Issues and Perspectives

Development of a Species Status Assessment Process for Decisions under the U.S. Endangered Species Act

David R. Smith*, Nathan L. Allan, Conor P. McGowan, Jennifer A. Szymanski, Susan R. Oetker, Heather M. Bell

D.R. Smith, U.S. Geological Survey, Leetown Science Center, 11649 Leetown Road, Kearneysville, WV 25430; drsmith@usgs.gov; 304-724-4467; ORCID 0000-0001-6074-9257

N.L. Allan, U.S. Fish and Wildlife Service, 10711 Burnet Road #200, Austin, TX 78758; nathan_allan@fws.gov; 512-490-0057

C.P. McGowan, U.S. Geological Survey, Alabama Cooperative Fish and Wildlife Research Unit, Auburn University, Auburn, AL 36849-5418; cmcgowan@usgs.gov; 334-844-4796

J.A. Szymanski, U.S. Fish and Wildlife Service, 555 Lester Avenue, Onalaska, WI 54650; jennifer_szymanski@fws.gov; 608-783-8455

S.R. Oetker, U.S. Fish and Wildlife Service, 1875 Century Blvd NE, Atlanta, GA 30345; susan_oetker@fws.gov; 404-679-7050

H.M. Bell, U.S. Fish and Wildlife Service, 134 S. Union Blvd, Lakewood, CO 80228; heather_bell@fws.gov; 303-236-4514

Abstract. Decisions under the U.S. Endangered Species Act (ESA) require scientific input on the risk that the species will become extinct. A series of critiques on the role of science in ESA decisions have called for improved consistency and transparency in species risk assessments and clear distinctions between science input and policy application. To address the critiques and document the emerging practice of the U.S. Fish and Wildlife Service, we outline an assessment process based on principles and practices of risk and decision analyses that results in a scientific report on species status. The species status assessment (SSA) process has three successive stages: 1) document the species' life history and ecological relationships to provide the foundation for the assessment, 2) describe and hypothesize causes for the species' current condition, and 3) forecast the species' future condition. The future condition refers to the species' ability to sustain populations in the wild under plausible future scenarios. The scenarios help explore the species' response to future environmental stressors and to assess the potential for conservation to intervene to improve its status. The SSA process incorporates modeling and scenario planning for prediction of extinction risk and applies the conservation biology principles of representation, resiliency, and redundancy to evaluate the current and future condition. The SSA results in a scientific report distinct from policy application, which contributes to streamlined, transparent, and consistent decision making and allows for greater technical participation by experts outside of the USFWS, for example by state natural resource agencies. We present two case studies based on assessments of the eastern massasauga rattlesnake (*Sistrurus catenatus*) and the Sonoran Desert tortoise (*Gopherus morafkai*) to illustrate the process. The SSA builds upon the past threat-focused assessment by including systematic and explicit analyses of the species' future response to stressors and conservation, and as a result, we believe it provides an improved scientific analysis for ESA decisions.

Key words: species status assessment; SSA; risk assessment; conservation; U.S. Endangered Species Act; decision making

Received May 12, 2017; Accepted: January 18, 2018; Published Online Early: January 2018; Published: xxx

Citation: Smith DR, Allan NL, McGowan CP, Szymanski JA, Oetker SR, Bell HM. 2018.?. Development of a species status assessment process for decisions under the U.S. Endangered Species Act, *Journal of Fish and Wildlife Management* 9(1):xx-xx; e1944-687X. doi:10.3996/052017-JFWM-041

*Corresponding author: drsmith@usgs.gov

Running Title: Species Status Assessment for Endangered Species Decisions

Introduction

Decisions that support the purpose of the Endangered Species Act (ESA 1973, as amended) require a scientific assessment of the species' risk of extinction or, conversely, its probability of persistence (Carroll et al. 1996; Doremus and Tarlock 2005; Waples et al. 2013; definition of *species* from the ESA (1973) is presented in Table 1 along with definitions for other ESA-related terms, which are shown in italics.) The purpose of the ESA (ESA 1973, 16 USC. §1531-1544) is to conserve *threatened* and *endangered species* and the ecosystems upon which they depend (section 2(b) of the ESA; Table 1). The Secretaries of the Interior and Commerce acting through the U.S. Fish and Wildlife Service (USFWS) and National Oceanic and Atmospheric Administration (NOAA) make decisions to fulfill that purpose. Analysts present results from a species assessment to decision makers who then make a policy judgment based on the ESA and in light of that scientific information (Shaffer 1987; Rohlf 1991; Carroll et al. 1996; Doremus 1997; Vucetich et al. 2006; Gregory et al. 2013).

The past practice for species assessment within the USFWS often focused on *threats* without an explicit analysis of the species' response to the *threats* (Andelman et al. 2004). Andelman et al. (2004) reviewed nine protocols for assessing species risk and concluded that the *threats*-focused assessment used at the time by the USFWS had low repeatability and transparency and was based on *threat* occurrence without explicit prediction of the species' future response to those *threats*. Also, a series of critiques on the role of science in ESA decisions have called for improved consistency and transparency in the procedures for assessing species status (Carroll et al. 1996; Andelman et al. 2004; Waples et al. 2013; Lowell and Kelly 2016; Murphy and Weiland 2016). Lowell and Kelly (2016) and Murphy and Weiland (2016) questioned the degree to which the mandate for *best available science* has been met under past practice. A team from USFWS and U.S. Geological Survey (including the authors and those listed in the Acknowledgements) with experience in species assessments, ESA applications, and decision analysis developed an assessment process to address issues raised by the critiques and to update the process to reflect the advances in scientific practices. The new process, termed the species status assessment or SSA, is framed around a systematic analysis of the species' current and future responses to stressors and conservation efforts. The USFWS directed the use of the SSA for ESA decisions through a USFWS Director's memorandum (USFWS 2016a). In this paper we document the SSA process, which is now an emerging practice within the USFWS (Earl et al. 2017; McGowan et al. 2017; Evansen et al. 2017). The SSA is relevant to species risk assessment broadly, but our primary focus is on ESA decisions conducted by USFWS.

At its most basic level, the SSA is species focused rather than *threat* focused, and it is designed to analyze the available scientific information to estimate the species' current condition and forecast the species' future condition for ESA decisions (Figure 1). *Threats* are potential explanations for current and future condition rather than the end point of the assessment. Sequentially, the SSA results in a description of the fundamental ecological and evolutionary relationships between the species and its environment to hypothesize explanations for the current condition and provide a foundation for forecasting future condition. In the development of the assessment process, we incorporated the recommendations from current literature on species assessment regarding population viability analysis (Shaffer and Stein 2000; Redford et al. 2011; Waples et al. 2013; Wolf et al. 2015; Murphy and Weiland 2016), a broad role for modeling (Starfield 1997; Ruhl 2004; Addison et al. 2013), and scenario planning relevant to the decision context (Peterson et al. 2003; Duinker and Greig 2007; Goodwin and Wright 2014; IPBES 2016).

As pointed out by Doremus (1997), Rohlf (2004), Vucetich et al. (2006), Woods and Morey (2008), Waples et al. (2013), and others, ESA decisions involve both scientific and normative (policy) dimensions (Table 2). Conflating the roles of science and policy can create unnecessary confusion both within the agencies charged to make ESA decisions and with the public and partners who are affected by those decisions (Robbins 2009; Wilhere 2011; Waples et al. 2013; Boyd et al. 2016). The SSA results in a scientific report distinct from the application of policy, which is a departure from past USFWS practice in many instances (Waples et al. 2013).

In this paper, we focus on the conceptual framework for the SSA and present two case studies to illustrate how the process can be implemented. We avoid being overly prescriptive so that practices can adapt to specific cases and adopt innovative techniques and methodologies; SSA guidance is kept up-to-date and available online (USFWS 2016b). The structure of this paper first presents a process followed by example applications of that process (case studies). The case studies on eastern massasauga rattlesnake (*Sistrurus catenatus*) and Sonoran Desert tortoise (*Gopherus morafkai*) are among early applications, and insights gained have been used to improve the SSA process. We point out the links between the SSA and the five-factors of the ESA (Table 1) and conservation efforts. Finally, we discuss the roles of science and policy in ESA decisions.

