants growing in prairie strips can contain neonicotinoids and the likely source of soil and plant exposure is treated seed used in crops adjacent to the prairie strips. In prairie strips, the exposure to the monarch could be to adults foraging on flowering plants or to the larvae feeding on milkweed. Although there could be exposure, the authors suggest that the risk to monarchs using the agricultural prairie strips is low due to the levels detected in plant tissues from their study, which were below reported toxicity thresholds of concern for larvae.

Effects of Insecticides to Monarchs

Insecticides are pesticides with chemical properties that are designed to kill insects. Their main uses are to control insect pests in agricultural production, natural habitats, lawns and gardens, and in and around households and buildings. The U.S. Environmental Protection Agency

(USEPA), under the authority of the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), regulates and registers pesticides for use in the United States. To evaluate the environmental risk of proposed pesticide use as part of the registration process, the USEPA requires laboratory studies of toxicity to select non-target species. The non-native honeybee (*Apis mellifera*) is currently the primary invertebrate surrogate used in testing to evaluate risks to non-target terrestrial insects. If negative effects to non-target species are anticipated from the proposed use of a pesticide, the USEPA may choose to not approve the pesticide for registration, or to require restrictions on pesticide labels to help minimize anticipated impacts. However, under FIFRA risk management, a degree of non-target risk may be deemed acceptable if the risks are outweighed by the potential benefits of use of a pesticide. Therefore, risk to non-target species, including monarchs, cannot be ruled out simply because a pesticide has undergone the registration process and is used according to the label.

Most insecticides considered herein are non-specific and broad-spectrum in nature. That is, insects exposed to insecticides are broadly susceptible to mortality and sublethal effects. Furthermore, the larvae of many insects in the Order Lepidoptera are considered major pest species, especially in agricultural and forested areas, and insecticides are tested specifically on this taxon to ensure that they will effectively kill individuals at labeled application rates. Therefore, it is reasonable to presume that monarchs exposed to insecticides within areas of use are likely to be killed or otherwise affected following an application. Monarchs exposed in areas outside insecticide use where drift occurs may also be affected depending on the concentration of the pesticide to which they are exposed.

Scientific data documenting insecticide effects to lepidopterans are largely limited to: (1) laboratory dosing studies on larvae to investigate the toxicity of an insecticide with various endpoints measured, (2) modeling studies predicting the extent of insecticide threat to individuals or populations, and (3) field-based studies that investigate insecticide concentrations in plant tissues (as described above) and/or attempt to measure effects to populations in treated and untreated areas. All three types of studies have their limitations. For example, standardized methods of laboratory toxicity testing have not yet been adopted for lepidopteran species, resulting in inconsistencies in exposure regimes (e.g., duration, contact vs ingestion, life stage) and reporting of toxicity values (e.g., units of measurement). Lack of accepted testing protocols confound the ability to make comparisons across studies and species. Given such variability, this section presents a brief summary of select information from published literature on the effects and toxicity of three widely-used classes of insecticides to monarchs or other lepidopteran species: organophosphates, pyrethroids, and neonicotinoids. Conclusions are noted where possible. Other classes of insecticides and other types of pesticides can be similarly investigated.

Organophosphates and pyrethroids

Information on direct toxicity of organophosphate and pyrethroid insecticides to lepidopteran species is available from efficacy studies on target pest species (particularly *Pieris brassicae* and related species, reviewed in Braak et al. 2018, pp. 507–518). In this assessment, we generally focus on toxicological effects to non-target species, with data available within the families Nymphalidae, Lycaenidae, Papilionidae, Hesperiidae, and Pieridae (Salvato 2001, pp. 9–12; Hoang et al. 2011, pp. 998–1004; Eliazar and Emmel 1991, pp. 18–19; Hoang and Rand 2015, pp. 715–718; Bargar 2012a, pp. 2125–2128; Davis et al. 1991, pp. 151–161). Most studies measured the acute toxicity of insecticides to various lepidopteran species and report median lethal dose values (LD50s) for dietary or contact exposure pathways. Methods varied across studies in relation to length of exposure, life stage, chemical form (active ingredient vs formulated product), and exposure regime. In general, while toxicity was exhibited across all species and chemicals, no consistent patterns emerged either within or across studies that demonstrated sensitivity was related to species (or species group), life stage, or size of adults, though inconsistency in testing regimes may limit the ability to detect patterns that exist. Of the organophosphates tested (dichlorvos, malathion, naled, and dimethoate) species tended to exhibit the greatest sensitivity to naled and the least to malathion, though these results were not always consistent across species and methods. For pyrethroids, toxicity values were reported for two insecticides, permethrin and resmethrin. However, resmethrin testing was performed in formulation with piperonyl butoxide, a synergist that is combined with pesticides to enhance toxicity and comparisons cannot be made between relative toxicity of these two insecticides. Based on the available data from these insecticide studies, there is no evidence to imply that a particular species or family of lepidopterans is expected to exhibit more or less sensitivity to a particular organophosphate or pyrethroid than others, including targeted pest species.

Only two studies looked specifically at effects to monarchs within these classes of insecticides. Both studies found that monarchs exposed to pyrethroids at concentrations expected following field applications could experience mortality. Oberhauser et al. (2006, pp. 1629–1631) found that larvae that consumed milkweed leaves treated with permethrin in dilutions of field operable solutions (dilutions 0.5 and 0.1%) had significantly reduced rates of survival. Of the 60 larvae exposed to the two treatments, 37 died (33 as larvae and 4 as pupae) and larval stage development time was significantly delayed. Survival rates were lower for first instar larvae compared to later instar larvae. In the same study, effects to female oviposition choice, the number of eggs laid, and survival 1, 8, and 15 days after the initial spray event. Females were placed in enclosures that contained milkweeds exposed across three treatment groups: (1) milkweed plants sprayed with operational solutions of permethrin, (2) milkweed sprayed with operational solutions of permethrin, treated with oil solution, and untreated, and (3) milkweed plants that were untreated. Overall female survival was low for the two permethrin treatments (8–16 %) compared to 92% survival for the untreated treatment; with the lowest survival rate 1 day after the initial spray event. In addition, the studies found that ovipositing females did not discriminate amongst treatment groups, but fewer eggs were laid on permethrin treated plants 1 day after initial spray date compared to treated plants 8 and 15 days later.

