

**Species Status Assessment for the  
Eastern Massasauga Rattlesnake (*Sistrurus catenatus*)**

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**Authors: Jennifer Szymanski, Cathy Pollack, Laura Ragan, Mike Redmer, Louise Clemency, Kristen Voorhies, Jonathan JaKa**

## Executive Summary

This report summarizes the results of a species status assessment conducted for the eastern massasauga rattlesnake (EMR), *Sistrurus catenatus*, to assess the species' overall viability. The EMR historically occupied sections of western New York, western Pennsylvania, southeastern Ontario, the lower peninsula of Michigan, the northern two thirds of Ohio and Indiana, the northern three quarters of Illinois, the southern half of Wisconsin, extreme southeast Minnesota, east central Missouri, and the eastern third of Iowa.

The species status assessment (SSA) begins with a description the snake's ecological requirements for survival and reproduction as they relate to its overall viability. We generally defined viability as the ability of the species to maintain self-sustaining populations over the long-term. Using the principles of resiliency, representation, and redundancy, we considered the species' needs at the individual, population, and species scales. We also identified the beneficial and risk factors influencing the species' viability. We considered the degree to which the species' ecological needs are met both currently and as can be forecasted into the future, and assessed the consequences of any unmet needs as they relate to species viability.

The EMR species ecology is summarized in Chapter 2, risk and beneficial factors analyses in Chapter 3, the analyses for current condition are summarized in Chapter 4, and future condition analyses are summarized in Chapter 5. In Chapter 6 we summarize our analyses of current and projected future EMR population conditions, and provide additional context for interpreting those results through comparison to the results of another population model that incorporates climate change predictions, and by highlighting the sources and effects of uncertainty in our models and methods.

For survival and reproduction at the individual level, the EMR requires appropriate habitat, which varies depending on the season and its life stage. During the winter (generally October through March), they occupy hibernacula, such as crayfish burrows. Intact hydrology at EMR sites is important in maintaining conditions that support their over-winter survival. During their active season (after they emerge from hibernacula) they require low canopy cover and sunny areas (intermixed with shaded areas) for thermoregulation (basking and retreat sites), abundant prey (foraging sites), and the ability to escape predators (retreat sites). Habitat structure, including early successional stage and low canopy cover, appears to be more important for EMR habitat than plant community composition or soil type.

At the population level, the EMR requires sufficient population numbers and population growth (controlled by survivorship, recruitment, population structure, and size). Populations also require a sufficient quantity of high quality microhabitats with intact hydrology and ecological processes that maintain suitable habitat, and connectivity among these microhabitats.

We define a self-sustaining population as one that is demographically, genetically, and physiologically (DGP) robust with a high level of persistence given its habitat conditions and the risk or beneficial factors operating upon it. We define a DGP robust population as one that has an adult female population size greater than 50 and has a stable or increasing growth rate; we defined high persistence as a probability of persistence greater than 0.90.

We relied on a population-specific model developed by Faust *et al.* 2011 (hereafter referred to as the Faust model) to assess the health of populations across the EMR range. Faust and colleagues developed

a generic, baseline model for a slightly growing EMR population. Using this baseline model and site-specific information, including population size estimate, risk factors operating at the site, and potential future management changes that might address those factors, the Faust model forecasted the future condition of 57 EMR populations over different time spans (10, 25, and 50 years). We extrapolated the Faust results and supplemental information gathered since 2011 to forecast the future conditions of the remaining (non-modeled) EMR populations.

At the species level, the EMR requires multiple (redundancy), self-sustaining (resiliency), populations distributed across the full gradient of genetic and ecological diversity (representation). Using the literature on distribution of genetic diversity across the range of this species, we identified three geographic “analysis units” corresponding to genetic variation patterns across the EMR populations. We assume these genetic variation patterns represent areas of unique adaptive diversity. We subsequently use these analysis units (Eastern, Central, and Western) to structure our analysis of viability.

The most prominent risk factors include habitat loss and fragmentation, especially through development and vegetative succession; road mortality; hydrologic alternation resulting in drought or flooding; persecution; collection; and mortality of individuals as a result of post-emergent prescribed fire and mowing.

The population model developed by the Faust *et al.* (2011), indicates that the risk factors most likely to push a population to quasi-extirpation within 25 years (high magnitude risk factors) are late-stage vegetative succession, high habitat fragmentation, moderate habitat fragmentation, total habitat loss, and moderate habitat loss or modification. Our analysis of 57 EMR populations modeled by Faust *et al.* (2011) and an additional 165 populations for which we have risk factor information, indicates that of 222 EMR populations, 84% are impacted by at least one high magnitude risk factor and 63% are threatened by multiple high magnitude risk factors.

Assessing occurrence of these risk factors, we found that 97% of EMR populations have at least one risk factor currently affecting the site. Vegetative succession is the most commonly occurring risk factor, with 75% of sites being impacted by succession. Vegetative succession makes EMR habitat unsuitable by reducing or eliminating thermoregulatory and retreat areas. Post-emergent fire is the second most common factor (69% of sites), and fragmentation is the third most common factor (67% of sites). Some form of habitat loss or modification is occurring at 52% of the sites; 17% of these sites are at risk of total habitat loss. Among the other factors considered, water fluctuation, collection/persecution, and road mortality occur at 38%, 35%, and 15% of the sites, respectively.

We assumed these risk factors are chronic and are expected to continue with a similar magnitude of impact into the future, unless ameliorated by increased implementation of conservation actions. Disease, new or increasingly prevalent, is another emerging and potentially catastrophic threat to EMR populations. Due to a lack of information on the potential emergence or future spread of disease among EMR populations, we did not model this threat in forecasting future conditions for the rattlesnake. Our analysis also did not consider two other prominent risk factors, road mortality and persecution.

Of 267 sites with extant populations, 64% occur on land (public and private) that is considered protected from development. Two of these populations have signed Candidate Conservation Agreements with the U.S. Fish and Wildlife Service. These plans include actions to mediate the stressors acting upon the populations and provide management prescriptions to perpetuate EMRs on these sites. At an additional 22 sites, habitat restoration is occurring. We do not have information at these sites to know if

restoration has mediated the current threats acting upon the populations; the Faust model, however, included these restoration activities in the projections of trends, and thus, our future conditions analyses considered these activities and assumed that ongoing restoration would continue into the future. Lastly, another 18 populations have conservation plans in place. Although these plans are intended to manage for EMR, we did not have sufficient site-specific information to assess whether these restoration activities are currently ameliorating the stressors acting upon the population. Thus, we were unable to include the potential beneficial impacts into our analyses. On the remaining protected sites, non-development stressors such as fragmentation, succession, exotic species invasion, dam construction, water level manipulation, and other incompatible habitat modifications are likely to continue.

As a result of the risk factors acting upon EMR populations, the resiliency of the EMR across its range and within each analysis unit has declined. Rangewide, there are 581 known historical EMR populations of which 267 are known to still be extant, 163 are likely extirpated or known extirpated, and 121 are unknown status. For the purposes of this assessment, we considered all populations with extant or unknown status as currently extant (referred to as presumed extant,  $n = 388$ ). Of those populations presumed extant, 40% are likely quasi-extirpated (had 25 or fewer adult females).

The number of extant populations has declined rangewide by 33% and another 21% have unknown status. Of those populations presumed extant, 156 (40%) are presumed to be quasi-extirpated while 99 (26%) are presumed to be DGP robust. Of these presumed DGP robust populations, 29 are presumed to have threat conditions suitable for maintaining populations over time, and thus, are self-sustaining. The greatest declines in resiliency occurred in the WAU, where only 21 populations are presumed extant and of these only 1 is presumed to be self-sustaining. Although to a lesser degree, loss of resiliency has occurred in the CAU and EAU, where 22 and 6 populations, respectively, are presumed to be self-sustaining.

According to our analysis, 20 of the 222 populations (9%) for which we have pertinent data have habitat and threat (manageable) conditions suitable for maintaining a self-sustaining population. Assuming the 222 populations are a representative sample for all EMR populations, of the 388 presumed current populations, about 35 have habitat and threat conditions suitable for maintaining a self-sustaining population.

The degree of representation, as measured by spatial extent, across the EMR range has declined as noted by the northeasterly contraction in the range and by the loss of area occupied within the analysis units. Overall, there has been more than 46% reduction of extent of occurrence (EoO) rangewide. This loss has not been uniform, with the WAU making up most of this decline (nearly 70% reduction in EoO in the WAU). However, losses of 43% and 32% in the CAU and EAU, respectively, are notable as well. Assuming that loss of range equates to loss of adaptive diversity, the degree of representation of the EMR has declined since historical conditions.

The redundancy of the EMR has also declined since historical conditions. Potential catastrophic events relevant to EMR populations include flooding, disease, and drought. We were unable to find sufficient information on the likelihood of disease outbreaks, the factors that affect disease spread, and the magnitude of impact on EMR populations to assess the risk from a catastrophic disease outbreak. Similarly, we were unable to assess flooding as a catastrophic risk. Thus, we assessed the vulnerability of unit-wide extirpated (AUE) due to varying intensities of drought.

The risk of extirpation of all populations within an analysis unit varies by analysis unit and by the level of drought considered. The risk of AUE due a catastrophic drought varies by analysis unit and by the level of drought considered. In the central and eastern analysis units, the  $\lambda$  rates for D3 and D4 droughts are 0.0, so there is little to no risk of AUE regardless of spatial dispersion. Portions of the central analysis unit are at risk of a D2 level catastrophic drought; populations in the southern portion of the CAU and scattered portions in the north are at risk. In the western analysis unit, the risk of AUE using D3  $\lambda$  rates is low, but the risk of losing clusters within the western analysis unit is notable; 5 of the 8 population clusters are vulnerable to a catastrophic drought. The probability of AUE is notably higher with D2  $\lambda$  rates; 7 of the 8 clusters are at risk of D2 level catastrophic drought. Thus, the probability of losing most populations within the WAU due to a catastrophic drought is high.

To assess the future resiliency, representation, and redundancy of the EMR, we used the Faust model results to predict the number of self-sustaining populations likely to persist over the next 10, 25, and 50 years, and extrapolated those proportions to the remaining presumed extant populations to forecast numbers of self-sustaining populations likely to persist at the future time scales. We then predicted the change in representation and redundancy.

Rangewide, 7 of the 57 modeled EMR populations (Faust model) are projected to be self-sustaining at years 10, 25, and 50. Extrapolating to all presumed extant populations, by year 50 in the western analysis unit, one population is likely to be self-sustaining, 15 of the 21 presumed current populations will be extirpated and another two quasi-extirpated. In the central analysis unit, 54 populations of the 294 presumed extant populations are likely to be self-sustaining while 174 are forecasted to be extirpated and another 22 quasi-extirpated by year 50. In the eastern analysis unit, 6 populations are forecasted to be self-sustaining by year 50, and 55 of the 73 presumed extant populations are forecasted to be extirpated and another 6 quasi-extirpated.

We calculated the future extent of occurrence (representation) for the 57 modeled populations (Faust model) and for the populations forecasted to persist at years 10, 25, and 50 by using the counties occupied by populations to evaluate the proportions of the range falling within each analysis unit and the change in spatial distribution within each analysis unit. Our results indicate that EMR populations are likely to persist in all three analysis units; however, the distribution of the range is predicted to contract northeasterly and the geographic area occupied will decline within each analysis unit over time. The results predict a 65% reduction of the area occupied rangewide by year 50, with a predicted 83% reduction in the western analysis unit. These losses in extent of occurrence are likely underrepresented given that we used the entire area of counties in which EMR populations will continue to occur for our analysis, so losses of other populations within those counties are not reflected. These losses represent a loss in adaptive diversity for the species.

We assessed the ability of EMR populations to withstand catastrophic events (redundancy) by predicting the number of self-sustaining populations in each analysis unit and the spatial dispersion of those populations relative to future drought risk. Modeling D3 drought, at year 10 the probability of extirpation of all populations in the western analysis unit likely ranges from 22% to 40%, and at years 25 and 50 it ranges from 47% to 63%. Using the frequency rates of a less severe but more likely D2 drought, the probability of extirpation of all populations within an analysis unit is more likely in all three analysis units, though still low in the central and eastern analysis units. In the western analysis unit, the risk is much higher: using average annual rates, the range of probability of extirpation of all populations at year 10 ranges from 85% to 94%, at years 25 and 50 ranges from 92% to 97%.

Given the loss of populations to date, portions of the EMR range are in imminent risk of extirpation in the near term. Specifically, our analysis suggests there is a high risk of extirpation of the WAU and southern portions of CAU and EAU within 10-25 years. Although self-sustaining populations are expected to persist, loss of populations within the CAU and EAU are expected to continue as well, and thus, are at risk of extirpation in the future. These losses have led to reductions in resiliency and redundancy across the range and may lead to irreplaceable loss of adaptive diversity across the range of the EMR, thereby leaving the EMR less able to adapt to changing environment into the future. Thus, the viability of the EMR has and is projected to continue to decline over the next 50 years.

Our analyses are predicated on a host of assumptions, which likely lead to both over- and underestimate estimates of risk. In total, however, we believe our predictions are optimistic, especially in light of the results of recent climate change modeling that suggests past EMR extirpation has been caused in part by extreme drought and flooding events. As the frequency of these events is predicted to increase in the future, the observed extirpation front is predicted to continue into the future.

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## Chapter 1: Introduction, Analytical Approach, and Methods

This report summarizes the results of a species status assessment (SSA) conducted for the eastern massasauga rattlesnake (EMR), *Sistrurus catenatus*. The species was given federal candidate status in 1999; *i.e.*, the EMR warranted protection under the Endangered Species Act of 1973, as amended (ESA), but was precluded by higher priority actions. As part of the multiple district litigation court settlement for listing actions, we agreed to make listing determinations pursuant to the ESA for 251 species that were federal candidates as of 2010. Pursuant to that settlement agreement, a proposed rule or not warranted finding for the EMR is due in Fiscal Year 2015. Thus, we conducted a SSA to determine whether the EMR still warrants protection under the ESA.

The intent of the SSA is to assess the ability of the EMR to maintain self-sustaining populations over time (*i.e.*, viability). To assess EMR viability, we applied the conservation biology principles of resiliency, representation, and redundancy (Shaffer and Stein 2000, pp. 308-311). As described more fully below, resiliency is the ability of the species to withstand annual variation in the environment; representation is the ability of the species to adapt to long-term changes (*i.e.*, evolutionary potential); and redundancy is the ability of the species to withstand catastrophic events. A species' viability can be measured by its degree of representation, resiliency, and redundancy.

Our analytical approach for assessing EMR viability involved 3 stages. In Stage 1, we described the species ecology in terms of the 3Rs. Specifically, we identified the ecological requirements for survival and reproduction at the individual, population, and species levels. In Stage 2, we determined the baseline condition of the species using the ecological requirements previously identified in Stage 1. That is, we assessed the species' current condition in terms of 3Rs and identified past and ongoing factors (beneficial and risk factors) that led to the species' current condition. In Stage 3, using the baseline conditions established in Stage 2 and the predictions for future risk and beneficial factors, we projected future conditions of the EMR.

The species ecology (Stage 1) is summarized in Chapter 2, risk and beneficial factors in Chapter 3, the current conditions (Stage 2) in Chapter 4, and the future conditions (Stage 3) in Chapter 5. Lastly, a synthesis of EMR viability given our analyses of current conditions and projections of future conditions relative to historical conditions is provided in Chapter 6.

### 1.1 Resiliency, Representation, and Redundancy

To assess the viability of EMR, we used the conservation biology principles of resiliency, representation, and redundancy. These principles, referred to as the 3Rs, are distinct yet interrelated concepts (Figure 1.1). Viability is not a static state, but rather there are degrees of viability. Generally speaking, the more resiliency, representation, and redundancy a species has, the more protected it is against the vagaries of the environment, the more it can tolerate stressors, and thus, the more viable it is. The 3Rs framework is useful for comparing the degree of viability of a species through time.

### 1.1.1 Resiliency

Resiliency is the ability of a species to respond to and recover from disturbances and perturbations. These include the normal year-to-year variation in rainfall and temperatures and stochastic events such as fire, flooding, and storms. Simply stated, resiliency is having the means to recover from “bad years.” To be resilient, a species must have healthy populations; that is, have populations that are able to sustain themselves through good and bad years. The healthier and the greater number of healthy populations, the more resiliency a species possesses. For many species, resiliency is also affected by the degree of connectivity among populations and the diversity of ecological niches occupied. Connectivity among populations increases the genetic health of individuals (heterozygosity) within a population and bolsters a population’s ability to recover from disturbances via rescue effect (immigration). Diversity of ecological niches improves a species resiliency by guarding against disturbances and perturbations affecting all populations similarly (*i.e.*, decreases the chance of all populations experiencing bad years simultaneously or to the same extent).

### 1.1.2 Representation

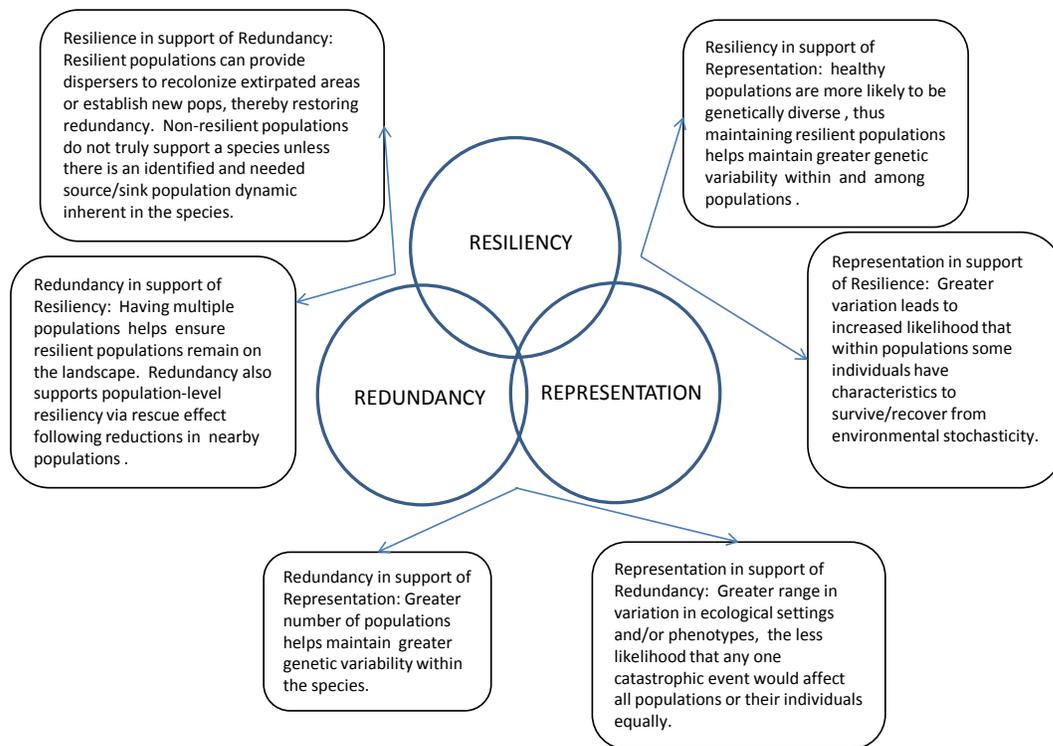
Species level representation is the ability of a species to adapt to long-term changes in the environment; it’s the evolutionary potential or flexibility of a species. Representation is the range of variation found in a species, and this variation (called adaptive diversity) is the source of species’ adaptive capabilities. Representation can be measured through the breadth of adaptive diversity of the species. The greater the adaptive diversity, the more responsiveness and adaptable the species will be over time, and thus, the more viable the species is. Maintaining adaptive diversity includes conserving both the ecological diversity and genetic diversity of a species. By maintaining these two sources of adaptive diversity across a species’ range, the responsiveness and adaptability of a species over time is preserved. Ecological diversity is the heritable physiological, ecological, and behavioral variation exhibited by a species across its range. Genetic diversity is the number and frequency of unique alleles within and among populations.

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

### 1.1.3 Redundancy

Species-level redundancy is the ability of a species to withstand catastrophic events. Redundancy protects species against the unpredictable and highly consequential events for which adaptation is unlikely. In short, it is about spreading the risk. Redundancy is best achieved by having multiple populations widely distributed across the species' range. Having multiple populations reduces the likelihood that all populations are affected simultaneously, while having widely distributed populations reduces the likelihood of populations possessing similar vulnerabilities to a catastrophic event. Given sufficient redundancy, no single or multiple catastrophic events are likely to wipe-out a species. Thus, the greater redundancy a species has, the more viable it will be. Furthermore, the more populations and the more diverse or widespread that these populations are, the more likely it is that the adaptive diversity of the species will be preserved. Having multiple populations distributed across the range of the species, will help preserve the breadth of adaptive diversity, and hence, the evolutionary flexibility of the species.

Figure 1.1. The interrelationships among resiliency, representation, and redundancy (3Rs). The 3Rs, in combination, reflect the species' viability.



## 1.2 Methods

The analysis-specific methods are detailed in the each Chapter, but here we describe, in brief, the methods we used for assessing current and future conditions of EMR (Chapters 4 and 5). We also

describe the Faust *et al.* (2011, pp. 1-66) population model and how we applied their results in our analyses. Lastly, we provide a note on the various population numbers reported in the document.

### 1.2.1 SSA Analyses

The information presented and analyses conducted in this SSA are relevant to a listing decision. The SSA report will be updated as needed for other ESA decision contexts.

Our analysis entailed first assessing the population status (whether the population is extant, extirpated, or unknown) of all known historical populations, and then, assessing the degree of resiliency, representation, and redundancy for those populations believed to be extant or that have unknown status. For our 3R analyses, we assumed populations with unknown status are extant, and collectively referred to these extant and unknown populations as “presumed extant” populations.

To assess population status of each historically known EMR population, we garnered information from the States and Ontario as to whether the population was currently extant, known or likely extirpated, or of unknown status. The criteria used to classify the population status are not standardized across the range. Due to time constraints, we were unable to reconcile differences in methods, and thus we, although cognizant of the differences in classification, applied the population status as it was reported to us.

To assess the current and future degree of resiliency, we evaluated the health (whether the populations are self-sustaining) of those populations classified as presumed extant. We defined self-sustaining as one that: 1) is demographically, genetically, and physiologically robust (DGP robust); 2) has suitable habitat conditions; and 3) is free of, or has manageable, threats acting upon it. More specifically, a self-sustaining robust population is one that has an adult female population size greater than 50 ( $N_f > 50$ ), has a positive population growth rate ( $\lambda \geq 1$ ), and has a high probability of persistence ( $p(P) > 0.90$ ) despite the stressors acting upon it. We relied on the modeling work by Faust *et al.* (2011) to assess the health of the populations presumed extant presently and at years 10, 25, and 50.

To assess the current and future degree of representation, we determined the current and projected future evolutionary potential of EMR across its range. Specifically, we evaluated the changes in adaptive diversity over temporal and spatial scales. We measured the change in adaptive diversity by calculating the change in the spatial distribution (extent of occurrence) of the populations presumed extant presently and at years 10, 25 and 50.

To assess the degree of redundancy, we determined the current and projected vulnerability of EMR to catastrophic events. Specifically, we evaluated the vulnerability to wide-spread extirpation of EMR populations given the number and distribution of populations presumed extant presently and at years 10, 25, and 50.

### 1.2.2 Faust Model

Faust *et al.* (2011) built a customized, age-based, stochastic population model for a hypothetical EMR population, and then applied this model across the range using site-specific information. The demographic parameters underlying the model were derived from empirical data and expert judgment. Faust and colleagues convened experts in 2008 from across the range to facilitate data and information sharing and, ultimately, to elicit expert judgments based on empirical data and collective knowledge.

These experts characterized a “hypothetical healthy population”; that is, a population that has ample habitat such that its vital rates are not affected by density-dependence, has reliable and abundant resources (prey base, hibernacula, etc.), and has high quality habitat with enough open canopy to meet EMR metabolic and ecological needs (Faust *et al.* 2011, p. 3). This healthy population dynamic was referred to as the baseline model. Two versions of the baseline model were developed, an early-maturing and a late-maturing model. The former represents the dynamics of most populations, while the latter is characteristic of populations in the northern portions of the range.

The species experts also identified, based on collective knowledge and empirical data, the most prominent factors (stressors and beneficial actions) that are impacting EMR populations across the range. The experts provided the magnitude and direction of change in the vital rates upon exposure to each factor and the frequency of occurrence of such factors. Faust and colleagues also queried population-specific experts to garner information at extant EMR sites across the range. Data collected included a population size estimate, factors operating at the site, and potential future management changes that might address those factors. Inserting these population-specific data into the baseline model, Faust generated estimates of population growth rate ( $\lambda$ ), ending population size (N), and probability of quasi-extirpation [(p(QE))<sup>1</sup>] for all populations with sufficient data. Probability of quasi-extirpation was defined as the probability of ending with 25 or fewer adult females ( $N_f \leq 25$ ). Faust and colleagues garnered sufficient information to model 57 populations.

As with all models, the Faust model is predicated upon several assumptions; as our analyses rely heavily on the results of the Faust model, the assumptions associated with the Faust model also apply to the results of the SSA. The notable assumptions are listed below; a more thorough accounting of the assumptions and caveats is given in the Faust *et al.* (2011, p. 56) paper. Key assumptions of the Faust *et al.*, (2011 p. 56) model include:

- 1) Males are not limiting the population dynamics.
- 2) Environmental stochasticity/variation is not correlated across vital rates, *e.g.*, a “bad” year for one vital rate does not imply a “bad” year for other vital rates.
- 3) Vital rates are not autocorrelated among years, *e.g.*, a “bad” year in a vital rate does not influence the next year’s rate.
- 4) Density dependence is not impacting population dynamics.
- 5) There is no spatial variation in vital rates at sites across the range, other than those considered in the early-maturing and late-maturing models.
- 6) The definition of each factor is consistent across the range, *e.g.*, “pre-emergent fire” has the same meaning across sites.
- 7) The magnitude of impact is the same across sites, *e.g.*, at all sites, moderate fragmentation decreases first-year survival by 0.3.

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<sup>1</sup> Faust *et al.* (2011), as well as other authors cited within, used the terms extinct, extinction, and quasi-extinct(ion). Based on feedback from one reviewer and a decision maker, we replaced these terms with extirpated, extirpation, and quasi-extirpated(extirpation) to better clarify that the results are referring to losses of portions of the species’ range (populations), not the species as a whole.

8) The impact from multiple factors is additive rather than multiplicative or synergistic, i.e., if multiple factors occur at a site, the effects of those factors are added together.

9) For factors that have a frequency of 1, it is assumed that impact across all model years is constant; in reality, factors such as succession may have an increasing impact over time, or other factors may have a strong immediate effect that then fades over time.

10) For factors with frequencies less than 1 (“proportional” factors), the model randomly determines with a specific frequency whether that factor is applied. For factors such as prescribe fire, which may be done on a regular schedule, such as every 3 years, is applied as a 33% chance of a factor occurring in any given year. Consequently, by chance, a factor may occasionally be applied for multiple years in a row. In general, across the 3000 iterations the effects of this should not be large.

11) Populations with initial adult female numbers less than or equal to 25 are automatically assigned a  $p(QE) = 1$ , even if these populations may be projected to grow over time.

### 1.2.3 Extrapolation Approach

In our analyses, we used the information garnered from experts as well as the results from the population-specific Faust modeling. Specifically, to assess the current and future health of all presumed extant (known extant and unknown) populations, we extrapolated the results of the 57 modeled populations to the additional presumed extant populations rangewide by multiplying the proportion of modeled populations meeting a specific condition (*e.g.*, self-sustaining) by the number of presumed extant populations. Although not ideal, we believe extrapolation is a valid approach for evaluating populations for which we lacked data. Our belief is it is reasonable to assume that the populations for which we have data are representative of what the majority of EMR populations are experiencing. This is particularly true given that the results do not give precise estimates of the future status, but rather, the results provide insights into the extent factors can affect EMR demography and provide a general sense of population health given known factors that are occurring at sites across the range.

### 1.2.4 Note on Numbers of Populations

In conducting our assessment, we used the “best available” information. This information, however, varies greatly from site to site. For some sites we have highly detailed current information, for some we have information from several years ago, and for others we have only vague historical information.

For this reason, we used different subsets of information dependent on the type of analysis and the information that was available. As a result, the number of populations we report varies among our analyses. For example, we have demographic information for 57 populations but have threat data for 165 populations. Thus, our extrapolation for population health and for our factor analyses are based on different populations numbers, 57 and 165, respectively.

As a reference for the reader, Table 1.1 provides information on the data sets that are used in various analyses and referenced throughout this report.

Table 1.1. Tallies of EMR population numbers used throughout this EMR SSA report.

<b>Datasets used in EMR SSA analyses</b>	<b># of Populations</b>
Historical populations	581
2014 Extant populations	267
2014 Unknown population	121
2014 Likely extirpated populations	41
2014 Extirpated populations	152
Presumed extant (extant + unknown status) populations	388
Modeled populations (States and Canada) from Faust <i>et al.</i> 2011 (p. 16)	57
Populations with State and Province reported data on threats	165
Populations with State and Province reported data on threats and 57 modeled populations (States and Canada) from Faust model	222
Presumed extant populations excluding 57 modeled populations from Faust <i>et al.</i> 2011	331

## Chapter 2: Species Description, Distribution & Ecology

In this chapter, we describe the description and taxonomy, the distribution, and life history of EMR. We also provide a summary of the pertinent ecological requirements of the EMR at the individual, population, and species levels. These ecological requirements form the basis of our analyses conducted in Chapters 3 through 5.

### 2.1 Species Description & Taxonomy

The EMR is a small, heavy-bodied snake with a heart-shaped head and vertical pupils. The average length of an adult is approximately 0.6 meter (two feet), with a maximum length of approximately one meter (three feet). Adult EMRs are most often gray or light brown with large, light-edged chocolate brown to black blotches on the back and smaller blotches on the sides, though in some areas (especially in northeast Indiana, southeast Michigan, and northern Ohio) significant numbers of individuals in populations may be nearly or completely black in color. The belly is marbled dark gray or black and there are brown stripes on the sides of the head, each of these bordered by a narrow, white stripe. Its tail has several dark brown rings and is tipped by gray-yellow keratinized rattles. Young snakes have the same markings as adults, but are paler, and have bright yellow tails that darken in color as age progresses. Until the first time the neonates (newborns) shed their skin, the rattle is represented by a single “pre-button” and between the first and second time they shed, the rattle is represented by a complete terminal segment called a “button.” As pitvipers, this species, and all rattlesnakes have an extrasensory “pit” located on each side of the head between the eyes and nares. These pits allow thermal sensing of the environment, potential prey, and other objects. This species and the related western massasauga rattlesnake (*Sistrurus tergeminus*) can be difficult to distinguish based on external characteristics (Gloyd 1948, p. 55). While recent studies have used genetic techniques to determine identity, morphological characteristics used to distinguish the two have included: 1) the number of ventral (belly) scales; 2) the ventral coloration/pattern; 3) the number of and shape of dorsal blotches; and 4) markings and patterns on the nape of the neck and head (Gloyd 1940, pp. 36, 38-40, 42-44, 46-49, 52-55; Evans and Gloyd 1948, pp. 3-6).

The EMR, described by Rafinesque in 1818, has had a variety of locally used common names including: eastern massasauga rattlesnake, eastern massasauga, prairie rattlesnake, spotted rattler, and swamp rattler (Gloyd 1940, p. 44; Minton 1972, p. 315). While the current recommended standard name is Eastern Massasauga (Crother 2012, p. 68), the U.S. Fish and Wildlife Service (Service) has chosen to use the common name eastern massasauga rattlesnake, since the word “massasauga” is not by itself in plain usage nor widely recognized, and because we feel the inclusion of the word “rattlesnake” allows us greater transparency in communicating to the public.