The SSA Process

An SSA is intended to be initiated when a species is first considered for *endangered species* designation and to follow the species (USFWS 2016a) so that the information and analyses are available for subsequent ESA determinations (Figure 1). The SSA results in a scientific report that describes the risk of extinction, which a decision maker can then use along with policy judgment to determine legal status under the ESA (Doremus 1997; DeMaster et al. 2004; Doremus and Tarlock 2005; Vucetich et al. 2006; Robbins 2009). The initial SSA report can be adapted to the needs of a particular decision (*listing*, *recovery planning*, *consultation* and *permitting*; Table 1) and updated with new data and information (Figure 1). The level of detail in an SSA will depend on the amount and quality of available data (Doremus and Tarlock 2005). In some cases, the available information is sufficient for assessing risk on a continuous scale. In other cases, because of data limitations only categorical levels of risk can be assessed.

The three stages of an SSA

The SSA process involves three successive stages: 1) the species' ecology, 2) the species' current condition, and 3) the species' future condition.

Stage 1: The species' ecology. The first stage of an SSA is an exploration of the species' life history and ecology, which lays the foundation for the next stages of the process. Stage 1 results in a description of the life history, including trophic niches, reproductive strategies, biological interactions, and habitat requirements to determine how individuals at each life stage survive and reproduce. The SSA identifies areas representing significant ecological, genetic, or life history variation (i.e., the ecological settings) informed by historical as well as present distribution. The entire range of historical conditions under which the species was presumably self-sustaining serves as a starting point to understand how the species functions (or functioned) to maintain populations across its range (Seminoff et al. 2015)

Stage 2: The species' current condition. The next stage of the SSA is to describe the current condition of the species' habitat, demographics, and distribution. Stage 2 results in an empirical description of the current 1) population structure, distribution, abundance, demographic rates, diversity (ecological, genetic, life-history), and habitat, 2) changes from historical to current condition (i.e., trends), and 3) explanations or hypotheses of the causes and effects of stressors and conservation efforts that resulted in the current condition.

Stage 3: The species' future condition. In the final stage, an SSA results in the prediction of the species' response to a range of plausible future scenarios of environmental conditions and conservation efforts. This step entails an analysis of future plausible scenarios of stressors and conservation efforts to project consequences on the species' ability to sustain populations in the wild over time. The predictions start at the current condition estimated in Stage 2 and project forward based on the information developed in Stage 1 on how the species interacts with its environment. The metrics used for future condition align with metrics used in the prior stages and include demographics (abundance and population growth or productivity), distribution, and diversity (ecological, genetic, life-history), which are core autecological parameters that measure the relationships between a species and its environment. The numerical resolution and spatial and temporal scale of the metrics will depend on data availability and the information needed for the decision context.

The future condition is unavoidably uncertain because (1) future events are inherently probabilistic (aleatory uncertainty) and (2) the knowledge of the species' response to future scenarios is imperfect due to sampling and measurement error, competing hypotheses about ecological relationships, and imprecisely defined terms and categories (epistemic and linguistic uncertainty; Taylor et al. 2002; Carey and Burgman 2008; Lukey et al. 2010; McGowan et al. 2011; Phillips-Mao et al. 2016; Murphy and Weiland 2016). The scenarios developed in Stage 3 represent an important tool for incorporating uncertainty in species risk (Peterson et al. 2003; Duinker and Greig 2007; Goodwin and Wright 2014; Rowland et al. 2014). Scenarios are designed to explore the species' response to environmental stressors (including climate change) and interventions by conservation efforts that could ameliorate the stressors (Duinker and Greig 2007; Fordham et al. 2013; Gregory et al. 2013; Phillips-Mao et al. 2016; IPBES 2016). Uncertainty in forecasts within scenarios comes from variability in model predictions or expert judgments (Taylor et al. 2002; Refsgaard et al. 2007; McGowan et al. 2011; Drescher et al. 2013). The combination of variation among scenarios and uncertainty in forecasts within scenarios is used to explore the species' risk profile, in other words, the plausible range in the species' response to future stressors and conservation efforts.

Principles and practices

Representation, resiliency, and redundancy. The SSA process applies the conservation biology principles of *representation*, *resiliency*, and *redundancy* (we refer to them here as the 3Rs) to evaluate the current and future condition of the species (Shaffer and Stein 2000; Redford et al. 2011; Waples et al. 2013; Wolf et al. 2015; Earl et al. 2017). In general, species risk will decrease, or at least does not increase, with increases in *representation*, *resiliency*, and *redundancy*. Shaffer and Stein (2000) composed a hierarchical order to the 3Rs as they relate to viability; first, conserve some of everything (i.e., *representation*) and then save enough to last (i.e., *resiliency* and *redundancy*). From a decision analysis perspective, the 3Rs can be viewed as means objectives for the overarching fundamental objective of sustaining populations in the wild. The fundamental objective is what we want to achieve, and the means objectives are essential ways to achieve what we want (Gregory et al. 2012). Shaffer and Stein (2000) related *representation* to the conservation of a species within the array of different environments in which it occurs or areas of significant ecological, genetic, or life-history variation, termed here as ecological settings (Carroll et al. 2010; Wolf et al. 2015). We suggest that for ESA decisions, *representation* uses diversity as a proxy for adaptive capacity. *Resiliency* refers to the ability of a population to withstand stochastic disturbance events; thus, *resiliency* is related to the demographic ability to absorb and bounce back from disturbance and persist at the population or meta-population scale. *Redundancy* spreads risk among multiple populations or areas to minimize the risk due to large-scale, high-impact (i.e., catastrophic) events. Thus, the 3R concept helps to construct a risk assessment that takes into account demographic factors, distribution or spatial structure, and diversity. Demographic factors (abundance, survival, productivity, and ultimately intrinsic population growth rate) contribute to the ability to absorb disturbance and persist (*resiliency*). Spatial structure contributes to *redundancy* through increased distributional extent by spreading risk across the broader landscape and adds to *resiliency* by increasing connectivity among meta-populations. Diversity, as represented in genetic, geographic, or life-history variation, contributes to adaptive capacity and can inform decisions related to the ESA concepts of distinct population segment (USFWS and National Marine Fisheries Service 1996) and significant proportion of the range (USFWS and National Marine Fisheries Service 2014; Earl et al. 2017).

The NOAA use abundance, productivity, spatial structure, and diversity as the criteria to determine viable populations in ESA-relevant assessments (Waples et al. 2013). The criteria relate to the 3Rs in this way: abundance and productivity correspond to *resiliency*, spatial structure contributes to *resiliency* and *redundancy*, and diversity relates to *representation*. Wolf et al. (2015) recommended the following metrics within the *recovery planning* context: abundance, productivity, and connectivity for *resiliency*; number of populations for *redundancy*; and occupancy across the gradient of genetic, ecological, or life-history diversity for *representation*.

Link to the five factors. The ESA identifies five statutory factors to consider as causes for endangerment (see section 3 for definition of endangerment and section 4(a) for the five factors; Patrick and Damon-Randall 2008). The *five factors* are listed in Table 1. The ESA requires that a *listing* determination identify the factors causing endangerment, but the law does not specify how the factors should be considered. The SSA approach within each stage of the assessment is to hypothesize and evaluate the causal relationships between the factors and the species' response. Stage 1 describes the influence of habitat (factor A) and other natural factors (e.g., factor C: disease and predation) on the species' ecology. Stage 2 identifies and evaluates any of the factors that are hypothesized to have led to the species' current condition. Stage 3 incorporates the factors, which are hypothesized to have population-level effects, into scenarios used to forecast the species' future condition. The SSA considers not just the presence of the factors, but assesses to what degree they influence risk. Because the SSA uses metrics for demographics, distribution and diversity, the effect of multiple stressors is inherent in the assessment and helps to assess how populations and ultimately the species responds cumulatively to the interactive effects of stressors and conservation efforts included in the future scenarios.

Link to conservation efforts. The SSA incorporates conservation efforts in the assessment, and results in a description of how conservation efforts influence the species' current condition. The SSA can incorporate conservation efforts in the scenarios used to forecast the species' future condition. In the context of a *listing* decision, for example, the SSA can be used to evaluate sufficient regulatory mechanisms that satisfy factor D of the *five factors* (Table 1). An evaluation of the species' response to conservation efforts, including those with uncertain implementation and effectiveness, may help to develop and evaluate conservation strategies. Prior to *listing*, the assessment of what could be done to improve the species' condition provides opportunities to carry out *candidate conservation* (Table 1) actions in advance of future ESA decisions. In addition, if the species is subsequently listed for protection under the ESA, then the conservation strategies evaluated in the SSA can be used in *recovery planning*.

Role of models and modeling. Projecting the species' future condition, which is integral to all ESA decisions, and thus to an SSA, relies broadly upon models and modeling (Starfield 1997). The resolution of the available information (including covariates, response variables, and uncertainties) and purpose of the modeling determine the type and complexity of the models. The utility of a model to an SSA and an ESA decision depends on how the available information, derived from data or expert judgment, are analyzed and how the outputs are interpreted.