Oberhauser et al. (2009, pp. 84–90) exposed adult and larval monarchs to ultra-low volume (ULV) applications of resmethrin (as the formulated product Scourge, which contains resmethrin plus the synergist piperonyl butoxide) to evaluate the effects of mosquito control on monarchs. Three experiments examined impacts to survival in adults and larvae subject to direct spray at varying locations upwind and downwind, and in larvae consuming previously exposed milkweed. Monarch mortality varied with conditions of experimental design, but significant increases over controls were found at distances up to 120 m downwind from the application site over the three experiments. Milkweed plants sprayed one day prior to monarch exposure resulted in significant mortality to larvae as compared to controls. In one of three experiments, adult mass was negatively affected by exposure to resmethrin. One experiment exposed house fly (*Musca domestica*) and milkweed bug (*Oncopeltus fasciatus*) larvae to resmethrin under conditions that caused monarch mortality and found no effects to survival of either species.

Neonicotinoids

There are published studies examining the toxicity of neonicotinoids to monarchs (described herein). A summation of toxicity values of neonicotinoids across taxa (insects, birds, fish, molluscs, mammals, annelids) found insects to be the most sensitive taxa when exposed via contact or the dietary/ingestion pathway with LD50s ranging from 0.82 to 88 ng per insect (Goulson 2013, pp. 983–984). The variation in LD50 values is attributed to size of the insect, with the most sensitive insect being the brown planthopper (*Nilaparvata lugens*; a native species) weighing 1 mg, and the least sensitive insect being the Colorado potato beetle (*Leptinotarsa decemlineata*; a crop pest and non-native species) weighing 130 mg.

Several lab dosing studies looked specifically at neonicotinoid effects to monarchs at varying life stages. While reduced survival was detected in most treatments, results of each study were influenced by differences in pesticide tested, life stage, exposure regime, and experimental methods. Pecenka and Lundgren (2015, pp. 2–3) attempted to mimic a pulsed exposure in the field by feeding swamp milkweed leaves dosed with clothianidin to larvae for 36 hours during the first stadium, and then observing effects up to the third instar. Each larva was fed a single 1 cm milkweed disk with an aqueous solution of clothianidin on agarose gel on the leaf. Once that disk was consumed, the larvae were then fed clean milkweed leaves until the end of the experiment in the third instar. Increasing mortality was observed with increasing dose, measured in µg/L (ng/g) clothianidin in the 10 µL of solution applied to each leaf disk: the LC10, LC20, LC50, and LC90 concentrations were found to be 7.72, 9.89, 15.63, and 30.70 ng/g, respectively. Significant effects to development time, body length, and weight for newly eclosed second instars were observed at doses as low as 0.5 ng/g. This study reveals effects to monarchs at seemingly low environmental concentrations of clothianidin; however, concentrations as reported (ug/L of solution per leaf disk) are not easily extrapolated to typical concentration units for a dietary testing exposure scenario (gram per leaf or ng/g ww of leaf). Therefore, it is difficult to make a direct comparison to concentrations expected to be found on milkweed leaves in the environment.

Krischik et al. (2015, pp. 3–11) investigated imidacloprid rates for greenhouse/nursery use. The authors suggest that this particular use of the insecticide can result in higher concentrations of residues found in flowering plants compared to imidacloprid used as a seed treatment; therefore, it was selected for the study. Multiple experiments were conducted using Mexican milkweed (*Asclepias curassavica)* plants with imidacloprid applied to the soil to investigate dietary exposure pathways from whole flowers or plant tissues to insects. Mexican milkweed flowers grown in soils treated with imidacloprid at labeled rates reduced survival in 3 of 4 lady beetle species, in some cases as soon as two/three days after treatment. Adult monarch and painted lady butterflies either free-ranging or force-fed imidacloprid in solution showed no effects to survival, fecundity, or egg hatch at either labeled rates or twice labeled rates. However, larval survival of both species was reduced by day 7, with few monarchs surviving past this point. Authors hypothesized that adult butterflies may not metabolize the insecticide, instead excreting it unchanged.

James (2019, pp. 3–5) examined the effects of nectar dosed with imidacloprid on monarch longevity and egg production. For the 28-day study, adult monarchs (11 males, 11 females) were consistently fed a sugar-water solution containing 23.5 ng/g imidacloprid, a concentration within the range detected in nectar of crop plants. Mortality occurred in dosed monarchs and individuals exhibited behavioral effects by day 12 (uncoordinated flapping of wings and uncontrolled vibrating of body and wings). Sample sizes throughout the study were low: At 12 days post eclosion, 4 males and 4 females remained in the dosed group with 4 males and 4 females in the control. At 22 days post eclosion, 2 individuals remained in the dosed group with 3 males and 5 females in the control. No effects were detected in mass, forewing length, oocyte development, and growth. This study tested one scenario in which adult monarchs feed on the nectar of crop plants treated with imidacloprid under certain conditions. It is uncertain the degree and frequency to which monarchs nectar on crop plants, the full range of concentrations likely to be present in treated plants, and if the imidacloprid concentration tested is representative of what could be expressed in the nectar of native flowering plants.

To determine the residue level in the milkweed tissue that leads to an adverse effect to monarchs, Bargar et al. (2020; pp. 2–15) conducted three experiments that estimated the dietary exposure level of clothianidin associated with adverse effects in monarch butterflies. Results showed transfer of clothianidin from soil to milkweed plant (swamp milkweed- *Asclepias incarnata*), to larvae and to adult – this is the first study to show lifestage transfer from soil to adult. In the experiments, swamp milkweed plants were dosed (via soil treatments) with five concentrations, each experiment increasing dose levels, and larvae were exposed via dietary exposure from the time they hatched from eggs until pupation. Endpoints measured included larval survival and growth, pupation success, and adult mass. Experiment 1 consisted of concentrations that included the label rate for application of a clothianidin product, while Experiments 2 and 3 included only concentrations greater than the label application rate. In Experiment 1, clothianidin was measured in the milkweed leaves and detected in the larvae only at the two dose levels greater than the label rate, with concentrations in leaves measured at 11 ng/g (SD = 3.6) and 54 ng/g (SD = 27) in the two dose groups and in larvae at 6.0 ng/g (SD 3.3) and 13 ng/g (SD 3.4). Two of the three surviving adult butterflies from the highest dose group had detectable concentrations of clothianidin (3.1 and 5.2 ng/g). At the label application rate, concentrations in leaves, larvae, and adults were all below the detection limit and no significant effects to larval growth and survival, adult mass, pupal were observed. For Experiments 2 and 3, dose levels were all greater than label application rates for several clothianidin products. The greater dose levels resulted in detectable concentrations in leaves and larvae from all treatments. Experiment 3 was conducted to eliminate the possible effect of aphids that infested plants during Experiment 2; therefore, only the results for Experiment 3 are reported herein; however, the elevated exposure in both experiments led to adverse effects on survival and growth. Clothianidin was detected in the milkweed leaves at measurable concentrations ranging from 54 (SD = 42) to 1,545 ng/g (SD = 481), and larval consumption of the contaminated leaves negatively affected larval growth and adult survival. Larval growth was affected at 1,154 ng/g leaf and no larvae in this highest dose level reached the pupal stage. Larval mortality ranged from 50% in the lowest dose level (54 ng/g leaf) to 100% in the highest dose level, and 33–50% of the monarch butterflies died at the pupal stage in both of the lowest dose levels tested. Four adult monarchs successfully eclosed, three in the control and one in the lowest dose level. Due to the results of the three consecutive experiments, the authors suggest that clothianidin concentrations expected from applications that follow the label in wild milkweed plants are generally not high enough to adversely affect monarch butterflies and that monarchs may be relatively insensitive to clothianidin at label application rates.