The Service has previously recognized the eastern massasauga rattlesnake as a subspecies (*Sistrurus catenatus catenatus*) of a wider ranging species (*Sistrurus catenatus*). Due to recently published scientific information on the phylogenetic relationships of the massasaugas, we recognize the EMR as a distinct species (*Sistrurus catenatus*). As previously recognized, the massasauga, *Sistrurus catenatus* was one of two species of rattlesnakes within the genus *Sistrurus*, and included three recognized subspecies: *S. c. catenatus* (eastern massasauga rattlesnake), *S. c. tergeminus* (western massasauga rattlesnake), and *S. c. edwardsii* (desert massasauga rattlesnake) (Gloyd 1940, pp. 44-55; Minton 1983, pp. 332.1-332.2), Conant and Collins (1998, pp. 231-232). It was long thought that populations of *S. c. catenatus* and *S. c. tergeminus* intergraded in central Missouri, southwestern Iowa, Kansas, and Oklahoma (Conant

and Collins 1998, pp. 231-232; Evans and Gloyd 1948, pp. 225-232; Gloyd 1940, pp. 44-55). Recent phylogenetic analyses of the genus *Sistrurus*, as well as morphological differences and allopatric (occurring in clearly defined and different areas) distributions between these two taxa provide multiple lines of evidence to indicate that the EMR is distinct from the remaining two subspecies (Kubatko *et al.* 2011, p. 404; Gibbs *et al.* 2011, 433-439). In addition, populations occurring in central and northwestern Missouri and extreme southwest Iowa were formerly considered part of the eastern massasauga rattlesnake distinct population segment (Szymanski 1998). However, recent evidence suggests these populations cluster genetically with the western and desert massasaugas (Chiucci and Gibbs 2010, pp. 5345-5358; Gibbs *et al.* 2011, pp. 433-439; Gerard *et al.* 2011, p. 291; Gibbs 2011, pers. comm.). Similar phylogenetic results have been suggested by other researchers working on this group of snakes (Douglas 2010, pers. comm.; King 2011, pers. comm., Ray *et al.* 2013, pp. 106, 109-111). An article by Holycross *et al.* (2008, pp. 421-424) examined taxonomic conflicts with an old and unused scientific name and the names traditionally used for the three North American massasaugas. In 2011, the International Commission for Zoological Nomenclature (ICZN) was petitioned to conserve binomial usage of, and designate neotype specimens for both *Sistrurus catenatus* and *Sistrurus tergeminus* (Crother *et al.* 2011, pp. 271-274). The ICZN issued opinion 2328 in favor of that petition (ICZN 2013, pp 282-283), and based on the best available information, the Service now recognizes the eastern massasauga rattlesnake as a distinct species, *Sistrurus catenatus*.

## 2.2 Historical & Current Distribution

The documented historical range of the EMR included sections of western New York, western Pennsylvania, southeastern Ontario, the upper and lower peninsulas of Michigan, the northern two thirds of Ohio and Indiana, the northern three quarters of Illinois, the southern half of Wisconsin, extreme southeast Minnesota, east central Missouri, and the eastern third of Iowa. The limits of the current range of the EMR resemble the boundaries of its historical range. However, the geographic distribution of extant localities has been restricted by the loss of the populations from much of the area within the boundaries of that range. Extant populations in central and western Missouri previously considered to be EMR are now known to cluster genetically with the western massasauga rattlesnake (Kubatko *et al.* 2011, p. 404; Gibbs *et al.* 2011, 433-439). However, the non-extirpated populations in the St. Louis metropolitan area of east central Missouri are believed to be EMRs (Evans and Gloyd 1948 pp. 3-10). This determination is based entirely upon phenotypes of museum specimens, but because no viable tissues are available to confirm their phylogeographic relationships through molecular techniques (as the species is extirpated throughout Missouri), we include these populations within the historical range of the EMR, based on the Evans and Gloyd study. Based on the information that we have collected, the EMR is also likely extirpated from Minnesota (USFWS 1998, p. 7).

## 2.3 Life History

Life history includes the annual events in species' life and characteristics that affect the likelihood that individuals (as portions of populations) will survive and contribute to the population from one year to the next. The annual cycle of EMR is characterized by 2 seasons; the active season and the inactive or winter dormant season. The start of the active season varies by latitude but generally it begins in March or April when EMRs emerge from their winter hibernacula and move to their summer habitat where mating and parturition occurs in later summer. Generally, males and non-gravid females spend the active season foraging, while gravid females thermoregulate to obtain optimal body temperatures for young development. In fall, EMRs return to their winter areas to hibernate.

### 2.3.1 Reproduction & Recruitment

Massasauga populations demonstrate considerable variation in reproductive traits throughout the range (Jellen 2005, p. 43). Also, as summarized under the Taxonomy Section (above), until recently (2013), taxonomic practice was to include three subspecies within one wide-ranging species, the massasauga rattlesnake (*Sistrurus catenatus*). Thus, some key literature articles on the life history or other characteristics pooled data for the subspecies as recognized at the time, or considered populations now considered to be the closely related western massasauga rattlesnake *Sistrurus tergeminus*. Among these was the first population viability model for massasaugas (Seigel and Sheil 1999, pp. 17-22). Despite now considered to be western massasaugas, the results of this PVA, as well other similar studies, provide useful information on the life history parameters that could affect population viability.

Early reports were unclear or speculative with regard to the timing of mating. For example, Crawford (1936, pp. 49-50) speculated mating occurred upon emergence from hibernation in spring, because young were born in August and early September in Ohio. Similarly, Atkinson and Netting (1927, pp. 40-43) speculate that the presence of well-developed embryos in July indicated mating was likely in April of early May. Other observations made (*e.g.*, Guthrie 1927, p. 13) were based on specimens held in captivity and therefore may not mimic the timing of wild breeding snakes. Most recent data indicate that mating is actually most prevalent in summer or early autumn, though it may rarely occur in spring (Table 2.1) (Aldridge and Duvall 2002, p. 6; Aldridge *et al.* 2008, p. 405; Jellen 2005, p. 41; Johnson 1995, p. 109; Johnson 2000, p. 189; Reinert 1981, pp. 383-384; Swanson 1933, p. 37). Under captive conditions, massasaugas may mate once from March through May, and again in August through September (Johnson 1989, p. 73).

Table 2.1. Reported dates of birth in the EMR.

State	Date(s)	Source
Illinois	July 29 to August 23	Anton 1993:76, 2000: 248
	August 12	Bielma 1973: 40.
	Late July to early August	Jellen 2005: 37, 40
	August 19 to 30	Tobiasz 1941: 269
	August 17 to September 1	Wright 1941: 666
Indiana	September 1	Hay 1887: 216
	August 14	Adler 1960: 38
Iowa	August 8 to 22	VanDeWalle 2014 (pers. comm).
Michigan	July 26 to August 14	King and Hileman 2013: 26
New York	August 16 to 27	Johnson 1995: 110-111
Ontario	August	Johnson 1998: 71
Pennsylvania	July	Atkinson and Netting 1927: 40-43.
	July 26 to September 7	Swanson 1933: 37

The mating system of many pitvipers includes ritualized male-male aggression, sometimes called “combat” to assert dominance, though it is not as well known in the genus *Sistrurus* (Aldridge and Duvall 2002, p. 20). However, there are published observations of male massasaugas behaving aggressively towards one another (Shepard *et al.* 2003, pp. 155-156; VanDeWalle 2004, pp. 196-197). These observations took place in tall vegetation (Aldridge *et al.* 2008, p. 409), which may explain the rarity of similar observations. The behavior has been commonly observed in North American zoos that maintain breeding groups (Johnson 1989, p. 73); although combat rituals do not appear to be necessary to ensure mating success in captives (Andrew Lentini personal communication to M. Redmer October 29, 2014).

Males may also use chemical cues to simultaneously trail and pursue individual females during the mating season (Johnson 1989, p. 71). Because mature male EMRs often occur at a higher ratio to receptive females (Table 2.3), competition for mates can be intense. Males may exhibit prolonged periods of mate searching, longer daily movements, and defensive female polygyny during the mating season (Jellen 2005, p. 9; Johnson 2000, p. 189).

Like most pitvipers, the EMR is ovoviviparous, meaning embryos develop within eggs held by the female, and gives birth to live young. Data indicate average brood size varies greatly across the range (Table 2.2). While the average brood size was reported as 9.3 (Aldridge *et al.* 2008, p. 404; Jellen 2005, p. 47), there is also a significant relationship between brood size increasing at higher latitudes (Aldridge 2008, pp. 404-406; Jellen 2005, p. 36). This trend may be explained by longer activity seasons at the southern portion of the range as well as the longer time required to reach the size of sexual maturity at lower latitudes (Aldridge *et al.* 2008, pp. 404-406).

Table 2.2. Brood sizes across the range of the EMR.

State/Province	Brood Size	Mean	Source
Illinois	3-11	8.3	Aldridge <i>et al.</i> , 2008: 404-406.
	8-11	9	Anton 1993: 76
	20	--	Anton 2000: 248
	9	--	Bielma 1973: 40
	2-11	6.7	Jellen 2005: 36
	11	11	Tobiasz 1941: 269
	5-14	9.5	Wright 1941: 666
Indiana	7	--	Adler 1960: 38
	5-6	5.5	Hay 1887: 216
		8.8	Kingsbury 2014 (pers. comm.)
Iowa	4-14	9.5	VanDeWalle 2014, pers. communication
Michigan	2-13	6.5	King and Hileman 2013: 4
New York	6-13	9.3	Johnson 1995: 11, 110
Ohio	9	9	Watkins-Colwell 1995: 40
Ontario	9-19	13.3	Parent and Weatherhead, 2000: 175
Pennsylvania	5-9	7.3	Kowalski 2007
	5-7	6.5	Reinert 1981:393
	3-8	6	Swanson 1933: 37
Wisconsin	6-19	11.1	Keenlyne 1978:372-374

The male reproductive cycle begins when the testicular segment of the kidneys of mature male EMRs starts to thicken in June, with the first sperm appearing by mid-June, and maximum spermatogenesis

occurring in July through September (Aldridge *et al.* 2008, p. 405). Captive male EMRs follow a similar cycle (Lentini 2014 pers. comm.).

Female EMRs mature during the summers of their second or third full growing seasons following birth (Aldridge *et al.* 2008, p. 406; Keenlyne 1978, p. 372; Reinert 1981, pp. 393-394), but may take as long as five or six years to reach maturity in more northern parts of the range (Johnson *et al.* 2000, p. 9). Captive EMRs appear to become sexually mature at 25 months following birth (Johnson 1989, p. 73).

With few exceptions (Bielma 1973, p. 62; Keenlyne 1978, p. 372), the female reproductive cycle in EMRs follows a biennial pattern in which mating and parturition are skipped in years following ones where broods are born (Aldridge *et al.* 2008, pp. 407-408; Jellen 2005, p. 41; Johnson 1995; Reinert 1981, pp. 393-394). The average size of adult EMRs is 24 inches (61.0cm) and the largest individual recorded was 32 inches (81.3cm) Crawford 1936, p. 31). Wright (1941, p. 664) considered adult EMRs to be those over about 21 inches (54.1cm) in length. Based on extrapolation of data from average growth rates, it was estimated that females in an Illinois population would mature at an age of two years (Bielma 1973, p. 46). Although a captive female reportedly produced two broods in approximately six months (Johnson 1989, p. 77). The cycle was described in detail by Jellen (2005, p. 41) and Aldridge *et al.* (2008, pp. 404-406) and includes mating during mid to late summer, after which the females store sperm for the remainder of the active season and throughout hibernation. Ovulation and fertilization take place following emergence from hibernation during the following spring, with embryonic heartbeats detectable by late May. Skeletal formation, growth of the embryos, and birth follows from late July to August. Following giving birth in summer, females enter a non-reproductive state characterized by the presence of non-vitellogenic oocytes, and may forage heavily before entering hibernation (Aldridge *et al.* 2008, p. 405; Jellen 2005, p. 41).

The physiological costs of reproduction result in reduced mating events by female vipers, which mate only after they have sufficient energy reserves (Aldridge and Duvall 2002; p.; Bull and Shine 1979, pp. 279-282). The time needed to forage and reallocate nutrient resources expended while carrying broods may be great, and the success in replenishing lost energy reserves may not only determine whether they will survive hibernation, but also whether they will mate or reproduce the following year (Aldridge *et al.* 2008, pp. 406-407). Because of this, receptive females may occur in lower proportion to males during the breeding season, even in populations in which there is a female bias. In one case a captive female produced two broods in only six months, though it was suggested that optimal captive conditions contributed to this (Johnson 1989, p. 77). Nonetheless, researchers in Grayling, MI observed both male and female adult EMRs who had either mated or available to mate that spent time foraging at the expense of time looking for a mate (Tetzlaff *et al.* 2015b).

Correlations between brood size and female length have been reported in some populations (Parent and Weatherhead 2000, p. 175), but not in others (Aldridge *et al.* 2008, p. 405). Strong relationships between maternal prepartum mass and both litter size and litter mass are also known (Aldridge *et al.* 2008, p. 405). Similarly, females in the genus *Sistrurus* and other vipers show high investment in reproduction as measured by pre-birth mass lost by the female during birth of a brood. In southcentral Illinois, females lost between 24.4-55.5% (mean = 43.6%) of their prepartum mass (Aldridge *et al.* 2008, p. 405). Females of the related pygmy rattlesnake (*Sistrurus miliarius*) lost 45% of their pre-partum mass (Farrell *et al.* 1995, p. 23). While few studies on EMRs have quantified the percent of mass lost due to birthing, observations of body condition of post-partum females have indicated an emaciated appearance (Bielma, 1973, p. 41; Reinert 1981, p. 393). In the largest reported EMR brood (20),

produced by a female from northeast Illinois, the female lost 55.7% of her pre-birth mass by passing both neonates and other embryonic-associated tissues (Anton 2000, p. 248).

Female EMRs may occasionally re-absorb developing embryos which could be a result of ectopic pregnancy indicating passage out of the uterus due to injury or other mechanism (Aldridge *et al.* 2008, p. 406; Jellen *et al.* 2007, pp. 343-344). Similar unexplained disappearance of previously detected embryos (without subsequent birth) in captive female EMRs has been documented (D. Boehm, Curatorial Manager, Lincoln Park Zoo, personal communication to M. Redmer USFWS November 2014).

### 2.3.2 Survivorship

Survival of EMRs was summarized in two papers; Bailey *et al.* (2011, pp. 170-171) and Jones *et al.* (2012, pp. 1581-1583). Using survival rates of radio telemetered EMRs on managed habitat in southwest Michigan, short-term survival was calculated to provide a daily survival estimate (0.9997), a ninety-day survival estimate (0.9714), and a 168-day survival estimate (0.9742) for the site (Bailey *et al.* 2011, p. 170). Similarly, pooled datasets from radio telemetered studies across the range of the EMR, and three populations of western massasaugas from Missouri were used to calculate survivorship (Jones *et al.* 2012). When human caused sources of mortality were excluded, weekly active-season survival averaged 0.99, cumulative active-season survival averaged 0.77, and quasi-winter survival averaged 0.89 (Jones *et al.* 2012, p. 1581). Jones *et al.* (2012, p. 1582) also reported that survival was positively correlated along a southwest to northeast geographic gradient.

### 2.3.3 Population Structure & Sex Ratio

Studies can be found that indicate the age class of captured EMRs (Davis 2008, pp. 22-25) at a particular site, but studies on the actual numbers of individuals in each stage class that are necessary to sustain a population is lacking. Most studies including samples of multiple broods have found nearly equal sex ratios (Keenlyne and Beer 1973, p. 381; Kowalski 2007, p. 18) as is common in snakes (Shine and Bull 1977, pp. 231-233). Other studies indicate that some individual broods may strongly bias one sex but over time the ratio still approaches 1:1 (Jellen, 2005, p. 43; Aldridge *et al.* 2008, p. 407). However, adult portions of populations are often biased significantly to more females than males (Table 2.3), and this could lend to the idea that the greater vagility of males incurs greater risk of mortality (especially during the mating season). For example, there have been documented increases in road mortality among males during the mating season (Shepard *et al.* 2008, p. 293). Conversely, a male-biased sex ratio at birth may counter increased male mortality so the sex ratio is more even among adults (Jellen 2005, p. 43). The sex ratio in a Missouri population of the closely related western massasauga (*Sistrurus tergeminus*) reportedly shifted (from female biased to male biased) following a severe flood (Seigel *et al.* 1998, pp. 129-130). It was suggested that the reduced ability of locomotion in gravid females as compared to males and non-gravid females may have made them more vulnerable to the effects of the flood.

Table 2.3. Reported sex ratios of individual populations across the range of the EMR.

State/Province	Sex Ratio M:F	Type	Source
Illinois	1:5.7 (3:17)	Single brood	Anton 2000: 248
	1:0.8	Entire sample	Dreslik <i>et al.</i> , 2011
	1.3:1 (13:10)	Adult, entire sample	Mauger and Wilson 1999: 113
	1.8:1 (14:8)	Two broods	Tobiasz 1941: 269
	1:3.5 (2:7)	Adult, one season captures	USFWS Chicago Field Office unpublished data
Michigan	1:1.6 (119:186)	Adult, entire sample	King and Hileman 2013: 4
New York	1:1.48 (29:43)	Entire sample	Johnson 1995: 107-108
Pennsylvania	1:3.7	Adult, one site.	Kowalski 2007
	1:1.6	Adult, one site.	
	1:2.6	Adult, entire sample	
	1:1.57	Juvenile, one site	
	1:1.03	Juvenile, one site	
	1.18:1	Juvenile, one site	
	1:1 (80:81)	Juveniles/broods, entire sample	
Wisconsin	1.1:1 (107:100)	Broods, entire sample	Keenlyne and Beer 1973:381

## 2.4 Species Ecological Requisites

This section summarizes the key ecological requirements for survival and reproduction of EMR at the individual, population, and species levels.

### 2.4.1 Individual Level Ecology

The EMR is active in the spring, summer, and fall and inactive in the winter when it hibernates. Therefore, depending on whether the snake is active or inactive determines what type of habitat is required. The EMR also requires different habit types during its various life stages (i.e., neonate, juvenile, adult, gravid female). The individual level ecological requisites are described below and summarized in Table 2.4.

#### a. Active Season (Spring, Summer, and Fall) Habitat

The EMR has a broad range both latitudinally (occurring from approximately 38° in south central Illinois to 46° north latitude in Ontario, Canada) and longitudinally (occurring from approximately 76° in New York to 94° west longitude in Iowa). As a result, some plasticity in the active seasons is expected. The EMR can be out of the hibernacula from approximately March to November in the southern part of their range (Dreslik 2005, p. 37) and April to October in the middle and northern parts of the range (Beltz 1993, p. 16; Mauger and Wilson 1999, p. 120; Smith 2009, p. 14). Different dates of emergence and return have been cited in the literature and presumed determined by local climate factors and local microhabitats (Beltz 1993, p. 16; Mauger and Wilson, 1999, pp. 118-120; Marshal *et al.* 2006, p. 145). In one Indiana fen habitat, snakes emerged from hibernation in mid-April but could be seen as early as late March (Marshal *et al.* 2006 p. 145, Kingsbury *et al.* 2003 p. 7). Soil temperature may be used as a predictor of whether or not a snake may be out of its hibernaculum. Mauger and Wilson (1999, p. 120)

report that snakes had not been found out of burrows at soil temperature less than 10.0<sup>0</sup> C, even under ideal basking conditions. After emergence, the snakes move to basking areas within a few meters of their hibernacula and remain there for about a week (Marshall *et al* 2006, p. 145), sometimes using the hibernacula as refuge during inclement weather (DeGregorio 2008, p. 10). Observations in Iowa show that this may be from one to several weeks depending on weather conditions (personal communication from VanDeWalle to Clemency 06/09/2015). Over the following two weeks, males and non gravid females move 200 to 600 meters away from their hibernacula to habitats where they establish activity centers (Marshall *et al.* 2006, p. 145). Generally, in the spring through early summer, males and non-gravid females tend to move from their hibernacula into upland and structurally complex vegetation communities for foraging, and often, courtship and mating. In Canada, snakes hibernate in forested areas and continue to use forested habitats throughout the active season (Weatherhead and Prior 1992, p. 450), but increase their use of open, wetland, and edge habitat to a peak in mid-summer (Harvey and Weatherhead 2006, p. 214).

The type of habitat used during the active season generally consists of high, dry habitats, open canopy wetlands and adjacent upland areas (Sage 2005, p. 32; Lipps 2008, p. 1). Active season habitat use varies regionally and this variability has been observed in multiple EMR populations across its range (Reinert and Kodrich 1982, p. 169; Johnson *et al* 2000, p. 3). Because of this, individual snakes can be found in a wide variety of habitats including: old fields (Reinert and Kodrich 1982, p. 163; Mauger and Wilson 1999, p. 111) bogs, fens (Kingsbury *et al* 2003, p. 2; Marshall *et al.* 2006, p. 142), shrub swamps, wet meadows, marshes (Wright 1941, p. 660; Sage 2005, p. 32), moist grasslands, wet prairies (Seigel 1986, p. 334), sedge meadows, peatlands (Johnson and Leopold 1998, p. 84), forest edge, scrub shrub forest (DeGregorio *et al.* 2011, p. 378) floodplain forests (Moore and Gillingham 2006, p. 745) and coniferous forests (Harvey and Weatherhead 2006, p. 207). At the eastern range periphery, (New York and southern Ontario) EMR prefer peatland community types (Johnson 1995, p. 54). In an Indiana fen habitat, the snakes prefer emergent wetland and meadows with woodlands being least preferred (Marshall *et al.* 2006, p. 148). However, in Pennsylvania, the snakes use some woodlands, wetter grasslands, and upland grasslands, with all areas having a shrub component (Reinert and Kodrich 1982, p. 166-170). A southeastern Michigan population was found to use open canopy wetland and dry herbaceous grassland as active season habitat (Sage 2005, p. 32).

Active season habitat use can also be site dependent even within a particular region (Reinert and Kodrich 1982, p. 169; Kingsbury 2002, p. 9). For example, one population in the upper Bruce Peninsula, Ontario, Canada is associated with sand dunes, wetlands, deciduous forest and coniferous forest whereas another EMR population 12.5 km (7.8 mi) away from the first location is associated with wetlands, deciduous forest, coniferous forest, and a large successional field (Harvey and Weatherhead 2006, p. 207).

Active season habitat consists of thermoregulatory or *basking* sites, *retreat* sites, and *foraging* sites.

#### *Basking Sites*

Basking sites are critically important for snakes because thermoregulation is closely associated with the physiological functions of shedding, digestion, locomotion, and gestation. Basking sites are generally open, sunny areas in higher and drier habitats than used in fall or winter. Basking area vegetation is usually short but it is adjacent to taller vegetation. Shorter vegetation may be more desirable for thermoregulation whereas taller vegetation may provide better cover (Marshall *et al.* 2006, p. 148). Within this habitat of open vegetative structure, individual EMRs often select sites near isolated trees or

shrubs which may occur due to either the shade provided by the trees or shrubs or for protection from aerial predators (Johnson *et al.* 2000, p. 3).

### *Retreat Sites*

The availability of retreat sites is important to the snake at all times of the year. Retreat sites are generally used by the snake to hide from potential predators, but are also important to gain shelter from extreme temperatures because these sites are more thermally stable than surface habitat (Shoemaker 2007, p. 9-10). Retreat sites can be hibernacula, rock crevices, hummocks, live or dead tree root systems, mammal holes, crayfish burrows, shrubs, boards, burn piles before burning, or any structure that a snake can crawl into or under. Potential predators of the EMR are carnivorous mammals, birds-of-prey, and ophiophagous (prey upon snakes only) snakes (Szymanski 1998, p.11). One study in Ontario (Harvey and Weatherhead 2006, p. 213) found that snakes remained close (within ~0.5 m (1.6 ft)) to retreat sites regardless of the time of year, perhaps to reduce the risk of predation. The authors concluded that the most consistently preferred habitat among EMRs was a nearby retreat site (Harvey and Weatherhead 2006, p. 212).

### *Foraging Areas*

Foraging habitat provides opportunities for snakes to encounter and take prey species. For species such as rattlesnakes that hunt by ambushing prey, habitat selected in which to wait for prey must simultaneously satisfy multiple needs (feeding, thermoregulation, predator avoidance) for longer periods of time than locations used by species that are active hunters (Harvey and Weatherhead 2006, p. 207), further increasing the importance of habitat selection. Foraging habitat can be floodplain, riparian, lowland, and upland forest or any area that provides an adequate abundance of suitable prey. Foraging habitat usually has an open canopy and a sedge or grass ground cover (Johnson *et al.* 2000, p. 4). In Illinois, the EMR showed a strong habitat preference for early successional grasslands (Dreslik 2005, p. 155 and 183), presumably for foraging.

#### b. Diet

The diet or prey of the EMR varies across the species range (Holycross and Mackessy 2002, pp. 454-464; Weatherhead *et al.* 2009, p. 693) but may consist primarily of small mammals (*Microtus*, *Peromyscus*, and *Blarina*) (Mauger and Wilson 1999, p. 113; Holycross and Mackessy 2002, p. 454). Juvenile EMR occasionally feed on snake species (VanDeWalle and VanDeWalle 2008, p. 358; Shepard *et al.* 2004, p. 365). Mauger and Wilson (1999, p. 113) found that subadult EMR size classes showed an equal proportion of both mammal and snake prey.

The EMR occupies a variety of ecologically distinct habitats (*e.g.*, prairie, bog, fen). Each habitat supports different prey communities, and thus, habitat drives a shift in prey species for the EMR. Diet may be stable over short periods of time (one year) but highly variable among years and populations across the range of the EMR. The importance of specific prey items varies by population (Chucchi *et al.* 2012, p.2). This sit and wait rattlesnake predator (like EMR) appears to forage on whatever prey species is most easily available within its habitat. In Grayling, MI, researchers observed the opportunistic and diverse diet of adult EMR who had foraged a variety of items including an American red squirrel, an adult Eastern Garter snake, two Northern Red-bellied snakes, a nestling Brown Thatcher, and a mostly digested small murid mammal (Tetzlaff *et al.* 2015b). At a Wisconsin EMR population, Keenlyne and Beer (1973, p. 383) found that 95% of all food items were endotherms with over 85% of the diet

consisting of one species of vole (*Microtus pennsylvanicus*). Results from another study, which sampled the diet of five different EMR populations (three in Ohio, one in New York, and one in Illinois), found that shrews (*Blarina* and *Sorex*) were the most common prey (making up an estimated 58% of the total diet) followed by voles (15%), snakes (12%), frogs (8%), and mice (7%) (Chucchi *et al.* 2012, p. 9). The taking of frogs and small snakes by the EMR as food items is believed to be geographically variable and may be determined by habitat differences and/or abundance of prey in a particular population (Shepard *et al.* 2004, p. 364; Chucchi 2011, p. 105). Chucchi (2011, p. 109) theorized that although shrews represent a much smaller meal, they may be preferred over other mammalian prey for the following two reasons: 1) shrews have a much higher metabolic rate and are more active than other mammals which may cause the rate at which EMR encounter shrews to be higher than the encounter rate for voles and mice despite the lower abundance of shrews in many populations; and 2) EMR venom may be more efficient against smaller species with a high metabolism by increasing the speed with which the prey item is incapacitated thereby limiting the distance a prey item can travel after injection of venom and thus increasing the likelihood that the snake can find this prey item and ingest it (Chucchi 2011, p. 109). In three Ohio EMR populations, Chucchi (2011, pp. 109-110) reported avoidance of voles as prey, which he explained is a function of the different natural histories. *Microtus* species prefer to use runways as methods of travel through habitats, while EMR are sit-and-wait predators that seek cover to ambush their prey. Thus, unless EMR choose ambush sites along runways (which would increase their risk of being preyed upon), their likelihood of encountering voles may be much lower than the likelihood of encountering shrews (Chucchi 2011, p. 110).

### c. Gravid females

The reproductive condition of females is a factor that influences habitat requirements. Gravid females require gestation sites and retreat sites but do not require foraging areas because they do not feed until after parturition. Females are often emaciated following parturition and must forage to replenish energy reserves before hibernating (Harvey and Weatherhead 2010, pp. 65-66). Therefore, foraging areas are important to gravid females only after parturition. Non-feeding gravid female EMRs were found to feed ravenously after giving birth (Keenlyne and Beer 1973, p. 383).

Gravid females select sites with a more open canopy to regulate their body temperature and thus facilitate embryo development (Marshall *et al.* 2006, p. 148; Shoemaker 2007, p.14). They thermoregulate much more than nongravid females with their activity restricted to shifting between basking locations and overnight refugia (Marshall *et al.* 2006, p. 148; Harvey and Weatherhead 2011, p. 65; DeGregorio *et al.* 2011, p. 77). Gravid females maintain significantly higher mean body temperatures above ambient temperature compared with non-gravid females (Foster *et al.* 2009, p. 48). Gravid females tend to be relatively sedentary, centering their activities near various types of cover (*e.g.*, large rocks, stumps, brush, or debris piles) in relatively open areas with prolonged exposure to direct sunlight. Gestation sites must have low crown closure, surrounding vegetative ground cover, and a feature that provides relatively warm refuge during cool weather such as a large rock, stump, beaver lodge, brush or debris pile. Harvey and Weatherhead (2006, p. 210) found that gravid females preferred locations with more nearby rock cover, closer retreat sites, and less canopy cover than non-gravid females. A critical component of suitable gestation sites appears to be open-canopy areas.

Thermoregulation is so important to gravid female EMRs that they spend the majority of the gestation period within open-canopy areas (Reinert and Kodrich 1982, p. 169). The consistent selection of early to mid-successional upland vegetation by gravid females in Michigan (Foster *et al.* 2009, p. 52), Pennsylvania (Reinert and Kodrich 1982, p. 165), and Ontario (Harvey and Weatherhead 2006, p. 213)

suggests that this type of vegetation is most likely preferred by gravid female EMRs rangewide. This type of habitat has significantly higher mean soil temperature than early to mid-successional wetlands (Foster *et al.* 2009, p. 48). Depending on the location of the population, gestation habitat of gravid female EMRs could be rock outcroppings, open grassland, shoreline, sedge meadow, barrens, or any suitable land characteristic that provides the snake the ability to thermoregulate and avoid predators. Usually gravid females will remain near their winter hibernacula until parturition in late July or early August and then move to other foraging locations (Marshall *et al.* 2006, p. 145; Johnson 1995, p. 23). In New York, males and nongravid females left peatlands after hibernation and used adjacent wooded swampland, whereas gravid females remained in peatlands until parturition (Johnson 2000, p. 8). In Canada, gravid females used a beaver lodge in a marsh, a large flat rock in a field, a juniper on an alvar, and the edge of a gravel path in a forest (Harvey and Weatherhead 2006, p. 214). In northern Michigan gravid females selected barrens (with a nearly complete lack of over story canopy) as their gestation sites (DeGregorio 2008, p. 17). In Indiana gravid females chose shoreline which consisted of a narrow band of habitat along the shore of a lake which was dominated by rushes (*Juncus spp.*) and shrubby cinquefoil (*Potentilla fruticosa*) (Marshall *et al.* 2006, p. 142). At one site in Pennsylvania, gravid females preferred the driest and most open sites available (Reinert and Kodrich 1982, p. 165).

Local gestation sites may be used by several females in a given season and appear to be used by the same individuals in successive breeding years. While at their chosen gestation sites, gravid female snakes generally engage exclusively in basking behavior, forfeiting opportunities for other essential behaviors such as feeding (Keenlyne and Beer 1973, p. 384; Marshall *et al.* 2006, p. 141; Weatherhead and Prior 1992, p. 451). Since gravid females feed very little, if at all, it appears that they maintain themselves on reserved body energy (fat) throughout their pregnancies (Keenlyne and Beer 1973, p. 384). In the fall, gravid females continue to thermoregulate more than males or non-gravid female snakes, despite giving birth in late summer (Harvey and Weatherhead 2011, p. 65).

#### d. Neonates

Neonates, which are born near the end of summer (late July early August) have a short active season before their first hibernation, even so they require basking sites, retreat sites, and foraging areas as described above. There is a difference in the type of prey consumed by neonates as opposed to adult or subadult EMRs. Mauger and Wilson (1999, p.113) found that younger age classes of EMRs (i.e. juvenile and subadult) ingested a higher proportion of snake prey. In contrast, the prey recovered from free ranging neonate EMRs at a site in southern Illinois consisted primarily of southern short-tailed shrews (*Blarina carolinensis*) (Shepard *et al.* 2004, p.360). In feeding trials, neonate EMRs showed a preference for snake prey, disinterest in frog and insect prey and indifference toward mammal prey (Shepard *et al.* 2004, p.360). It is theorized that due to gape limitations neonates may have difficulty ingesting small mammals, but snakes are easier to ingest and are the most common prey item in young EMRs (Shepard *et al.* 2004, p. 365). The presence of garter snakes (*Thamnophis sirtalis*) within EMR habitat may be an indicator of the possible survival of the neonate age class as the garter snakes appear to be used as a neonate EMR food source (King *et al.* 2004, p. 436). Because garter snakes also bear young in the fall, this particular food source may be more available and easier to ingest for neonate EMRs (Keenlyne and Beer 1973, p. 384).

Most neonates consume a prey item, at least occasionally, which suggests that successful foraging prior to hibernation may be important for survival (Baker *et al.* 2010, p. 4). After the EMR's first hibernation, their diet appears to shift primarily to endotherms.