Uncertainty and low quality data do not prohibit the utility of a model as long as the sensitivity of model outputs to violations of the underlying assumptions or sources of uncertainty are assessed and effectively communicated to the decision makers. Even simple models can be quite useful. For example, conceptual models (expressed as influence diagrams) are useful for illustrating life history or graphically relating environmental factors to a species' condition. Also, McCarthy et al. (2004) found that predictions of extinction risk were less biased when based on an explicit model compared to subjective judgment, even if the model is necessarily simplistic because of sparse data. Habitat models can translate ongoing stressors to future habitat upon which the species depends (Copeland et al. 2009). Population models can project future condition as a function of future stressors and conservation (Akçakaya and Sjögren-Gulve 2000; Runge et al. 2007; Murphy and Weiland 2016). Models provide an explicit, transparent and, repeatable method of analysis, which facilitates a thorough peer review of both the SSA methodology and results (Rohlf 1991; Starfield 1997; Ruhl 2004; Addison et al. 2013). Models also provide a structure to integrate new information in subsequent assessments. Within an SSA the specific models should be parsimonious and built to meet the specific needs of the SSA within the decision context (Starfield 1997; Burgman and Yemshanov 2013).

Role of expert judgment. As Burgman (2016) advised, only after all other sources of data are exhausted should an assessment turn to expert judgment to fill in gaps. But data gaps are common in *endangered species* assessments. Formal elicitation of expert judgment is a complicated endeavor that involves the careful determination of specific information needs that may call for expert judgment, identification and preparation of experts, and elicitation and characterization of uncertainty in judgments. Fortunately, recommended practices for eliciting expert judgment have been recently published (U.S. Environmental Protection Agency 2011; Drescher et al. 2013; Burgman 2016). The first considerations when eliciting expert judgment is to determine the information needs and the format to acquire that information – workshop, interviews, or questionnaires. Identification of experts can start with a review of relevant literature. Professional credentials, position, the areas of expertise, and relevant experience can be used for selection criteria to help ensure that scientific experts are familiar with the topic and that the choices were transparent, unbiased, and captured a broad diversity of expertise and professional judgments related to the topic. Detailed technical questions are developed on topics germane to the assessment. After establishing a common understanding of the information context, experts are asked, facilitated through in-person, phone, or online discussions, about facts and information based on their individual, professional knowledge on specific topics. Group consensus is not sought. Instead, the variation in judgment among experts is an important source of uncertainty. Techniques to capture uncertainty within and among experts include 4-point elicitation or likelihood point method coupled with the Delphi method (Burgman 2016).

Communicating SSA results to decision makers. Decision makers are informed by the SSA results and apply ESA policies to the decision at hand (Table 2). The decision makers compare the risk inferred from the SSA to the relevant ESA regulatory standards and definitions (Waples et al. 2013; Murphy and Weiland 2016). To take full advantage of the SSA, the decision makers need to accept the overall analytical process and understand the SSA results, as well as the strength of data and any assumptions used to develop any models used for estimation or prediction. Therefore, it is important that SSA biologists and decision makers discuss and agree to the metrics, future scenarios, and time frames as influenced by the policy on *foreseeable future* (Table 1). It is incumbent upon the analyst to present the levels of uncertainty for the future condition to the decision makers in the agreed upon metrics. A practical interpretation is that the uncertainty represents the plausible range in the species' future condition across the scenarios including prediction variance within each scenario. The decision maker also needs to understand the underlying assumptions and data limitations to avoid inappropriate conclusions and inference (Murphy and Weiland 2016). The characterization of uncertainty, such as the probability associated with confidence level or the quantitative interpretation of what is plausible, relies on scientific judgments that can be based on professional norms (Doremus and Tarlock 2005; Anderson et al. 2001; Refsgaard et al. 2007). For instance, Anderson et al. (2001) advised that a confidence level (1-α)% should be explicitly reported but acknowledged that there are acceptable options for the particular level (e.g., 90, 95, 99%). Heuristically, the level of uncertainty is related inversely to the quality of available data and information (Runge 2011; Williams and Johnson 2015). More and better data would, in principle, reduce uncertainty.

However, policy judgments have to be made in the face of that uncertainty due to data limitations or time constraints or because some uncertainty (e.g., environmental variability) is irreducible even with more research (Murphy and Weiland 2016). Furthermore, policy judgment may not be sensitive to the level of uncertainty. For example, the appropriate policy choice can be evident in spite of the uncertainty, in which case, allocating time or funds to reducing the uncertainty would not provide value to the decision making process (Williams and Johnson 2015).

Case Studies

Eastern massasauga rattlesnake

The eastern massasauga rattlesnake (EMR) became a candidate for *listing* due to multiple factors associated with habitat modification and loss of populations across its range. Given the species' broad distribution and the inconsistency in amount and quality of demographic information across the various populations, the SSA was considered to be fairly complex. The assessment was documented in a peer-reviewed report (USFWS 2016c).

Stage 1. The species' ecology. Eastern massasauga rattlesnake occupies wet meadows, fens, and bogs in the midwest and northeast US and southern Ontario, Canada (Seigel 1986; Kingsbury et al. 2003). The particular ecological needs of EMR vary with season. During the hibernation period, EMR requires a moist subterranean space below the frost line to avoid desiccation and freezing (Sage 2005), while during the active season, EMR needs a mosaic of shaded and sunny areas for thermoregulation, abundant prey, and areas to escape predators (Johnson 2000).

To assess EMR viability, we analyzed its historical, current, and projected future abundance and distribution. We began with describing the breadth of adaptive capacity across the EMR range. We investigated variation in habitat use, prey, venom, climate, and genetics as potential indicators of variation in adaptive capacity. We ultimately determined that breadth of adaptive capacity can be captured by a wide distribution of populations within three genetically diverse regions identified by Ray et al. (2013): 1) the western analysis unit (WAU) consisting of populations in Minnesota, Missouri, Iowa, Wisconsin, and Illinois, 2) the central analysis unit (CAU) consisting of populations in Indiana, southern and central Michigan, Ohio, and far southwestern Ontario, and 3) the eastern analysis unit (EAU) consisting of populations in New York, Pennsylvania, northern Michigan, and the remaining portions of Ontario.

Next, we assessed the change in the number and distribution of populations from before 2014 to current (2014-2016) and future (10, 25 and 50 years) time-periods in each of the three analysis units (AUs). We assessed the status of historical populations (extant, extirpated, or unknown) and the health of the extant populations. We relied on the results of Faust et al. (2011) and supplemental information garnered since 2011 to assess the health of the populations. Faust et al. (2011) built an age-based, stochastic population model for a hypothetical, healthy EMR population and determined how various influences affect EMR vital rates. The demographic parameters, prominent influences, and the effect of such influences on vital rates were derived from empirical data and expert judgment. The prominent influences identified and analyzed were: habitat loss, vegetative succession, fragmentation, road mortality, hydrologic alteration, human harassment, collection, ineffective management regimes, and habitat restoration. Using elicited site-specific information, Faust et al. (2011) generated estimates of population growth rate, ending population size, and probability of quasi-extirpation (adult female population size ≤ 25) for all populations with sufficient data (57 populations). We used these results to identify populations considered healthy, i.e., self-sustaining. We defined a self-sustaining population as having: 1) an adult female population size > 50, 2) a positive population growth rate, and 3) a probability of persistence greater than 0.90 over 25 years despite the stressors acting upon it. We extrapolated the results to infer the health of the non-modeled populations by multiplying the proportion of modeled populations meeting our self-sustaining criteria by the number of extant populations in each AU.

We evaluated the change in adaptive capacity over time by calculating the spatial extent of occurrence (EoO) rangewide and within the three AUs. We used ArcGIS to draw polygons around clusters of counties with EMR populations and summed the area of all polygons within and across AUs. We evaluated EMR *redundancy* by assessing the vulnerability to catastrophic events. We consulted the literature and species experts to identify the natural and anthropogenic catastrophic events that would likely lead to population extirpation. Experts identified drought, flooding, and disease (rapid and widespread epidemic) as potential catastrophic events. We had insufficient information on flood (specifically, the magnitude of flood that would lead to extirpation) and disease risk (notably, the likelihood of disease outbreaks, the factors that affect disease spread, and the magnitude of impact on EMR populations) to include either in our analysis. Thus, drought was the only catastrophic event analyzed. To calculate the risk of extirpation due to drought, we used an extinction risk model developed by Ruckelshaus et al. (2002).