Krishnan et al. (2020, pp. 924–931) conducted contact (cuticular) and dietary toxicity tests on monarch butterfly larvae at each lifestage for five insecticides that are registered for use as foliar applications on maize and soybean: a pyrethroid (beta-cyfluthrin), an anthranilic diamide (chlorantraniliprole), an organophosphate (chlorpyrifos), and two neonicotinoids (imidacloprid and thiamethoxam). For the dietary assays, larvae were reared on insecticide-treated tropical milkweed (*Asclepias curassavica*) leaves for 48 or 24 hours. Contact and dietary LD50s differed among larval stages with first instars being the most sensitive followed by third and fifth instars. The LD50 concentrations for beta-cyfluthrin and chlorantraniliprole ranged from 9.2 to 480 ng/g larva and 12.0 to 190 ng/g larva. respectively, and were the most toxic insecticides across all instars. Chlorpyrifos was the least toxic to first instars (LD50 of 79,000 ng/g larva). For the neonicotinoids, clothianidin was more toxic to larvae than both imidacloprid and thiamethoxam. In a separate study, Krishnan et al. (2021b, pp. 1773–1775) compared monarch insecticide exposure rates from previous studies to modeled spray drift predictions to determine risk. The authors suggest that the overall risk to monarch from foliar applications of insecticides is high and expected to cause downwind mortality compared to risk from neonicotinoid seed treatments which is expected to cause little to no downslope mortality.

Several studies previously summarized that suggest risk of neonicotinoids to monarch. Poutry et al. (2022, pp. 533–540) suggest that the risk of systemic neonicotinoids and exposure to adult monarchs is relatively low, concurrent with findings by Krishcik et al. (2015, pp. 3–11). The study tested the toxicity of two neonicotinoids (imidacloprid and clothianidin) fed to adult monarchs through a nectar-dosed solution at field relevant concentrations and at exceedances. Adverse effects (reduced nectar consumption, survival, and flight performance time) were detected at concentrations that exceeded field relevant levels by 22 and 99 times. There were no adverse effects detected at doses representing field relevant concentrations. The authors suggest that monarchs are possibly more tolerant of ingested clothianidin and imidacloprid than what has been reported as risk in previous literature.

Monarchs may be exposed to neonicotinoid drift that results in residues on the surface of plants or is systemically incorporated into plant tissues (e.g., milkweed leaves, forb pollen and nectar plants). Several studies report that neonicotinoids are present in samples from milkweed plants indicating that drift can occur off intended use sites (Pecenka and Lundgren 2015, pp. 3–4; Olaya-Arenas and Kaplan 2019, pp. 10–14; Halsch et al. 2020, pp. 7–9). These studies do not discern the contribution of drift onto plant tissues versus systemic incorporation to the concentrations of neonicotinoids detected. Some studies found that monarch dietary exposure is generally not high enough to adversely affect larvae feeding on milkweed plants or adults nectaring on forbs (Bargar et al. 2020, pp. 15–18; Hall et al. 2022, pp. 160–167; Poutry et al. 2022, pp. 538–541) while at least one study found reduced survival of larvae from exposure at labeled application rates (Krischik et al. 2015, pp. 10–11). Halsch et al. (2020, pp. 5–6) reported that concentrations of pesticides in milkweed plants sampled from retail, urban, and agricultural sites did have exceedances over the reported honeybee toxicity thresholds for several pesticides (including one neonicotinoid). Limited studies have reported that neonicotinoid exposure to larvae can arrest pupal ecdysis (Bargar et al. 2020, pp. 14–15; Krishnan et al. 2021a, pp. 2–6; Krishnan et al. 2021b, pp. 1767–1768). Generally, available data suggest that exposure to neonicotinoids can affect adult, larval, and pupae survival, though in many cases the connection to field-relevant concentrations expected to result from direct application, drift, or systemic incorporation is unclear. In addition, the studies described herein differ considerably with respect to experimental design, with variables such as life stage, milkweed species, pesticide tested, exposure regime, and endpoints monitored potentially contributing to measurements of toxicity. Given the complexity of the monarch life cycle, exposure potential and biological effects of neonicotinoids to the monarch are not fully understood. At present we extrapolate the available information to exposure scenarios or life events where data are currently lacking; further studies of these areas may contribute to an increased understanding of the risk of neonicotinoids to the monarch.

Risk Evaluation

Ecological risk assessment to evaluate potential insecticide effects to monarchs can be assessed by (1) comparing laboratory-derived toxicity values to environmental concentrations of insecticides (based on either predictive modeling or post-application sampling), and/or (2) studying effects to individuals exposed to insecticide applications in the field. Effects (lethal and sublethal) are then characterized and a determination is made as to the extent of risk. Additional (unknown) risk in the field can be caused from indirect effects of insecticides, such as susceptibility to disease or predation, and the potential for additive or even synergistic effects from exposure to multiple pesticides in the field. The lack of standardized toxicity testing and limited monarch-specific data limit a definitive risk assessment for monarchs. Accordingly, available assessments generally center on other lepidopteran species, from which risk to monarchs can be extrapolated.

Organophosphates and pyrethroids

Though organophosphate and pyrethroid insecticides are used in all facets of pest control, there has been particular interest in performing risk assessments based on exposure scenarios from mosquito control applications, as lepidopterans can be exposed within the site of application (i.e., they occur in areas where mosquitoes are treated). In particular, the need for mosquito control in southern Florida has led to concerns regarding the effects on native lepidopterans. The few studies described below indicate that mosquito adulticide applications may pose risk to lepidopteran species, but results differed across studies, pesticide type, and species. Mosquito adulticide treatment differs from other treatments in that application rates tend to be lower than other uses, and pesticide is applied in a ULV spray designed to maximum time before deposition so as to encounter airborne mosquitoes. For these reasons, factors such as application rate and environmental transport should be considered when relating the risk assessments and field studies of mosquito adulticides described below to other uses of these insecticides (e.g., cropland, natural areas, and residential settings).