#### e. Nonactive Season / Winter and Hibernation Habitat

After the active season, EMRs move to low wet areas for overwintering or hibernation (Reinert and Kodrich 1982, p. 164 and 169; Johnson *et al.* 2000, p. 3; Harvey and Weatherhead 2006, p. 214; Mauger and Wilson 1999, p. 117). Seasonal habitat use is driven by the inability of a single area to meet all of a snake's habitat requirements throughout the entire year (Johnson *et al.* 2000, p. 3). However, some populations of EMR do not appear to show a seasonal shift in habitat use, with individuals remaining in wetlands all year (Wright 1941, p. 662; Weatherhead and Prior 1992, p. 450).

To survive the winter, each individual EMR requires a suitable hibernation site which is critical to avoid lethally low temperatures and reduce the risk of desiccation (Reinert and Kodrich 1982, p. 169). Consequently, hibernation sites must provide insulated and moist subterranean spaces below the frost line where individuals can avoid freezing and dehydration (Sage 2005, p. 56). Most EMRs will either return to the same hibernacula annually (Johnson *et al.* 2000, p. 26) or to an area within roughly 100 m (328 ft) of their previous hibernation site (Sage 2005, p. 61; Harvey and Weatherhead 2006 p. 213). It has been postulated that the reason for the return may be that either suitable hibernation sites are not generally available and/or that there is a cultural component to site selection (Johnson *et al.* 2000, p. 26). Because of their preference to return to the same hibernacula, the snakes become highly concentrated from the time they return to a hibernation area until the time they disperse in the spring (Johnson *et al.* 2000, p. 26). These hibernation sites can occur in wetland, wetland edges, wet prairie, closed canopy forests with mossy substrates (DeGregorio 2008, p. 20), wet grassland, and sedge meadow (Mauger and Wilson, 1999 p. 116). In Canada, snakes hibernate in forested areas but those that were tracked in forest habitat were found to occasionally frequent small openings presumably to thermoregulate (Weatherhead and Prior 1992, p.450).

Across its range, EMRs have been reported to hibernate for up to six months of the year, and have used crayfish burrows, mammal burrows, rocky crevices, rodent holes, hummocks, old stumps, rotten logs, and tree and shrub root systems (Wright 1941, p. 660; Johnson 1995, p. 36; Mauger and Wilson 1999, p. 117; McCumber and Hay 2003, p. 5; Dreslik 2005, p. 96; Harvey and Weatherhead 2006, p. 214; Johnson and Leopold 1998, p. 84; Sage 2005, p. 35) or any excavation that reaches the water table (Reinert 1978, p. 63). The snakes hibernate either singly or in small groups or clusters, aggregating where favorable microhabitats occur. The habitat needs while hibernating seem to be determined by features below ground (water table depth, water table dissolved oxygen, temperature, and pH) which differ from the above-ground habitat used by snakes while active (*e.g.*, Sage 2005, p. 70-73). Sage (2005, pp. 70-73) found that snakes may be selecting hibernation sites where water tables are near, but not at the surface, and where a buffer exists between the water table and the surface. This buffer suggests that the snakes need some level of protection against flooding during the hibernation period (Sage 2005, p. 71). The ability of a snake to tolerate flooding depends on the amount of oxygen available within groundwater, the metabolic demand for oxygen (influenced by temperature), and the rate of transfer across the integument. The EMR has evolved three strategies to manage oxygen demands during hibernation. Because hibernating snakes must spend most of the winter fully submerged in water, they may tolerate anoxic conditions, rely on extrapulmonary respiration, or periodically surface and breathe air (Sage 2005, pp. 70, 71). Sage (2005, pp. 72, 73) observed that EMRs must periodically breathe throughout the winter, but can tolerate short periods where access to air is limited. Sage (2005, p. 72) also observed that snakes position themselves near the surface of the water at different times during hibernation and occasionally stick their heads out into the air column. Most of these observations occurred during the fall, and near emergence in the spring, when groundwater temperature and snake

body temperatures were highest (Sage 2005, p. 72). It is presumed that metabolic activity and thus oxygen demand are at their peak during these periods as well.

Baker *et al.* (2010, p. 5) found that all, with the exception on one, neonate hibernation sites were shared with either eastern garter snakes or another (either neonate or juvenile) EMR. Similarly, King *et al.* (2004, p. 436) reported that in both years of their study (2000 and 2001) garter snakes were seen emerging from every hibernaculum shared with a neonate EMR which was tracked during the study. Neonate EMRs may use the presence of other snakes including other snake species as an indicator of the suitability of a particular burrow (Baker *et al.* 2010 p. 5; King *et al.* 2004, p. 436).

#### f. Microhabitat Connectivity

All EMR need corridors between microhabitats (basking sites, retreat sites, and foraging areas) and between seasonal habitats. Eastern massasauga rattlesnakes can traverse corridors most successfully (reduced likelihood of mortality) between habits when there are no barriers such as roads, rivers, or anything that can act as a barrier to snake movement. The absence of roads is an important criterion because roads are a strong barrier to EMR movement due to road mortality (Shepard *et al.*, 2008a., pp. 290-293; Shepard 2008b. pp. 352-356; Choquette 2011, pp. 63-65) or road avoidance behavior.

#### g. Summary

Individual EMRs require open, sunny areas, intermixed with shaded areas, for thermoregulation (basking sites), abundant and available prey (foraging sites), the ability to escape both temperature extremes and predators (retreat sites), presence of the water table near the surface for hibernation (hibernation sites), and connectivity between each of these habitats.

Particular plant species or soils within EMR habitat do not seem to be as important as the structure of the habitat to the EMR (Beltz 1993, p. 21). Preferred habitats tend to have a generally open vegetative/shrub structure, where trees and shrubs are thinly distributed, relative to surrounding areas (Johnson *et al* 2000, p.1). This open vegetative structure provides the desirable thermoregulatory areas, increases prey densities for the snakes by enhancing the growth of sedges and grasses (Johnson *et al.* 2000, p. 3), and provides retreat sites. Regardless of the season, all EMR habitats include sunny and shaded areas, and an open vegetative or early successional structure (Beltz 1993, p. 21; Reinert and Kodrich 1982, p. 169; Johnson *et al.* 2000, p. 3).

Table 2.4 Individual-level Ecology of EMR: the requisites needed for survival and reproduction.

Season	Life Stage	Survival & Reproduction Requisites	Description
Spring	Adults and Juveniles	access to basking or thermoregulatory sites	generally higher, drier habitats than used in fall and winter: upland
			sunny areas
			adjacent to hibernaculum sites
		access to retreat sites	hibernacula
		access to foraging sites	foraging site must have retreat sites and basking sites prey must be available
Summer	Adults and Juveniles	suitable thermoregulatory sites	upland habitat
			shorter vegetation but adjacent to taller vegetation
			retreat sites nearby
		suitable foraging habitat	floodplain, riparian, lowland, and upland forest
			suitable and adequate abundance of prey throughout the summer period
		suitable retreat sites	cover during extreme temperatures
			cover (tree, shrub, rocks, etc.) to escape predators
		adequate connectivity among habitat areas	habitat connectivity among foraging areas, thermoregulatory areas, and retreat sites
	suitable gestation sites	thermoregulation and predator avoidance for gravid females & for embryo development	
	suitable retreat sites	rock crevices, shrubs, crayfish burrows, other	
	Neonates	suitable foraging habitat ☐ suitable thermoregulatory sites suitable retreat sites	foraging sites for prey
			upland habitat
rock crevices, shrubs, boards, burn piles before burning, etc.			
Fall	All Snakes	access to retreat sites	for thermoregulation
			predator avoidance
		corridor from summer habitat to hibernation sites	barrier-free (road free) pathways to hibernacula
Winter	All Snakes	access to basking or thermoregulatory sites	sunny areas adjacent to hibernaculum
			suitable hibernacula

### 2.4.2 Population Level Ecology

In this section, we describe the ecology needs of a healthy population (*i.e.*, what a population requires to sustain itself over time). Self-sustaining populations are those that are able to respond to and recover from disturbances (stochastic events such as fire, flooding, storms) and perturbations (normal year-to-year environmental variation in temperature and rainfall). Simply said, healthy populations are

those able to sustain themselves through good and bad years. To be self-sustaining, populations must: 1) be demographically, genetically, and physiologically robust, 2) occupy areas with suitable habitat conditions for all life stages and seasons, and 3) have only manageable threats acting upon them. These three population level requisites are discussed below and summarized in Table 2.5.

a. DGP Robust

To respond and recover quickly from disturbances and to adapt to normal environmental variation, populations need to be demographically, genetically, and physiologically (DGP) robust (adapted from Redford *et al.* 2011), which means having sufficient numbers of individuals and positive growth rates. The number of individuals is important because as the population size declines, a host of challenges arise. Smaller (reduced in size) populations behave differently; their population dynamics become strongly influenced by demographic (*e.g.*, higher chance of skewed sex ratios), environmental (*e.g.*, greater vulnerability to fluctuations due to bad years that lead to extinction), and genetic (*e.g.*, higher chance of losing genetic variation, and hence, increase homozygosity) stochasticity.

As a population becomes smaller, adverse genetic consequences are more likely to occur. Specifically, as a population decreases, its gene pool becomes smaller and loss of genetic diversity more likely. As genetic variation declines, so too does a population's ability to respond to changes (disturbances and perturbations) in its environment. Genetic variation (which is the source of adaptability for a species) is determined by the joint action of natural selection and genetic drift (chance). In small populations, genetic drift is higher which can lead to deleterious alleles becoming more frequent and fixed in population due to chance alone. Also, in small populations, related individuals are more likely to breed, which can lead to increase homozygosity (decreased genetic variation) in individuals. Similarly, as a population becomes smaller, greater fluctuation in population demography (birth and death rates, sex ratio) due to chance alone occurs. Greater fluctuation in demographic rates leads to higher chances of population extirpation. Lastly, as the population becomes smaller, they become more vulnerable to environmental variation; they are less able to recover from bad years. Thus, to be DGP robust, population numbers should be comfortably above the species-specific population size in which small population vulnerabilities (genetic, demographic, and environmental consequences) begin to exert control over the population dynamics.

Studies from across the range provide insights into EMR sensitivity to population size. Middleton and Chu (2004, p. 41) showed even the loss of a single adult individual from a population can strongly increase extirpation risk. Miller (2005, p. 22) found a considerable increase in extirpation risk when population size less than 50 individuals. A study of population in southwestern Michigan found population persistence declined as the numbers of individuals declined, and population persistence may be in peril if individual numbers decline below 130 (Bailey 2010, p. 40). A study in Missouri of the closely related western massasauga (*Sistrurus tergeminus*) found that populations with 50 or fewer individuals may have high rates of extirpation (Seigel and Sheil 1999, p. 20). Faust *et al.* (2011, p.4) elicited expert judgments from EMR researchers across the range of the species. Based empirical data and experience, these experts agreed that 25 adult females is a reasonable threshold in which a population might begin experiencing negative consequences associated with small populations (Faust *et al.*, 2011, p. 3). From these data, it appears small population dynamics begin exerting control when EMR populations are reduced to 25 to 35 adult females. Thus, to be DGP robust, a population must exceed this adult female population size threshold.

Population size (whether it increases or decreases) is influenced by a population's growth rate ( $\lambda$ ). For a population to grow, it must have a positive growth rate ( $\lambda \geq 1$ ). Population growth rate is primarily a function of reproduction, survival, and mortality rates of the individuals comprising the population, and the age-structure of the population.

Jones *et al.* (2012, p. 1583) found a relationship between adult survival and a latitudinal-longitudinal gradient (with lower survival in the southwest, higher in the northeast), and suggested that some of the variation can be accounted for by climate. Pooling from multiple datasets, they summarized documented survival rates across the range. The mean survival rate for 499 individuals at 16 sites was 0.67 (0.35 – 0.95); the wide variation, especially the lower rates, are likely an artifact of some of the sampled populations having a declining trend. A long-term study of a Georgian Bay population in Ontario with a stable population trend, the average adult survival was 0.65 ( $v=0.038$ ) over 30 years. Faust and colleagues (2011, p. 7-8) built a baseline model for a hypothetical healthy population (growing population,  $\lambda = 1.03$ ). Adult non-postpartum survival was 0.7 ( $v=0.10$ ), postpartum survival was 0.6 ( $v=0.016$ ), and probability of breeding was 0.5 ( $v=0.003$ ).

Fundamentally, to sustain a population, recruitment needs to be greater than mortality. Reinert (1981, pp. 393-394) reported that for population sustainability, reproductive females must breed every other year with an average litter size of 8 individuals, and with at least 50% of those surviving in the first year and 65% surviving the second year. Further, at least 70% of non-reproducing adults must survive and 60% of reproducing adults must survive every year. Middleton and Chu (2004, pp. 40-41) found low or no risk of extirpation for most demographic modeling parameters. Even conservative numbers such as first reproduction at age 6, litter size of 10, and 70% neonate mortality resulted in EMR populations that grow away from extirpation slowly. They also found that losing a single individual from a population can increase the risk of extirpation to over 20% and losing greater than 20% of a single age class per year can raise the extirpation risk of that population to 30% or more. Bailey (2010, p. 20) found that for an EMR population in Michigan the mortality of juvenile snakes influenced population persistence more than the mortality of adult snakes.

The seemingly discordant results are not surprising as it is likely that there isn't a single set of demographic rates needed to ensure a stable to growing population. Instead, trade-offs among reproductive, survival, and mortality rates can be made to produce healthy population dynamics. However, there is a limit to each demographic rate and likely a range of rates that occurs in the wild.

#### Suitable Habitat Conditions

Self-sustaining populations require sufficient quality and quantity of habitat to support a DGP robust population and intact ecological processes that maintain suitable conditions over time. Specifically, populations require high quality microhabitats (active season and winter habitat) of adequate quantity (patch size) and connectivity among these microhabitats. To ensure suitable habitat conditions into the future, habitat areas must also have intact hydrological and ecological processes that create and maintain suitable habitat conditions.

#### *High Quality Microhabitats*

Within a habitat patch, EMR require suitable hibernation, foraging, and gestation habitat. Suitable hibernation sites are critical to protect the entire population as the snakes spend four to six months in a physiologically susceptible condition when dormant. Suitable gestation sites are needed for thermoregulation by females to ensure proper development of their young (Reinert and Kodrich 1982,

p. 170). Suitable foraging areas are needed to provide sufficient prey for growth of young and maintenance of adults.

The actual size of a habitat patch that is needed to support an EMR population depends upon the quality and distance between microhabitats. Furthermore, the home range sizes of individuals vary by life stage and reproductive condition. relatively unknown but may be related to the size of a snake's home range which varies with ecologically significant factors such as the availability of resources (basking sites, hibernation sites, gestation sites, prey, mates) as well as biological factors such as the life stage of the snake (adult, juvenile, neonate) and the reproductive condition (gravid or non-gravid) (Johnson *et al.* 2000, p. 5). For example, neonate EMRs have smaller movement patterns than adults and shorter distances traveled (Jellen and Kowalski 2007, p. 996-998). For these reasons, movement and resource use varies within and among populations. Indeed, studies have documented home range sizes for EMRs varies from 1.0 ha (2.5 ac) to 136 ha (336.1 ac) (Durbian *et al.* 2008, p. 757; Moore and Gillingham 2006, p. 748; Reinert and Kodrich 1982, p. 166 and 167; Marshall *et al.* 2006, p. 148; Bissell 2006, p. 31; Mauger and Wilson 1999, p. 115; King 1999, p. 80; Dreslik 2005, p. 124; Johnson 2000, p. 4-9; DeGregorio *et al.* 2011, p.74; Weatherhead and Prior 1992, p. 451). An important caveat, to bear in mind when drawing conclusions from home range studies, is that many of these are based on small study sites, and thus, careful consideration is needed as the sustainability of these populations is in doubt (Durbian *et al.* 2008, p. 757). Although no minimum size at which a DGP robust population can be supported is known, generally speaking, the larger the patch size the larger the population that can be supported.

The actual size of a patch may not be as important as whether the particular habitat can provide the snakes the ability to persist by meeting all of their life requisites. However, Johnson *et al.* (2000, pp. 20-21) suggested that the value of suitable habitat tends to decrease as it becomes more isolated. For example, if EMR habitat becomes subdivided or fragmented, the two or more habitat patches (which together equal the size of a larger patch) may not be as valuable as the larger patch (Johnson *et al.* 2000, p. 21). Reasons for this include that the small fragment size is inadequate to support EMRs and certain individual snakes can have substantial spatial demands. Home range size is likely a function of the relative proximity of the requisite habitat components, and reflect hibernation and reproductive strategies rather than food preference (Reinert and Kodrich 1982, p. 170; Durbian *et al.* 2008, p. 757).

#### *Connectivity Among Microhabitats*

Connectivity between the active season (summer) habitat and inactive season (winter) habitat is crucial for the population sustainability. Similarly, when temperatures shift the snakes must have the unimpeded ability to either access or retreat to a particular (summer or winter) habitat. EMR populations require access to connected habitats where they are not subject to road mortality while traversing from one habitat to the other.

#### *Intact Hydrology and Ecological Processes*

Lastly, the EMR occupies an early successional habitat and depends upon particular hibernation conditions. Suitable microhabitats are early successional stage habitats that are needed for thermoregulation and for foraging. Thus, the ecological processes that create and maintain an early successional stage need to remain functional. The EMR also requires specific over-wintering conditions; namely, it needs a stable water table. Thus, for a population to be sustained over time, the ecological processes that maintain and create suitable habitat conditions must remain intact and functional.

b. Manageable Threats

Even if a population is DGP robust population and it has enduring habitat conditions, external threats can exceed the intrinsic capability of the species to recover. Stressors can lead to unsustainable mortality by directly killing individuals or indirectly through habitat modification. Thus, for populations to be self-sustainable, they must be free of threat or at least exposed to only manageable threats.

c. Summary

To be self-sustaining over time, the population must be large enough to avoid genetic, demographic, and environmental consequences. Self-sustaining populations also require high quality summer and winter habitat, connectivity among these habitats, intact ecological processes that maintain suitable summer and winter habitats. Lastly, self-sustaining populations must have only manageable stressors acting upon them.

Table 2. 5. Population-level Ecology: the requisites for a self-sustaining population.

	Requisites LT viability	Description
Demography	Population growth	Need a stochastic $\lambda \geq 1$ , which is a function of: survivorship, recruitment, and population structure
	Population size	Minimum N required; results vary, but N should exceed 35 to 50 adult females
Habitat	High quality microhabitats	Hibernacula, basking, gestation, foraging sites - see individual needs for description of high quality habitat
	Sufficient quantity of microhabitats	No minimum size is known but the larger the patch size, the larger the population size that can be supported: Homeranges 1-135 ha been documented but many of these populations are considered imperiled.
	Connectivity among microhabitats	Seasonal habitats must be connected via safe corridors
	Intact hydrology and ecological processes	To maintain early successional habitats, normal water level fluctuations should be maintained and processes that create the microhabitats

2.4.3 Species Level Ecology

In this section, we describe the ecology needs at the species-level using the conservation principles of resiliency, representation, and redundancy (3Rs). These needs describe what is required for the species to maintain self-sustaining populations over a biologically meaningful timeframe, i.e., needed for viability. The species level ecological needs are discussed below and summarized in Table 2.6.

a. Resiliency

Resiliency is the ability of a species to respond to and recover from disturbances and perturbations. Disturbances include stochastic events such as fire, flooding, and storms; perturbations include normal year-to-year variation in rainfall and temperatures. In short, resiliency is the ability to recover from bad years and stochastic events. To be resilient, a species must have healthy populations; that is, populations that are able to sustain themselves through good and bad years. We detailed the requirements for a self-sustaining population above, under the *Population Level Ecology* section.

For many species, resiliency also requires connectivity among populations for gene flow and demographic rescue. As explained further under the *Evolutionary Processes: Gene Flow, Genetic Drift, and Natural Selection* section below, low connectivity among populations is likely the natural state for EMR populations (Gibbs *et al.* 1997, p. 1130) with limited dispersal likely a long-standing biological trait of this taxon (Chiucchi and Gibbs 2010, p. 5354). Thus, connectivity among populations may not have been a historical requisite for species level resiliency.

#### b. Representation

Representation is the ability of a species to adapt to long-term changes in its environment; in short, it is the evolutionary potential of a species. To preserve a species' evolutionary potential the adaptive diversity (ecological diversity and genetic diversity) and the evolutionary processes that drive evolutionary change need to be maintained. Generally speaking, this means to maintain populations across longitudinal and latitudinal gradients, and to maintain the historical patterns of gene flow, natural selection, and genetic drift.

##### *Adaptive Diversity: Ecological Diversity*

Populations of a species adapted to different environments represent ecological diversity. To maintain ecological diversity, the habitat niches where the populations still survive need to be conserved. To identify the breadth of ecological diversity of EMR, we looked for differences in hibernacula use, venom, and age of maturity, across its range.

The EMR has been reported to hibernate in crayfish burrows, rocky crevices, rodent holes, hummocks, old stumps, rotten logs, and tree and shrub root systems, mole tunnels, or any excavation which reaches the water table (Wright 1941, p. 660; Reinert 1977, p. 63; Johnson 1995, p. 36; Mauger and Wilson 1999, p. 117; Harvey and Weatherhead 2006, p. 214; Johnson and Leopold 1998, p. 84; Sage 2005, p. 35). It appears that crayfish burrows are the typical choice, but EMR will also use other refugia as described above when these are lacking (Reinert 1977, p. 63; Jellen 2005). It appears that the variation in hibernacula structures is not an adaptive trait of the EMR, but rather an artifact of what suitable structures are available.

In venomous snakes, considerable within-species geographic variation in the chemical composition of the venom reflects natural selection for venom best able to act on the vital systems of, and subdue, locally available prey (Barlow *et al.* 2009: 2443; Calvete *et al.* 2009, p. 1736-1743; Daltry *et al.* 1996, pp. 537-539). Snake venom proteins are one of the most studied types of animal toxins (Gibbs *et al.* 2013, p. 1). With only three recognized species, the genus *Sistrurus* represents a small but widely distributed clade with diets including a diverse range of prey across their composite ranges (Holycross and Mackessy 2002, pp. 454-464; Weatherhead *et al.* 2009, p. 693). Consequently, snakes in the genus *Sistrurus* (EMR and pygmy rattlesnakes) have been the subject of several studies into venom variation. These studies have demonstrated that genetic mechanisms play a key role in venom composition and the genes that regulate venom coding evolve rapidly (Gibbs and Rossiter 2008, pp. 152, 153, 159-160; Gibbs *et al.* 2009, pp. 118-199; Gibbs *et al.* 2013, pp. 4-10) and toxic effects of venom are prey specific and influenced by the local availability of prey species (Sanz *et al.* 2006, pp. 2104-2107; Gibbs and Mackessy 2009, pp. 673-675; Gibbs and Chiucchi 2011, p. 384). However, there is considerable population-level variation in individual venom proteins and the evolutionary cause remains unclear (Gibbs and Chiucchi, 2011, p. 383-397). While population-specific differences in prey may explain

differences in venom composition, the authors were unable to test this explanation (Gibbs and Chiucci 2011, p. 393), and in fact an earlier study showed similar diets between two populations with substantially different venom compositions (Weatherhead *et al.* 2009, pp. 693-697). Thus, differences in venom do not appear concordant with geographic patterns. Within population diversity, however, warrants further study.

We also evaluated data on temperature, precipitation, and period of maturation (early maturing or late maturing) across the EMR range to determine if there were any discernible correlations. Early maturing populations occur in the more southerly locations most likely due to overall warmer temperatures earlier and for a longer time period each season. Late maturing populations occur in the more northerly locations most likely due to overall cooler temperatures for longer time period each season. It is unknown whether age to maturity is a heritable adaptation or simply a plastic response to local environmental conditions.

#### *Adaptive Diversity: Genetic Diversity*

In an effort to identify areas of genetic diversity across the range of the EMR, Gibbs *et al.* (1997, p. 1130) analyzed microsatellite DNA data from five EMR populations (two from the U.S. and three from Ontario, Canada). Their results indicated a high degree of genetic separation among EMR populations, and thus low levels of gene flow (Gibbs *et al.* 1997, p. 1130). As microsatellite data are sensitive to population level variation (Gibbs *et al.* 1997, pp. 1130-1131), these results suggest that gene flow among the studied populations is very restricted. Furthermore, Gibbs and colleagues contend that this isolation among the populations is not a new phenomenon but rather these populations have likely been genetically isolated from each other for some time (Gibbs *et al.* 1997, p. 1130).

Chiucci and Gibbs (2010, pp. 5346 , 5350) corroborate results from Gibbs *et al.* (1997) through their study of genotyped 388 adult EMRs (19 locations) at 19 polymorphic microsatellite loci. Their findings report low rates of gene flow on both contemporary and historical time scales (Chiucci and Gibbs 2010, p. 5354). The authors also report that geographically discrete EMR populations exhibit high levels of genetic differentiation and structure at both rangewide and regional spatial scales (Chiucci and Gibbs 2010, p. 5354).

Andre (2003, p. 24) results comport with findings of the studies above. Andre (2003, pp. 22-24) investigated the genetic differences in three sub-populations of the EMR in Illinois, and compares the results to two other populations Indiana and Ohio. Genotypes were determined at three microsatellite DNA loci for 77 EMRs from the Illinois population, 10 from Indiana population, and 11 from Ohio population. Results indicate the presence of unique alleles among the Illinois, Indiana and Ohio populations (Andre 2003, p.25), as well as, within-population genetic differentiation among the three Illinois sub-populations.

Most recently, Ray *et al.* (2013, p. 109) used mitochondrial DNA (mtDNA) sequence data from 179 EMRs at 34 locations, and identified 18 haplotypes. Three geographical areas emerged as genetically distinct units: 1) the western unit consists of populations in Iowa, Wisconsin, and Illinois, 2) the central unit consists of populations in Indiana, southern and central Michigan, Ohio, and far southwestern Ontario, and 3) the eastern unit which consists of populations in New York, Pennsylvania, northern Michigan, and the remaining portions of Ontario.

Although seemingly in conflict, USFWS geneticists (Greg Moyer and Meredith Barton) clarified that the Ray *et al.* results do not conflict with those of Gibbs *et al.* and Chiucchi and Gibbs. The differences are due to differences in methodology, and more specifically, to differences in scale. Ray *et al.* used methods that discern differences at a broader, regional scale, while Gibbs, Chiucchi, and Andre used methods that are able to discern genetic differentiation at a finer, local scale. Thus, a reasonable conclusion from the composite of genetic studies is that there are broad-scaled genetic differences across the range of EMR and within these broad units, there is genetic diversity among populations comprising the broad units.

Adaptive diversity is one of two aspects to consider when devising strategies for conserving the representation of a species. The second is maintaining the evolutionary processes that drive evolutionary change.

#### *Evolutionary Processes: Gene Flow, Genetic Drift, and Natural Selection*

The primary drivers of evolutionary change are gene flow, natural selection and genetic drift (Crandall *et al.*, 2000, p. 291). Gene flow is expressed through the physical transfer of genes or alleles from one population to another through immigration and breeding. As previously described, the EMR exhibits low levels of gene flow among populations. The results from Chiucchi and Gibbs (2010, p. 5354), for example, strongly imply that the high levels of population structure currently observed in EMR populations are not a consequence of living in a highly fragmented habitat, but rather it is consequence of limited dispersal. Andre (2003, p. 23) also concludes that because genetic differences were detected in snakes only a few kilometers apart, inbreeding is most likely taking place within these populations due to limited natal dispersal as well as limited adult movement. This pattern of limited movement may eventually lead to a loss of genetic variability in these populations should they become too small in number, however, at present, such loss of genetic variability (and resulting inbreeding depression) is not evident from genetic analyses at these sites (Gibbs *et al.* 1997; Andre 2003, p. 25). Chiucchi and Gibbs (2010, p. 5354) further hypothesize that EMR populations have a long history of living in relatively small isolated populations that have been dominated by the effects of genetic drift and these populations may have a history of small to moderate levels of inbreeding but the effects of inbreeding depression are minimized due to the repeated exposure of deleterious recessive alleles. A more recent study by Gibbs and Chiucchi (2012, p. 1138) which sampled 14 populations of EMRs from Ohio, Pennsylvania, and Canada, found evidence for inbreeding in only one population suggesting that inbreeding is generally rare. Although three of the sample populations had large populations and inbreeding would be expected to occur infrequently, all others had smaller populations in which inbreeding effects would be expected (Gibbs and Chiucchi 2012, p. 1139). In light of this information, it does not appear that gene flow is a primary driver in evolutionary change.

In summary, the available data indicates high levels of genetic differentiation (genetic diversity) among populations despite the close geographic proximity of some populations. It also appears that low levels of gene flow and genetic isolation may be the natural state for EMR populations and that these populations have been isolated for longer than the several hundred years that define the period of European settlement (Gibbs *et al.* 1997, p. 1130), and thus, gene flow among populations is not a driver in evolutionary change. The sources of the genetic differences detected are likely due to natural selection (source of among regional differences) and genetic drift (source of within regional differences). Given the analysis above, conserving the full breadth representation of EMR means maintaining populations across and within the three genetically distinct regions.

### c. Redundancy

Redundancy is the ability of a species to withstand catastrophic events; it is about spreading the risk. Spreading the risk entails having multiple populations with a sufficient spatial spread to minimize the overlap of populations and catastrophic events. Moreover, as it is important to preserve the breadth of adaptive diversity, redundancy means having a sufficient number and distribution of populations across the three broad genetically diverse regions and within each of these regions.

### d. Summary

The species level needs for long-term viability requires having multiple (redundancy), self-sustaining populations (resiliency) distributed across longitudinal and latitudinal gradients (representation) to maintain the ecological and genetic diversity of EMR. Information to date suggests that EMRs are highly genetically structured across their range and among closely located populations.

#### 2.4.4 Analysis units

Given the data available, the breadth of adaptive diversity can likely be captured by wide distribution of populations within the three broad regions identified by Ray *et al.*, (2013). Thus, to facilitate our analyses, we used these three geographic units as our analysis units to evaluate past, current, and future representation of EMR. The three units are: 1) the western analysis unit (WAU) consists of populations in Minnesota, Missouri, Iowa, Wisconsin, and Illinois, 2) the central analysis unit (CAU) consists of populations in Indiana, southern and central Michigan, Ohio, and far southwestern Ontario, and 3) the eastern analysis unit (EAU) consists of populations in New York, Pennsylvania, northern Michigan, and the remaining portions of Ontario (Figure 2.1).

Figure 2. 1. Shaded areas showing the three genetically distinct subunits (Western, Central, and Eastern) adapted from Ray *et al.* 2013.

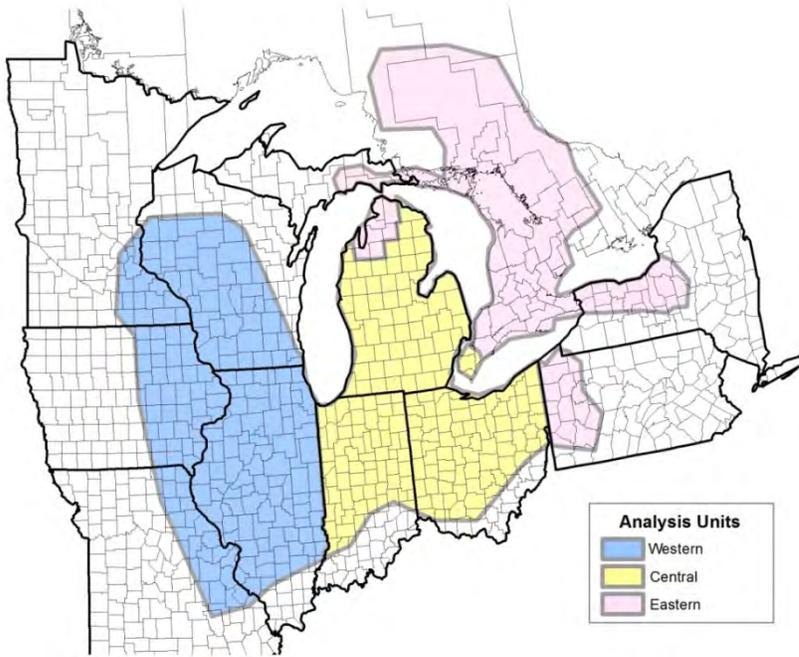


Table 2.6. Species-level ecology: Requisites for long-term viability (ability to maintain self-sustaining populations over biologically meaningful timeframe).