Stage 2. The species' current condition. Historically, there were 558 EMR populations scattered across parts of Ontario, New York, Pennsylvania, Ohio, Michigan, Indiana, Illinois, Missouri, Iowa, Wisconsin, and Minnesota. Currently, there are 347 extant populations in 10 states (Figure 2). Within the WAU, 20 of 72 historical populations are extant, and of these, six are self-sustaining. In the CAU, 256 of 350 historical populations are extant, of which 65 are self-sustaining. In EAU, 71 of the 136 historical populations are extant, 30 of which are self-sustaining. Range-wide, EMR spatial extent declined from the historical period by 41%, with 70%, 33%, and 26% decreases in the WAU, CAU, and EAU, respectively.

Stage 3. The species' future condition. Due to time constraints, we ran only one future scenario; we assumed the magnitude of impact and frequency of the prominent influences would continue into the future. To identify the number of populations likely to persist under the continuation scenario, we assumed that populations that met the criteria for self-sustaining at years 10, 25 and 50 would persist for those three time periods. Using these results for the 57 modeled populations, we then extrapolated to the remaining extant populations by multiplying the proportion of modeled populations that were self-sustaining by the total number of currently extant populations in each AU to estimate the number of EMR populations that are projected to be self-sustaining at years 10, 25 and 50.

Range-wide, population losses were forecasted to continue into the future (Figure 3), with 263 populations forecasted to be extirpated by year 50. Of the populations projected to persist, one population in WAU, 47 in CAU, and six in EAU were forecasted to be self-sustaining by year 50. The spatial extent of EMR was projected to decline. Range-wide, the EoO was forecasted to decline by 80% by year 50; within the AUs, EoO was projected to decline by 91%, 64%, and 89% in the WAU, CAU, and EAU, respectively. The risks of AU-wide extirpation due to catastrophic drought remained near zero for CAU and EAU due to a combination of low drought risks and the number of populations, but the probability of WAU-wide extirpation within 25 years ranged from 0.02 to 0.82 depending upon the drought severity.

The abundance and distribution of EMR have declined from its historical condition and is forecasted to continue to decline into the future. Relative to historical conditions, currently there is a 38% reduction in the number of extant populations with predicted reductions to reach 85% by year 50. These losses have not been uniformly distributed. The WAU, which historically represented 28% of the EMR range, today represents 14% of the species' range, and by year 50, is predicted to represent 12% of the range with one self-sustaining population persisting. Catastrophic drought greatly increases the risk of extirpation resulting in a 0.96 probability of extirpation of the WAU. Similarly, the EAU historically comprised 36% of the range, but by year 50 it is projected to comprise 19% with six self-sustaining populations persisting, representing a 96% loss of the historical populations. In the CAU, 78% of the historical populations are predicted to be extirpated, with 47 populations projected to be self-sustaining. Although populations are projected to persist, EMR range is projected to contract, with high likelihood for the extirpation of populations from the western portion of the range and substantial losses in eastern portion of the range. These losses are likely to lead to considerable decreases in adaptive capacity, which may impair the ability of EMR to adapt to near-term and long-term changes in its environment (e.g., novel diseases and predators, habitat alteration due to invasion of exotic species), thereby increasing its vulnerability to extinction.

Communicating SSA results to decision makers. Prior to a 2-day in-person meeting with the USFWS decision makers, the analysis team provided a written SSA report (USFWS 2016c) and presented a summary via a webinar of the methods, results, and the implications of uncertainty. This provided an opportunity for the decision makers to ask questions, request inclusion of alternative scenarios, and explore different assumptions prior to making a determination of whether EMR meets the definition of threatened or endangered under the ESA. The decision makers asked about the rationale underlying the definition of a self-sustaining population and our extrapolation approach. The decision makers were satisfied with the rationale for underlying assumptions and with the range of uncertainty we modeled. The decision makers recommended that EMR warranted protection under the ESA, and it was designated a *threatened species* (USFWS 2016c). The determination attributed habitat loss as the greatest cause of current and future condition and noted that emergent disease, collection and persecution of individuals, and climate change contribute to the risk of extinction.

Sonoran Desert tortoise

The Sonoran Desert tortoise (SDT) was determined to be a candidate for *listing* under the ESA in 2010 due to a preponderance of different potential *threats* to the species (USFWS 2010). We anticipated this to be a complex SSA as the geographic scope encompassed Arizona, US, and Sonora, Mexico, which we incorporated into geospatial and demographic modeling (USFWS 2015a, McGowan et al. 2017). Interest in the decision from external parties was quite high due to the implications of a *listing* determination.

Stage 1. The species' ecology. Sonoran Desert tortoise is a long-lived tortoise that ranges from Arizona, US, to central Sonora, Mexico. Recent genetic analysis delineated the SDT as a species distinct from the Mojave Desert tortoise (Murphy et al. 2011). Sonoran Desert tortoise utilize rocky slopes at higher elevations and soil types that facilitate excavation of burrows for shelter and nesting, though they sometimes use natural cavities (Van Devender 2002). The occurrence of drought has potentially large effects on tortoise demographics, as seasonal monsoon rains and annual green ups provide forage resources that improve survival and reproduction (Averill-Murray et al. 2002; Sullivan et al. 2014).

Spatial sub-divisions cannot clearly delineate population boundaries for SDT. Though potential barriers to movement exist, there was not sufficient genetic evidence of spatial structure to guide population delineation (Edwards et al. 2004; Edwards 2015). Therefore, we divided the species' range into two populations for this analysis, separating the US from Mexico because the species faces different *threats* and management strategies in these two regions.

Stage 2. The species' current condition. The geographic range of the SDT is unchanged compared to historical conditions. The abundance of the two populations was estimated by first evaluating habitat quality across the range with a geospatial model that used land cover vegetation type, slope, and elevation to classify areas into primary, secondary, and tertiary potential habitat classes (Figure 4A). Estimated population densities vary considerably (from two to 20 adult tortoises per km^2). We assumed that primary habitat would sustain densities at the high end of that range (17 adult tortoises per km^2), secondary in the middle (9 per km^2), and tertiary at the low end (2 per km^2). Based on a range of habitat qualities and reported population density, we extrapolated Arizona abundance ranged from ~310,000 to ~640,000 and Mexico from ~160,000 to ~330,000 adult males and females (USFWS 2015a). Using a population projection model, we ran different scenarios with low and high starting population sizes to convey to decision makers how uncertainty in current population size is expected to influence future condition.

We elicited conceptual models of SDT ecology from species experts to identify key demographic parameters, habitat requirements, and environmental factors that might affect tortoise populations (Figure 4B). Through a series of meetings with federal, state, and academic biologists we mapped the environmental factors, such as food availability, precipitation, invasive grasses, and cattle grazing, that influence tortoise demographics and, therefore, their population viability. We initially used the conceptual modeling to explore all the factors that might affect tortoises, but focused on the most important components that would affect viability. For example, though we initially explored the effects of increasing regional temperatures due to anticipated climate change on sex ratio in tortoise populations (nest temperature determines sex), experts agreed that temperature effects on adult survival via drought would manifest much sooner and more severely than sex ratio effects on the populations, so we eliminated temperature effects on sex ratio from our stochastic simulation model.

Our analysis identified potential primary *threats*: drought and expected decreases in precipitation due to climate change, increases in frequency and intensity of wildfire, and habitat loss and degradation due to urban expansion and human population growth and invasion of nonnative plants (Figure 4A). Literature indicates that tortoise survival is highly susceptible to drought (Zylstra et al. 2013), causing up to a 10% decline in annual survival of adult and larger juvenile tortoises. Urbanization and wildfire could limit habitat availability and reduce the quality of remaining habitat through associated disturbances (Figure 4). Nonnative plants degrade habitat quality and increase wildfire intensity and effects. Because the rate of change in climate or urbanization in the future are uncertain, we made these effects variable over time in the projection model and used the model to explore multiple scenarios of climate change and habitat loss.