In an assessment of the risk of naled, deposition was measured 50 minutes following a single pre-dawn ULV spray for mosquito control (applied as Trumpet EC at a rate of 70 g a.i./ha; Bargar 2012b, pp. 886–889). These results were combined with morphometric data for 22 species within 5 families to estimate deposition onto butterflies roosting in the application area during a pre-dawn spray. Using lepidopteran toxicity values from the literature (described above), a 67–80% chance of exceeding the mortality estimate for the butterflies was predicted following such a spray. Assuming equivalent sensitivity, the greatest risk was estimated for butterflies within the Lycaenidae family, and the lowest risk for those within the Hesperiidae family; relative risk to butterflies within Papilionidae, Nymphalidae, and Pieridae families was considered to be intermediate.

Another risk assessment examined potential effects to native Florida caterpillars from the mosquito control pesticides permethrin, naled, and dichlorvos (Hoang and Rand 2015, pp. 715–717). Exposure data for this analysis were taken from a report generated from a field monitoring program in Big Pine Key, Florida in 2007–08, though measured values on leaves were not presented directly in Hoang and Rand (2015). The joint probability analysis in the risk assessment revealed that permethrin concentrations on host plants had a 42% chance of exceeding the lowest observed adverse effects dose (LOAED) for native Florida caterpillars and a 0.02% chance of exceeding acute LD50 values. Probabilities of exceedance for diclorovos were 11% and 2.2% for its LOAED and LD50, respectively, and the probability of exceedance was 11% for the LD50 for naled. The authors indicated that these values may underestimate actual risk in the field as they are based solely on 24-hour dietary exposure and do not consider the influence of direct topical exposure from drift or chronic exposure from insecticide persistence on leaves.

Two other field studies also examined native butterfly populations in areas with mosquito control. Population surveys in the rock pinelands of south Florida (Long Pine Key) and the Lower Florida Keys (Big Pine Key) were conducted in areas that receive year-round application of pesticides (pyrethroids and organophosphates) for mosquito control and those without such treatment (Salvato 2001, pp. 11–12). Adult densities of Florida leafwing (*Anaea troglodyta floridalis,* family Nymphalidae) were significantly lower in treated areas than in control areas. Population counts of Bartram's scrub-hairstreak (*Strymon acis bartrami*, family Lycaenidae) and Meske's skipper (*Hesperia meskei*, family Hesperiidae) did not appear to be reduced following pesticide application. In a second study, insecticide residue deposition and butterfly survival were monitored following a spray of naled during routine mosquito control in North Key Largo, Monroe County, Florida (Zhong et al. 2010, pp. 1963–1968). Sampling stations were set up within the spray zone, drift zone, and control areas (>25 miles away). Survival rates of 5th instar Miami blue butterfly caterpillars (*Cyclargus thomasi bethunebakeri*, family Lycaenidae) were 52–98% at sampling stations within the spray zone, and did not differ between drift and control zones. Naled was recorded in a remote drift zone 12 miles from the application area causing mortality to test mosquitoes in sampling stations, but not to butterfly larvae similarly exposed. Naled concentrations greater than 1000 ug/m^2 were associated with dramatically reduced larvae survival rates, though larvae surviving to the pupal stage successfully emerged. Wind speed was associated with higher deposition and larval mortality.

Neonicotinoids

While no field studies exist to assess the population effects of neonicotinoids, modeling studies have attempted to relate monarch declines to this class of pesticides. Forister et al. (2016, p. 4) investigated neonicotinoid use and butterfly declines at four sites in Northern California that have been monitored for four decades. The model indicated an association between declining butterfly numbers and increasing neonicotinoid use, suggesting that neonicotinoids could influence populations occurring close to application sites. Similarly, Thogmartin et al. (2017, pp. 8–11) analyzed multiple threats to monarchs including climate, habitat loss, disease, and insecticides in a time series analysis using partial least squares regression models. Glyphosate and neonicotinoid use in monarch breeding habitat were both correlated with the observed monarch population decline. Gilburn et al. (2015, pp. 5–9) modeled neonicotinoid usage on agricultural lands and population estimates for 17 species of butterflies in the UK from 1985 to 2012. A negative correlation was indicated for hectares of farmland that used neonicotinoid pesticides and butterfly population declines. The authors determined that more studies are needed to determine if there is a causative link between neonicotinoid usage and the decline of butterflies, or whether the negative correlation represents a proxy for other environmental factors associated with intensive agriculture practices. Van Deynze et al. (2023, pp. 12–13) predicted the net effects of several stressors identified for the monarch butterfly and through modeled estimates, predicted that pesticide and neonicotinoid-treated seed use drive butterfly declines in the Midwest over other stressors such as herbicides, land use and management, and climate change variables. Authors suggest that more comprehensive data is needed on pesticide use across geographic areas to fully realize the risk to monarch butterfly populations.

In an assessment broadly examining insecticides, DiBartolomeis et al. (2019, pp. 10–19) incorporated existing toxicity data (honeybee LD50 data for contact and oral toxicity), persistence (soil half-life), and mass applied (estimated total pounds per acre used for foliar and seed treatments) to model pesticide loading (defined as acute insecticide toxicity loading, AITL) in agricultural land and surrounding areas. The model suggests that from 1992 to 2014, the AITL in the United States increased 4-fold based on contact toxicity and 48-fold based on oral toxicity. The authors attribute this change to an increase in pesticide loading from neonicotinoids beginning in 2004. Three neonicotinoids (imidacloprid, thiamethoxam, clothianidin) combined to contribute 91.8% of the total AITL for oral toxicity. As presented, the AITL is a measure of raw insecticide toxicity in the environment and does not take into account how non-target species such as monarchs may be exposed to these chemicals. As previously discussed, factors such as accumulation in exposure media (e.g., nectar, leaf, direct spray) and the location and timing of application can be highly influential in estimating effects to individuals and populations, and may differ across classes of insecticides. Environmental persistence, as measured by a chemical’s half life in soil, appears to be a significant driver in results, yet its relationship to pesticide availability to nontarget target species in unclear. As such, it is difficult to translate the conclusions of this assessment to potential effects to monarchs.