<b>3 Rs</b>	<b>Requisites long-term viability</b>	<b>Decsription</b>
<b>Resiliency</b> <i>(populations able to withstand stochastic events)</i>	Self-sustaining populations across the range	Self-sustaining populations are demographically, genetically, and physiologically robust, have sufficient quantity of high quality habitat, and are free of, or have manageable, threats.
<b>Representation</b> <i>(genetic &amp; ecological diversity to maintain adaptive potential)</i>	Maintain adaptive diversity of the species	Pops maintained across longitudinal and latitudinal gradients to maintain the ecological & genetic diversity of EMR
	Maintain evolutionary processes	Maintain evolutionary drivers--gene flow, natural selection, genetic drift--to mimic historical patterns
<b>Redundancy</b> <i>(number &amp; distribution of populations to withstand catastrophic events)</i>	Sufficient distribution of populations to spread risk	Sufficient distribution to guard against catastrophic events wiping out portions of the species adaptive diversity and the species as a whole, i.e., to reduce covariance among populations; spread out geographically but also ecologically (different ecological settings).
	Sufficient number of self-sustaining populations	Adequate number of self-sustaining populations to buffer against catastrophic losses of adaptive diversity

## Chapter 3: Threats and Conservation Actions

In this chapter, we review the negative and beneficial factors affecting the historical, current, and future conditions of the EMR. Factors may be of natural and human-made origin. Those factors that have a negative impact on EMR individuals are referred to as risk factors (also as stressors or threats); factors that have a beneficial effect are referred to as conservation actions. We begin with describing generally the most prominent risk factors and conservation actions, and end with a summary of our analysis of the impact that these factors are having on EMR populations and the projected impact into the future.

### 3.1 Risk Factors

#### 3.1.1 Habitat Loss & Modification

The loss of habitat was historically, and continues to be, the primary threat affecting this species (Szymanski 1998, p. 15; Johnson 2000, p. 16; Lipps 2008, p. 8). Habitat loss includes direct habitat destruction of native land types (*e.g.*, grassland, swamp, fen, bog, wet prairie, sedge meadow, marshland, peatland, floodplain forest, coniferous forest) due to conversion to agricultural land, development, and infrastructure associated with development (roads, bridges). For example, Illinois was once covered by an estimated 200 million acres of prairie or grassland habitat, but now only 0.01% of these habitat types remain (Ellis 2010, pp. 2-3). Because EMR habitat varies seasonally and also varies over its range, the destruction of even a portion of a population's habitat (*e.g.*, hibernacula or gestational sites) causes a negative effect to individual snakes, thus reducing the numbers of individuals in a population and, in turn, reducing the viability of that population.

Habitat loss also includes modification due to fragmentation, succession, exotic species invasion, dam construction, fire suppression, water level manipulation, and other incompatible habitat modifications (Jellen 2005, p. 33). These non-development related habitat losses continue even in publicly held areas protected from development.

#### a. Vegetative Succession

Vegetative succession is a major contributor to EMR habitat loss (Johnson and Breisch 1993, pp. 50-53; Reinert and Buskar 1992, pp. 56-58). The open vegetative structure, typical of EMR habitat, provides the desirable thermoregulatory areas, increases prey densities by enhancing the growth of sedges and grasses, and provides retreat sites. Degradation of EMR habitat typically happens through woody vegetation encroachment or the introduction of non-native plant species. These events alter the structure of the habitat and make it unsuitable for EMR by reducing and eventually eliminating thermoregulatory and retreat areas. Fire suppression has led to the widespread loss of open canopy habitats through succession (Kingsbury 2002, p. 37). Alteration in habitat structure and quality can also affect EMR by reducing the forage for its prey base (Kingsbury 2002, p. 37). Succession decreases the available habitat to EMR prey, reducing the densities of EMR prey and thereby reducing the subsequent numbers of EMR that the site can support. Jellen (2005, p. 30) postulates that the current greatest threat facing the continued existence of EMR in Pennsylvania is woody vegetation encroachment (Jellen 2005, p. 30). Of all possible threats considered by the Faust model, succession was the one most commonly cited (81%) by experts familiar with extant EMR sites.

## b. Habitat Fragmentation

Roads, bridges, and other structures constructed in EMR habitat fragment the snakes' habitat and impact the EMR both through direct mortality as snakes are killed trying to cross these structures (Shepard *et al.* 2008b, p. 6), as well as indirectly through the loss of access to habitat components necessary for the survival of the snakes.

These barriers also prevent the snakes from intermixing with other individuals or populations (Kingsbury 2002, p. 39). The EMR exhibits high site fidelity to hibernation, gestation, and foraging areas and this high site fidelity may override the aversion to crossing roads (Rouse *et al.* 2011, p. 454 and 455), but crossing the roads will increase an individual snake's chance of being killed by vehicular traffic. Roads contribute to individual snake mortality through vehicular crushing (Bailey *et al.* 2011, p. 170; Lipps 2008, p. 9) and an increased exposure to predation by the snake being more out in the open if basking on a road or trying to cross it.

Barriers often hinder normal snake movement, creating smaller patches of habitat and restricting movement across those habitats (Forman *et al.* 2003, p. 137). Particularly roads, but any type of infrastructure can divide a previously continuous population into smaller fragments causing insufficient numbers of individual snakes to successfully cross and thus maintain necessary population level dynamics jeopardizing the long-term viability of that population (Shine *et al.* 2004, p. 10; Andrews and Gibbons 2005, p. 780). This is especially serious when the snake species is rare or has a high conservation interest (Forman *et al.* 2003, p. 119) such as in the case of EMR. Kingsbury (2002, p. 39) theorizes that single lane roads with little traffic might act as a minor barrier, but roads with more lanes and higher traffic may form a complete barrier.

## c. Hydrologic Alternation/Water Fluctuation

Extreme fluctuations in the water table may negatively affect body condition the following active season, cause early emergence, or direct mortality (Harvey and Weatherhead 2006, p.71; Smith 2009, p. vii, 33, 38-39). Changes in water levels under certain circumstances can cause mortality to individual EMR, such as during hibernation (Johnson *et al.* 2000, p. 26; Kingsbury 2002, p. 38) when the snakes are underwater hibernating. The water in the hibernacula protects the snakes from dehydration and freezing, thus dropping the levels in the winter leaves the snakes vulnerable to both (Kingsbury 2002, p. 38; Moore and Gillingham 2006, p. 750; Smith 2009, p. 5). Because individual EMR often return to the same hibernacula year after year, dropping water levels in EMR hibernacula could potentially decimate an entire population if the majority of individuals in that population hibernate in the same area. At least one concentrated area of crayfish burrows used regularly by massasaugas as a hibernaculum was abandoned by both crayfishes and snakes alike during consecutive years of extreme drought in northeast Illinois (M. Redmer, U.S. Fish and Wildlife Service, Chicago Field Office, personal observation 2005-2006). Other massasauga species experts have shared similar observations, though this threat is incompletely documented in the literature. Raising water levels may not be as detrimental to the snakes provided that the duration is not more than a few days or perhaps weeks, and that all suitable habitats in an area are not simultaneously submerged (Kingsbury 2002, p. 39). Other water level changes such as those that may occur downstream from hydroelectric dams during periods of peak electric demands, are harmful to EMR (Johnson *et al.* 2000, p. 26) because the snakes are not adapted to this variation (Kingsbury 2002, p. 39). Individual EMR located directly downstream from the dam would be impacted immediately while those individuals located many miles downstream would experience the

effects later even in areas where water levels change as little as a few inches (Kingsbury 2002, p. 39). Kingsbury (2002, p. 3) explains how the construction of a hydroelectric dam in the Huron-Manistee National Forest flooded extensive areas of EMR habitat, and how pulsed discharges led to repeated oscillations of wetland water levels destroying or substantially degrading downstream habitat. Affects to the individual snakes from this practice would include stress, inadequate basking areas, inadequate hibernation areas, inadequate foraging areas, temporary displacement, or permanent displacement during the active season. All of these affects to individuals can lead to mortality and may lead to the mortality of an entire population depending on the level of impact of water level manipulation (Seigel *et al.* 1998, pp. 128-130).

### 3.1.2 Management Practices

#### a. Post-emergent Prescribed Fire

An effective tool for controlling vegetative succession is the use of prescribed fire which kills or temporarily sets back the growth of woody vegetation, retards the growth of undesirable species, and stimulates the response of prairie species (Johnson *et al.* 2000, p. 25). However, direct mortality of EMR can result from exposure to fire if burning occurs when the snakes are out of their hibernacula (post-emergent fire) (Cross 2009, pp. 18, 19, 24; Cross *et al.* 2015, p. 355; Dreslik 2005 p. 180; Dreslik *et al.* 2011, p. 22). In Missouri, Durbian (2006, p. 329) observed the mortality of 8 western massasauga rattlesnakes on a 16.6 ha (41 ac) prairie after a burn conducted on April 18, 2000.

#### b. Mowing

Mowing and herbicide application are two additional strategies, often used in conjunction with prescribed burning, to control woody vegetation and invasive species encroachment. Durbian and Lenhoff (2004, p. 21) postulated that pre-burn mowing may potentially reduce fire related mortality of EMRs and other snake species by negatively modifying the occupied habitat forcing the snakes to leave the area or seek refuge below ground. Durbian (2006, p. 329) subsequently found that pre-burn mowing at a height of 20 cm (8 in) resulted in the direct mortality of 3 of 7 radio-marked EMRs. After the burn, 3 unmarked individuals in the burned area were killed by the fire itself indicating that a number of EMRs did not leave the site after mowing as hypothesized by Durbian and Lenhoff (2004, p. 21). Durbian (2006, p. 333) concluded that mowing prior to burning results in additional direct mortality to EMRs beyond that incurred by prescribed burning and advises to conduct burns while EMRs are hibernating until methods that effectively reduce mortality while achieving the treatment objectives are identified (Durbian 2006, p. 333).

### 3.1.3 Road Mortality

Although viperids like EMR are more sedentary than other snakes, they also move more slowly which increases the probability of being killed while crossing roads (Andrews and Gibbons 2005, p. 779). Shepard *et al.* (2008b, p. 6) found that although individual EMR avoid crossing roads, when they do attempt to cross roads they have a high probability of being killed. In one EMR population in Illinois, Shepard *et al.* (2008a, pp. 353-354) found a seasonal pattern in road mortality, with the highest mortality occurring in August and September. In August, adult males comprised the majority of dead snakes found which coincided with the peak of the mating season (Shepard *et al.* 2008a, p. 353). Movement is a major determinant in the success of males finding mates (Jellen *et al.* 2007, p. 455). Within the Illinois population, after birthing in August, the road mortality of juveniles and neonates was

highest in August and September which coincided with the timing of neonate dispersal and foraging behavior (Shepard *et al.* 2008a, p. 354).

Although limited, the patterns that point to vulnerability of EMR to road mortality are supported by other research on snakes and reptiles. Studying six species of snakes (n=652) from two families (Viperidae and Colubridae) in Europe, Bonnet *et al.* (1999, pp. 40-44) found that the snakes' vulnerability to being run over by a vehicle is highest when they travel outside of their normal home range with the highest mortality occurring in adult males during the mating season, neonates or hatchlings immediately after birth or hatching; and adult females on egg laying migrations (Bonnet *et al.* 1999, pp. 39, 47). Using imitation reptiles (turtle and snake) Ashley *et al.* (2007, p. 140) found that reptiles were struck by vehicles at a greater rate than would be expected by chance suggesting some drivers (~2.7%) intentionally targeted reptiles on the road. Indeed, several drivers were observed to speed up and position their vehicles to hit the reptiles (Ashley *et al.* 2007, p. 142). Although 2.7% is a small percentage of the total drivers, given moderate to heavy traffic volumes, intentional hits could be a significant component of the total road mortality experienced by a particular reptile population (Ashley *et al.* 2007, p. 141). Ashley *et al.* (2007, p. 141) theorized that the actual rate of vehicle-reptile collisions would be considerably greater than their study documented because they only accounted for those collisions that were intentional and did not account for accidental collisions. Ashley *et al.* (2007, p. 141) further suggested that if roads experience high traffic volumes (such as during holidays) that coincide with high reptile activity (mate searching or neonate dispersal) then reptile populations would experience unusually high road mortality. Shepard *et al.* (2008a, p. 357) found that the intensity of traffic on a road did not affect the level of road mortality, but the quality of the habitat through which the segment ran was a factor. Langen *et al.* (2009, p. 110) found that the configuration of wetlands within 100 m (328 ft) of the road is a valid indicator of a reptile and amphibian road mortality hot spot and if wetlands occur on both sides of the road, the road kill will be further exacerbated. Reptiles and amphibians are underrepresented in the road kill mortality record because they are often overlooked due to their small size (especially when run over) and because they are easily and readily scavenged after being road killed (Stoner 1936, p. 279).

#### 3.1.4 Persecution

Kingsbury (2002, p. 36) contends that compared to other venomous snakes EMR is not threatening and that to protect themselves, individual snakes rely on their cryptic coloration to keep them hidden as opposed to aggressive attack. Kingsbury (2002, p. 36) also observed that individuals typically do not rattle, strike, or even move unless they are physically disturbed. Bielema (1973, p. 28) also considers the EMR non-aggressive. Despite the behavioral traits of EMR, most people are fearful of snakes (Ohman and Mineka 2003, p. 7), whether the snake is venomous or not, making snakes the most unpopular wildlife species. Humans often perceive snakes as vile and loathsome creatures (Alves *et al.* 2014, p. 2). Because of this fear and negative perception of snakes, many people have a low interest in snakes or their conservation and consequently large numbers are deliberately killed (Whitaker and Shine 2000, p. 121; Alves *et al.* 2014, p. 2). Human-snake encounters frequently result in the death of the snake (Whitaker and Shine 2000, pp. 125-126).

Whitaker and Shine (2000, p. 122) examined the sources of mortality in two species of large elapid snakes (eastern brown snakes, *Pseudonaja textilis*, and common black snakes, *Pseudechis porphyriacus*) in Australia and found that of the 58 brown snakes tracked, eight were killed, with humans responsible for half of those killings. Adult snakes in the surrounding area were also killed by motor vehicles (39 brown snakes and 36 blacksnakes) with some of these deaths undoubtedly unintentional, but 29

appeared to be deliberate killing as they were found dead close to the road side, suggesting the drivers had swerved in order to hit the snakes (Whitaker and Shine 2000, p. 124). The authors also found that half of the large elapids seen by people were approached, and one in three was killed regardless of the species and despite the fact that all snakes in Australia are legally protected under New South Wales legislation (Whitaker and Shine 2000, p. 126). Their data also revealed that humans are 100 times more likely to attack a snake than a snake was to attack a human (Whitaker and Shine 2000, p. 126).

Evidence of human persecution of EMR is documented in the literature. On public land owned by the state of Pennsylvania, in what is thought to be the largest remaining contiguous EMR habitat in that state, Jellen (2005, p. 9) describes documented direct mortality of EMR by game hunters. There have been observations of EMR intentional mortality and collection in other areas of Pennsylvania as well (Jellen 2005, p. 11). At an EMR site in Michigan, Bailey *et al.* (2011, p. 171) includes human persecution as a direct source of this population's mortality. In northeast Illinois, a population of EMR near Chicago was first subjected to early settlers deliberately turning hogs loose into the habitats occupied by the snakes (Bushey 1985, p. 10). Later, as the habitat occupied by the same population was fragmented for road construction and other development, snakes were also deliberately killed, in some cases up to 100 were killed annually (Bushey 1985, pp. 10-11). Finally this population was for a number of years subject to organized rattlesnake hunts (Wheeling Historical Society Website Accessed March 26, 2015). Evidence of ongoing poaching has been documented by trail cameras placed in massasauga habitat in Ohio, as has publicizing massasauga locations on internet sites which could increase risk of additional poaching (Doug Wynn, EMR Expert, Russell's Point, Ohio, personal communication to FWS June 1, 2015).

### 3.1.5 Collection

The American Pet Products Manufacturers Association reports that > 3.9 million U.S. households contain one or more reptiles or amphibians and that the retail trade in live reptiles, amphibians and related products is worth a minimum of two billion dollars annually (Franke and Telecky 2001, p. vii). Exotic pet ownership is an increasingly important part of the wildlife trade business (Courchamp *et al.* 2006, p. 2408) with local amphibian and reptile populations being exploited for use in the pet trade (Schlaepfer *et al.* 2005, p. 263). Declaring a species as threatened or endangered by a conservation organization provides official proof that the species is rare making it more desirable, more valuable, and increasing the likelihood of exploitation (Courchamp *et al.* 2006, p. 2408). Internationally, species that have a Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) status are significantly more expensive than species with no CITES status, presumed to be a consequence of being considered more valuable due to their rarity (Courchamp *et al.* 2006, p. 2408). In the summer, gravid female EMR are more likely to be found than any other classes of snake because their focus is on selecting suitable basking areas often in open habitats (Kingsbury 2002, p. 36). The propensity for EMR to be in more open areas makes them more vulnerable to collection as they are easier to find (Kingsbury 2002, p. 36).

Wildlife crimes are committed by people with knowledge of the laws with the intent to break them, usually for possession or monetary gain (USFWS 2010). On the illegal black market, spotted turtles sell for \$125 to \$400 each, timber rattlesnakes sell for \$175 to \$300 each, a bog turtle can bring as much as \$2,000, and an EMR is considered priceless (USFWS 2010). In 2006, an undercover investigation was initiated by the Division of Law Enforcement of the New York State Department of Environmental Conservation, working closely with the New York State Police, the Pennsylvania Fish and Boat Commission, the United States Fish and Wildlife Service, the United States Immigration and Customs Enforcement, Environment Canada, and the Ontario Ministry of Natural Resources. This undercover

investigation found a very active illegal market for native species at shows and on internet markets which originated outside of New York but influenced the illegal collection within New York (USFWS 2010). Investigators identified large reptile shows as sites of illegal transactions of amphibians and reptiles (USFWS 2010). One of their more noteworthy discoveries was that an entire population (N=35) of EMR was poached from a site in Canada and smuggled into the United States for subsequent sale (USFWS 2010; Gary Allen, personal communication 09/29/2014). The investigators successfully traded native venomous snakes with the smuggler to recover the wild population of EMR and then used DNA testing to determine the snakes' original location for subsequent return (USFWS 2010).

Collection and killing are two factors that result from the close proximity of a snake population to urbanization. Collection and killing are forms of loss to both the snake individually and to the population; the animals are removed from the population either through death (persecution) or disappearance (collection).

Collection of EMR for research is also known to result in the death of individual snakes and has been noted as early as 1936 (Crawford 1936, p. 38). Crawford (p.38) describes taking 45 EMRs one summer from his study population (Ohio) and did not mention whether any of these were subsequently released and further mentioned that persistent collecting had reduced the number of these snakes and he had to find a new area to supply him with new specimens. Crawford collected 44 EMR in 1931 at one particular site, but in 1935 with intense searching, only 8 were found (Crawford 1936, p. 39). Crawford (1936, p. 47) explains how the time of emergence, when the snakes remain close to their hibernacula for a period of time until the weather gets consistently warm, is the most opportune time for collection, as the snakes are more easily seen and concentrated in an area, and thus more vulnerable to collection and subsequent mortality. For their study in determining the diet of EMR, Keenlyne and Beer (1973, p. 382) collected, killed, and autopsied 323 EMR specimens from a 9-mile area in Wisconsin. Depending on the number of snakes in that population, which is not known, this effort may have extirpated that population.

Radio telemetry studies have been used extensively in studying the EMR with the assumption that the transmitter has no effect on the study animal (Lentini *et al.* 2011, p. 107). Studies from other taxa suggest that carrying a transmitter may have adverse effects, including energy costs (Gessamen *et al.* 1991, pp. 551-553), increased stress (Suedkamp Wells *et al.* 2003, p. 809), impaired mobility, changes in behavior, nutrition, and reproductive success (Reed *et al.* 2005, pp.96-98), and even death (Burger *et al.* 1991, p. 696). Field studies that use implanted transmitters in snakes can give invalid results if the implant influences the behavior or survival of the subject (Lentini *et al.* 2011, p. 122). A commonly used implantation technique involves inserting the antenna subcutaneously (under the skin) (Reinert and Cundall 1982, pp. 703-704). This technique can lead to antenna erosion and skin penetration, which can predispose the snake to local infections which become systemic and eventually lead to death (Lentini *et al.* 2011, p. 122). Lentini *et al.* (2011, p. 122) removed 7 intracoelomic (inside the body cavity) transmitters previously implanted (using the method described by Reinert and Cundall (1982, pp.703-704) in adult EMR by researchers in Ontario and found over 85% of the snakes had a severe inflammatory or infectious reaction associated with the surgically implanted transmitter (Lentini *et al.* 2011, p. 122). Because of this finding, Lentini *et al.* (2011, p. 122) went on to examine the effects of intracoelomic implants in a group of 24 EMRs finding that inflammation and infection occurred despite careful surgical procedures and advanced veterinary care (antibiotics) in 33% (4 out of 12) of transmitter-equipped snakes (Lentini *et al.* 2011, p. 107-123). Evidence of the long-term effects that implanted radio transmitters may have on the EMR comes from information on 3 adult EMR implanted for tracking in a 1999 and 2000 field study (Lentini *et al.* 2011, p. 122). After the 1999-2000 study, these

3 snakes were transferred to a captive environment and subsequently necropsied upon their deaths in 2005 and 2006 with results showing that their cause of death was attributed to severe bacterial infection and systemic inflammation associated with the implanted transmitter (Lentini *et al.* 2011, p. 122). These snakes died 6 to 7 years after implantation of transmitters because of implantation of the transmitters. There have been many studies of the spatial ecology and behavior of EMR that have used implanted radio transmitters. Thus potential health risks associated with future use of this technique to research EMR should be carefully considered with the benefits of such studies, as well as the possibility that implanted transmitters or infections resulting from their implantation may affect the behavior of the individual snakes studied.

### 3.1.6 Predation

Predation of EMRs is a natural event that results in the loss of individuals from a population. However, there is little data on mortality rates, and most of what is known about predation is based on only a few direct observations. One such observation was made by Sage (2005, p. 67) who found snake mortality in the early spring at hibernation burrows that was attributed to striped skunk (*Mephitis mephitis*) predation. It appeared that an animal had dug out the mouths of hibernacula and that the snakes were pulled out of their hibernacula and eaten (Sage 2005, p. 67). Mauger and Wilson (1999, p. 119) also found evidence of EMR mortalities when they found two snakes just outside of a hibernacula that had been predated. For one population in Illinois, Wylie *et al.* (2011, p. 6) found high neonate predation rates before their first overwintering. For a species already experiencing stress from numerous sources (*e.g.*, habitat loss, habitat fragmentation, collection, persecution, disease), like EMR, normal effects from natural predation may contribute to population losses.

### 3.1.7 Disease

In reptiles, stress can lead to immune suppression which can further lead to an increase in mortality from infectious disease or parasitic infections (Allender 2006, p. 31). Many factors can induce stress in reptiles such as habitat loss or alteration (degradation), environmental contamination (Allender 2006, p. 31; Wylie *et al.* 2014, p. 1), or human harassment (research, collection, persecution). Habitat loss or alteration can result in the inability of the current habitat to meet all of the snake's physiological requirements (Allender 2006, p. 31). The prevalence of disease within a snake population can cause differences in the health and body condition of snakes in the population. For example, the inadequate condition of female snakes in a population due to disease can also affect reproductive rates of that population and eventually negatively affect the viability of that population (Allender 2006, p. 31).

Fungal infections in reptiles are described as opportunistic pathogens, infecting animals with depressed immune systems (Wylie *et al.* 2014, p. 2). Snake fungal disease (SFD) is an emerging disease found in certain populations of wild snakes in the eastern and Midwestern United States (Sleeman 2013, p. 1). The National Wildlife Health Center reports that although fungal infections have been reported in wild snakes prior to 2006, the number of reported cases of snakes with fungal dermatitis has increased recently (Sleeman 2013, p. 1). The fungus, *Ophidiomyces ophiodiicola* is associated with SFD, but other fungi are also isolated in affected snakes (Sleeman 2013, p.1). And although The National Wildlife Health Center has confirmed fungal dermatitis in wild snakes from 9 states (Illinois, Florida, Massachusetts, Minnesota, New Jersey, New York, Ohio, Tennessee, and Wisconsin), it is suspected that SFD is more widespread in the United States than is currently known (Sleeman 2013, p. 1). The EMR is one species of snake that has only recently been diagnosed with SFD (Sleeman 2013, p. 1; Allender *et al.* 2011, p. 2383). Clinical signs of SFD include scabs or crusty scales, subcutaneous nodules, premature

separation of the outermost layer of the skin (or abnormal molting), opaque cloudiness of the eyes (not associated with molting), localized thickening or crusting of the skin, skin ulcers, swelling of the face, and nodules in the deeper tissues of the head (Sleeman 2013, p. 2). In 2008, three EMRs from one Illinois population died from SFD (Wylie *et al.* 2014, p. 1). These snakes had severe facial swelling and disfigurement and died within 3 weeks of discovery (Allender *et al.* 2011, p. 2383). Later examination determined that at least 32 individuals with SFD were found between 2000 and 2014 (Drs. Matthew Allender and Michael Dreslik, University of Illinois at Urbana-Champaign, personal communication March 30-31, 2015). All known individual EMRs with SFD that have been brought into captivity in Illinois died and five additional EMR from three sites in Michigan tested positive for SFD in 2013 and 2014 (Jackson 2015, pp. 1). In Grayling, MI in 2015, two adult male EMRs were confirmed to be infected with *Ophidiomyces* fungus using qPCR on a tissue biopsy after they were found with deformed scales and mandibles in the field (Tetzlaff *et al.* 2015a). The two individuals died within two weeks despite receiving treatment, a notably higher mortality rate for EMR than previously observed in the Timber rattlesnake (*Crotalus horridus*) (Tetzlaff *et al.* 2015a). The authors note that this was the furthest north that SFD and *Ophidiomyces* has been detected (Tetzlaff *et al.* 2015a). However, our understanding of this disease is still in its early stages. Snakes that have tested positive have developed external symptoms and experienced high mortality, but conversely other snakes that have tested positive remained asymptomatic at some sites. Researchers are unsure if the primary vector is direct contact with an infected individual or contact with fungal spores in the soil (Wylie *et al.* 2014, p. 9). Wylie *et al.* (2014, pp. 6-7) found that *O. ophiodiicola* can be transferred between mothers and neonates.

*Ophidiomyces ophiodiicola* may also indirectly cause mortality through behavioral changes. When infected with a pathogen, many reptile species induce fever through basking (Burns *et al.* 1996, p. 138) which is thought to increase their immune response. Increased basking time leads to an increase the amount of time spent exposed to visual predators and thus increased mortality (Webb and Whiting 2005, p. 521). In addition, increased overall basking time increases the snake's metabolic rate resulting in faster utilization of resources but reducing the amount of time the individual can spend on foraging. Increased metabolic rate with less time spent foraging can lead to declines in body condition or death (Wylie *et al.* 2014, p. 8).

Several EMRs in southern Illinois have also tested seropositive for the pathogen ophidian paramyxovirus (Allender 2006, p. 42; Allender *et al.* 2006, p. 111), meaning that these individuals have been exposed to the virus (Allender *et al.* 2006, p. 112). The authors' noted that it is possible that the seropositive snakes were exposed to a different species of paramyxovirus that is similar enough to cause a positive reaction in the ophidian paramyxovirus assay (Allender *et al.* 2006, p. 112). Allender *et al.* (2008, p. 361) described the inherent limitations of hemagglutination inhibition assays in sensitivity and specificity of different ophidian paramyxovirus, which can result in excessive numbers of false-positive samples which they indicate could be attributed to different lab analyses.

The recent documentation of disease in EMR populations is an additional threat to the species. Although disease in any snake population could be considered a normal life event, a fatal disease outbreak in a population with low population size may compromise the viability and integrity of the entire population. Although we provide documentation of disease in EMR populations in Illinois only, we suspect that rangewide other EMR populations may also exhibit effects from SFD and/or ophidian paramyxovirus.

## 3.2 Assessing the Impact of Risk Factors on EMR Populations

### 3.2.1 Assessing Magnitude of Impact of Stressors on EMR Populations

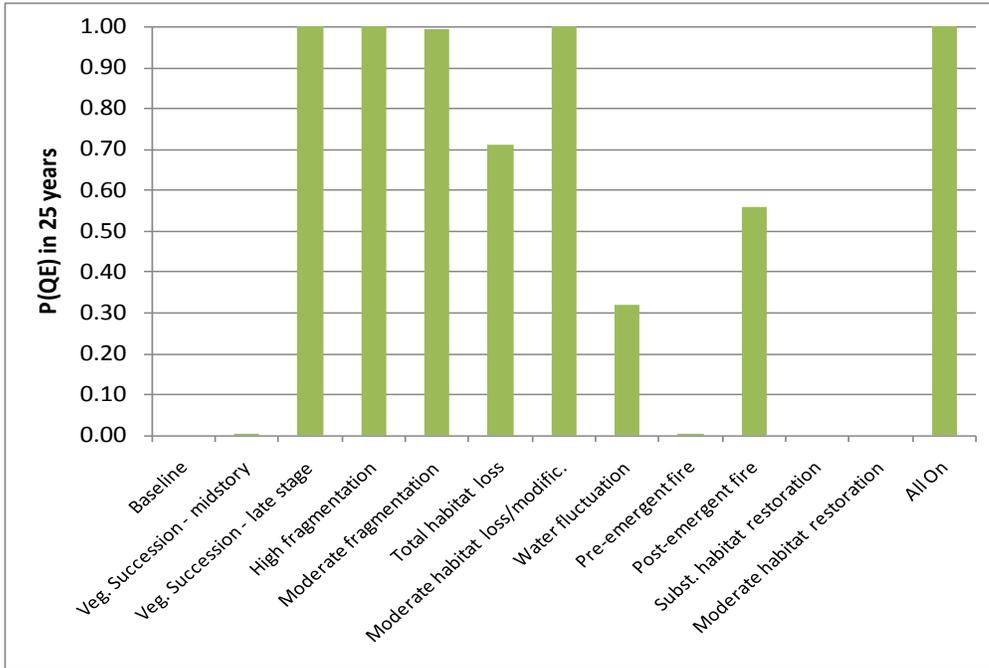
In general, any stressor that kills individuals, especially adult females, will cause stress on a population. The factors described above have the potential to direct or indirectly kill individuals. Because many of the remaining populations are assumed to be small in terms of the numbers of individuals, loss of individuals could be detrimental to the long-term existence of entire populations and eventually, to the species. A modeling exercise by Middleton and Chu (2004, pp. 40-41), using demographic modeling parameters from Canadian EMR populations and from experts' estimates, found low or no risk of extirpation for most demographic modeling parameters, but conversely, found that incidental mortality, from road kill, direct human persecution, or other sources, can have a strong effect on EMR populations. Even a single individual lost in this way per year can increase the risk of extirpation from zero to over 20%.

We relied on the work of Faust and colleagues (2011, pp. 13,14) to quantify the magnitude of impact each factor is having on populations. They identified, through expert input, the most prominent and likely factors acting upon populations across the range as: mid-story vegetative succession, late stage vegetative succession, habitat fragmentation (at high and moderate levels), total habitat loss, moderate habitat loss or modification, water fluctuation, pre-emergent fire, and post-emergent fire. They elicited information from experts regarding the magnitude of impact and the pervasiveness of each factor. To assess the magnitude of impact from the various factors, they identified whether each factor would have a positive or negative impact, degree of impact, and the frequency (or likelihood) of each factor occurring (*e.g.*, if total habitat loss occurs, the population is extirpated for the that model iteration, but the frequency sets the chance of this occurring in any given model year). They used these magnitudes of impacts to model the probability of quasi-extirpation ( $p(QE)$ ,  $N \leq 25$  females) within 25 years on a hypothetically healthy EMR population (Figure 3.1). Five of the risk factors resulted in a high  $p(QE)$  in 25 years ( $p(QE) \geq 0.7$ ): late-stage vegetative succession, high habitat fragmentation, moderate habitat fragmentation, total habitat loss, and moderate habitat loss or modification. Two factors resulted in a moderate impact ( $p(QE)=0.3-0.7$ ): water fluctuation and post-emergent fire. The remaining 2 risk factors resulted in low impact ( $p(QE) \leq 0.3$ ).

Based on current information, we believe the risk factors analyzed by Faust *et al.* (2011, pp. 12,13) are ongoing and are expected to continue with a similar magnitude of impact into the future, unless they are addressed by increased implementation of conservation actions (see Conservation Actions section below). In addition to the nine factors mentioned above, several other risk factors were also noted to occur at some sites, but information about the magnitude or impact of these factors was lacking. These factors include invasive species, ATV use, logging, road mortality, and persecution.