Stage 3. The species' future condition. We built a population viability model to simulate tortoise populations and measure population *resiliency* into the future under stochasticity and parametric uncertainty (McGowan et al. 2017). The matrix population model accounted for three life stages (small juveniles, large juveniles, and breeding adults; McCoy et al. 2014). The parameters in the model (e.g., survival, fecundity, etc.) varied annually to represent environmental variability and also applied parametric uncertainty functions to account for imperfect data and observations errors that affect parameter estimates (McGowan et al. 2011). To represent climate change, we incorporated a randomized drought function that determined what proportion of the population experienced drought and the magnitude of the drought effect on the survival rates.

$$S_t^{A,d} = \left(P_{drought} \times S_t^A \times DE_t\right) + \left(\left(1 - P_{drought}\right) \times S_t^A\right)$$

where $P_{drought}$ is the proportion of the population exposed to drought and $S_t^{A,d}$ is the survival rate of adults for the full population, given the proportion that was exposed to drought. DE_t is the drought effect in a specific year which was modeled as a random variable between 0.8 and 0.99 which simulates a 1% to 20% reduction in survival due to the drought in any given year, to represent differing drought severity (spatially and magnitude) from year to year. This drought function enabled simulations with increasing drought frequencies that could result from future climate change. We also incorporated into the model a ceiling-type density dependence function that prevented the population from exceeding the abundance allowed by the available habitat (Morris and Doak 2002). The density ceiling would reduce productivity to 0 if the population exceeded the threshold, allowing us to limit population growth without speculating on the mechanisms of density dependence. The density threshold was set at the maximum population size possible if all the available habitat were occupied at the highest empirically observed densities (USFWS 2015a; McGowan et al. 2017). We used the model to simulate the effects of nine scenarios related to climate-driven drought effects, habitat loss from urban expansion, and potential benefits of proposed management actions. Habitat loss scenarios lower the density dependent ceiling over time to mimic loss of habitat that could be occupied, and positive management scenarios were simulated by counteracting or stabilizing habitat loss (i.e., increasing or stabilizing changes in the density dependent ceiling). Also, the model included random variation in annual parameter values to represent environmental stochasticity and added parametric uncertainty to survival and fecundity parameters (McGowan et al. 2011).

Model outputs included the median population trajectories and the 2.5 and 97.5 percentile of the population trajectories (Figure 5), and we also reported the probability of quasi-extinction under each scenario over a 200-year time span. We measured the probability of quasi-extinction as the proportion of simulation replicates that fell below a predetermined minimum population size (e.g., with 1000 replicates and a quasi-extinction probability of 0.05, 50 replicates fell below the threshold and 950 did not). We evaluated both 2% and 4% of the initial population estimates as quasi-extinction thresholds, offering two quasi-extinction thresholds to the decision makers to more fully describe risk of extinction. In other words, we predicted the probability that the population would decline to 2% or 4% of its initial size estimate in the future. Quasi-extinction thresholds are theoretically supposed to represent the point at which a population is so small that extinction is unavoidable, however we do not know what that threshold is for SDT populations, so we presented results for two different thresholds. We selected the 2% and 4% thresholds using input from species experts and managers. At 2% or 4 % of current abundance, we presumed that population densities would be so low or the population would be so patchily distributed that population would be ecologically and functionally extinct.

Our habitat analysis predicted reductions in overall potential habitat available and degradation in habitat quality over time. Our population model predicted that SDT abundance in both populations would, on average, decline through time with some continual habitat degradation; however, there was wide variation in predicted outcomes (Figure 5). We ran the model under multiple future scenarios that varied maximum habitat availability based on potential habitat conditions and management actions, and we varied survival rates under different magnitudes of climate change effects. In all, we ran the model under nine different scenarios for each of the two populations. The model results also indicated that extinction probability was very low (<0.02) under most scenarios over the next 100 years (Figure 5) and virtually no extinction probability within 50 years (USFWS 2015a). Small population declines (measures of *resiliency*) are predicted due to some habitat loss and degradation and drought impacts from climate change. However, relatively small changes in the overall distribution of the species (measures of *redundancy* and *representation*) were predicted.

Communicating SSA results to decision makers. In addition to an extensive written SSA report detailing population assessments, the habitat modeling and population viability modeling (USFWS 2015a), we held a two-day, interactive meeting with the analysis team and USFWS decision makers to present the results of the SSA. In the meeting, analysts presented information on species biology, model structure, projection scenarios, and figures and tables of model output, and they explained their rationale for any scientific judgments. For example, we presented figures depicting the future median population size, the 95% confidence interval of population size and the proportion of trajectories that declined to the quasi-extinction threshold for 18 different future scenarios (Figure 5). The analysis team responded to decision makers' clarifying questions and explained the many areas of uncertainty, using figures and projection scenarios. The decision makers settled on a *foreseeable future* of 50 to 75 years and considered predicted changes in abundance, distribution and diversity over that time frame.

The decision makers recommended that SDT was not warranted for *listing* under the ESA as a threatened or *endangered species*, and it did not receive ESA protection (USFWS 2015b).

Discussion

We presented case studies to illustrate how the SSA process can be implemented. The spatial distributions for both EMR and SDT cover multiple states crossing international borders and some population data were available. For EMR, representative areas were identified by three genetically-informed regions (Ray et al. 2013). *Resiliency* was assessed using population-specific modeling; EMR condition was projected at 10, 25, and 50 years under the scenario of a continuation of existing stressors and conservation efforts. *Redundancy* was evaluated by assessing the likelihood of losing all populations within a representative area due to catastrophic drought. The analysts gave decision makers the opportunity to run additional scenarios to describe the full risk profile for the species. Going through the SSA iteratively can reveal that the scenarios had not sufficiently captured the risk profile suggesting additional scenarios for analysis.

For SDT, evidence indicated an absence of strong genetic or population structure. However, exposure to *threats* and management strategies differed across the international border into Mexico. Thus, the assessment was structured within representative areas defined by the international border. The density of adult tortoises was assumed to be a function of habitat category as influenced by precipitation, wildfire, and urban expansion. A demographic model was used to project SDT abundance and quasi-extinction during a 200-year time span for combinations of drought, urban development, and conservation effort scenarios.

Although the case studies were in the context of *listing* decisions, the intent of the SSA is to develop a scientific analysis, which can then be used as a basis for informing the various ESA decisions (Figure 1). Much of the information of an SSA represents the state of knowledge of a species and its ecology, which would change with new data but not with a particular decision. However, from a decision analysis perspective, the SSA predicts the consequences that arise from the decision options, which depend on the particular ESA decision (Runge 2011; Gregory et al. 2013). For example, the decision to protect a species under the ESA relies on an assessment of the likelihood of extinction assuming levels of stressors and conservation efforts without ESA protections in place (Doremus 1997; Waples et al. 2013). In contrast, for an already protected species, decisions relate to *recovery planning* or *consultation* and permitting (Figure 1; Steiger 1994; McGowan 2013). *Recovery planning* (Table 1) requires a comparison of actions to identify the set of actions or strategy that offers the best chance to reduce species risk to a level that recovers the species (Boor 2013; McGowan et al. 2014). Interagency *consultation* on federal actions and permitting of nonfederal actions (cf Consultations and Permits in Table 1) requires an evaluation of species risk with and without a proposed project (Runge et al. 2008; McGowan and Ryan 2010). Thus, an SSA adapts to the decision context (Figure 1). Scenarios can be used to adapt an SSA to a particular decision context. For example, scenarios for interagency *consultation* compare a species' response to alternative project designs and conservation measures in combination with future *threats*. Scenarios for *recovery planning* incorporate alternative recovery actions to identify those that will most likely achieve the goal of species recovery.

The cost of completing an assessment is an important consideration regardless of regulatory context because of workload relative to available agency capacity (Rohlf 2004; Murphy and Weiland 2016). The SSA explicitly analyzes a species' response to stressors and conservation, which was not always included in the past *threat*-focused process. We suggest that the additional analytical demands will be at least offset by the efficiencies produced by 1) relying on the analysis for multiple decisions as the SSA follows the species and 2) helping to defend decisions due to the improved consistency and transparency of the supporting science. In our experience, the initial SSA usually takes longer than the past *threat*-focused process, but the SSA report is available for decisions on the same species (Figure 1). However, efficiency comparison of the past process and the SSA is not straightforward because science and policy were often conflated in the past, resulting in more extended Federal Register notices and allowing for few opportunities for input by outside experts. While the SSA completed before an *endangered species* designation may take longer than subsequent updates, it presumably leads to improved analyses, extensive expert input, clear decision processes, and short Federal Register notices. Nevertheless, insufficient institutional capacity can impede the use of *best* *available science* in decision making (Burgman 2015; Lowell and Kelly 2016; Murphy and Weiland 2016). Solutions to insufficient capacity include building analytical capacity through hiring, training, and collaboration with science institutions (Burgman 2015) and by ensuring that analytical practices are applied efficiently. It is reasonable to expect that, all else equal, the scientific rigor of an SSA is a direct function of analytical capacity relative to the demand on time, effort, and expertise; this capacity per demand relationship underlies the conclusions reached by Lowell and Kelly (2016). In our experience, the most demanding SSAs, which are associated with high data availability, wide range, and spatial complexity and extent of stressors, constitute a minority, perhaps about 10%, of the workload. In the interest of parsimony, an SSA, especially for low or moderate complexity situations, can characterize risk on a categorical rather than on a continuous scale and can reduce the number and complexity of future scenarios. However, simplification of an SSA potentially compromises scientific and legal defensibility.