In Krishnan et al. (2020, pp. 2–9), larval dose response curves generated from toxicity studies were used to model monarch mortality rates caused by insecticide drift exposure downwind from sprayed crop fields. Two scenarios were modeled: predicted spraying for (1) soybean aphid and (2) true armyworm - a pest of maize. The models took into account three application methods: aerial application, high ground boom, and low ground boom and predicted mortality rates (using both contact and dietary larval exposure data) between 0–60 meters from the edge of a sprayed field. Application rates based on the insecticide label were used in the models. Models for aerial applications using beta-cyfluthrin and chlorantraniliprole for the soybean aphid management scenario predicted larval mortality between 100 and 32% at distances 0–60 meters downwind from the agriculture field based on cuticular toxicity data. Based on dietary toxicity data, predicted larval mortality was between 100 and 10% for modeled distances downwind from the agriculture field. Larval mortality for chlorpyrifos, imidacloprid, and thiamethoxam, (using cuticular toxicity data) was 99, 91, and 67%. For the same insecticides, larval mortality was 96, 80, and 83% based on dietary toxicity data. Modeling for high ground boom applications produced similar predictions; however, lower mortality was predicted at distances 15, 30, and 60 meters downwind compared to aerial applications in which greater larval mortality was observed at 0 meters downwind. Across the scenarios, the mortality rates were generally highest for the first instars and lowest for fifth instars. The lowest percentage of monarch mortality was modeled at 60 meters downwind from the crop edge.

Summary and Conclusions

Despite inconsistencies in testing regimes (e.g., chemical concentrations, application methods and exposure routes, and life stage and species tested), studies presented here and in other reviews (Mule et al. 2017, pp. 2–5; Braak et al. 2018, pp. 507–518) demonstrate that insecticides can have negative effects on lepidopteran species. The majority of the studies evaluated for the Monarch Species Status Assessment are laboratory toxicity tests designed to identify the insecticide concentration that causes mortality or adverse effects. More recent laboratory toxicity studies have attempted to evaluate the effects at relevant environmental concentrations. Field studies are also available that measure insecticide concentrations in milkweeds or monitor effects to lepidopterans within and outside of an application site. Finally, modeling studies weigh the risk of insecticides amongst other threats to monarch populations. Many of these studies concluded that insecticide use may potentially have negative effects to lepidopterans, including monarchs. While these studies provide pieces of information to evaluate the risk of insecticides to monarchs, enough data gaps remain for the many variables involved to prevent a comprehensive analysis of effects.

As insecticides are generally likely to cause adverse effects to butterflies, exposure of monarchs (both adults and larvae) to these chemicals through diet and contact is the primary determinant of risk across a variety of land use sectors throughout the species' range. Monarch exposure to insecticides is not readily predictable, but dependent on individual monarchs encountering pesticide residues on or near the individual plants they use. In addition, exposure is influenced by factors such as the extent and frequency of insecticide use, timing of application, application rate and method, proximity of monarchs to the application site, contact with residues in the air or on plant surfaces, availability of residues in dietary items associated with lifestage present (leaves or nectar), and pesticide persistence.

One primary limiting factor in understanding the risk of pesticides to the monarch is the lack of information regarding which chemicals are applied to landscapes across the country in all monarch use areas (urban, rural, agricultural, forested, residential, parks and recreational areas). A lack of these data prevents the opportunity to understand the extent and volume of pesticide use across the landscape to allow for comprehensive risk assessment. The extent and manner of insecticide use itself is not regularly monitored or easily predicted in any given area. Insecticide use can vary both temporally and spatially, and is subject to regional or broad scale changes from disease and pest outbreaks, and emerging pest pressure. The toxicity of insecticides present on the landscape to lepidopterans may change based on the development and use of new insecticides, the regulation of older insecticides, the unknown effects of pesticide mixtures in the environment, and the advent of new technologies to prevent drift and reduce nontarget exposure.

Despite the challenges to determine a quantifiable extent to which insecticides impact the monarch population, and to determine a specific cause and effect relationship of insecticide effects to monarchs in environmental settings across various land use sectors, the substantial body of information available allows for a qualitative evaluation of the risk of insecticides to monarchs. Based on insecticide chemical characteristics and use; and the exposure potential, laboratory toxicity tests, field studies, and models presented herein, *insecticides are a threat to monarch populations*. This is primarily due to insecticides being used in areas on the landscape where monarchs occur; the fact that insecticides are designed to kill insects (and in many cases specifically target lepidopteran species); insecticides are likely to cause both lethal and nonlethal effects to non-target lepidopterans that are exposed in areas of application (such as crops fields, city parks, natural areas, residential areas, and yards and gardens); and may cause both lethal and nonlethal effects to non-target insects that are exposed from drift by droplet, vapor, and dust in areas outside of application sites and from systemic incorporation into non-target plant tissues.

Though many uncertainties (described throughout this assessment) regarding insecticide exposure and effects make it difficult to determine the *degree or extent of risk* to both individuals and at the population level, there are some factors that contribute to this uncertainty that are manageable and can be addressed through conservation actions, toxicity and exposure research and methodologies, and outreach/education programs. Manageable factors include:

- General awareness of insecticide use (e.g., ornamental plants and other consumer products that may contain neonicotinoids), and public policy affecting insecticide registration and use.
- Extent of development and adoption of best management practices for insecticide use, including Integrated Pest Management (e.g., establishing “acceptable levels” of pest pressure) and drift control measures.
- Extent of agricultural land uses with monoculture systems that increase the potential for, and frequency of, insect pest outbreaks and the economic need for chemical control.
- Societal expectations for widespread use of mosquito control insecticides.
- Technological capability to develop chemical insect pest controls which are more selective for the pest species, short-lived in the environment, less mobile, etc.
- Lack of standardized toxicity testing protocols to determine effects to the monarch and other non-target lepidopterans.
- Lack of standardized methods for field studies to determine the extent of exposure to the monarch population and other non-target lepidopterans.
- Field measurements of insecticide residues in select components of monarch habitat across a variety of land use sectors (i.e., quantified exposure).
- Lack of studies that clearly relate laboratory trials and field studies to realistic field exposure and effects to monarch butterflies.

Additional research and monitoring of aspects associated with these factors can provide the information necessary to reduce the uncertainties, and to determine which factors are the most important to manage risk. Most of these factors directly relate to insecticide exposure – *managing exposure manages risk*.

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[2] The Risk of Direct Toxicity of Herbicides to the Monarch Butterfly

Herbicides are widely used throughout the range of the monarch, and can cause mortality and reduced vitality to milkweed host plants and nectar source plants. However, plants may survive exposure if the herbicide has no toxicity to the plant (e.g., it is selective only for certain plants) or if concentrations are not high enough to elicit an effect (e.g., exposure from drift). In these cases, monarch caterpillars may retain use of the host or forage plants, but may be directly exposed to herbicides through contact or diet.