Disease is a new, or increasingly prevalent, and potentially catastrophic threat to EMR populations, as discussed above. Because we lack information on the potential future emergence or spread of disease among EMR populations, we did not model this threat in forecasting future conditions of EMR populations. This remains a gap in our analyses that should be addressed as new information is obtained, especially as we anticipate that the level of disease risk is likely to increase in the future.

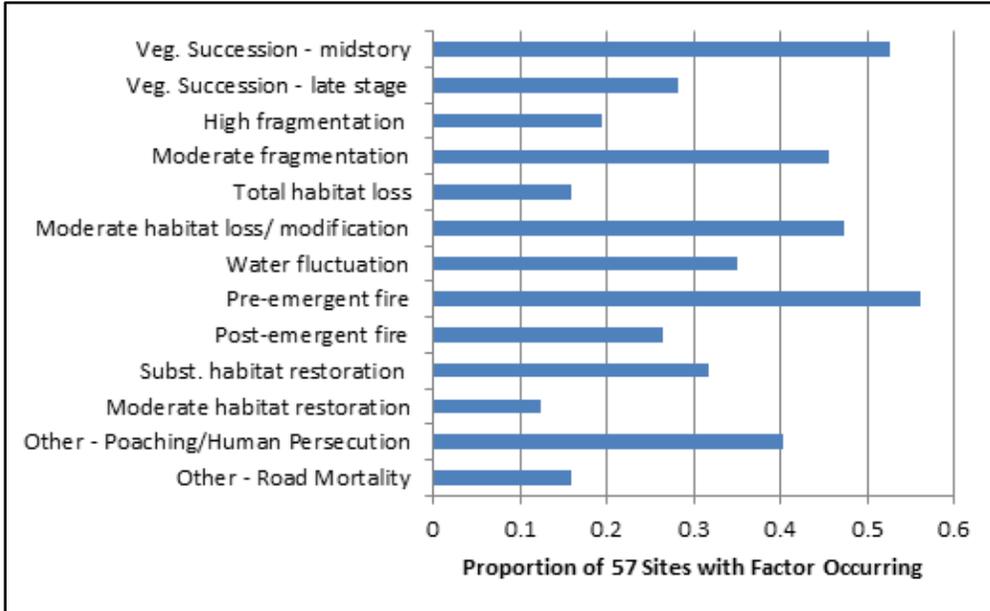
Figure 3.1. Probability of quasi-extirpation, (p(QE)), in 25 years for a population experiencing each individual factor and all modeled factors combined (taken from Faust *et al.*, 2011, p.15). The p(QE) is a function of the magnitude of impact and the frequency of the factor occurring.



### 3.2.3 Assessing the Pervasiveness of Factors Affecting the Species

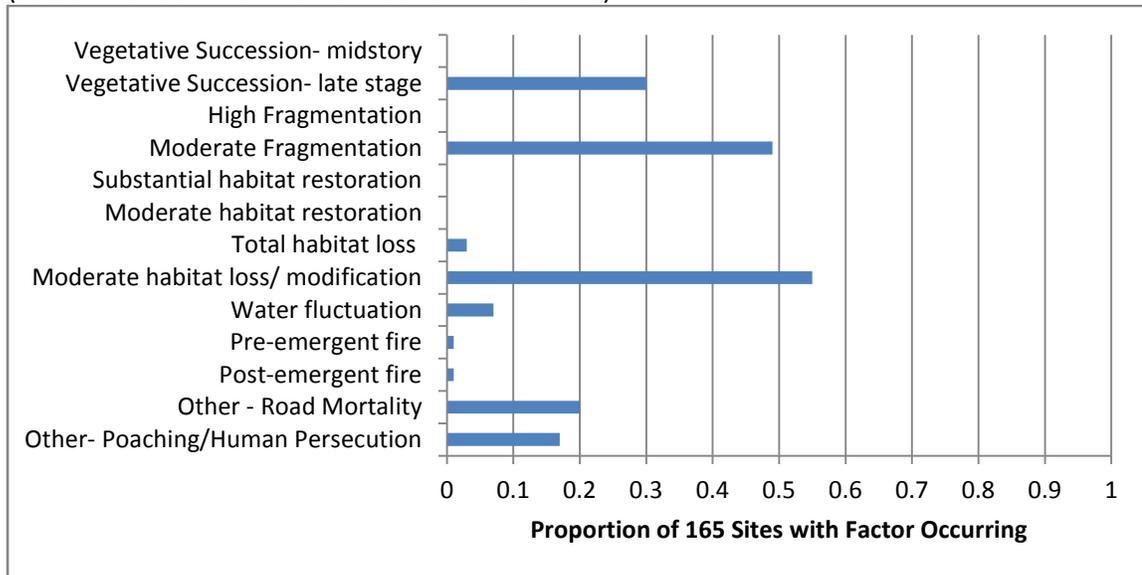
Faust *et al.* (2011, p. 59) also examined how pervasive these risk factors were across the EMR range. They included 2 additional risk factors (in addition to the 9 discussed above) in their pervasiveness analysis, collection/persecution and road mortality, however these factors were not included in the model. They found that 95% of the populations (all but 3 populations) have at least one factor currently affecting the site. Of those populations with at least one factor, all but one have multiple factors acting at the site, and the majority (77%) of populations are experiencing 3 or more factors in combination. In general, of the risk factors considered, vegetative succession (including both midstory and late-stage) is the most common factor, occurring at 81% (N=46) of the populations; fire management (including both pre-emergent and post-emergent) and fragmentation (including both high and moderate) are the second most commonly occurring factors, each occurring at 65% (N=37) of the populations. Some form of habitat loss or modification (either total loss or moderate loss/modification) is occurring at 50% (N=28) of the populations; 16% (N=9) of these populations are at risk of total habitat loss. Among the other factors considered, water fluctuation, collection/persecution, and road mortality occur at 35%, 40%, and 16% of the populations, respectively. Figure 3.2 illustrates the proportion of the 57 modeled populations from the Faust model with each individual risk factor; some populations are experiencing multiple factors (*e.g.*, a population might be experiencing both midstory and late stage succession, thus the proportion of populations experiencing some form of vegetative succession is lower than adding the proportions for midstory and late stage succession together).

Figure 3.2. Proportion of populations with factors occurring at the 57 modeled populations (Faust 2015, pers. comm.).



In addition to the information garnered by Faust and colleagues, we received site-specific information from the States and Ontario regarding threats for 165 of 388 presumed extant EMR populations. For these populations, habitat loss or modification is the most common factor, occurring at 55% of populations, with 3% of populations at risk of total habitat loss (Figure 3.3). Fragmentation is the second most common factor (49% of sites), and succession is the third most common factor (31% of sites). Among the other factors, road mortality occurs at 20%, collection/persecution at 17%, water fluctuation at 7%, and pre- or post-emergent fire at less than 1% of the populations.

Figure 3.3. Proportion of EMR populations with factors occurring at 165 sites across the range (Factor data obtained from the States and Ontario).



Applying the magnitudes of impact reported by Faust *et al.*, (2011), 187 (84%) of the 222 populations (57 modeled + 165 additional populations) are impacted by at least one high magnitude factor; 104 (63%) of these populations have more than one high magnitude factors occurring. Two populations are impacted by a moderate magnitude factor (1%), and 19 populations (9%) are impacted by only low level factors. None of the risk factors analyzed by the Faust model are impacting 15 (5%) populations; however, 14 of those populations are experiencing at least one other non-modeled risk factor (*e.g.*, collection, disease), based on the information received from the States (Table 3.1). Thus, for those 14 populations, we are unable to assess the magnitude of impacts.

If we break down these factors by analysis unit, in the WAU, 95% of the populations are impacted by high magnitude factors; the other population is impacted by low magnitude factors. In the CAU, 82% are affected by high magnitude factors, 0.5% by moderate magnitude factors, 9% by low magnitude factors, and 8.5% by none of the modeled factors (or other factors). In the EAU, 88% populations are experiencing high magnitude factors, 4% moderate level factors, and 8% low magnitude factors. In the EAU, no sites are free from effects due to risk factors.

Table 3.1. The number and degree of impact of the modeled risk factors affecting EMR populations (includes 57 populations from the Faust model and the 165 presumed extant populations for which we received information on site-specific threats from the States). WAU=western analysis unit, CAU=central analysis unit, EAU=eastern analysis unit. High=  $p(QE) \geq 0.7$ ; Moderate=  $p(QE)=0.3-0.7$ ; Low=  $p(QE) \leq 0.3$ .

	Level of Magnitude of Impact			
	High	Moderate	Low	None
WAU	20	0	1	0
CAU	144	1	16	15
EAU	22	1	2	0

### 3.3 Beneficial Factors (Conservation Actions)

The EMR is State-listed as endangered in IA, IL, IN, NY, OH, PA and WI, and is listed as endangered in Ontario. In Michigan the EMR is listed as “special concern,” and a Director of Natural Resources Order (No. DFI-166.98), prohibits take of EMR except by permit.

#### 3.3.1 Conservation Actions & Plans

Of the 267 sites with information indicating the presence of an extant EMR population, 164 occur on public land in the United States or in Canada that is considered protected from development. Seven more of the 267 populations occur on private land with protection from development. Of the remaining 93 populations, 79 exist on private land without protection or on a mixed private and public land area with limited protection, and 17 exist on land with unknown ownership or protection status. As discussed in the threats section above, however, non-development habitat losses from fragmentation, succession, exotic species invasion, dam construction, water level manipulation, and other incompatible habitat modifications continue even on lands protected from development.

Many EMR populations occur on lands managed by State conservation agencies and many of these sites, as well as a few privately owned sites, receive ongoing conservation management to address some of the threats previously discussed. We asked the States which EMR populations occurred on sites that had management plans specifically addressing the conservation needs of the snake. States reported conservation plans for 25 sites out of 267 sites with extant EMR populations: 13 sites in Ontario (out of 40), 2 sites in Iowa (out of 6), 1 site in Indiana (out of 19), 2 sites in New York (out of 2), 2 Sites in Ohio (out of 17), 2 sites in Pennsylvania (out of 5), and 2 sites in Illinois (out of 4). Habitat restoration is occurring at 5 of these populations and at 19 additional populations without conservation plans in place (Table 3.2).

### 3.3.2 Candidate Conservation Agreements

Sections 2 and 10 of the Endangered Species Act (ESA) of 1973, as amended, allow the U.S. Fish and Wildlife Service to enter into Candidate Conservation Agreements with Assurances (CCAA). Section 2 of the Act states that encouraging interested parties, through Federal financial assistance and a system of incentives, to develop and maintain conservation programs is a key to safeguarding the Nation's heritage in fish, wildlife, and plants. Section 10(a)(1)(A) of the ESA authorizes the issuance of permits to enhance the survival of a listed species.

In developing CCAs, we must ensure they meet the recovery standard. As protected properties harbor the core populations, the recovery standard on these sites is to ensure the long-term protection of the population. The specific conservation measures required at each protected property to achieve this standard will vary, however, in general the need is to protect requisite habitat components (foraging, gestational, and over-wintering habitats) in sufficient quantities and qualities and to implement management practices that promote EMR welfare. A CCA will meet the recovery standard if it ensures EMR persistence by committing to: (1) implement management that promotes the well-being of EMRs; (2) restore or enhance habitat to support a viable population; (3) protect such habitat; and (4) reduce threats and minimize take especially of the adult age classes.

The Wisconsin Department of Natural Resources (DNR) developed a CCA for one EMR population in Wisconsin. Through the agreement, existing savanna habitat on state land, especially important to gravid (pregnant) females, will be managed to maintain and expand open canopy habitat, restore additional savanna habitat, and enhance connectivity between habitat areas. Periodic prescribed burns in EMR occupied or previously occupied habitats will be conducted to control or reverse the loss of open canopy caused by natural succession and to restore connectivity between occupied habitats. Burning of occupied habitat will occur only during the EMR's non-active period (during hibernation). In addition, surveys will be conducted for the snake every 3 years to evaluate the species population status.

In Ohio, a CCA for a State Nature Preserve population addresses threats from habitat loss from the prevalence of late-stage successional vegetation, the threat of fire both pre and post emergence of EMR individuals, and limited connectivity through habitat fragmentation. The Ohio Division of Natural Areas and Preserves will take steps to ensure the continued persistence of the EMRs by adopting an adaptive management approach. Management activities will be highly dependent upon weather conditions. Winter activities may include mowing and/or burning when appropriate. During the summer, management will be limited to manual cut and treat methods and basal bark or foliar spraying. Protection measures that will address threats from persecution and collection will include heightened security during the spring and fall, installation of a gate or similar device at the entrance to the site to deter poachers, and regulating public use of the preserve by restricting access.

The State of Michigan drafted a proposed CCAA for agency-managed lands in the State. Some of these proposed management actions are ongoing, but we do not have site-specific data on these management actions to include them in our analysis. Nonetheless, we believe the management actions proposed will address some of the threats (*e.g.*, habitat loss, vegetative succession) impacting populations. Two CCAAs were drafted for sites in Iowa. Neither was finalized but some of the proposed management actions are being implemented at both sites.

We assessed the degree to which conservation actions included in approved and proposed CCAAs address the known threats affecting the EMR populations. In the WAU, one site has a CCAA that will address threats at that site to the extent that we would expect an increase in the viability of that population. Additionally, in the CAU, one site has a CCAA that will address threats at that site to the extent that we would expect an increase in the viability of that population, and additional sites are potentially covered by a CCAA that would address some of the threats. There are no sites in the EAU that are covered by a CCAA.

Table 3-2. Summary of Factor Analysis. Site-specific data on factors affecting EMR populations within a given state (or province) and analysis unit. Data are collected from Faust et al. (2011) and State/Ontario reported data. When unknown threat severity was assigned to moderate category. Threats not addressed by pre-determined categories are included in the “Other Habitat Threats” column. States and populations of EMR with existing Candidate Conservation Agreements (CCAAs) were evaluated for the effectiveness of management practices relative to the threats reported for that state or population.

Analysis Unit	State	Veg. Succession: midstory   late	Habitat Loss: Moderate   Total	Fire: post emergence	Frag: moderate   high	Road mortality	Water fluctuation	Persecution	Other Habitat Threats	Habitat Restoration: substantial   moderate	Conservation Plan	Will CCAA Address?
WAU	IA	5   1	0   0	6	7   1	2	3	0		1   1	2	Unknown (still draft) at 2 sites
	IL	1   5	3   3	0	2   4	2	3	6		4   0	2	No
	WI	1   4	4   0	0	1   0	0	3	4	red canary grass	2   0	0	Yes, at 1 site
CAU	OH	4   11	6   6	3	3   2	5	7	6	incompatible land use, invasive species, hydrologic alteration	1   2	2	Yes, at 1 site
	IN	0   4	1   1	0	4   0	1	2	1		1   0	1	No CCAAs
									invasive species, logging, ATV/OHV use, incidental take from management activities	7   2	1	Unknown (still draft)
EAU	MI	0   0	1   0	0	0   0	1	0	1		0   0	0	No CCAAs
	ON	2   1	5   4	1	5   0	2	1	2		1   1	13	No CCAAs
	PA	3   4	6   2	2	5   2	0	5	7	mining	0   0	2	No CCAAs
	NY	1   1	0   0	0	0   0	1	0	0	disease, invasive species	0   0	2	No CCAAs

## Chapter 4: Past and Current Conditions

In this Chapter, we describe our analysis of the current condition of the EMR relative to historical conditions. Specifically, we report the population status of all known populations, and we characterize the current degree of resiliency, representation, and redundancy of EMR relative to historical conditions.

To assess population status of each known EMR population, we garnered information from the States and Ontario. We requested the following information: the name of the population, population status (extant, likely or known extirpated, or unknown), the date a snake was last observed, the date of the last survey, land ownership (i.e. public, private, both), whether the site had a conservation plan specifically for the EMR, whether there was any documented illegal collection or persecution, other threat information, and any additional comments regarding the population.

The criteria used to classify population status by the States and Ontario are not standardized across the range. For example, one State may deem a population extirpated if EMR has not been observed at the site within 20 years, while another State may assign unknown status to a population under the same scenario. Due to time constraints, we were unable to reconcile differences in methods, and thus we, although cognizant that differences in classification exist, used population status as reported to us.

To assess the resiliency, representation, and redundancy of EMR, we assumed populations with unknown status are extant, and collectively referred to these extant and unknown populations as “presumed extant” populations. As explained in Chapter 2, we delineated 3 broad geographic areas (analysis units) that are important for preserving EMR adaptive diversity. Our analyses, therefore, were conducted at both the rangewide and analysis unit scales.

### 4.1 Resiliency

Resiliency is necessary to ensure that EMR can withstand annual environmental variation such as fluctuation in rainfall, temperature, and prey availability, for example. For EMR to have sufficient resiliency it must have healthy (self-sustaining, resilient) populations. To assess the current resiliency of the EMR, we evaluated the health of all presumed extant populations across the range.

#### 4.1.1 Methods

We evaluated the health of EMR populations by assessing whether populations are self-sustaining. We defined self-sustaining as one that: 1) is demographically, genetically, and physiologically robust (DGP robust)<sup>2</sup>; 2) has suitable habitat conditions; and 3) is free of, or has manageable, threats acting upon it. More specifically, we defined self-sustaining population as one that has an adult female population size greater than 50 ( $N_f > 50$ ), has a positive population growth rate ( $\lambda \geq 1$ ), and has a high probability of persistence ( $p(P) > 0.90$ ) despite the stressors acting upon it.

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<sup>2</sup> Adapted from Redford *et al.* 2011

We used the information garnered by Faust *et al.* to assess the health of populations. Faust and colleagues developed a stochastic, matrix model to project population trend over time for 57 populations. Population-specific inputs included current adult female population size ( $N_0$ ) and factors (stressors and beneficial actions) acting upon the population. Model outputs included median ending adult female population size ( $N_F$ ), population growth rate ( $\lambda$ ), and probability of quasi-extirpation<sup>3</sup>. Quasi-extirpation was defined as the population threshold at which demographic and genetic stochasticity dominate the dynamics of the population, and it was set at 25 or fewer adult females. We used the Faust model values for  $N_0$ ,  $N_F$ , and  $\lambda$  directly in our analyses, and used the  $1-p(QE)$  to calculate the probability of persistence (i.e., a population with a high  $p(P)$ , is one with a low probability of becoming quasi-extirpated).

Information on population growth rates ( $\lambda$ ) and probability of persistence [ $p(P)$ ] estimates for current conditions, however, were not available<sup>4</sup>. Thus, to assess the current health of EMR populations, we relied solely upon initial population size ( $N_0$ ) to determine whether a population is currently DGP robust ( $N_0 > 50$ ). We also used the factor data (Chapter 3) gain insights into whether threats are also manageable, and therefore, the population could be self-sustaining.

Using these results, we then extrapolated to the remaining presumed extant populations by multiplying the proportion of modeled populations that were DGP robust by the total number of presumed extant populations to estimate the number of EMR populations that are currently DGP. Similarly, we estimated the number of EMR populations that manageable threats by multiplying the proportion of populations with no or only low magnitude of stressors acting upon it by the number of presumed extant populations.

All assumptions and caveats pertinent to the Faust model results also apply our extrapolation results (see Chapter 1 for a list of key assumptions underlying the Faust model). Additional assumptions we applied in our current conditions analyses include:

1. Populations with unknown status were assumed extant although many of these populations may indeed be extirpated.
2. Populations with  $N_0 > 50$  were considered DGP despite lacking information on population growth rate.
3. There were 21 populations in which it was uncertain of whether the population possesses early or late maturing dynamics. We used the results from the early maturing model for these 21 populations.
4. Stressors not considered in the Faust model have a low magnitude of impact, and hence, not considered in our analysis.

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<sup>3</sup> Faust *et al.* (2011, p. 1-66), as well as other authors cited within, used the terms extinct, extinction, and quasi-extinct(ion). Based on feedback from one reviewer and a decision maker, we replaced these terms with extirpated, extirpation, and quasi-extirpated(extirpation) to better clarify that are results are referring to losses of portions of the species' range (populations), not the species as a whole.

<sup>4</sup> Calculating population growth rates requires having information on past population sizes, which neither Faust nor us had. Estimating probability of persistence entails projecting the probability of quasi-extirpation given current population sizes but no future impacts from factors. Although technically feasible to compute, these scenarios were not run by Faust and colleagues, and thus, the  $p(P)$  estimates for current conditions could not be calculated.

- The 57 populations modeled by Faust *et al.*, and the 165 populations with threat data are representative of threats acting upon the remaining presumed extant populations.

We use several terms to characterize current conditions. For clarification, a list of these terms and their definitions are provided here:

Term	Definition
<b>Healthy population</b>	A population that is self-sustaining
<b>Self-sustaining</b>	A population that is DGP robust, has suitable habitat conditions, and manageable threats
<b>DGP</b>	A population with more than 50 adult females ( $N_F > 50$ ) and has a positive population growth rate ( $\lambda \geq 1$ )
<b>Presumed extant</b>	A population currently classified as extant or unknown status
<b>Quasi-extirpated</b>	A population with 25 or fewer adult females ( $N_F \leq 25$ )
<b>Modeled population</b>	One of the 57 populations in the Faust model
<b>Presumed QE</b>	A population assumed to be QE through extrapolation
<b>Presumed DGP</b>	A population assumed to be DGP through extrapolation

#### 4.1.2 Results

Rangewide - There are 581 known populations, of which 267 are extant, 163 are likely to be or known extirpated, and 121 have unknown status (Table 4.1). Faust and colleagues received sufficient information on 57 of the 388 populations presumed extant (267 extant + 121 unknown). Of these 57 populations, 25 (44%) are quasi-extirpated ( $N_F \leq 25$  females), 7-8 populations are considered small ( $N_F = 26-50$ ), 4-5 populations medium sized ( $N_F = 51-100$ ), 3 populations medium to large ( $N_F = 100-200$ ), and 7 large populations ( $N_F \geq 200$ ) (Figure 4.1) and 14 (25%) of the 57 populations currently meet the criterion for being DGP robust. Extrapolating from the 57 modeled populations to all presumed extant populations, 156 populations are presumed to be quasi-extirpated (Table 4.2) and 99 populations are presumed to be DGP robust (Table 4.3).

From the threats analysis summarized in Chapter 3, 20 (9%) of the 222 populations (57 populations from the Faust model, + additional 165 populations with data provided from the States) have manageable threats (no or only low magnitude stressors occurring). Assuming these 222 populations are a representative sample for all EMR populations, 35 of the 388 presumed extant populations are presumed to have manageable threats.

Table 4.1. Current population status of all historical populations of EMR based on data from the States and Ontario.

Current Rangewide Status		
Status	# of pops	%
Extant	267	46%
Likely extirpated	41	7%
Extirpated	152	26%
Unknown	121	21%

Figure 4.1. The distribution of the initial population class sizes of the modeled (n=57) EMR populations. QE=quasi-extirpated ( $N_0 \leq 25$ ), QE-S= quasi-extirpated to small ( $N_0=1-50$ ), S=small ( $N_0=26-50$ ), S-M=small to medium ( $N_0=26-100$ ), M=medium ( $N_0=51-100$ ), M-L=medium to large ( $N_0=101-200$ ), L=large ( $N_0=200+$ ), and UK= unknown size.

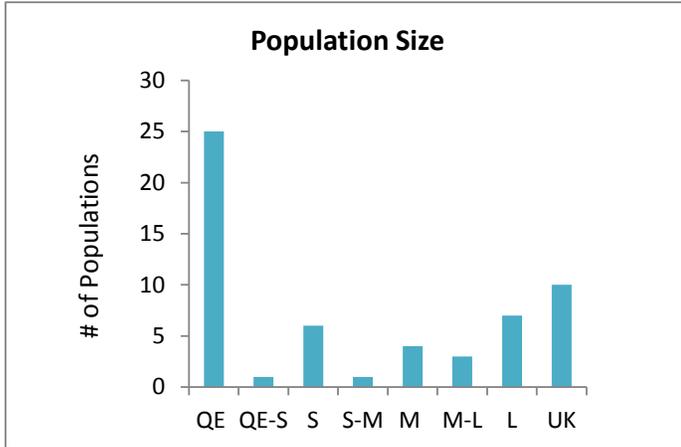


Table 4.2. The extrapolated number of populations rangewide and by analysis unit that are quasi-extirpated (QE). # Presumed Extant = the number of populations with extant and unknown population status. # Modeled = the number of populations with known population size. #QE = the number of populations quasi-extirpated,  $N_0 \leq 25$ . Prop. QE = the proportion of the # of modeled populations that are QE. # Extrapolated = the number of presumed extant populations that are presumed likely to be QE (# Presumed Extant \* Prop. QE).

	# Presumed Extant	# Modeled	# QE	Prop QE	# QE Extrapolated
WAU	21	18	10	0.56	12
CAU	294	27	11	0.41	120
EAU	73	12	4	0.33	24
Rangwide	388	57	25		156

Table 4.3. The extrapolated number of populations that are demographically, genetically, and physiologically (DGP) robust rangewide and by analysis unit. # Presumed Extant = the number of populations with extant and unknown status. # Modeled = the number of populations with known population size. #DGP = the number of populations with  $N_0 \geq 50$ . Prop. DGP = the proportion of # modeled populations that are DGP robust. # Extrapolated = the number of presumed extant populations that are likely to be DGP robust (# presumed extant \* Prop DGP).

	# Presumed Extant	# Modeled	# DGP	Prop DGP	# DGP Extrapolated
WAU	21	18	3	0.17	4
CAU	294	27	6	0.22	65
EAU	73	12	5	0.42	30
Rangwide	388	57	14		99

Analysis units - In the WAU, there are 65 known populations. Of these, 17 populations are extant, 44 are likely or known extirpated, and 4 have unknown status (Table 4.4). Of the 18 WAU populations modeled by Faust and colleagues, 10 (56%) are quasi-extirpated while 3 (17%) are DGP robust (Tables 4.2 and 4.3). Extrapolating to the 21 presumed extant populations in WAU, 12 populations are presumed quasi-extirpated, and 4 populations are presumed DGP robust (Tables 4.2 and 4.3).

We had habitat and threat conditions data for all 21 presumed extant populations. All but 1 presumed extant population in the WAU is impacted by high magnitude stressors; the 3 known DGP robust populations are also impacted by moderate or high magnitude stressors. At 2 populations of these populations and at 4 others, however, substantive habitat restoration is ongoing. At one site a CCAA is in place, which suggests that the plan adequately addresses all stressors facing the population. We currently do not know whether the habitat restoration at the other 5 sites is sufficient to mediate the stressors. Thus, 1 to 3 populations of the 3 DGP robust populations could have habitat and threats conditions suitable for maintaining a self-sustaining population.

In the CAU, there are 383 historical populations. Of these, 199 populations are extant, 89 are likely or known extirpated, and 95 have unknown status (Table 4.4). Of the 27 CAU modeled populations, 11 (41%) are quasi-extirpated while 6 (22%) are DGP robust. Extrapolating to the 294 presumed extant populations in the CAU, 120 are presumed to be quasi-extirpated and 65 are presumed to be DGP robust (Tables 4.2 and 4.3).

Extrapolating from the 176 CAU populations with threat data, 241 (82%) are impacted by high magnitude stressors, 1 (0.5%) by moderate magnitude stressors, 26 (9%) by low magnitude stressors, and 25 (8.5%) by no stressors. At 13 populations, habitat restoration is occurring, and extrapolating to all presumed current populations, 142 populations may have restoration occurring. At 1 additional site has a CCAA in place, which is attempting to address the stressors acting upon the population. Additional populations have conservation plans in place, so it likely that land managers at these sites may be actively working to mediate stressors. We do not have, however, site-specific information at these sites about what activities are ongoing and to what extent they mediate the stressors.

All 6 of the known DGP populations have at least 1 high magnitude factor occurring, but at 2 of these populations, habitat restoration is also occurring. We are uncertain as to whether the restoration has or will mediated the threats operating on these sites, but if so, 33% of the known DGP populations also have manageable threats. Extrapolating to all presumed DGP populations, 22 of the DGP populations have habitat and threats conditions suitable for supporting self-sustaining populations. In addition, as there are 13 populations with ongoing restoration occurring, the number of populations with suitable habitat and threat conditions could increase over time.

In the EAU, there are 133 historically known populations. Of these, 51 populations are extant, 60 are likely or known extirpated, and 22 have unknown status. Of the 12 EAU populations modeled, 4 (31%) are quasi-extirpated and 5 (42%) DGP robust. Extrapolating to 73 presumed extant populations, 24 populations are presumed to be quasi-extirpated and 30 are presumed to be DGP robust.

Extrapolating from the 25 EAU populations with threat data, 64 (88%) of the populations are impacted by high magnitude stressors, 3 (0.4%) by moderate magnitude stressors, and 6 (9%) by low magnitude stressors. At 3 populations, habitat restoration is occurring, and extrapolating to all presumed extant populations in EAU, 18 populations may have restoration occurring. There no CCAAs in place, but at 15 additional sites conservation plans are in place, so it likely that land managers at these sites may be

actively working to mediate stressors. We do not have site-specific information so we do not know whether or to what extent stressor management is occurring.

All 5 of the known DGP robust populations are impacted at least 1 high or moderate risk factor; at 1 of these populations and at 1 other population habitat restoration occurring. Extrapolating to all presumed DGP populations, 6 of the 30 DGP populations have manageable threats. As many sites are presumed to have habitat restoration occurring, the number of populations with suitable habitat and threat conditions could increase over time.

Table 4.4. The number of populations by status within each analysis unit and rangewide. WAU=western analysis unit, CAU = central analysis unit, EAU= eastern analysis unit.

Status	Analysis Unit		
	WAU	CAU	EAU
Extant	17	199	51
Likely extirpated	18	14	9
Extirpated	26	75	51
Unknown	4	95	22
Rangewide	65	383	133

#### 4.1.3 Summary

The resiliency of the EMR across its range and within each analysis unit has declined since historical conditions. The number of extant populations has declined rangewide by 33% and another 21% have unknown status. Of those populations presumed extant, 156 (40%) are presumed to be quasi-extirpated while 99 (26%) are presumed to be DGP robust. Of these presumed DGP robust populations, 29 are presumed to have threat conditions suitable for maintaining populations over time, and thus, self-sustaining. The greatest declines in resiliency occurred in the WAU, where only 21 populations are presumed extant and of these only 1 is presumed to be self-sustaining. Although to a lesser degree, loss of resiliency has occurred in the CAU and EAU, where 22 and 6 populations, respectively, are presumed to be self-sustaining.

#### **4.2 Representation**

To preserve a species' evolutionary potential, meaning its future ability to respond to ongoing and novel stressors, it is important to preserve the species' adaptive diversity and the processes that drive evolutionary change. To capture the full gradient of EMR adaptive diversity, it is necessary to maintain widely distributed populations in each of the 3 analysis units (See "Species Ecology" in Chapter 2 for a full discussion of adaptive diversity with regard to the EMR and how the analysis units were delineated). Thus, loss of populations within the units is likely to lead to loss of adaptive diversity. The degree of adaptive diversity loss is an irreducible uncertainty (cannot be directly measured), however, we presume that the more populations that are lost and the greater the range reduction, the greater the loss of EMR adaptive diversity.

To assess the degree of adaptive diversity persisting today, we evaluated changes in spatial extent from historical conditions to current conditions. Specifically, we assessed the change in spatial extent rangewide and within each analysis unit.

The assumptions applied in our analysis include:

1. The 3 analysis units and a wide distribution within each analysis unit capture the breadth adaptive diversity of EMR.
2. The loss of an analysis unit and loss of extent of occurrence within an analysis unit constitutes loss of adaptive diversity.

#### 4.2.1 Methods

We measured the historical and current spatial extent of EMR populations by drawing polygons (using ArcGIS) around clusters of counties with known EMR populations and summing the area of all polygons within an analysis unit and across units. The sum total within an analysis unit is referred to as the “extent of occurrence” or EoO. For historical conditions, all counties with at least one population documented were included in the EoO calculation. For current conditions, all counties with at least 1 presumed extant population were included in the calculation. Because of this, losses in EoO are not captured in counties that have at least 1 presumed extant population. Thus, the estimates of the area occupied by EMR are inflated, and the percent change from historical to current condition is underestimated. Our analysis, therefore, provides only a coarse comparison between historical and current conditions.

#### 4.2.2 Results

Currently, populations persist throughout the historical range of the EMR and within each analysis unit; however the spatial distribution within these units has changed. The spatial distribution has contracted northeasterly, which is evident both visually (Figure 4.2) and quantitatively (Table 4.5). Historically, the WAU represented 27% of the range of EMR, whereas currently, it represents 16% of the range (Table 4.5).