The SSA process is based on methods for risk assessment and scenario planning, which have been developed over the past decades (Shaffer 1981 and 1987; Carroll et al. 1996; Akçakaya and Sjögren-Gulve 2000; Shaffer and Stein 2000; Peterson et al. 2003; DeMaster et al. 2004; Keith et al. 2004; Duinker and Greig 2007; Runge et al. 2007, 2008; Patrick et al. 2008; McGowan and Ryan 2009; McGowan et al. 2013; Fordham et al. 2013; Doak et al. 2015; Wolf et al. 2015; IPBES 2016; Murphy and Weiland 2016; Phillips-Mao et al. 2016). Fundamental to all ESA determinations is a basic understanding of the species' ecology, current condition, and future condition, which is what the SSA is designed to provide. Importantly, the SSA does not stop at an assessment of *threats* but moves to answer the next natural question, what do the projected stressors and conservation efforts mean for the species' future condition or the risk of extinction?

Recognizing the distinct roles of science and formal policy (e.g., application of regulatory standards) is a prerequisite to providing a transparent and consistent species assessment (Doremus 1997; Ruhl 2004; Doremus and Tarlock 2005; Gregory et al. 2013; Waples et al. 2013). The SSA results in a scientific report to the decision maker about a species' condition, and then the decision maker applies the policy to make the decision. Conflating the roles of science and policy can create unnecessary confusion both within the agencies charged to make ESA decision and with the public and partners who are affected by those decisions (Waples et al. 2013; Boyd et al. 2016).

While it is important to distinguish the roles of science and policy, it is essential that the scientific information matches the decision context. When decision makers understand the analytical processes and results then the assessment be more useful in making ESA decisions. Therefore, regular communication between the decision makers and the analysts to achieve a common understanding of metrics, future scenarios, time frames, and implications of uncertainty helps to ensure that the assessment is informative to the decision makers. Any scientific analysis involves judgments, which determine underlying assumptions and parameters (Doremus and Tarlock 2005). For example, confidence levels, quasi-extinction thresholds, and stressor levels to define scenarios are judgments to be evaluated, peer-reviewed, and communicated to the decision maker with the aim of providing a transparent assessment. In the case studies, future condition under multiple quasi-extinction thresholds and the rationale underlying the choice of population metrics were communicated to the decision makers.

An SSA empirically evaluates species risk (Table 2). However, whether that level of risk is deemed to be unacceptably high, leading to ESA protection, is an inherently normative determination (Doremus 1997; Vucetich et al. 2006). Consistency and transparency in ESA decisions emerge from two sources: the scientific analysis and the policy application. In the context of ESA determinations, normative judgment is expressed through the ESA legislation, guidance for interpreting the ESA, agency practice, and clarifying case law (Rohlf 2004). In total, these policies provide an understanding of the ESA's regulatory standards, which are then applied to make the decision. The SSA affects only the consistency and transparency of the risk assessment and does not determine policy standards and definitions. Currently, the policies do not contain explicit standards for making management judgments, and policies can change over time.

In summary, the design of the SSA process is intended to improve the consistency and transparency of the scientific analysis of the available biological information to support policy-based ESA decisions. The degree that the SSA represents progress can be gauged relative to the baseline, which is the earlier *threat*-focused analysis (Andelman et al. 2004). The SSA includes explicit analyses of the species' response to stressors through a description of the ecology, estimation of the current condition, and forecasts of the future condition under multiple scenarios. Decision makers apply the policy-guided interpretation of the ESA to the SSA results to make ESA determinations. Both science input and policy application contribute to consistency in ESA decisions. The SSA results in a scientific report distinct from policy judgment, which contributes to streamlined, transparent, and consistent decision making and allows for greater technical participation by experts outside of the USFWS. As a consequence, we believe the SSA provides better scientific analysis that will in turn improve ESA decisions.

Supplemental Material

Reference S1. Faust L, Szymanski J, Redmer M. 2011. Range wide extinction risk modeling for the eastern massasauga rattlesnake (*Sistrurus catenatus catenatus*). Alexander Center for Applied Population Biology, Lincoln Park Zoo, Chicago, Illinois, USA. Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S1 (2430 KB PDF); also available at https://www.researchgate.net/publication/299645160_Range-wide_Extinction_Risk_Modeling_for_the_Eastern_massasauga_Rattlesnake_Sistrurus_catenatus_catenatus_Technical_Report_to_the_US_Fish_and_Wildlife_Service?showFulltext=1&linkId=5703be2908ae646a9da9bf00 (2,431 KB PDF).

Reference S2. [IPBES] Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. 2016. Summary for policymakers of the methodological assessment of scenarios and models of biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. S. Ferrier, K. N. Ninan, P. Leadley, R. Alkemade, L.A. Acosta, H. R. Akçakaya, L. Brotons, W. Cheung, V. Christensen, K. A. Harhash, J. Kabubo-Mariara, C. Lundquist, M. Obersteiner, H. Pereira, G. Peterson, R. Pichs-Madruga, N. H. Ravindranath, C. Rondinini, B. Wintle, editors. Secretariat of the

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany. 32 pp.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S2 (10989 KB PDF); also available at http://www.ipbes.net/sites/default/files/downloads/pdf/SPM_Deliverable_3c.pdf (10,990 KB PDF).

Reference S3. Rowland ER, Cross MS, Hartmann H. 2014. Considering multiple futures: scenario planning to address uncertainty in natural resource conservation. Washington, DC: U.S. Fish and Wildlife Service, 162 pp.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S3 (12714 KB PDF); also available at http://www.cio.noaa.gov/services_programs/prplans/pdfs/ID232_Final_Product_Green_Turtle_Technical_Memorandum_SWFSC_No_539.pdf (12,715 KB PDF).

Reference S4. Runge MC, Sanders-Reed CA, Fonnesbeck CJ. 2007. A core stochastic population projection model for Florida manatees (*Trichechus manatus latirostris*). U.S. Geological Survey Open-File Report 2007-1082. 41 pp.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S4 (544 KB PDF); also available at https://pubs.er.usgs.gov/publication/ofr20071082 (545 KB PDF).

Reference S5. Seminoff, JA, Allen CD, Balazs GH, Dutton PH, Eguchi T, Haas HL, Hargrove SA, Jensen MP, Klemm DL, Lauritsen AM, MacPherson SL, Opay P, Possardt EE, Pultz SL, Seney EE, Van Houtan KS, Waples RS. 2015. Status review of the green turtle (*Chelonia mydas*) under the U.S. Endangered Species Act. La Jolla, California, NOAA Technical Memorandum, NOAANMFS-SWFSC-539, National Marine Fisheries Service 571 pp.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S5 (6467 KB PDF); also available at http://www.cio.noaa.gov/services_programs/prplans/pdfs/ID232_Final_Product_Green_Turtle_Technical_Memorandum_SWFSC_No_539.pdf (6,468 KB PDF).

Reference S6. U.S. Department of the Interior. 2009. The meaning of "Foreseeable Future" in Section 3(20) of the Endangered Species Act. Solicitor memorandum M-3702, Office of the Solicitor, Washington, D.C.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S6 (886 KB PDF).

Reference S7. U.S. Environmental Protection Agency. 2011. Expert elicitation task force white paper. Washington, DC: U.S. Environmental Protection Agency Science Policy Council.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S7 (1561 KB PDF); also available at http://www.fws.gov/endangered/esa-library/pdf/ESAall.pdf (1,562 KB PDF).

Reference S8. [USFWS] U.S. Fish and Wildlife Service. 2015a. Species status assessment for the Sonoran Desert tortoise. Version 1.0, September 2015. Albuquerque, New Mexico: Southwest Region U.S. Fish and Wildlife Service. 35 pp.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S8 (18235 KB PDF); also available at https://www.fws.gov/southwest/es/arizona/Documents/SpeciesDocs/SonoranTort/SDT_SSA_Report_September2015_Chp1-3.pdf (18,236 KB PDF).