As with insecticides, the potential for direct effects of herbicides on monarchs can vary by active ingredient, product additives used (e.g. surfactants), exposure pathway, life history phase exposed, timing of application, and the amount of chemical exposed to the monarch. Herbicides work by interacting with the cellular structure or biochemical pathway of the target plant, and by causing tissue damage and plant mortality. Some herbicides are enzyme inhibitors acting on the enzymes that are important for plant growth and development. Although the mode of action for herbicides is to target specific pathways for plants, there are similarities between some plant enzymes that herbicides target and insect enzymes. For example, some herbicides target acetyl CoA carboxylase in plants, an enzyme important for plant growth but also for protein synthesis in insects (Lou et al. 2001, pp. 120–127; Goldring and Read 1993, pp. 855–858). Other herbicides can target glutamine synthetase, an enzyme critical for photorespiration in plants and ammonia detoxification and reassimilation in insects (Kutlesa and Caveney 2001, pp. 27–28). For the vast majority of herbicides, the mode of action and influence on lepidopteran biological systems remains unknown.

We are unaware of published data testing the direct effects of herbicides to monarchs. This section provides an evaluation of the risks of herbicides to monarchs based on a brief summary of herbicide-lepidopteran toxicity studies; it does not include an exhaustive review of the available science. Effects to monarchs from loss of milkweed due to herbicides is addressed in the SSA Chapter 5.

Herbicide Concentrations in Milkweed Leaves

As with insecticides, oral exposure of monarchs to herbicides is dependent on residues being present on or within dietary items. Olaya-Arenas and Kaplan (2019, pp. 8–10) detected herbicides in leaves of milkweed (*A. syriac*a) within 100 m of crop fields in northwest Indiana. Atrazine was the most frequently detected herbicide, in 80–87% of the samples and at the highest concentrations (2015: 6.84 ng/g mean, 0.52 ng/g median, 238.7 ng/g maximum; 2016: 37.0 ng/g mean, 4.73 ng/g median, 1352.9 ng/g maximum), followed by s-metolachlor (in greater concentrations early in season) and acetochlor.

Herbicide Toxicity to Lepidopterans

Studies suggest that the active ingredients in some herbicide formulations have the potential to cause lethal and sublethal effects in lepidopterans under certain exposure scenarios. Schultz et al. (2016, pp. 868–873) tested the direct effects of graminicides fluazifop-p-butyl, sethoxydim, clethodim mixed with the adjuvant NuFilm on three *Euphydryas* species in the 2nd instar larval phase under two different scenarios. In the first experiment, *E. colon* larvae were directly exposed to the treatments at labeled rates for habitat types which could be treated for invasive plants, placed in individual rearing containers, and fed until entering diapause. Control groups received a NuFilm only treatment and a water only treatment. This experiment found that contact treatment with sethoxydim reduced survivorship of pre-diapause *E. colon* larvae by 20% compared to the water only control, while there was no observed effect to larval survival from fluazifop-p-butyl, clethodim, and the NuFilm treatments. In the second experiment, all three *Euphydryas* species were exposed to fluazifop-p-butyl mixed with NuFilm; hostplants were also treated, with larvae and host plants placed within a mesocosm study design. Survival, larval development time, and feeding behavior were observed. This experiment found no effects of fluazifop-p-butyl on larval survival or development time; however, feeding group size (number of gregarious larvae) was reduced by exposure to the herbicide.

Stark et al. (2012, pp. 25–27) examined the individual effects of three formulated herbicide products containing triclopyr (Garlon 4 Ultra - a selective herbicide used to control woody plants and broad leaved plants), sethoxydim (Poast - a selective herbicide used to control grasses), and imazapyr (Stalker - a non-selective herbicide used to control grasses) directly applied to 1st instar Behr's metalmark (*Apodemia virgulti*) and their food source (buckwheat) at labeled field rates. Larvae were then fed treated plants and allowed to develop into adults. Triclopyr, sethoxydim, and imazapyr products each reduced the number of pupae (and consequently the number of adults) produced compared to the control by 24%, 27%, and 36%, respectively.

To investigate the most likely and worst case scenarios for herbicide exposure to lepidopterans, Russell and Schultz (2009, pp. 5–9) assessed the biological effects of two herbicides to the 3rd instar phase of the Puget blue (*Icaricia icarioides blackmorei*) and the cabbage white (*Pieris rapae*). The timing of the 3rd instar larval phase corresponds to when herbicides are most likely to be used in the field. Survival, development time, and growth were measured in the larvae after the exposure of two grass-specific herbicides and one surfactant (Preference) in mixtures: fluazifop-*p*-butyl and surfactant, sethoxydim and surfactant, fluazifop-*p*-butyl and water, sethoxydim and water, a water control, and an untreated control. A backpack sprayer was used to administer the treatments to simulate ground application; maximum labeled spot spraying recommended rates were applied. To test most likely scenarios, larvae were placed on host plants (*Lupinus albicaulis*) and the herbicide mixtures for each treatment were directly sprayed on the plants. Larvae were exposed to the residues via contact and dietary exposure. To test for the worst case scenario, larvae and the host plants were separately sprayed with the herbicide mixtures and the larvae were then placed on the plant to simulate maximum direct contact and dietary exposure. The study found that survival was reduced for *P. rapae* (but not for *I. i. blackmorei*) when exposed to fluazifop-*p*-butyl plus surfactant (21% reduction) and sethoxydim plus surfactant (32% reduction) compared to the control. Development time to eclosion for *I. i. blackmorei* occurred earlier in all treatment groups compared to the controls, but this was not observed for *P. rapae*. Wing area was smaller for female *P. rapae* when exposed to fluazifop-*p*-butyl plus surfactant (10% reduction) and sethoxydim plus surfactant (14% reduction) compared to the controls. Males exhibited a 9% reduction in total wing area in the sethoxydim plus surfactant treatment.

Kutlesa and Caveney (2001, pp. 27–31) found the herbicide glufosinate-ammonium (GLA), a non-selective post-emergence contact herbicide that competitively inhibits the enzyme glutamine synthetase, to cause lethality to Brazilian skippers (*Calpodes ethlius*) from dietary exposure from concentrations calculated to be similar to field application rates. 5th instar caterpillars were placed in petri dishes on moistened filter paper and fed leaf discs from the plant species Canna lily that were treated with acute doses of GLA to determine an LD50. Each caterpillar received one treated leaf disc and were observed until it was completely consumed (approximately 24 hours) and then provisioned with untreated leaves until pupation or death. The LD50 for GLA was calculated to be slightly lower than expected residues on leaves after field application. For behavioral studies, caterpillars were fed leaves that had high and low concentrations of GLA and mass and general behaviors were recorded daily. A decline in normal activity was observed 2–3 days after treatment with a daily dose of 5 mmol and the caterpillars stopped feeding altogether after 3–4 days. Multiple normal behaviors were observed to be altered and the caterpillars died after 6–7 days after exposure.