In addition to this northeasterly range contraction, the distribution within all analysis units has contracted as well. Within each of the analysis units, the EoO has declined. The WAU has experienced the greatest decline in extent of occurrence, with at least a 69% reduction in spatial extent (Table 4.5). The reductions in EoO in the CAU and EAU are not as severe, but yet still notable, with a 43% reduction in the CAU and 32% reduction in the EAU (Table 4.5). These within analysis unit losses have occurred throughout in all 3 units but a range contraction is most evident in the southern portions of the WAU and CAU (Figure 4.2).

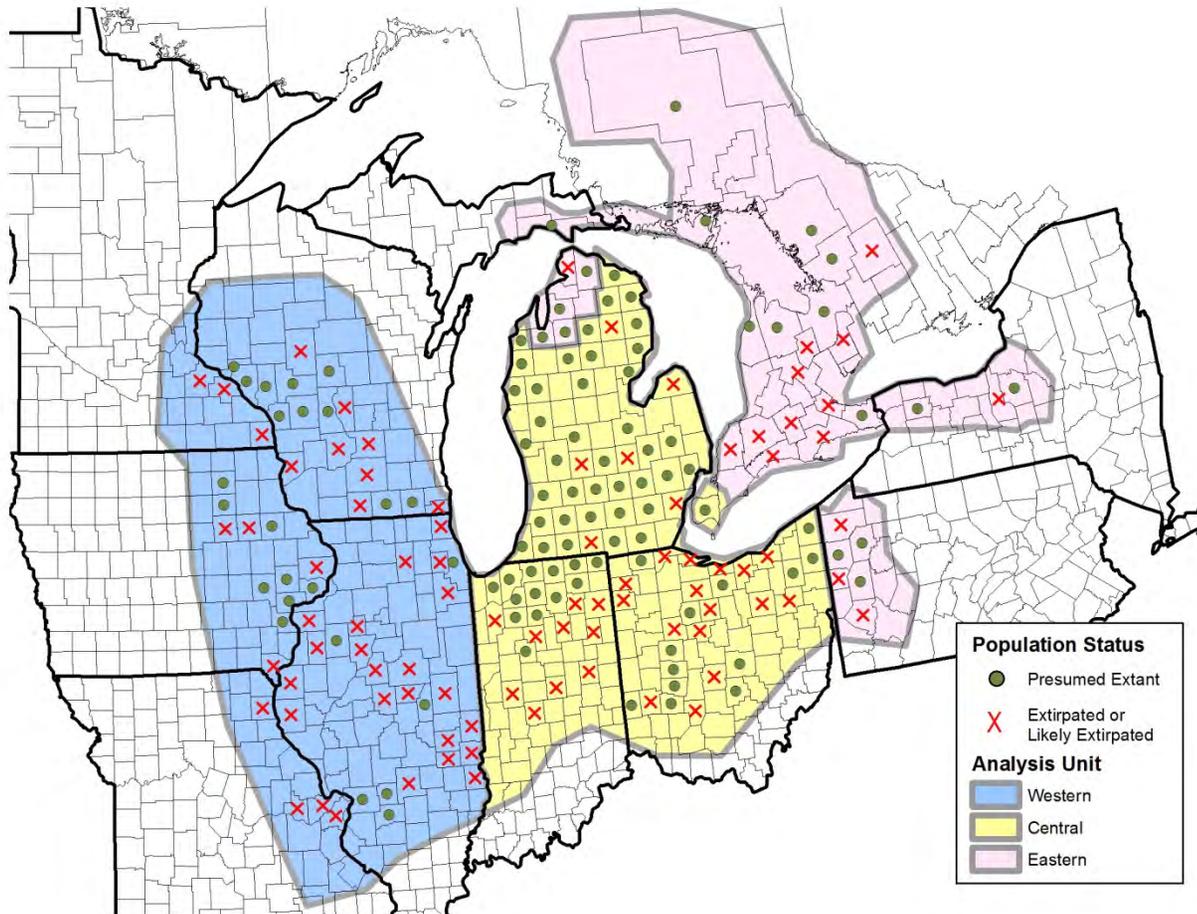
Table 4.5. The percent of range falling within each analysis unit and the percent reduction in EoO from historical to present day.

Analysis Unit	% Range within AU		% Reduction
	Historical	Current	
WAU	27.3%	15.8%	68.9%
CAU	37.5%	39.6%	43.2%
EAU	35.2%	44.7%	31.7%
Rangewide			46.2%

### 4.2.3 Summary

The degree of representation, as measured by spatial extent, across the EMR range has declined as noted by the northeasterly contraction in the range and by the loss of area occupied within the analysis units. Overall, there has been more than 46% reduction of EoO rangewide. This loss has not been uniform, with most of this decline occurring in the WAU (nearly 70% reduction in EoO in the WAU). However, losses of 43% and 32% in the CAU and EAU, respectively, are notable as well. Assuming that loss of range equates to loss of adaptive diversity, the degree of representation of the EMR has declined since historical conditions.

Figure 4.2. The geographical distribution of presumed extant (extant and unknown status) and extirpated EMR populations within each analysis unit.



## 4.3 Redundancy

Species-level redundancy is necessary to guard against catastrophic events simultaneously affecting all populations within an analysis unit. Redundancy is best achieved by having multiple, self-sustaining populations widely distributed across the gradient of adaptive diversity (i.e., within each analysis unit). Having multiple populations reduces the likelihood that all populations are affected simultaneously, while having widely distributed populations minimizes the likelihood of covariance among populations, thereby ensuring that all populations will not be similarly exposed nor similarly respond to a catastrophic event. To assess the current level of redundancy relative to historical conditions, we used the number of DGP robust populations within each analysis unit to evaluate the vulnerability of an analysis unit to extirpation due to a catastrophic event.

### 4.3.1 Methods

To assess the degree of redundancy in each analysis unit, we queried USFWS experts (Frank Durbian, Richard S. King, and Trisha Crabill) for the types of catastrophic events to consider. The experts identified drought, flooding, and disease as the most prominent, potentially catastrophic, events. Similarly, Pomara *et al.* (2014, p. 2095-2097) found extreme droughts and flooding as contributing factors in EMR extirpation events. However, we had insufficient information on flood risk (the magnitude of flood that would lead to extirpation) and disease risk (the likelihood of disease outbreaks, the factors that affect disease spread, and the magnitude of impact on EMR populations) to include it in our redundancy analysis. Thus, drought was the only catastrophic factor analyzed.

We consulted the U.S. Drought Monitor<sup>5</sup> for conducting our analyses. The Drought Monitor classifies general drought categories by intensity, with D1 being the least intense drought and D4 being the most intense drought. Based on feedback from internal and external experts, we analyzed the vulnerability of EMR extirpation due to D2-D4 drought intensities.

We used the following equation to calculate the probability of extirpation of all populations within an analysis unit (Ruckelshaus *et al.* 2002, p.312):

$$p(AUE) = (1 - e^{-\lambda * t})^n,$$

where  $\lambda$  is the annual rate (frequency) of a catastrophic drought event,  $t$  is the number of years of concern, and  $n$  is the number of populations in the analysis unit. We determined  $\lambda$  by calculating the frequencies of a D3 drought occurring from 2000 through 2015 for the periods of November through March for all counties occupied by EMR historically and currently. Drought data were unavailable for Ontario, so we assumed the drought frequency ( $\lambda$ ) to be the same as that of Michigan. We used the total number of presumed extant populations for  $n$  (see the Resiliency section of this chapter for further details on the methodology behind estimates of  $n$ ), and 25 years as the duration of time,  $t$ .

The  $p(AUE)$  is the probability of all populations within an analysis unit being extirpated ( $N=0$ , no individuals survive) by a catastrophic drought. We calculated the probability of extirpation,  $p(X)$ , of the

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<sup>5</sup> The U.S. Drought Monitor, established in 1999, is a weekly map of drought conditions that is produced jointly by the National Oceanic and Atmospheric Administration, the U.S. Department of Agriculture, and the National Drought Mitigation Center (NDMC) at the University of Nebraska-Lincoln.

populations within each State and then multiplied these State probabilities to obtain the  $p(\text{AUE})$  for the analysis units.

This model assumes identical and independent risks (i.e., catastrophic events are uncorrelated). Spatial dispersion and life history and genetic diversity, however, influence the likelihood of a single event extirpating multiple populations (Ruckelshaus *et al.* 2002, p. 314). Thus, to account for spatial distribution, we also incorporated a spatial dispersion analysis.

To evaluate the aspect of spatial dispersion, and thus vulnerability due to covariance, among populations of an analysis unit, we overlaid the current distribution of presumed extant populations with drought risks (drought frequencies) to evaluate the potential to lose geographical areas within an analysis unit.

Fundamental assumptions applied in our redundancy analysis include:

1. Drought frequencies in Ontario are same as those in Michigan.
2. No autocorrelation among populations. For example, the probability of a drought occurring at one population does not affect the probability of drought occurring at another even if in close proximity to each other.
3. Drought frequencies in 2000-15 represent the true risk of drought over the next 25 years.
4. Drought is the only potential catastrophic event to impact EMR populations.

### 4.3.2 Results

Probability of Analysis Unit Extirpation  $p(\text{AUE})$  – The frequencies of a D4 are 0 for all analysis units and 0 for D3 frequencies in the CAU and EAU. Thus, the  $p(\text{AUE})$  for all units due to a D4 drought is 0 and 0 for a D3 drought event in the CAU and EAU (Table 4.6). Although the frequency of a D3 drought in the WAU ranges from 0.01 to 0.04, the  $p(\text{AUE})$  for WAU is essentially 0.0 as well (Table 4.6). Using D2  $\lambda$  values, again the results do not change for the CAU and EAU, but the  $p(\text{AUE})$  increases to 0.104 in the WAU; i.e., there is a 10% probability of extirpation due to a D2 level drought in the next 25 years in WAU (Table 4.7).

Table 4.6. The probability of extirpation of all extant populations within an analysis unit given D3 level drought frequencies.  $n$  = the number of presumed extant populations;  $\lambda$  = the drought frequency,  $p(\text{SX})$  = the state-specific probability of extirpation;  $p(\text{AUE})$  = the analysis unit-wide probability of extirpation; CA= Canada.

	Frequency of D3 Level Winter Drought									
	WAU			CAU			EAU			
	IA	IL	WI	IN	MI	OH	PA	NY	MI-E	CA
$n$	9	4	8	33	232	29	5	2	7	59
$\lambda$	0.040	0.013	0.023	0.000	0.000	0.000	0.000	0.000	0.000	0.000
$p(\text{X})$	0.016	0.006	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000
$p(\text{AUE})$	0.000			0.000			0.000			

Table 4.7. The probability of extirpation of all populations within an analysis unit given D2 level drought frequencies.  $n$  = the number of presumed extant populations;  $\lambda$  = the drought frequency,  $p(SX)$  = the state-specific probability of extirpation;  $p(AUE)$  = the analysis unit-wide probability of extirpation; CA= Canada.

	Frequency of D2 Level Winter Drought									
	WAU			CAU			EAU			
	IA	IL	WI	IN	MI	OH	PA	NY	MI-E	CA
$n$	9	4	8	33	232	29	5	2	7	59
$\lambda$	0.140	0.045	0.119	0.085	0.040	0.003	0.000	0.000	0.036	0.036
$p(X)$	0.759	0.208	0.658	0.015	0.000	0.000	0.000	0.000	0.000	0.000
$p(AUE)$	0.104			0.000			0.000			

The above analysis used the number of presumed extant populations within each analysis unit. However, as explained in the Species Ecology chapter (Chapter 2), redundancy requires having multiple *self-sustaining* populations in each representative area (analysis unit). To evaluate the risk of analysis unit-wide extirpation given the number of DGP robust populations, we used the average (and highest) drought frequency within the analysis unit as the  $\lambda$  rate<sup>6</sup>. The results for the CAU and EAU mirror the previous analysis,  $p(AUE)$  is 0.0 (Table 4.8). The  $p(AUE)$  for the WAU, however, is notably higher with 0.048 and 0.718, for D3 and D2 drought probabilities, respectively. This means that given the number of DGP robust populations within the WAU, there is a 4.8% chance of WAU-wide extirpation within 25 years from a D3 drought and a 71% chance of extirpation from a D2 drought (assuming drought frequencies remain constant).

Table 4.1 The probability of analysis unit extirpation,  $p(AUE)$ , using average drought frequencies ( $\lambda$ ) and the extrapolated number of DGP robust populations for  $n$ .

	WAU	CAU	EAU
$n$	4	65	30
$\lambda$ (D3)	0.025	0.000	0.000
$p(AUE)$	0.048	0.000	0.000
$\lambda$ (D2)	0.101	0.043	0.020
$p(AUE)$	0.718	0.000	0.000

Spatial Dispersion of Populations – Analysis unit extirpation is also influenced by the spatial dispersion (the likelihood of covariance among populations in an analysis unit) of populations within an analysis unit. Looking at the state-specific probabilities of extirpation,  $p(SX)$ , provide some insights of the potential vulnerability due to the spatial dispersion of populations within an analysis unit. In the WAU, the  $p(SX)$  estimates are notable in all 3 states, and particularly, for D2  $\lambda$  rates; Illinois  $\lambda$  rates are the lowest at 0.208, while Iowa and Wisconsin have considerably higher  $\lambda$  rates, 0.759 and 0.658, respectively. Overlaying the county drought  $\lambda$  rates with population locations, all populations, except those in the 3 southernmost counties in Illinois, are at risk of a catastrophic drought (Figures 4.3 and 4.4). Using D3  $\lambda$  rates, 5 of the 8 population clusters have a notable risk of a catastrophic drought, while 7 of 8 are at risk using D2  $\lambda$  rates.

<sup>6</sup> We do not know the locations of all extrapolated DGP robust populations, and therefore, could not calculate state-specific  $p(SX)$ . Instead, we used the State  $\lambda$  values to calculate an average  $\lambda$  for the 3 analysis units. We also used the highest state  $\lambda$  value as the analysis unit  $\lambda$ . The average and highest  $\lambda$  values give us a range of the potential vulnerability of analysis unit extirpation,  $p(AUE)$ .

In the CAU and EAU, the  $p(SX)$  is 0.0 or nearly so for all states; the lone exception being Indiana for D2  $\lambda$  rates (Tables 4.6 and 4.7). Overlaying the county  $\lambda$  rates and locations of populations, the risk from D3 droughts appears very low for all geographical areas within the CAU and EAU (Figures 4.3 and 4.4). However, using D2 drought frequencies, most of Indiana and portions of Michigan are at risk of extirpation due to a catastrophic drought. Thus, the southwestern portion and scattered portions in the north of the CAU are at risk of a catastrophic drought. The eastern and far northern portions of the CAU and all of EAU appear to have low to no risk of catastrophic drought.

#### 4.3.3 Summary

The risk of AUE due a catastrophic drought varies by analysis unit and by the level of drought considered. In the CAU and EAU, the  $\lambda$  rates for D3 and d4 droughts are 0.0, so there is little to no risk of AUE regardless of spatial dispersion. Portions of the CAU are at risk of a D2 level catastrophic drought; populations in the southern portion of the CAU and scattered portions in the north are at risk. In the WAU, the risk of AUE using D3  $\lambda$  rates is low, but the risk of losing clusters within the WAU is notable; 5 of the 8 population clusters are vulnerable to a catastrophic drought. The  $p(AUE)$  is notably higher with D2  $\lambda$  rates; 7 of the 8 clusters are at risk of D2 level catastrophic drought. Thus, the probability of losing most populations within the WAU due to a catastrophic drought is high.

Figure 4.3. The annual D2 frequencies for counties with presumed extant EMR populations. Darker shades represent higher annual frequency rates.

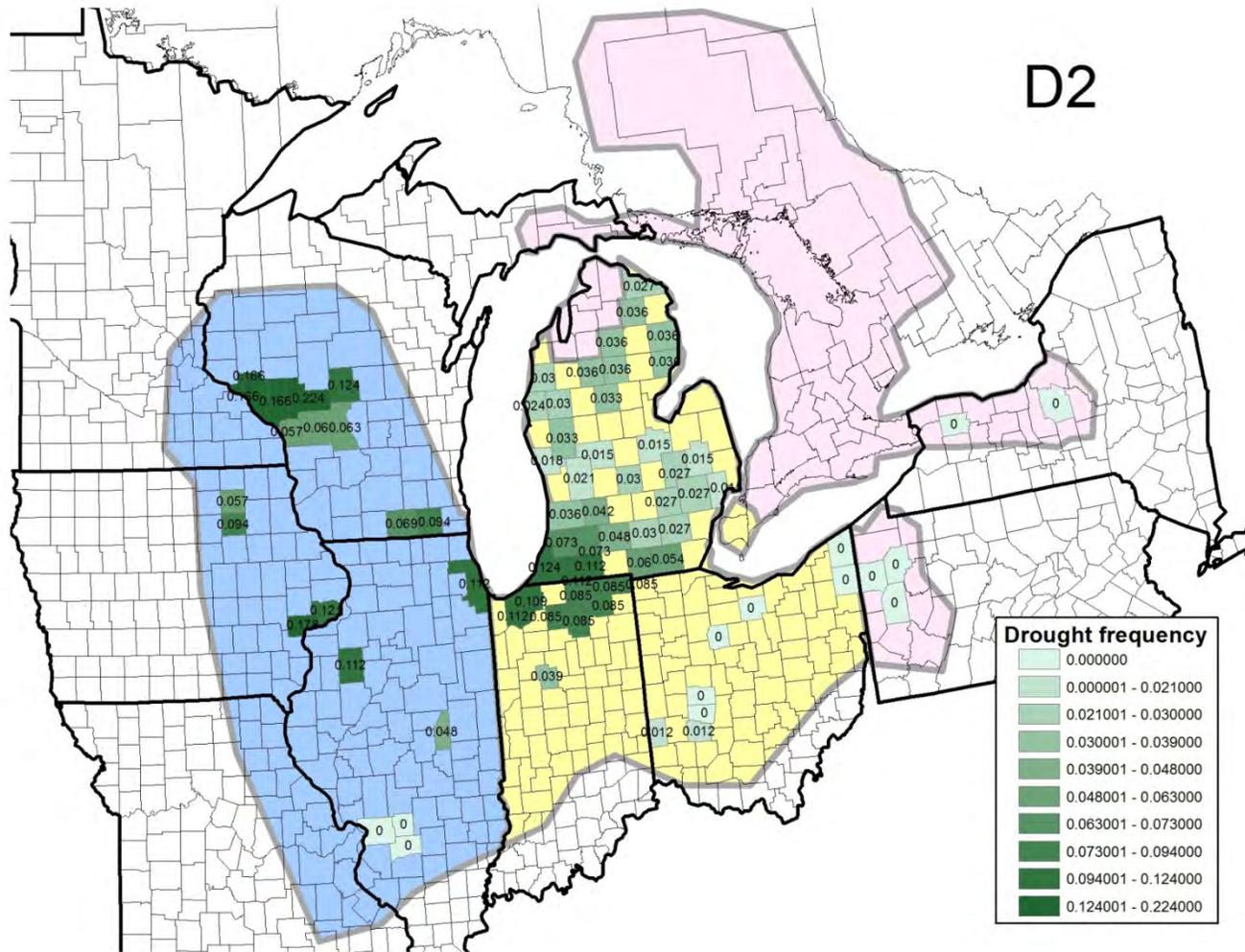
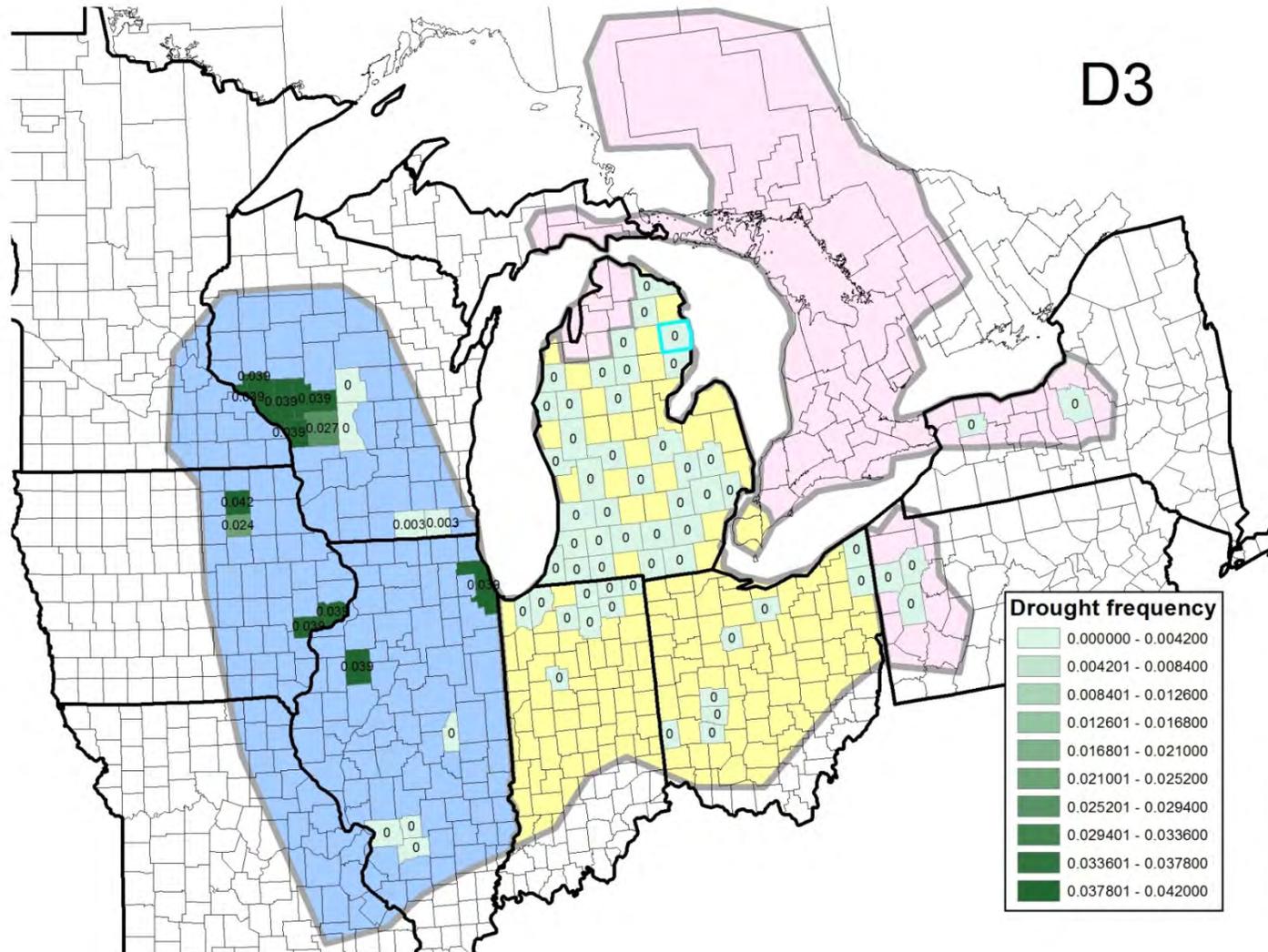


Figure 4.4. The annual D3 frequencies for counties with presumed extant EMR populations. Darker shades represent higher annual frequency rates.



## Chapter 5 Future Conditions

In the previous chapters, we evaluated the current condition of EMR and described and analyzed the risk and beneficial factors driving the historical, current, and future condition of the species. In this chapter, we predict the future conditions of EMR, in terms of its resiliency, representation, and redundancy, given its current conditions and the risk and beneficial factors acting upon the EMR populations.

To assess the resiliency, representation, and redundancy of EMR, we assumed populations with unknown status are extant, and collectively referred to these extant and unknown populations as “presumed extant” populations. As explained in Chapter 2, we delineated 3 broad geographic areas (analysis units) that are important for preserving EMR adaptive diversity. Our analyses, therefore, were conducted at both the rangewide and analysis unit scales.

### 5.1 Resiliency

Resiliency is necessary to ensure that EMR can withstand annual environmental variation such as fluctuation in rainfall, temperature, and prey availability, for example. For EMR to have sufficient resiliency it must have healthy (self-sustaining, resilient) populations. To assess the current resiliency of the EMR, we forecasted the number of populations likely to be extant into the future and evaluated the health of these populations given projections of future risk and beneficial factors occurring.

#### 5.1.1 Methods

To assess the future degree of resiliency, we evaluated future health (whether the populations are self-sustaining) of those populations classified likely to be extant at years 10, 25, and 50. We defined self-sustaining as one that: 1) is demographically, genetically, and physiologically robust (DGP robust); 2) has suitable habitat conditions; and 3) is free of, or has manageable, stressors acting upon it. More specifically, a self-sustaining robust population is one that has an adult female population size greater than 50 ( $N_F > 50$ ), has a positive population growth rate ( $\lambda \geq 1$ ), and has a high probability of persistence ( $p(P) > 0.10$ ) despite the stressors acting upon it.

We again used the results from the Faust model and supplemented our analyses with new information<sup>7</sup>. Faust and colleagues developed a stochastic, matrix model to project population trend over time for 57 populations. Population-specific inputs included current adult female population size ( $N_0$ ) and risk and beneficial factors acting upon the population. Model outputs included median ending adult female population size ( $N_F$ ), population growth rate ( $\lambda$ ), and probability of quasi-extirpation<sup>8</sup>.

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<sup>7</sup> Note, the 2011 Report provided summary statistics for year 25 only. Faust re-ran the model in March of 2015 to ascertain estimates for years 10 and 50.

<sup>8</sup> Faust *et al.* (2011, p. 1-66), as well as other authors cited within, used the terms extinct, extinction, and quasi-extinct(ion). Based on feedback from one reviewer and a decision maker, we replaced these terms with

(QE) was defined as the population threshold at which demographic and genetic stochasticity dominate the dynamics of the population, and it was set at 25 or fewer adult females. Specifically, we directly used median ending female population size ( $N_F$ ), and stochastic mean lambda ( $\lambda$ ) results in our analyses, and used the  $1-p(QE)$  to calculate the probability of persistence (i.e., a population with a high  $p(P)$ , is one with a low probability of becoming quasi-extirpated). Populations that are predicted to meet all 3 criteria at years 10, 25, and 50, were presumed to be self-sustaining at those 3 time periods. Using these results for the 57 modeled populations, we then extrapolated to the remaining presumed extant populations by multiplying the proportion of modeled populations that were self-sustaining by the total number of presumed extant populations to estimate the number of EMR populations that are projected to be self-sustaining.

To forecast the number of self-sustaining populations at years 10, 25 and 50, we first identified the number of populations that are likely to be extant in the future. We defined high probability of persistence as having less than 0.10 chance of being quasi-extirpated; so we used modeled generated population-specific probabilities of being quasi-extirpated [ $p(QE)$ ] to calculate population-specific probabilities of persistence [ $p(P)$ ,  $1-p(QE)$ ]. Using these modeled generated probability of persistence estimates and a random number generator, we identified the populations that are likely to be extant at years 10, 25, and 50.

Specifically, we ran a simulation of a 100 replicates using the random number generator function within Excel to determine whether a modeled population remains extant or is extirpated based on the its probability of persistence,  $p(P)$ . The simulation compares the estimated probability of persistence to the random number (simulating a possible extirpation event), drawn from a uniform distribution between 0 and 1. If the random number is greater than the modeled population's  $p(P)$  value, for that iteration, that population received a 0 and was considered extirpated. If the random number is less than the modeled  $p(P)$ , for that iteration, that population received a 1 and was considered extant. For each 100 replicates, we summed (by analysis unit) the number of extant modeled populations and calculated the average number of modeled populations that were extant across 100 replications (average number of replicates that had a 1 generated). We augmented the number of replications by using @Risk modeling software; running the simulation 1000 times, we calculated the average number of the modeled populations likely to be extant for the 3 analysis units. We used these averages to represent the forecasted number of modeled populations likely to be extant in the analysis unit.

For each analysis, we then divided the forecasted number of likely to be extant modeled populations by the total number of modeled populations to obtain the proportion of modeled populations that are likely to be extant. This proportion was multiplied by the number of presumed extant populations (388) to extrapolate the number of populations likely to be extant for the entire analysis unit at years 10, 25, and 50.

Using these predicted numbers of likely to be extant populations, we calculated the proportion of modeled populations that are projected to be self-sustaining in the analysis unit and multiplied this by the number of likely to be extant populations in the analysis unit. To forecast the number of quasi-extirpated populations, we calculated the proportion of modeled populations that are projected to be quasi-extirpated (based on the projected ending population sizes) and multiplied this proportion by the number of presumed extant populations in the analysis unit.

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extirpated, extirpation, and quasi-extirpated(extirpation) to better clarify that are results are referring to losses of portions of the species' range (populations), not the species as a whole.

All assumptions and caveats pertinent to the Faust model results also apply to the extrapolation results (see Chapter 1 for a list of key assumptions underlying the Faust model). Additional assumptions we applied in our current conditions analyses include:

1. Populations with unknown status were assumed extant although many of these populations may indeed be extirpated.
2. Risk factors (disease, road mortality and persecution) not considered in the Faust model were not considered in our analysis.
3. We lacked uniform data to indicate whether the factors evaluated in the model are likely to change in pervasiveness or magnitude of effect. Thus, in our predictive modeling of future resiliency, we assume the influence of future factors to remain constant over time.
4. Populations with  $N_F > 50$ ,  $\lambda \geq 1$ , and  $p(P) \geq 0.90$  were considered self-sustaining.
5. Populations that with  $N_0 \leq 25$  were automatically assigned  $p(QE)=1$  within the Faust model. Those initially quasi-extirpated populations but that met the DGP criteria,  $N_F > 50$ ,  $\lambda \geq 1$  at years 10, 25 and 50, were tallied as self-sustaining. These populations, although starting small, had strong population growth rates and either had beneficial factors or no to low magnitude stressors occurring. Through time these populations increased, and thus, had characteristics of a healthy population.

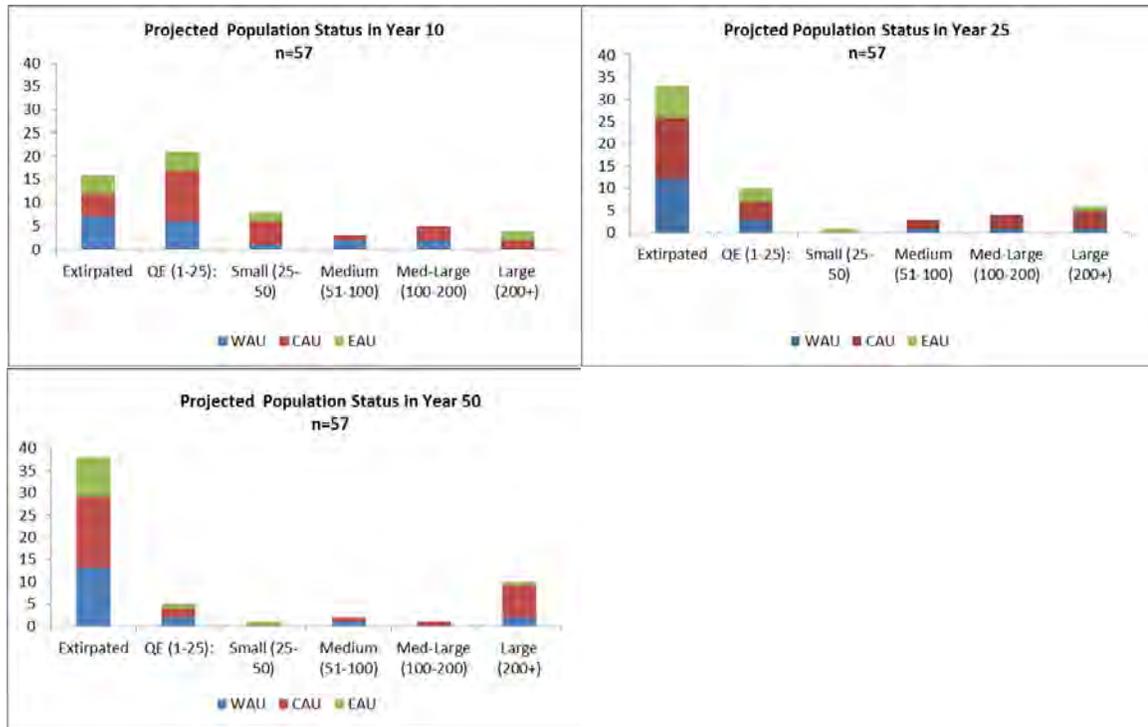
We use several terms to characterize future conditions. For clarification, a list of these terms and their definitions are provided here:

Term	Definition
<b>Healthy population</b>	A population that is self-sustaining
<b>Self-sustaining</b>	A population that is DGP robust and has suitable habitat conditions and manageable threats
<b>DGP</b>	A population with more than 50 adult females ( $N_F > 50$ ) and has a positive population growth rate ( $\lambda \geq 1$ )
<b>Manageable threats</b>	A population with a high probability of persistence ( $p(P) \geq 0.90$ )
<b>Presumed extant</b>	A population currently classified as extant or unknown status
<b>Quasi-extinct</b>	A population with 25 or fewer adult females ( $N_F \leq 25$ )
<b>Modeled population</b>	One of the 57 populations modeled by Faust <i>et al.</i> (2011, p. 1-66)
<b>Likely extant</b>	Populations projected, via extrapolation, to be extant in the future

### 5.1.2 Results – Modeled Populations

Rangewide - The number of modeled populations projected to be extirpated increases over time, with the number of populations in all other population size categories declining, except for the category of large population size ( $N_F > 200+$ ), which increases through year 25 and then stabilizes (Figures 5.1a-c). The reduction in the number of populations in a quasi-extirpated state over time is primarily due to populations becoming extirpated (20 of initial 25 quasi-extirpated populations are predicted to become extirpated); a few of the initially quasi-extirpated populations grow over time ( $n=3$ ). These latter populations, although initially small, quickly grew due to positive growth rates, risk factors having little or no impact on these populations, and beneficial factors occurring at these sites.