Reference S9. [USFWS] U.S. Fish and Wildlife Service. 2016a. Long-term listing transformation and five guiding principles of the Unified Listing Team. Director's memorandum 16 March 2016, Washington, D.C.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S9 (97 KB PDF).

Reference S10. [USFWS] U.S. Fish and Wildlife Service. 2016b. USFWS species status assessment framework. Falls Church, Virginia: U.S. Fish and Wildlife Service. 21 pp.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S10 (1326 KB PDF); also available at https://www.fws.gov/endangered/improving_ESA/pdf/SSA%20Framework%20v3.4-8_10_2016.pdf (1,327 KB PDF).

Reference S11. [USFWS] U.S. Fish and Wildlife Service. 2016c. Species status assessment for the eastern massasauga rattlesnake (*Sistrurus catenatus*). Falls Church, Virginia: U.S. Fish and Wildlife Service 117 pp.

Found at DOI: http://dx.doi.org/10.3996/052017-JFWM-041.S11 (2465 KB PDF); also available at https://www.fws.gov/midwest/endangered/reptiles/eama/pdf/SSAFinalV2July2016EMR.pdf (2,466 KB PDF).

Acknowledgments

We thank Greg Breese, Debby Crouse, Jonathan Cummings, Cat Darst, Beth Forbus, Carey Galst, Craig Hansen, Michael Horton, Donald Imm, Michelle Kissling, Mary Parkin, Mark Pavelka, Erin Rivenbark, Steve Morey, Marjorie Nelson, Tara Nicolaysen, Laura Ragan, Angela Romito, Justin Shoemaker, Jesse Wild and all additional members of the SSA Framework Implementation Team for their contributions to the development of the SSA. We thank Jeff Servoss, Michael Martinez, Shaula Hedwall, Brian Woolridge, Katie Boyer, Richard King, Frank Durbian, and Trisha Crabill for their contributions to the assessment case studies. We also thank Resit Akçakaya, Jean Cochrane, Jonathan Cummings, Holly Doremus, Marty Miller, Steve Morey, Jeff Newman, Mike Runge, Mark Shaffer, Robin Waples, Rollie White, an anonymous reviewer, and the Associate Editor for helpful comments on earlier versions of this paper. U.S. Fish and Wildlife Service funded this work.

References

Akçakaya HR, Sjögren-Gulve P. 2000. Population viability analysis in conservation planning: an overview. Ecological Bulletin 48:9-21.

Addison PFE, Rumpff L, Bau SS, Carey JM, Chee YE, Jarrad FC, McBride MF, Burgman MA. 2013. Practical solutions for making models indispensable in conservation decision-making. Diversity and Distributions 19:490-502.

Andelman SJ, Groves C, Regan HM. 2004. A review of protocols for selecting species at risk in the context of U.S. Forest Service viability assessments. Acta Oecologica 26:75-83.

Anderson DR, Link WA, Johnson DH, Burnham KP. 2001. Suggestions for presenting the results of data analysis. Journal of Wildlife Management 65:373-378.

Averill-Murray RC, Martin BR, Bailey SJ, Wirt EB. 2002. Activity and behavior of the Sonoran Desert tortoise in Arizona. Pages 135-158 in Van Devender TR, editor. The Sonoran Desert tortoise: natural history, biology, and conservation. Tucson, Arizona: The University of Arizona Press.

Boor GKH. 2013. A framework for developing objective and measurable recovery criteria for threatened and endangered species. Conservation Biology 28:33-43.

Boyd C, DeMaster DP, Waples RS, Ward EJ, Taylor BL. 2016. Consistent extinction risk assessment under the U.S. Endangered Species Act. Conservation Letters DOI: 10.1111/conl.12269

Burgman, MA. 2015. Governance for effective policy-relevant scientific research: the shared governance model. Asian and the Pacific Policy Studies 2(3): 441-451. DOI: 10.1002/app5.104

Burgman MA. 2016. Trusting judgement: how to get the best out of experts. Cambridge, England: Cambridge University Press.

Burgman MA, Yemshanov D. 2013. Risks, decisions and biological conservation. Diversity and Distributions 19:485-489.

Carey JM, Burgman MA. 2008. Linguistic uncertainty in qualitative risk analysis and how to minimize it. Annals of the New York Academy of Sciences 1128:13-17.

Carroll R, Augspurger C, Dobson A, Franklin J, Orians G, Reid W, Tracy R, Wilcove D, Wilson J. 1996. Strengthening the use of science in achieving the goals of the endangered species act: An assessment by the Ecological Society of America. Ecological Applications 6:1-11.

Carroll C, Vucetich JA, Nelson MP, Rohlf DJ, Phillips MK. 2010. Geography and recovery under the U.S.Endangered Species Act. Conservation Biology 24:395-403.

Copeland HE, Doherty KE, Naugle DE, Pocewicz A, Kiesecker JM. 2009. Mapping oil and gas development potential in the US intermountain west and estimating impacts to species. PLoS One 4(10):e7400 (http://dx.doi.org/10.1371/journal.pone.0007400)

DeMaster D, Angliss R, Cochrane J, Mace P, Merrick R, Miller M, Rumsey S, Taylor B, Thompson G, Waples R. 2004. Recommendations to NOAA Fisheries: ESA listing criteria by the Quantitative Working Group. NOAA Technical Memorandum NMFS-F/SPO-67. http://spo.nwr.noaa.gov/tm/tm67.pdf (October 2017)

Doak DF, Himes Boor GK, Bakker VJ, Morris WF, Louthan A, Morrison SA, Stanley A, Crowder LB. 2015. Recommendations for improving recovery criteria under the U.S. Endangered Species Act. BioScience 65:189-199.

Doremus H. 1997. Listing decisions under the Endangered Species Act: why better science isn't always better policy. Washington University Law Quarterly 75:1029-1153.

Doremus H, Tarlock DA. 2005. Science, judgment, and controversy in natural resource regulation. Public Land & Resources Law Review 26:1-38.

Drescher M, Perera AH, Johnson CJ, Buse LI, Drew CA, Burgman MA. 2013. Toward rigorous use of expert knowledge in ecological research. Ecosphere, art83 (http://onlinelibrary.wiley.com/doi/10.1890/ES12-00415.1/full).

Duinker PN, Greig LA. 2007. Scenario analysis in environmental impact assessment: improving explorations of the future. Environmental Impact Assessment Review 27:206-219.

Earl JE, Nicol S, Wiederholt R, Diffendorfer JE, Semmens D, Flockhart DTT, Mattsson BJ, McCracken G, Norris DR, Thogmartin WE, López-Hoffman L. 2017. Quantitative tools for implementing the new definition of significant portion of the range in the Endangered Species Act. Conservation Biology. doi:10.1111/cobi.12963

Edwards T, Schwalbe CR, Swann DE, Goldberg CS. 2004. Implications of anthropogenic landscape change on inter-population movements of the desert tortoise (*Gopherus agassizii*). Conservation Genetics 5:485–499.

Edwards T, Berry KH, Inman RD, Esque TC, Nussear KE, Jones CA, Culver M. 2015. Testing taxon tenacity of tortoises: evidence for a geographical-selection gradient at a secondary contact zone. Ecology and Evolution 5(10):2095-2114. DOI: 10.1002/ece3.1500

Evansen M, Li Y-W, Malcom J. 2017. Same law, different results: comparative analysis of Endangered Species Act consultations by two federal agencies.

https://www.biorxiv.org/content/early/2017/07/19/165647.full.pdf+html (October 2017)

Faust L, Szymanski J, Redmer M. 2011. Range wide extinction risk modeling for the eastern massasauga rattlesnake (*Sistrurus catenatus catenatus*). Alexander Center for Applied Population Biology, Lincoln Park Zoo, Chicago, Illinois, USA. (see *Supplemental Material*, Reference 1,); also available: https://www.researchgate.net/publication/299645160_Range-wide_Extinction_Risk_Modeling_for_the_Eastern_massasauga_Rattlesnake_Sistrurus_catenatus_catenatus_Technical_Report_to_the_US_Fish_and_Wildlife_Service?showFulltext=1&linkId=5703be2908ae646a9da9bf00 (October 2017).

Fordham DA, Akçakaya HR, Brook BW, Rodríguez A, Alves PC, Civantos E, Triviño M, Watts MJ, Araújo MB. 2013. Adapted conservation measures are required to save the Iberian lynx in a changing climate. Nature Climate Change 3:899-903.

Goodwin P, Wright G. 2014. Decision analysis for management judgment (5th edition). West Sussex, United Kingdom: John Wiley & Sons.