Bohnenblust et al. (2013, pp. 587–593) did not detect toxic effects of dicamba via contact or dietary exposure to 2nd and 3rd instar larvae of the corn earworm (*Helicoverpa zea*) and the painted lady (*Vanessa cardui)*. In contact exposure studies, larvae were placed in treatments and topically dosed with dimethylamine (DMA) and diglycolamine (DGA) formulation of dicamba within a range of the field application rate and placed in individual 50-mm petri dishes. Larvae were not provisioned during the toxicity studies and mortality was assessed at 4, 8, 12, 24, and 48-hour exposure durations. Percent mortality was equal across all treatments indicating that dose had little effect on survival for both species. To assess dietary exposure on the growth and development of *H. zea* and *V. cardui* larvae, soybean (*Glycine max*) and nodding plume thistle (*Carduus nutans*) were exposed to DMA formulation using a research grade automated sprayer at four rates that represent a range of 0.0001-0.1 of the current label rate of dicamba. After spraying, plants were isolated by treatment in a greenhouse. After three days, starved larvae (24 hours with no food provisions) were placed on the treated plants (*H. zea* on soybean and *V. Cardui* on thistle) and monitored until pupation or death. No differences in *H. zea* larval survival were detected across treatments and there was no relationship detected between number of days to pupation and herbicide dose. In the tests using thistle and *V. cardui* larvae, reductions in larval and pupa mass were observed.

LaBar and Schultz (2012, pp. 183–185) did not observe lethal or sublethal effects in a field study in which the habitat of the Puget blue was sprayed with sethoxydim and a non-ionic surfactant. During observational data collections, there was little to no observed impact on larval performance in the field or on oviposition for adults in the sprayed fields compared to non-treated fields.

Summary: Risk of Herbicides to Monarchs

In the herbicide toxicity studies summarized above, results 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. 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 maximum labeled rate) for this risk assessment.

For those herbicide-lepidopteran toxicity studies in which effects were observed, reductions in survival were generally between 20–40% of the exposed population. Effects were detected in a variety of herbicide types, including those that are non-selective, as well as those that are selective for monocots or dicots. However, results of these studies are mixed, and in a number of cases, no direct effects were found to lepidopterans from specific herbicides or particular exposure regimes.

In summary, herbicides have been detected in milkweed plants growing in proximity to agricultural fields and larval monarchs can be exposed by ingesting residues that are expressed in plant tissues; however, the direct effects of most herbicides to monarchs are unknown and likely to be highly variable. The toxicological information presented above represents a small percentage of all herbicide products used, and does not account for the most widely used herbicides such as glyphosate, atrazine, metolachlor, and 2-4 D. For those herbicides in which direct effects were detected, we are unable to elucidate the extent or specific circumstances of their use within the monarch range. 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 impacts due to pesticides (insecticides, herbicides, fungicides, rodenticides, etc.).

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Appendix 6. Changes to the SSA Report and Analyses

[1] Note on Changes from Version 2.1 (September 2020) to 2.2 (September 2023)

In 2023, we evaluated information that has become available since 2020, including updated population survey numbers for both the eastern and western North American monarch populations, relevant recent scientific literature, new analyses of monarch numbers, and restoration accomplishment reports provided to the Monarch Conservation Database (MCD). The team concluded that the reviewed information would not significantly change the analyses and future condition forecasts presented in the 2020 Species Status Assessment (SSA) report (version 2.1). However, we did incorporate references and new information into this updated SSA (version 2.2) that added significantly to our current understanding of relevant monarch biology.

Population Counts

Monarch populations are monitored annually via overwintering surveys in Mexico for the eastern migratory population (via hectares of monarch-occupied habitat) and at sites primarily along the California coast for the western migratory population. In the 2020 SSA report, we used the overall population growth rate (lambda; λ) and the most recent five-year average of the population numbers as the starting population size when projecting (modeling) future population trends. Below is a comparison of the population numbers used in the 2020 SSA report (version 2.1) with current population growth rates and five-year average population numbers.

Eastern Population (given three new years of data: '20–'21: 2.10 ha, '21–'22: 2.84 ha, and '22–'23: 2.21 ha):

Parameter	2020 (Last SSA Update)	2023
Population growth rate (λ)	0.960	0.956
5-year average population	3.66 ha	3.21 ha

Western Population (given three new years of data: 1,899 from 2020; 247,246 from 2021; and 335,479 from 2022):

Parameter	2020 (Last SSA Update)	2023
Population growth rate (λ)	0.878	0.890
5-year average population	168,365 individuals	128,356 individuals

Scientific Literature

We reviewed the scientific literature for papers relevant to monarch conservation published since the last monarch butterfly SSA report (version 2.1) in mid-2020. We identified over 300 papers and assessed their relevance to the SSA analyses, as well as their potential to inform ongoing and future conservation strategies. No papers significantly changed the main analyses and findings of the 2020 SSA report, but we incorporated the relevant information and added new references, where applicable, throughout this SSA report. This new information included data on the relative impact of different influences on the species, emerging threats, and novel analyses of monarch population dynamics. We did not cover topics that were outside of the scope of this version of the SSA, nor did we include similar findings to what were already in the SSA. These papers fall into the following main categories:

Main drivers of decline – Several papers have been published since 2020 that have included new models and analyses attempting to better understand monarch population drivers over various past time periods, and to predict drivers in the future. The main drivers explored include loss of milkweed due to herbicide use, impact of fragmented landscapes, climate change, mortality during fall migration and at overwintering grounds, effects of *Ophryocystis elektroscirrha* (*OE*) parasite, and impacts of non-migratory monarchs. Some of the papers have similar conclusions and others have potentially conflicting conclusions reflecting continuing uncertainty about the relative effects of various drivers. We reviewed the future scenarios used in the original SSA given the new literature and concluded that the projected range of scenarios continues to capture the potential new drivers of decline and does not substantially change our conclusions.