Figures 5.1.a-c. The number of modeled populations (n=57) projected to be extirpated and extant. The latter are divided into population size categories ranging from quasi-extirpated to large. WAU=western analysis unit, CAU= central analysis unit, and EAU = eastern analysis unit. The results are compiled from the median final population sizes across 3000 model iterations ending with extant populations.



Of the 57 populations modeled by the Faust model, only a subset of the projected extant populations is self-sustaining (Table 5.1a-c). Rangewide, 10 (17.5%) of the populations are projected to be DGP robust at year 10 but increase to 12 (21%) by year 25 (and remain at 12 to year 50). Seven (9%) of the populations<sup>9</sup> are projected to meet the probability of persistence threshold for a self-sustaining population at year 10, but by year 25, only 5 (19%) of the populations meet this threshold. Combining these data, 4 (7%) of the populations are projected to be self-sustaining at years 10, 25, and 50. An additional 3 (5%) of the populations are DGP robust, but have a low predicted probability of persistence because the initial population sizes ( $N_F$ ) are  $\leq 25$ , and thus, automatically, in the model, assigned a  $p(QE)$  of 1 (and thus, have  $p(P)=0$ ). These populations, however, are projected to increase in size over time, despite the low prediction of persistence, because of strong population growth. While not technically meeting the definition of self-sustaining (because  $p(P)<0.90$ ), we have included these populations in the tally of the total number of populations that are projected to be self-sustaining. Thus, 12% (7) of the 57 modeled populations are projected to be self-sustaining at years 10, 25, and 50 (Table 5.1a-c).

Analysis Units (modeled populations) – The projected number of extirpated ( $N_F = 0$ ) populations increases in all 3 conservation units (Table 5.2a-c). Within the WAU, of the 18 modeled populations, 7 are projected to be extirpated by year 10, 12 populations by year 25, and 13 populations by year 50.

<sup>9</sup> The populations that were DGP are not necessarily the populations that had high  $p(P)$ . Only a subset ( $n=4$ ) of the DGP populations also had high  $p(P)$ .

Two (11%) of the modeled populations are projected to be DGP robust at all 3 time periods and 1 (6%) to have high p(P) by year 10, and none at years 25 and 50; 2 (11%) populations are projected to be self-sustaining (adjusted) at year 10, and 1 (6%) at years 25 and 50 (Table 5.2a-c, Figures 5.2a-c).

Within the CAU, of the 27 modeled populations, 5 are projected to be extirpated by year 10, 14 populations by year 25, and 16 populations by year 50. Four (15%) are projected to be DGP robust and 4 (15%) to have high p(P) at all 3 time periods; 4 (15%) populations are projected to be self-sustaining (adjusted) by year 10, and 5 (19%) populations at years 25 and 50.

In the EAU, of the 12 modeled populations, 4 are projected to be extirpated by year 10, 7 populations by year 25, and 9 populations by year 50. One (8%) of the modeled populations is projected to be DGP robust at all 3 time periods and 2 (17%) populations to have high p(P) by year 10, and 1 (8%) at years 25 and 50; 1 (8%) populations is projected to be self-sustaining at all 3 time periods.

By definition, a self-sustaining should persist over time unless the magnitude of a threat increases in the future. This is true for both the CAU and the EAU, but not for the WAU. Within the WAU, one population meets the self-sustaining probability persistence threshold at year 10, but the threats acting upon it are slowly impairing its demography, and hence by year 25, it no longer meets the probability of persistence threshold for a self-sustaining population.

Table 5.1a-c. Projected results from 57 modeled populations. WAU=western analysis unit, CAU=central analysis unit, EAU=eastern analysis unit, RW=rangewide. The number of modeled populations: WAU=18, CAU=27, and EAU=12. DGP = demographically, physiologically, and genetically robust;  $N_F > 50$  and  $\lambda \geq 1$ .  $p(P)$  = probability of persistence  $\geq 0.90$ . S-S (self-sustaining)= DGP and  $p(P)$  thresholds met. The populations comprising DGP and  $p(P)$  may differ. Three populations with  $p(P)$  were QE at model year 0 ( $N_0 = QE$ ) and thus were automatically assigned  $p(P)=0$ ; these populations, however, had ending  $N_F > 50$  and  $\lambda \geq 1$  over time, so we classified them as S-S (tallied in Adjusted # of S-S column).

Self-sustaining Robustness at 10 years				
	DGP ( $N_F = >50$ & $\lambda \geq 1$ )	$p(P) \geq 0.10$	# of S-S	Adjusted # of S-S
WAU	3	1	1	2
CAU	6	4	2	4
EAU	1	2	1	1
RW	10	7	4	7

Self-sustaining Robustness at 25 years				
	DGP ( $N_F = >50$ & $\lambda \geq 1$ )	$p(P) \geq 0.10$	# of S-S	Adjusted # of S-S
WAU	3	0	0	1
CAU	8	4	3	5
EAU	1	1	1	1
RW	12	5	4	7

Self-sustaining Robustness at 50 years				
	DGP ( $N_F = >50$ & $\lambda \geq 1$ )	$p(P) \geq 0.10$	# of S-S	Adjusted # of S-S
WAU	3	0	0	1
CAU	8	4	3	5
EAU	1	1	1	1
RW	12	5	4	7

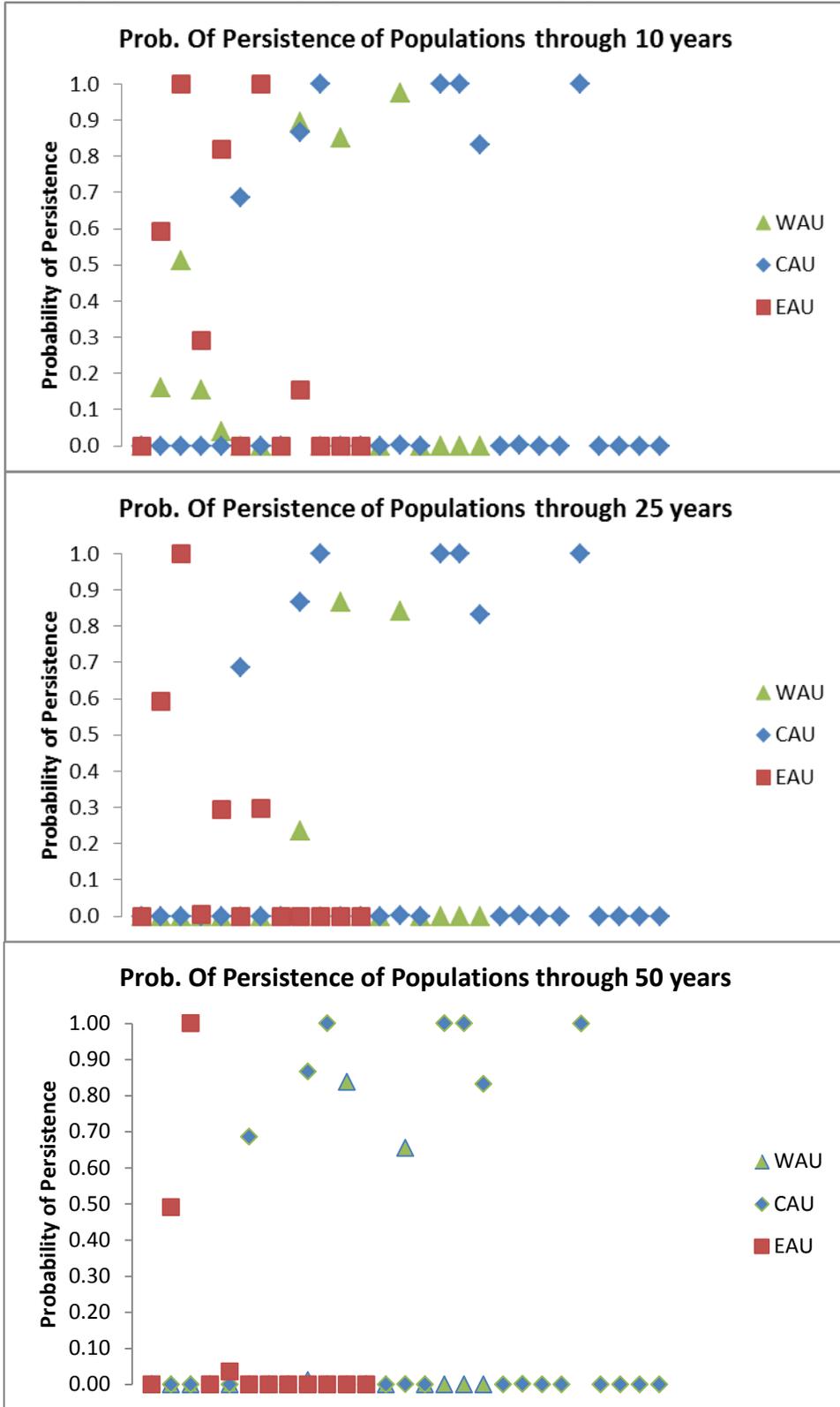
Table 5.2a-c. Projected population status of the 57 populations modeled by Faust *et al.*, (2011) at Year 10, Year 25, and Year 50. Population sizes were based on the median ending N values derived from Faust model.

Status	# of modeled populations at year 10		
	WAU	CAU	EAU
Extirpated	7	5	4
QE (1-25):	6	11	4
Small (25-50)	1	5	2
Medium (51-100)	2	1	0
Med-Large (100-200)	2	3	0
Large (200+)	0	2	2

Status	# of modeled populations at year 25		
	WAU	CAU	EAU
Extirpated	12	14	7
QE (1-25):	3	4	3
Small (25-50)	0	0	1
Medium (51-100)	1	2	0
Med-Large (100-200)	1	3	0
Large (200+)	1	4	1

Status	# of modeled populations at year 50		
	WAU	CAU	EAU
Extirpated	13	16	9
QE (1-25):	2	2	1
Small (25-50)	0	0	1
Medium (51-100)	1	1	0
Med-Large (100-200)	0	1	0
Large (200+)	2	7	1

Figure 5.2.a-c. Projected probability of persistence plots for the 57 modeled populations over time; a. Year 10 projections, b. Year 25 projections, and c. Year 50 projections.



### 5.1.3 Results - Extrapolating to All Populations

The above results pertain to the 57 populations modeled by Faust *et al.*, (2011). To discern the future conditions of all 388 presumed extant (i.e., populations with current (2014) extant and unknown status) populations, we extrapolated the results of the 57 modeled populations to all 388 populations. To extrapolate to all presumed extant populations, we first identified the populations that are likely to be extant in the future. We used the population-specific probability of persistence estimates to determine the number of populations within each analysis unit that are likely to be extant (see Methods subsection for further details).

Rangewide – Extrapolating from the population simulations, 87 of 388 presumed extant populations are forecasted to be extirpated and 151 quasi-extirpated by year 10, 209 populations extirpated and 65 quasi-extirpated by year 25, and 244 populations extirpated and 30 populations quasi-extirpated by year 50 (Table 5.3). Of those 388 populations that are not extirpated or quasi-extirpated, 132 populations are likely to be extant at year 10, 102 populations at year 25, and 96 populations at year 50. Of the 32 populations likely to be extant at year 10, 52 populations are likely to be self-sustaining. At years 25 and 50, 62 populations are likely to be self-sustaining (Table 5.3).

Analysis Units – Within the WAU, of the 21 presumed extant populations, 8 are likely to be extirpated and 7 quasi-extirpated by year 10, 14 populations extirpated and 4 quasi-extirpated by year 25, and 15 populations extirpated and 2 quasi-extirpated by year 50. Of the populations likely to be extant, 2 of 5 populations are likely to be self-sustaining at year 10, and 1 at years 25 and 50.

Within the CAU, of the 294 presumed extant populations, 54 are likely to be extirpated and 120 quasi-extirpated by year 10, 154 extirpated and 44 quasi-extirpated by year 25, and 175 extirpated and 22 quasi-extirpated by year 50. Of the 109 populations likely to be extant at year 10, 44 are likely to be self-sustaining, and at years 25 and 50, 54 of 87 populations are likely to be self-sustaining.

Within the EAU, of the 74 presumed extant populations, 24 are likely to be extirpated and 24 quasi-extirpated by year 10, 43 populations extirpated and 18 quasi-extirpated by year 25, and 55 populations extirpated and 6 quasi-extirpated by year 50. Of the 18 populations likely to be extant at year 10, 6 are projected to be self-sustaining, and these 6 populations continue to be self-sustaining at years 25 and 50.

Table 5.3. The forecasted number of populations extrapolated from modeled populations to the total number of presumed extant populations in the 3 analysis units. The numbers of “simulated” populations are those modeled populations forecasted to persist into the future (via a population simulator). The numbers of “projected” populations are outputs from the Faust model; those populations predicted to be self-sustaining ( $N_F > 50$ ,  $\lambda > 1$ ,  $p(P) \geq 0.90$ ); QE ( $N_F \leq 25$ ); or extirpated ( $N_F = 0$ ).

Analysis Unit/ Rangewide	# of Modeled Pops	# of Modeled Pops:			Proportion of Modeled Pops:			# of populations that are:			# extrapolated populations that are:		
		Yr 10	Yr 25	Yr 50	Yr 10	Yr 25	Yr 50	Yr 10	Yr 25	Yr 50	Yr 10	Yr 25	Yr 50
		<b>projected X</b>						<b>presumed extant</b>			<b>likely X</b>		
WAU	18	7	12	13	0.39	0.67	0.72	21			8	14	15
CAU	27	5	14	16	0.19	0.52	0.59	294			54	152	174
EAU	12	4	7	9	0.33	0.58	0.75	73			24	43	55
RW	57	16	33	38				388			87	209	244
		<b>projected QE</b>						<b>presumed extant</b>			<b>likely QE</b>		
WAU	18	6	3	2	0.33	0.17	0.11	21			7	4	2
CAU	27	11	4	2	0.41	0.15	0.07	294			120	44	22
EAU	12	4	3	1	0.33	0.25	0.08	73			24	18	6
RW	57	21	10	5				388			151	65	30
		<b>simulated to persist</b>						<b>presumed extant</b>			<b>likely extant</b>		
WAU	18	4	2	2	0.22	0.11	0.11	21			5	2	2
CAU	27	10	8	8	0.37	0.30	0.30	294			109	87	87
EAU	12	3	2	1	0.25	0.17	0.08	73			18	12	6
RW	57	17	12	11				388			132	102	96
		<b>projected self-sustaining</b>						<b>presumed extant</b>			<b>likely self-sustaining</b>		
WAU	18	2	1	1	0.11	0.06	0.06	21			2	1	1
CAU	27	4	5	5	0.15	0.19	0.19	294			44	54	54
EAU	12	1	1	1	0.08	0.08	0.08	73			6	6	6
RW	57	7	7	7				388			52	62	62

#### 5.1.4 Summary

The future resiliency—the number of self-sustaining populations— varies across EMR range. In the WAU, 83% of the modeled populations are projected to have a declining trajectory ( $\lambda < 1$ ) and 94% of the populations a low probability of persistence ( $p(P) < 0.90$ ) by year 25, and thus, the number of forecasted populations likely to be extant declines over time. By year 50, 17 of the 21 presumed extant populations are projected to be extirpated ( $n=15$ ) or 2 quasi-extirpated ( $n=2$ ), with only 1 population projected to be self-sustaining. The resiliency of the WAU is forecasted to decline over time. The situation is similar in the CAU and EAU but to a lesser degree. In the CAU, 70% of the modeled populations are projected to have a declining trajectory and 78% a low probability of persistence, and thus, by year 50, 196 of the 294 presumed extant populations are projected to be extirpated ( $n=174$ ) or quasi-extirpated ( $n=22$ ), and 54 populations to be self-sustaining. In the EAU, 83% of the modeled populations are projected to have a declining trajectory and 92% of the populations are projected to have a low probability of persistence, and thus, by 50, 61 of the 73 presumed extant populations are projected to be extirpated ( $n=55$ ) or quasi-extirpated ( $n=6$ ), and 6 to be self-sustaining.

## 5.2 Representation

To preserve a species' evolutionary potential, and its ability to respond to future changes in its environment, it is important to preserve the range of the species' adaptive diversity. This is done by maintaining the full gradient of genetic and ecological diversity, along with the processes that drive evolutionary change. To capture the full gradient of adaptive diversity for the EMR, it is necessary to maintain widely distributed populations in each of the 3 analysis units (See Species Ecology in Chapter 2 for a full description of the adaptive diversity of the EMR and how the analysis units were delineated). Therefore, we analyzed the change in the spatial distribution of the EMR rangewide and within each analysis unit over time.

### 5.2.1 Methods

To calculate the change in spatial distribution, we used the historical extent of occurrence (EoO) estimates derived in Chapter 4 and calculated the EoO for the 57 modeled populations and the subset of these modeled populations projected to be extant at years 10, 25, and 50. We calculated the EoO for the modeled populations by drawing polygons (using ArcGIS) around the counties occupied by these populations (the EoO for the 57 modeled populations is referred to as the baseline). To calculate EoO of the subset of modeled populations that are likely to be extant, we used the predicted number of modeled populations projected to be extant and drew polygons around the counties occupied by the populations with the highest probability of persistence (Chapter 4). We calculated the percent change in EoO from the baseline to years 10, 25, and 50. We multiplied the historical EoO estimate by these percent changes to calculate the EoO for all projected to be extant populations at years 10, 25, and 50. For example, suppose 3 of the modeled populations are projected to be extant at year 10, we would identify the 3 populations with the highest probabilities of persistence and calculate the EoO using the counties occupied by these 3 populations. We then calculate the percent change from the baseline model (57 model populations) to year 10. Assuming this is 29% reduction, we then multiply the historical EoO estimate by 29% to obtain the predicted EoO for all populations projected to be extant at year 10.

To calculate the portion of range falling within each analysis unit, we divided the calculated analysis unit EoO estimate by the sum of the EoO for all 3 units. We also used the predicted EoO values to calculate the percent decline in EoO rangewide and within each analysis unit at years 10, 25, and 50.

As explained in the Current Conditions (Chapter 4), the results are not a true measure of area occupied by the EMR but rather a coarse evaluation to make relative comparison among years. The reasons for this are twofold: (1) the calculations are done at the county, rather than the population, level and (2) if at least 1 population was projected to be extant, the entire county was included in the analysis, even if other populations in the county were projected to be extirpated. In other words, the EoO analysis likely does not capture the full extent of EoO decline that has occurred.

The assumptions applied in our analysis include:

1. The 3 analysis units and wide distribution within the analysis units capture the breadth of adaptive diversity of EMR.
2. The loss of an analysis unit and loss of extent of occurrence within an analysis unit constitutes loss of adaptive diversity.

## 5.2.2 Results – Extrapolation to All Populations

Rangewide - The spatial distribution of EMR is forecasted to continue to contract northeasterly over time (Figure 5.3), with the proportion of the range occurring within the WAU declining 14% by year 50 while the proportion of the range falling within the EAU increasing steadily over time and by year 50 containing 50% of the range of the species (Table 5.4). Rangewide, the EoO is predicted to decline 55% by year 10 and 65% by year 50 (Table 5.5).

Figure 5.3. The projected change in spatial distribution across the 3 analysis units over time.

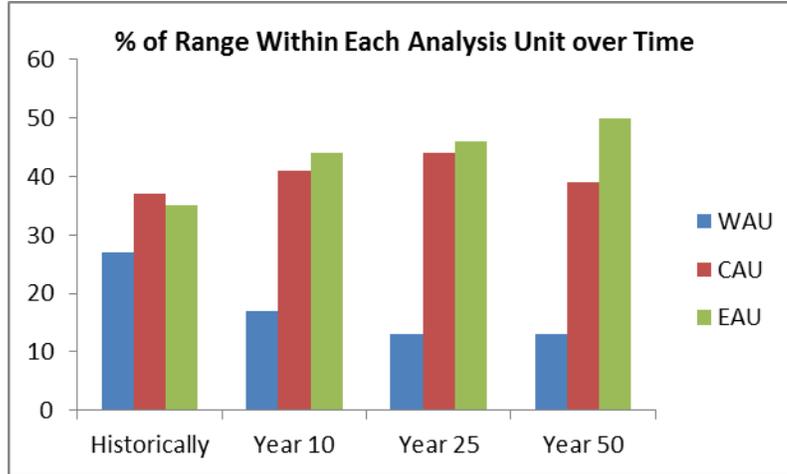


Table 5.4. The extrapolated proportion of range, as measured via EoO, of each analysis unit at years 10, 25, and 50.

	% of Range Occupied By Each Analysis Unit			
	Historically	Yr 10	Yr 25	Yr 50
WAU	27%	17%	13%	13%
CAU	37%	41%	44%	39%
EAU	35%	44%	46%	50%

Table 5.5. The extrapolated percent decline from historical conditions in Extent of Occurrence, EoO, among the 3 analysis units.

	% Reduction in EoO		
	Yr 10	Yr 25	Yr 50
WAU	71%	83%	83%
CAU	51%	57%	64%
EAU	44%	51%	51%
Rangewide	55%	63%	65%

Analysis Unit – A reduction in the spatial distribution within each analysis unit is also predicted to occur although the severity of decline varies across analysis units (Table 5.5). In the WAU, EoO declines 71% through year 10 and 83% by year 25. In the CAU and EAU, the predicted decline reaches 64% and 51%, respectively, by year 50.

### 5.2.3 Summary

Although EMR populations are predicted to be extant in all three analysis units, the range is projected to contract northeasterly and the area occupied within each analysis unit to declines over time. The results predict a 65% reduction of the area occupied by EMR rangewide, with the WAU comprising most of the decline (83% reduction within WAU by year 50). These predicted declines in extent of occurrence across the species range and within the analysis units suggest that loss of adaptive diversity is likely to occur.

## **5.3 Redundancy**

Species-level redundancy is necessary to guard against catastrophic events wiping out a unit of irreplaceable adaptive diversity. As explained in Chapter 1, the areas of unique adaptive diversity for EMR are captured by broad distribution of populations within the three analysis units. Sufficient redundancy for EMR viability is therefore best ensured by having multiple, self-sustaining populations widely distributed throughout each analysis unit. This reduces the degree of, and the potential for, covariance among populations, and thus, likelihood that all populations are simultaneously and identically impacted by catastrophic events. As explained in the Chapter 4, drought was the only catastrophic event we evaluated. Thus, we assessed the degree of redundancy across the EMR range by calculating the risk to extirpation at years 10, 25, 50 in each analysis unit and the spatial dispersion of those populations relative to future drought risk.

### 5.3.1 Methods

To assess the degree of redundancy in each analysis unit, we queried USFWS experts (Frank Durbian, Richard S. King, and Trisha Crabill) for the types of catastrophic events to consider. The experts identified drought, flooding, and disease as the most prominent, potentially catastrophic, events. Similarly, Pomara *et al.* (2014, 2095-2097) found extreme droughts and flooding as contributing factors in EMR extirpation events. However, we had insufficient information on flood risk (the magnitude of flood that would lead to extirpation) and disease risk (the likelihood of disease outbreaks, the factors that affect disease spread, and the magnitude of impact on EMR populations) to include it in our redundancy analysis. Thus, drought was the only catastrophic factor analyzed.

We consulted the U.S. Drought Monitor<sup>10</sup> for conducting our analyses. The Drought Monitor classifies general drought categories by intensity, with D1 being the least intense drought and D4 being the most intense drought. Based on feedback from internal and external experts, we analyzed the vulnerability of EMR extirpation due to D2-D4 drought intensities.

We used the following equation to calculate the probability of extirpation of all populations within an analysis unit (Ruckelshaus *et al.* 2002, p.312):

$$p(AUE) = (1 - e^{-\lambda*t})^n,$$

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<sup>10</sup> The U.S. Drought Monitor, established in 1999, is a weekly map of drought conditions that is produced jointly by the National Oceanic and Atmospheric Administration, the U.S. Department of Agriculture, and the National Drought Mitigation Center (NDMC) at the University of Nebraska-Lincoln.

where  $\lambda$  is the annual rate (frequency) of a catastrophic drought event,  $t$  is the number of years of concern, and  $n$  is the number of populations in the analysis unit. We determined  $\lambda$  by calculating the frequencies of a D3 drought occurring from 2000 through 2015 for the periods of November through March for all counties occupied by EMR historically and currently. Drought data were unavailable for Ontario, so we assumed the drought frequency ( $\lambda$ ) to be the same as that of Michigan. We used the forecasted number of self-sustaining populations for  $n$ , and 25 years as the duration of time,  $t$ .

Unlike our current conditions analysis in Chapter 4, the county locations for the extrapolated populations are unknown, and thus, we do not have state-specific  $n$  estimates and cannot model  $p(\text{AUE})$  based on the predicted probabilities of state-wide extirpation. Instead, we modeled 2 scenarios: 1) using the average  $\lambda$  for the states within an analysis unit, and 2) highest  $\lambda$  among the states within an analysis unit. The  $p(\text{AUE})$  is the probability of all populations within an analysis unit being extirpated ( $N=0$ , no individuals survive) by a catastrophic drought.

This model assumes identical and independent risks (i.e., catastrophic events are uncorrelated). Spatial dispersion, however, also influences the likelihood of a single event extirpating multiple populations (Ruckelshaus *et al.* 2002, 314). Thus, to account for spatial autocorrelation, we evaluated the predicted spatial dispersion of populations relative to drought risk. To do this, we overlaid the projected future distribution of likely to be extant populations with drought risks (drought frequencies) to evaluate the potential to lose geographical areas within an analysis unit.

Fundamental assumptions in our redundancy analysis include:

1. Drought frequencies in Ontario are same as those in Michigan.
2. No autocorrelation among populations. For example, the probability of a drought occurring at one population does not affect the probability of drought occurring at another even if in close proximity to each other.
3. Drought frequencies in 2000-15 represent the true risk of drought over the next 50 years.
4. Drought is the only potential catastrophic event to impact EMR populations.

### 5.3.2 Results - Extrapolating to All Populations

Probability of Analysis Unit Extirpation,  $p(\text{AUE})$  – The frequency of a D4 level drought is 0 for all analysis units, and thus the  $p(\text{AUE})$  is 0. The frequency of a D3 drought varies from 0.0 to 0.040 in the WAU, and the corresponding  $p(\text{AUE})$  using highest  $\lambda$  rates is 0.398 for year 10 and 0.631 for years 25 and 50; using average  $\lambda$  rates, the  $p(\text{AUE in 25 years})$  is 0.219 for 10 year and 0.468 for years 25 and 50 (Table 5.6). In the CAU and EAU, the  $p(\text{AUE in 25 years})$  is 0.0 for all time periods using average and highest  $\lambda$  rates (Table 5.6).

By using the highest and average  $\lambda$  rates, which are likely an over- and underestimate of risk; the true risk of AUE likely lies between these values. Thus, the  $p(\text{AUE})$  in WAU likely ranges from 0.219 - 0.398 for year 10 (0.468-0.631 for years 25 and 50)<sup>11</sup>. The  $p(\text{AUE})$  is likely 0.0 for the CAU and EAU because the  $\lambda$  rates—for both average and highest—are 0.0 or nearly so.

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<sup>11</sup> The pattern observed in the WAU—stabilizing  $p(\text{AUE})$  estimates following year 25—is expected. The  $p(\text{AUE})$  calculations are based on the forecasted number of self-sustaining populations; thus, as the forecasted number of self-sustaining populations is constant, so too are the  $p(\text{AUE})$  estimates.

Table. 5.6 a -b. a) The predicted p(AUE) is for 25 years (p(AUE) within 25 years of year 10, 25, and 50 years, respectively) given  $n$  # of populations and a D3 level drought.  $n$ = forecasted # of self-sustaining populations in each analysis unit at years 10, 25, and 50. b) The calculated frequencies of a D3 drought per State and per analysis unit. State frequencies are the average frequencies for the counties occupied by EMR within a State. Analysis Unit frequencies are the average frequencies for states within an analysis unit.

D3	p(AUE) given highest $\lambda$			p(AUE) given average AU $\lambda$			Average D3 Drought Risk		
	WAW	CAU	EAU	WAW	CAU	EAU	IA	IL	WI
Yr	$\lambda$	0.040	0.000	0.000	0.025	0.000	0.000	0.013	0.023
10		0.398	0.000	0.000	0.219	0.000	0.000	0.000	0.000
25		0.631	0.000	0.000	0.469	0.000	0.000	0.000	0.000
50		0.631	0.000	0.000	0.469	0.000	0.000	0.000	0.000
							IN	MI	OH
							0.000	0.000	0.000
							PA	MI-E	NY
							0.000	0.000	0.000
							CA		
									0.000

Using the frequency rates of a D2 drought, the p(AUE) is more likely in all 3 analysis units. In the WAW, the risk is very high: using average  $\lambda$  rates, p(AUE) is 0.847 for year 10 and 0.921 for years 25 and 50; using the highest  $\lambda$  rates, the p(AUE) is 0.940 for year 10 and 0.967 for years 25 and 50 (Table 5.7). The risk of p(AUE) in the CAU and EAU also increase but still remains low. In the CAU, the highest risk is 0.004 and in the EAU, 0.045 (Table 5.7).

Table 5.7a-b. Probability of Extirpation of all populations within the Western, Central, and Eastern Analysis units. The predicted p(AUE) is for 25 years (p(AUE) within 25 years of year 10, 25, and 50 years, respectively) given  $n$  # of populations and a D2 level drought.  $n$ = forecasted # of self-sustaining populations in each analysis unit at years 10, 25, and 50.

D2	p(AUE) given highest $\lambda$			p(AUE) given average AU $\lambda$			Average D2 Drought Risk		
	WAW	CAU	EAU	WAW	CAU	EAU	IA	IL	WI
Yr	$\lambda$	0.137	0.085	0.036	0.101	0.043	0.018	0.045	0.119
10		0.940	0.004	0.045	0.847	0.000	0.002	0.085	0.040
25		0.967	0.001	0.045	0.921	0.000	0.002	0.040	0.003
50		0.967	0.001	0.045	0.921	0.000	0.002	0.003	0.000
							PA	MI-E	NY
							0.000	0.036	0.000
							CA		0.036
									0.043
									0.100
									0.018

Spatial Dispersion of Populations – The above analyses give insights to the vulnerability of analysis unit-wide extirpation based solely on the number of populations. Analysis unit extirpation is also influenced by the spatial dispersion of populations. Thus, we also evaluated the spatial dispersion of populations relative to drought risk within an analysis unit.

Although we can predict the number of populations likely to be extant into the future, we are unable to predict specifically which populations are likely to be extant, and thus, the locations of the future populations. However, we know the locations of the presumed extant populations, so we can draw

inferences about what the future drought risk might be for future populations. If we assume the drought frequencies are unlikely to change, the additional risk due to inadequate spatial dispersion will, at best, stay constant, assuming no losses in spatial dispersion. If population clusters are lost, however, the risk may intensify over time. Given that the number of extirpated populations is projected to increase in time, it is reasonable to assume that population extirpations will lead to losses of population clusters.

In the WAU, using D3  $\lambda$  rates, 5 of the 8 populations clusters currently have notable risk of a catastrophic drought; using D2  $\lambda$  rates, 7 of 8 clusters are at high risk (Figures 4.3 and 4.4). Given the low number of populations likely to be extant in the WAU, the number of clusters that remain will likely be reduced, and this loss of spatial dispersion will exacerbate the already high p(AUE) estimates for the WAU. In essence, there will be fewer populations found in fewer locations, and all remaining populations will be located in areas prone to drought.

In the CAU and EAU, the risk of analysis unit extirpation from a D3 level drought will remain low unless the  $\lambda$  rates change into the future (see Climate Change section below). The risk from a D2 level drought, however, has implications for the CAU. Southern and northern portions of the CAU are at risk of a catastrophic D2 level drought occurring. Loss of populations in these areas may lead to portions of the CAU being extirpated.

### 5.3.3 Summary

The future redundancy—the number and spatial dispersion of self-sustaining populations—across the EMR range varies. In the WAU, the risk of analysis unit-wide extirpations from either a D2 or D3 catastrophic drought is high, given the low number of populations forecasted to be extant. Coupling this with a likely concurrent decline in population clusters (i.e., reduced spatial dispersion), the threat of analysis unit-wide extirpation is likely even higher. Thus, the level of redundancy in the WAU is projected to decline into the future.

Conversely, in the EAU, there is little to no risk of a D2 or D3 level drought, and consequently p(AUE) due to a catastrophic drought is very low. Thus, redundancy, from a catastrophic drought perspective, is not expected to decline over time in the EAU.

Similarly, in the CAU, there is little to no risk of a D3 catastrophic drought. The southern and northern portions of the CAU, however, are at risk of a D2 level catastrophic drought. Losses of populations in these areas, will increase the p(AUE) and may also lead to portions of the CAU being extirpated. However, the risk of analysis unit-wide extirpation will likely remain low given the presumed persistence of multiple populations scattered throughout low drought risk areas. Thus, from a drought perspective, the level of redundancy is not likely to be noticeably reduced in the CAU. A caveat to this conclusion, however, is that the forecasted decline in EoO suggests our data are too coarse to tease out whether the forecasted decline in populations will lead to substantial losses in spatial dispersion (increase covariance among populations), and thus, the risk of analysis-unit wide extirpation might be higher than predicted. Thus, the future trend in the level of redundancy in the CAU is less clear than for either the WAU or the EAU.

## Chapter 6: Synthesis

This Chapter summarizes the results from analyses and discusses the consequences for the future viability of EMR. We assessed viability for EMR by evaluating the ability of the species to maintain multiple (redundancy), self-sustaining populations (resiliency) across the full gradient of genetic and ecological diversity (representation) of the species by assessing the past, ongoing, and future factors operating on populations across the range.

Due to differences in methods, the results of the 1998 Rangewide Status Assessment (Szymanski 1998) cannot be directly compared to the SSA results, but the general conclusions and trends of 1998 comport with the findings of the SSA. As 1998, there was a 40% decrease in the number of counties occupied by EMR. Nine of 11 States/Provinces had lost more than 50% of their historical populations, and the remaining two states, 30% of their populations.

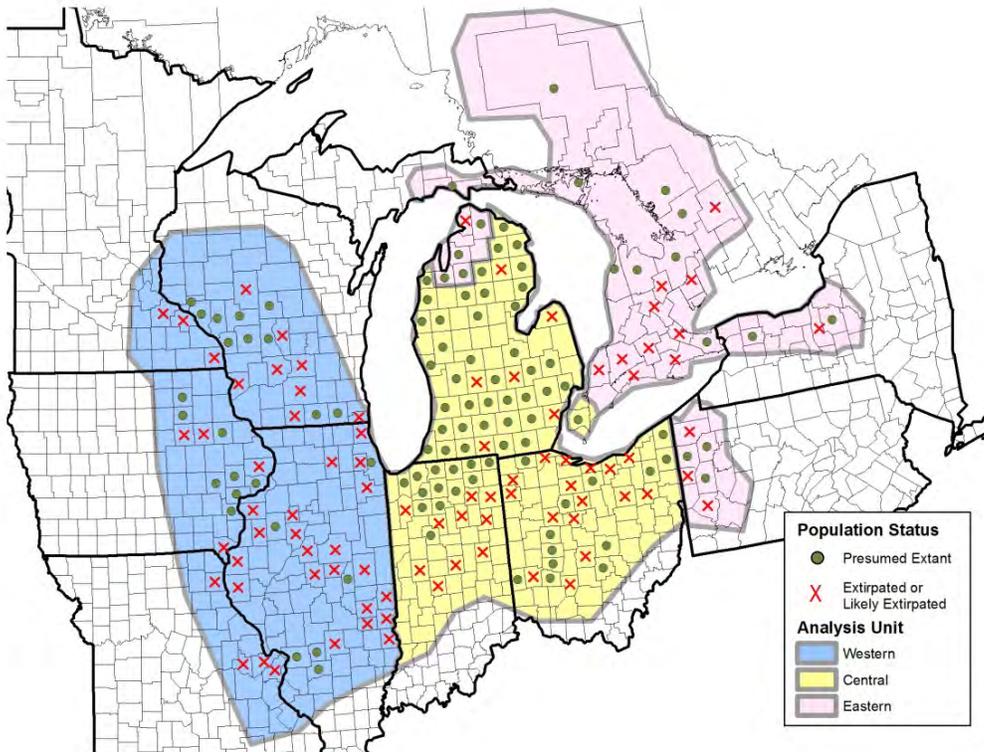
As predicted in the 1998 Rangewide Status Assessment, our analysis documented declines in EMR since historical time periods and predicts continued declines into the future. The EMR historically occupied sections of western New York, western Pennsylvania, southeastern Ontario, the lower peninsula of Michigan, the northern two thirds of Ohio and Indiana, the northern three quarters of Illinois, the southern half of Wisconsin, extreme southeast Minnesota, east central Missouri, and the eastern third of Iowa. Currently, EMR has been extirpated from Minnesota and Missouri and from counties in every State across the range including Ontario (Figure 6.1). The causes of the decline and the consequences for viability are described below.

### 6.1 Factors

Factors that affect EMR viability, in order of current relative significance to the species, include: habitat loss, vegetative succession, habitat fragmentation, road mortality, hydrologic alternation/water fluctuation, persecution, collection, habitat management practices including post-emergent prescribed fire and mowing, and disease. The loss of habitat was historically, and continues to be, the primary threat affecting this species either through development or vegetative succession. Habitat fragmentation is the second most commonly occurring factor at EMR sites, and vegetation succession is the third most common factor.

In addition to risk factors, beneficial actions are occurring at several sites. Conservation actions that reduce the risk factors include habitat management, site protection, and adoption of prescribed fire practices that avoid EMR mortality (*e.g.*, burning before snakes emerge from hibernacula). Currently there are two sites with Candidate Conservation Agreements committing the EMR site managers to conservation and management actions that fully address threats at those sites. Also, there are 22 sites with habitat restoration ongoing and another 18 sites with conservation plans in place. At these sites, however, it is unknown what actions are being implemented and to what extent the stressors might be mediated.

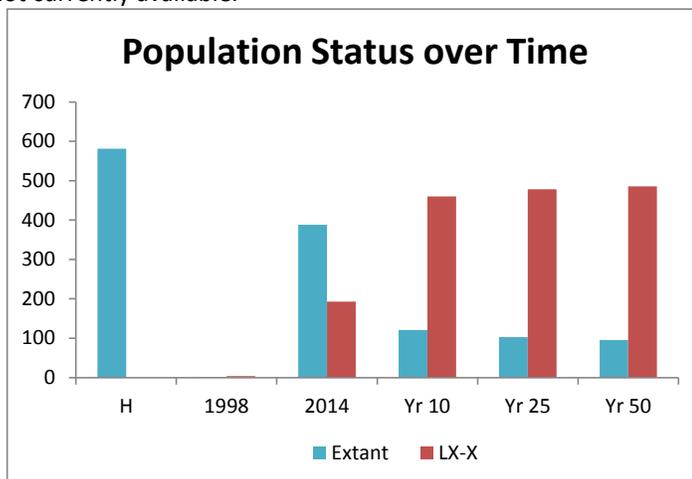
Figure 6.1. The geographical distribution of presumed extant (extant and unknown status) and extirpated EMR populations within each analysis unit.



## 6.2 Resiliency

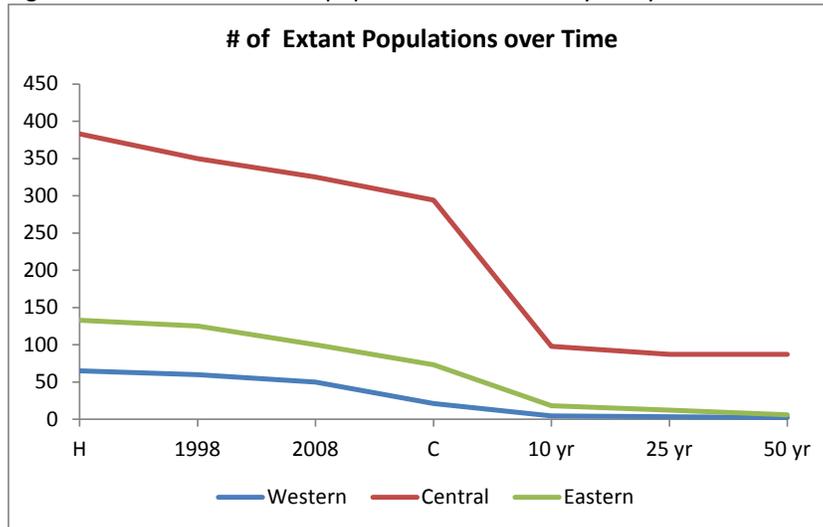
Rangewide, our analysis predicts continuation of the declining trend in number of extant populations (Figure 6.2).

Figure 6. 1. Predicted population status over time. The names of several populations have changed between 1998 and 2014; the 1998 data are being reconciled to comport with the 2014 data, and thus, are not currently available.



As of 2014, there is a 33% reduction in the number of populations from historical records and an 84% reduction in number of EMR populations is projected by year 50. The losses of extant populations occur throughout the species' range but are not uniformly distributed (Figure 6.3).

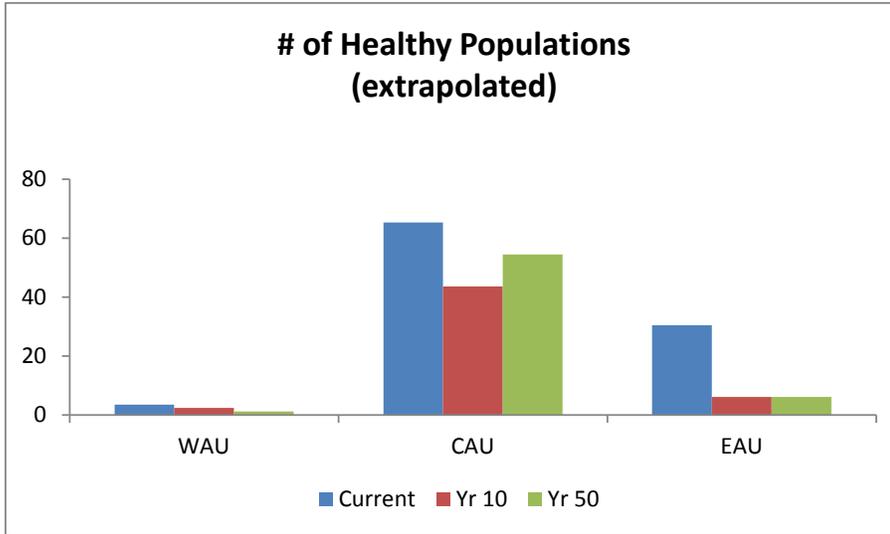
Figure 6.3. Number of extant populations over time by analysis unit



In the WAU, the number of populations has declined by 68%, and given the projected declining population trend and low persistence probabilities at many sites, by year 50, 96% of the historical populations are predicted to be extirpated. Only 1 population is likely to be self-sustaining (Figure 6.4). In the CAU, the number of populations has declined 23%, and by year 50, 77% of the historical populations are predicted to be extirpated and another 22 quasi-extirpated. Fifty-four populations are likely to be self-sustaining. In the EAU, the number of populations has declined 45%, and by year 50, 95% of the historical populations are forecasted to be extirpated and another 6 quasi-extirpated. Six populations are likely to be self-sustaining (Figure 6.4). The trend of increasing population extirpation is predicted rangewide, but losses are most marked in the western and eastern portions of the range.

These results do not include the impacts of disease, road mortality, persecution, and climate change (but see Climate Change section below). All of these factors are known to negatively affect EMR populations but these were not included in the Faust model, and thus, unable to be incorporated into our analyses.

Figure 6.4. Predicted future number of healthy populations in the analysis units. For the current time period, healthy refers to a population that is DGP robust. For the future time periods (years 10, 25 and 50), healthy refers to a population that is self-sustaining ( $N_f > 50$ ,  $\lambda \geq 1$ , and  $p(P) \geq 0.90$ )



### 6.3 Representation

The number of populations alone does not give a full picture of a species' viability. The distribution of these populations is important. Genetic studies indicate that EMR is highly genetically diverse across the range; if this variation represents evolutionary potential we should seek to preserve populations across the historical range. Losses of geographical areas may equate to irreplaceable losses of adaptive diversity of EMR.

Loss of populations from historical records to today has led to a loss of geographic range; EMR is extirpated from MN and MO. Rangewide, extent of occurrence is predicted to decline by 65% by year 50 (Table 6.1). Notably, this loss is likely greater than estimated because of the methodology used in our analysis (estimated at county level).

Table 6.1. Extrapolated percent reduction in Extent of Occurrence (EoO) from historical conditions.

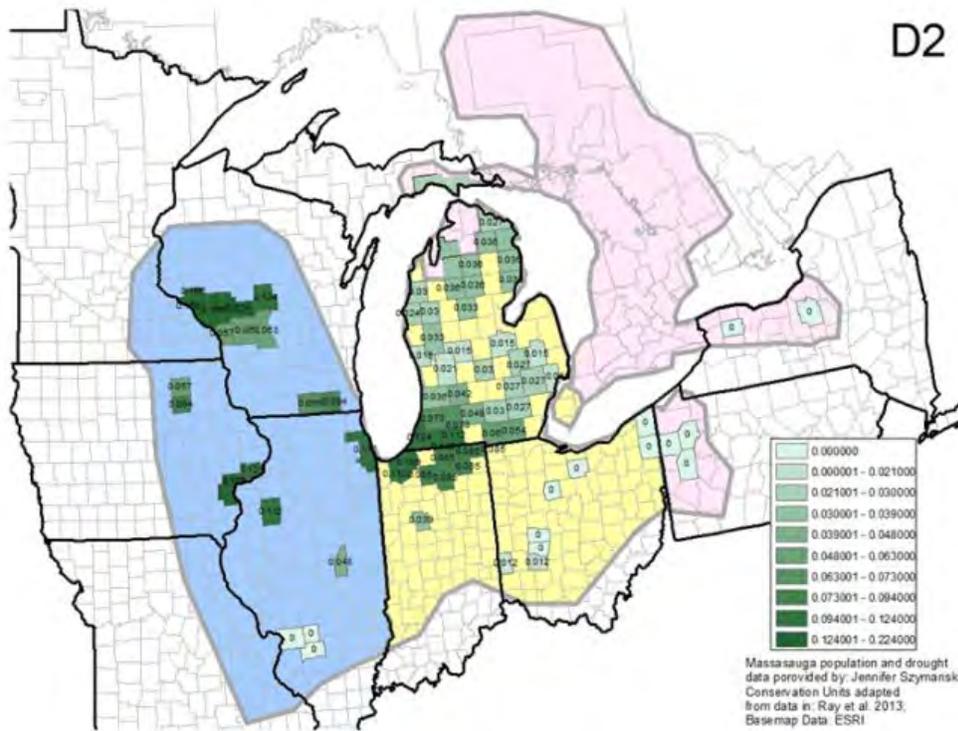
Analysis Unit	Current	Year 10	Year 25	Year 50
Western	63%	71%	83%	83%
Central	33%	51%	57%	64%
Eastern	25%	44%	51%	51%
Rangewide	46%	55%	63%	65%

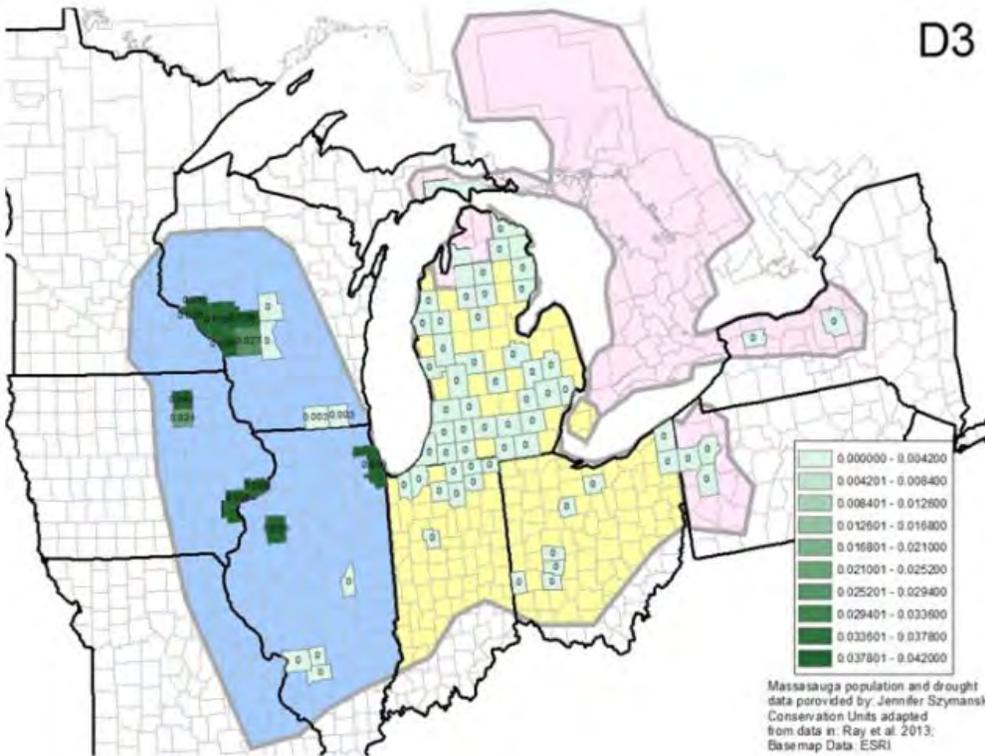
The reduction in extent of occurrence has not been uniformly distributed (Figure 6.1). The greatest losses have occurred in the western portion of the species range, which historically represented 27% of the EMR range. Today the western analysis unit represents only 19% of the species range, and by year 50, it is forecasted to represent only 13% of the range.

## 6.4 Redundancy

Redundancy is the ability to withstand catastrophic events, and is a function the number of populations and the spatial distribution of these populations relative to the spatial occurrence of catastrophic events. Drought was the only catastrophic event analyzed. Loss of resiliency has also led to loss of redundancy; again, the implications are not uniformly distributed across the analysis units, with the Western analysis unit being the most vulnerable. The WAU has a 42 to 92% chance of unit-wide extirpation from a D2 and D3 droughts alone, and higher probabilities of losing substantial portions of the WAU. The CAU ad EAU has lower probability, but losing portions of the CAU are possible.

Figure 6.5. Annual D2 and D3 frequencies for counties with EMR populations. Darker shades represent higher annual frequency rates





## 6.5 Consequences of Climate Change

A recent Climate Change Vulnerability Analysis (CCVA) prepared for the U.S. Fish and Wildlife Service Upper Midwest and Great Lakes Landscape Conservation Cooperative by Pomara and colleagues (Pomara *et al.*, Technical Report) evaluated the relationship between adult EMR survival and climate variables including summer flood risk and winter drought, as well as land cover, as an ongoing factor affecting recent historical patterns of persistence and extirpation of EMR populations. They found that winter drought and summer flood risk are the strongest climatic predictors of adult EMR survival according to the Climate Change Vulnerability Analysis. These drivers, along with land cover changes, are able to predict the historically observed extirpation front occurring along a gradient from the southwest to northeast parts of the current EMR range.

Demographic data on EMR populations were derived from 17 datasets from field-based studies of vital rates and survivability of adults in winter hibernation and summer active seasons throughout the range of the species. Climate variables were collected from various sources that could provide data for winter drought conditions (NCAR Palmer drought severity index), flood risk (maximum annual precipitation World climate Research Programme), extreme temperatures (maximum annual temperatures World Climate Research Programme), and elevation (NASA shuttle Radar Topography mission). In addition to climate, land cover was evaluated using the proportion of surrounding landscape dominated by agricultural and urban land (North American Land Change Monitoring System classification).

They found that low EMR adult survival rates were associated with high winter drought risk, high summer flood risk, and proportion of human-modified land cover (Figure 6.5). Using the predicted survival rates, they tested the relationship to climate and land cover by simulating (back-casting) adult survival at 189 sites with known extant and extirpated populations from 1950 to 2008 and compared the results to the actual population status data. The simulations correctly classified 75% of 189 sites examined as extant or extirpated/imminent risk. The most accurate predictor of current population status for EMR populations was a model with both climate (winter drought and summer flooding) and land cover variables included rather than a climate-only or land cover-only model, indicating that historical extirpations and current at-risk populations are linked to climate conditions.

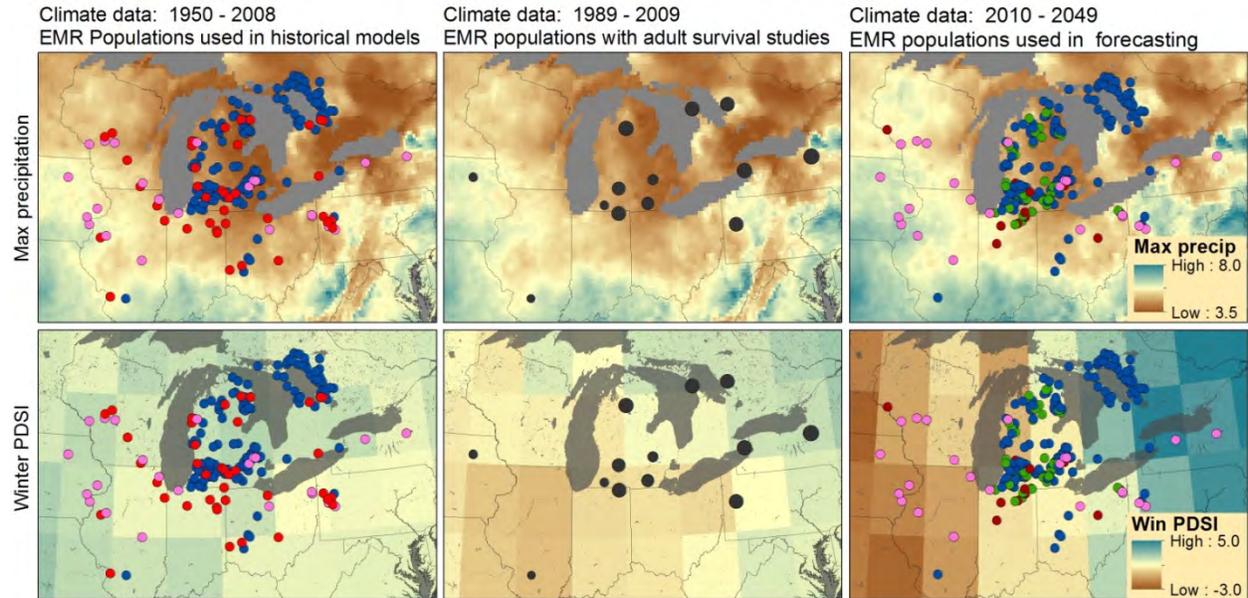
Results of the Climate Change Vulnerability Analysis indicate that populations in the southwestern parts of EMR range are extremely vulnerable to climate through increasing intensity of winter drought and increasing risks of summer floods. Populations in the eastern and central parts of EMR range are vulnerable to climate variables but less than the southwestern populations. Northeastern populations are least vulnerable to climate change.

For the purpose of this SSA we have relied on the results of the Faust model to predict population viability, and we have addressed climate change related threats of flooding and drought only to the extent that drought is modeled as a potential catastrophic event. The Pomara *et al.* analysis, which modeled survival rates using a different methodology and assumptions, and which treated effects of climate change as an ongoing, rather than catastrophic, stressor, produced comparable predictions of distribution and extent of future population persistence and loss across the species' range.

The Pomara *et al.* models also predicted declines—though more moderate—for many extant populations that are not otherwise considered to be in imminent risk, suggesting that climatic factors such as drought and flooding risk are also impacting these populations currently not believed to be at risk. This again suggests that our assessment may be optimistic in terms of the current resiliency of populations, especially in Indiana and Michigan.

Their results suggest that our future predictions may be optimistic as we: 1) looked only at drought risk, yet flooding risk is influential as well according to their results, and 2) we held drought frequencies constant through time, which is contrary to the likelihood of increasing drought frequency as predicted by climate models.

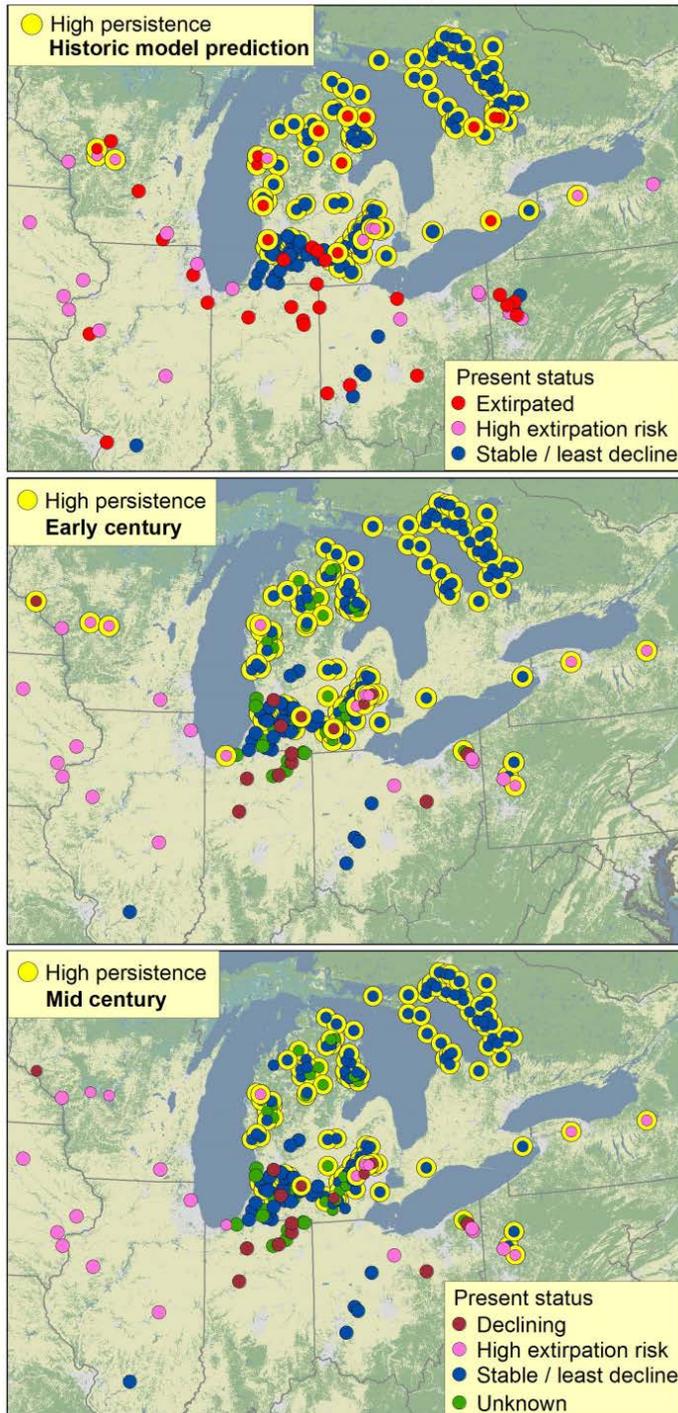
Figure 6.6. Climate data and the distribution of EMR populations. Reprinted with permission from Pomara *et al.*, Technical Report.



- Population sites
- Extant, showing least decline
  - Extirpated
  - Small, at high risk of extirpation
  - Status highly uncertain
  - Sites where survival studies occurred

**Climate data and the distribution of Eastern Massasauga (EMR) populations.** Winter drought and summer precipitation highs were strongly associated with EMR adult survival rates, reflecting depressed survival under conditions of drought and flooding. The top panels show mean precipitation highs (mm/day), and the lower panels show the Palmer Drought Severity Index (PDSI) during winter months (November-March); lower PDSI values indicate higher drought likelihood. Left panels show mean values across the years for which historical demographic models were tested; center panels show mean values for the period during which adult survival studies took place; and right panels show mean predicted values from business-as-usual climate projections for the forecasting period. In center panels, larger dot size indicates higher active-season survival. Sites with highly uncertain population status were not used for historical model testing, but forecasts were made at those sites.

Figure 6.7. EMR population forecasts based on climate forecasts and land cover. Reprinted with permission from Pomara *et al.*, Technical Report.



Eastern Massasauga population forecasts based on climate forecasts and land cover. Extirpations over the past five decades have mostly occurred in the southern and western parts of the range. Today, few robust populations remain in those regions. The top panel shows this pattern, with sites colored according to their 2008 status. Yellow circles indicate sites that were predicted by historical models to persist at least until 2008, giving an indication of model accuracy. The lower two panels show forecasts for the early (2020-29) and middle (2040-49) 21<sup>st</sup> century, using population models coupled with climate forecasts. Yellow again indicates high predicted persistence likelihood. All sites with extant populations in 2008 were included in forecasts; those with uncertain status were not used in historical models.

## 6.6 Concluding Remarks

In assessing the viability of species, it is insufficient to simply look at extinction risk based on numbers alone. It is important to understand what these numbers represent. How many populations are there? How healthy are these populations given ongoing and future stressors? Are there sufficient numbers of healthy populations to ensure irreplaceable losses of adaptive diversity do not occur due to environmental stochasticity and catastrophic events?

Given the loss of populations to date, portions of the EMR range are in imminent risk of extirpation in the near term. Specifically, our analysis suggests there is a high risk of extirpation of the WAU and southern portions of CAU and EAU within 10-25 years. Although self-sustaining populations are expected to persist, loss of populations within the CAU and EAU are expected to continue as well, and thus, at risk of extirpation in the future. These losses have led to reductions in resiliency and redundancy across the range and may lead to irreplaceable loss of adaptive diversity across the range of EMR, thereby leaving EMR less able to adapt to changing environment into the future. Thus, the viability of EMR has and is projected to decline over the next 50 years.

## 6.7 Uncertainty

The Endangered Species Act “best scientific data available” standard requires us to evaluate the viability of a species even in the situation where complete information may be lacking. For our analysis, therefore, we used expert judgment where empirical data was lacking, used modeling to form predictions about future population status, and extrapolated available information on some populations to a larger set of populations where we did not have any information. Our assessment and predictions of potential future population trends for this species are based on extrapolations from a model that is itself built on a combination of empirical data, expert judgment, and assumptions. In Table 6.2, we identify key sources of uncertainty and indicate the effect of the associated assumptions on our viability assessment.

Table 6.2 Effect of data limitations on species viability estimates (present and future). Both means we are uncertain of whether assumption will over or underestimate viability. NA = not applicable.

	Effect Upon Viability	
	Current Conditions	Future Conditions
<b>Faust Model Assumptions:</b>		
Vital rates are not autocorrelated among years, e.g., a “bad” year does not influence the next year’s rate.	NA	Overestimates
No spatial variation in vital rates at sites across the range, other than those considered in the early-maturing and late-maturing models.	NA	Overestimates
No spatial variation in the magnitude of impact.	NA	Either
Magnitude of impact from multiple factors is additive rather than multiplicative or synergistic	NA	Overestimates
No temporal variation in the magnitude of impact (successive years of a stressor occurring does not increase or decrease the magnitude of impact)	NA	Overestimates
<b>SSA Assumptions:</b>		
The 121 populations with unknown status are extant	Overestimates	Overestimates
Some $N_0$ were based on expert judgment	Either	Either
Populations with $N_0 > 50$ are considered DGP.	Either	NA
Populations with $N_F > 50$ , $\lambda \geq 1$ , and $p(P) \geq 0.90$ are self-sustaining.	NA	Either
Used the early maturing model results for the 21 populations for which the appropriate dynamic was unknown.	NA	Overestimates
The magnitude of impact and the pervasiveness of threats are constant through time	NA	Overestimates
Risk factors (e.g., road mortality, persecution, disease) not modeled by Faust et al., (2011) were not considered in our analysis.	NA	Overestimates
Conservation actions not part of a CCAA were not considered	NA	Underestimates
The 3 analysis units and wide distribution within the analysis units capture the breadth of EMR's adaptive diversity	Either	Either
The loss of an analysis unit and loss of extent of occurrence within an analysis unit constitutes loss of adaptive diversity.	Either	Either
Drought is the only climate change related impact	Overestimates	Overestimates
Drought frequencies remain constant through time	Overestimates	Overestimates

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