Population-level analyses – Multiple studies since 2020 have examined monarchs at the population level and attempted to ascertain population trends of North American monarchs. This research yielded mixed results on current trends throughout all and portions of North America and raised questions about the optimal approach for annual census of the populations (see Shirey and Ries 2023 and Vidal and Rendón-Salinas 2014). However, the convention of analyzing overwintering population data remains appropriate to assess trends in migratory monarch butterflies since monarch butterflies on the overwintering sites are guaranteed to be part of the migratory population and represent a concentration of the monarch populations in a relatively small area, providing insights into population trends at the most vulnerable part of the life cycle.

Pesticides – Several papers examined the concentrations of pesticides (i.e., insecticides, herbicides, and fungicides) occurring in different conditions and habitats in the environment, assessed the exposure of monarchs to pesticides at various life-stages, tested the effects of pesticides at different dose concentrations on monarch growth and survival, and further integrated consideration of pesticide exposure with models of monarch landscape use and overall survival. These papers are important, but do not change our conclusions on the last page of the SSA Insecticide and Herbicide Supplemental (Appendix 5): "Many of these studies concluded that pesticide use may potentially have negative effects to lepidopterans, including monarchs. While these studies provide pieces of information to evaluate the risk of pesticides to monarchs, enough data gaps remain for the many variables involved to prevent a comprehensive analysis of effects. Though many uncertainties regarding pesticide exposure and effects make it difficult to determine the degree or extent of risk to both individuals and at the population level, there are some factors that contribute to this uncertainty that are manageable and can be addressed through conservation actions, toxicity and exposure research and methodologies, and outreach/education programs."

Mexico Overwintering Habitat – Although much of the overwintering habitat is designated as a biosphere reserve, a law enforcement group designed to curb illegal activities in the reserve was recently disbanded. There are indications that illegal logging has again become a problem in some areas within the reserve. The overall situation across the overwintering grounds in Mexico remains socially and politically dynamic, and deforestation pressures may be compounded by expanding avocado farming in the region. However, model-based predictions of avocado farming under future climate conditions suggest that farm intrusions would not occur at levels that pose a systemic threat to overwintering habitat in Oyamel fir forests and deforestation levels do not appear to warrant adjustments to the status, threats, or future scenarios evaluated previously within the SSA.

Foundational Research – Understanding of monarch life history and conservation biology continues to be explored and expanded through research including: diversity and impact of monarch predators and parasites (wasps, *OE*, mice, etc.); eastern and western populations (interactions, differences and similarities); conservation (impact of disturbance on habitat; spatial arrangement, location, and size of patches for optimal oviposition and survival; and related social science); migration (diapause and orientation, captive rearing, and light pollution impacts); monarch genetics; invasive species; and vehicle mortality.

Monarch Conservation Database (MCD) Updates

As of June 27, 2023, conservation efforts recorded in the MCD totaled 10,457,316 acres, including 7,415,731 acres of completed and implemented efforts since 2014 with the remaining 3,041,585 acres consisting of planned, but not yet implemented actions. The most common conservation effort reported was direct planting of milkweed and other nectar resources. Conservation efforts implemented or entered into the MCD by State agencies accounted for nearly 50% of those acres (3,896,773 acres). The MCD also includes conservation efforts associated with the Nationwide Candidate Conservation Agreement (CCAA) for Monarch Butterfly on Energy and Transportation Lands (See *Conservation Efforts* in Chapter 5 for more information). As of April 9, 2023, there are approximately 850,000 "adopted" acres (acres with habitat conservation measures for monarch) through the CCAA. While we've incorporated these updated numbers into this version of the SSA report, this level of conservation effort does fall within the already modeled scenarios presented in the 2020 SSA.

The MCD also includes an estimate of acres enrolled in the Conservation Reserve Program. These acres had decreased between 2014 and 2018 and have continued to decrease between 2018 and March 2023. The current decline in Conservation Reserve Program (CRP) acreage relative to the 2014 baseline is approximately 25% (see Zulauf 2023, entire). As a result of the decrease in CRP acres, the net acres in certain geographies have also decreased, despite the contributions from other conservation efforts. However, overall, there has been a net increase in acres of conservation efforts, despite losses in CRP.

Extinction Risk Modeling

Based on our previous sensitivity analyses, we determined that our model results, in terms of *pE* estimates, were most sensitive to population growth (lambda) compared to the impact of starting population size or extinction threshold. Given the relatively small change in these values with the addition of the 2020–2023 data, we did not find it informative to re-run the model with these new values. Updated information on ongoing and expected future influences on monarch viability, including conservation efforts, habitat loss, and impacts of insecticides, also remained consistent with the information used to develop the future condition scenarios used in the 2020 SSA analysis. We also reviewed the new literature on monarch population modeling published since the 2020 SSA. These are informative in evaluating current stochastic risk to monarch populations and describing approaches to fitting data to alternative population models, but do not replace our modeling of future extinction risk. In sum, we did not re-run our model because we determined that any prospective changes to our previous statistical modeling framework, including the underlying data/assumptions, would likely be relatively minor in terms of their impact on the results and would not substantially alter the range of projected risks or the associated uncertainties previously analyzed, peer-reviewed, and published in the 2020 SSA.

[2] Note on Changes from Version 2.2 (September 2023) to 2.3 (February 2024)

Following a September 2023 peer review, we made many additions of citations to newer literature and many small language changes to better cite papers or to clarify our analysis.

We also edited for clarity and restructured the order in which information is presented to better reflect our analysis. The earlier versions of the SSA presented information in an order more related to the chronology of our early work rather than communicating our understanding of the analysis at the end of the process. Additionally, the earlier versions contained relics of analyses initially considered (e.g., "Adaptive Capacity Units (ACUs)"). We removed references to ACUs, because, while they represented our exploration of adaptive capacity as considered in the original framing of the analysis, they were not relevant to the analysis in the final draft. The population and genetics information presented with the ACUs was relevant, however, and was retained and presented in a different flow of the narrative.

Similarly, we changed the location of the presentation of information about the dispersed and non-migratory populations, initially termed the "worldwide" populations. In earlier versions of the SSA the worldwide information was presented early in the document since we considered that information early in our analysis. Since our analysis revealed that the dispersed populations were not significant to the viability of the ancestral and predominant (in abundance) monarch populations in North America, in this version we move that information to the end of each relevant section, and we moved the more detailed description of our analyses about those populations to an appendix.

We removed text boxes, figures, and other language that presented information about general principles of conservation biology or early analyses that that did not turn out to significantly inform our analysis, in order to focus the text on the significant analyses and results.

Monarch butterflies continue to be one of the most-studies insects in the world and our scientific understanding continues to progress, yet many significant uncertainties remain. We are very grateful for the extensive ongoing highly collaborative scientific contributions that form the basis of this assessment.