Gregory R, Failing L, Harstone M, Long G, McDaniels T, Ohlson D. 2012. Structured decision making: a practical guide to environmental management choices. West Sussex, United Kingdom: Wiley-Blackwell.

Gregory R, Arvai J, Gerber LR. 2013. Structuring decisions for managing threatened and endangered species in a changing climate. Conservation Biology 27:1212-1221.

IPBES. 2016. Summary for policymakers of the methodological assessment of scenarios and models of biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. S. Ferrier, K. N. Ninan, P. Leadley, R. Alkemade, L.A. Acosta, H. R. Akçakaya, L. Brotons, W. Cheung, V. Christensen, K. A. Harhash, J. Kabubo-Mariara, C. Lundquist, M. Obersteiner, H. Pereira, G. Peterson, R. Pichs-Madruga, N. H. Ravindranath, C. Rondinini, B. Wintle, editors. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany. 32 pp. (see *Supplemental Material*, Reference 2,); also available:

http://www.ipbes.net/sites/default/files/downloads/pdf/SPM_Deliverable_3c.pdf (October 2017).

Johnson G. 2000. Spatial ecology of the eastern massasauga (*Sistrurus c. catenatus*) in a New York peatland. Journal of Herpetology 34:186-192.

Keith DA, McCarthy MA, Regan H, Regan T, Bowles C, Drill C, Craig C, Pellow B, Burgman MA, Master LL, Ruckelshaus M. 2004. Protocols for listing threatened species can forecast extinction. Ecology Letters 7:1101-1108.

Kingsbury BA, Marshall JC, Manning J. 2003. Activity patterns and spatial resource selection of the eastern massasauga rattlesnake in northeastern Indiana. Fort Wayne, Indiana: Indiana-Purdue University 24p. https://core.ac.uk/download/files/145/22877099.pdf (October 2017)

Lowell N, Kelly RP. 2016. Evaluating agency use of “best available science” under the United States Endangered Species Act. Biological Conservation 196:53–59.

Lukey JR, Crawford SS, Gillis DJ, Gillespie MG. 2010. Effect of ecological uncertainty on species at risk decision-making: COSEWIC expert opinion as a case study. Animal Conservation 14:151-157.

McCarthy MA, Keith D, Teitjen J, Burgman MA, Maunder M, Master L, Brook BW, Mace G, Possingham HP, Medellin R, Andelman S, Regan H, Regan T, Ruckelshaus M. 2004. Comparing predictions of extinction risk using models and subjective judgement. Acta Oecologia 26(2):67-74.

McCoy ED, Aguirre G, Katzmeir RT, Tracy CR. 2014. Demography of North American Tortoises. Pages 134-142 in Rostal DC, McCoy ED, Mushinsky HR, editors. Biology and conservation of North American tortoises. Baltimore, Maryland: Johns Hopkins University Press.

McGowan CP. 2013. A structured approach to incidental take decision making. Environmental Management 51:241-250.

McGowan CP, Allan NA, Servoss J, Hedwall S, Woolridge B. 2017. Incorporating population viability models into species status assessment and listing decisions under the U.S. Endangered Species Act. Global Ecology and Conservation 12:119-130.

McGowan CP, Catlin DH, Shaffer TL, Gratto-Trevor CL, Aron C. 2014. Establishing endangered species recovery criteria using predictive simulation modeling. Biological Conservation 177:220-229.

McGowan CP, Runge MC, Larson MA. 2011. Incorporating parametric uncertainty into population viability analysis models. Biological Conservation 144:1400-1408.

McGowan CP, Ryan MR. 2009. A quantitative framework to evaluate incidental take and endangered species population viability. Biological Conservation 142:3128-3136.

McGowan CP, Ryan MR. 2010. Arguments for using population models in incidental take assessments for endangered species. Journal of Fish and Wildlife Management 1:183-188.

Morris WL, Doak D. 2002. Quantitative conservation biology. Sunderland, Massacheusetts: Sinauer Associates, Inc .

Murphy RW, Berry KH, Edwards T, Leviton AE, Lathrop A, Riedle JD. 2011. The dazed and confused identity of Agassiz's land tortoise, *Gopherus agassizii* (Testudines, Testudinidae) with the description of a new species, and its consequences for conservation. ZooKeys 113:39-71. DOI: 10.3897/zookeys.113.1353

Murphy DD, Weiland PS. 2016. Guidance on the use of best available science under the U.S. Endangered Species Act. Environmental Management 58: 1-14.

Patrick WS, Damon-Randall K. 2008. Using a five-factored structured decision analysis to evaluate the extinction risk of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). Biological Conservation 141:2906-2911.

Peterson GD, Cumming GS, Carpenter SR. 2003. Scenario planning: a tool for conservation in an uncertain world. Conservation Biology 17:358-366.

Phillips-Mao L, Galatowitsch SM, Snyder SA, Haight RG. 2016. Model-based scenario planning to develop climate change adaptation strategies for rare plant populations in grassland reserves. Biological Conservation 193:103-114.

Ray, J.W., R.B. King, M.R. Duvall, J.W. Robinson, C.P. Jaeger, M.J. Dreslik, B.J. Swanson, and D. Mulkerin. 2013. Genetic analysis and captive breeding program design for the eastern massasauga *Sistrurus catenatus catenatus*. Journal of Fish and Wildlife Management 4:104-113.

Redford KH, Amato G, Baillie J, Beldomenico P, Bennett EL, Clum N, Cook R, Fonseca G, Hedges S, Launay F, Lieberman S, Mace GM, Murayama A, Putnam A, Robinson JG, Rosenbaum H, Sanderson EW, Stuart SN, Thomas P, Thorbjarnarson J. 2011. What does it mean to conserve a (vertebrate) species? BioScience 61:39-48.

Refsgaard JC, van der Sluijs JP, Højberg AL, Vnrolleghem PA. 2007. Uncertainty in environmental modelling process- a framework and guidance. Environmental Modelling & Software 22:1543-1556.

Robbins K. 2009. Strength in numbers: setting quantitative criteria for listing species under the Endangered Species Act. 27:1-37.

Rohlf DJ. 1991. Six biological reasons the Endangered Species Act doesn't work - and what to do about it. Conservation Biology 5:273-282.

Rohlf DJ. 2004. Section 4 of the Endangered Species Act: top ten issues for the next thirty years. Environmental Law 34:483-553.

Rowland ER, Cross MS, Hartmann H. 2014. Considering multiple futures: scenario planning to address uncertainty in natural resource conservation. Washington, DC: U.S. Fish and Wildlife Service, 162 pp. (see *Supplemental Material*, Reference 3,); also available: https://www.fws.gov/home/climatechange/pdf/Scenario-Planning-Report.pdf (October 2017).

Ruckelshaus MH, McElhany P, Ford MJ. 2002. Recovering species of conservation concern: are populations expendable? Pages 305-329 in Kareiva PM, Levin S, editors. The importance of species: setting conservation priorities. Princeton, New Jersey: Princeton University Press.

Ruhl JB. 2004. The battle over the Endangered Species Act methodology. Environmental Law 34:555-603.

Runge MC, Sanders-Reed CA, Fonnesbeck CJ. 2007. A core stochastic population projection model for Florida manatees (*Trichechus manatus latirostris*). U.S.Geological Survey

Open-File Report 2007-1082. 41 pp. (see *Supplemental Material*, Reference 4,); also available: https://pubs.er.usgs.gov/publication/ofr20071082 (October 2017).

Runge MC, Sauer JR, Avery ML, Blackwell BF, Koneff MD. 2008. Assessing allowable take of migratory birds. Journal of Fish and Wildlife Management 73:556-565.

Runge MC. 2011. An introduction to adaptive management for threatened and endangered species. Journal of Fish and Wildlife Management 2:220-233.

Sage JR. 2005. Spatial ecology, habitat utilization, and hibernation ecology of the eastern massasauga (*Sistrurus catenatus catenatus*) in a disturbed landscape. Master's thesis. Ft. Wayne, Indiana: Purdue University.

Seigel RA. 1986. Ecology and conservation of an endangered rattlesnake, *Sistrurus catenatus*, in Missouri, USA. Biological Conservation 35:333–346.

Seminoff, JA, Allen CD, Balazs GH, Dutton PH, Eguchi T, Haas HL, Hargrove SA, Jensen MP, Klemm DL, Lauritsen AM, MacPherson SL, Opay P, Possardt EE, Pultz SL, Seney EE, Van Houtan KS, Waples RS. 2015. Status review of the green turtle (*Chelonia mydas*) under the U.S. Endangered Species Act. La Jolla, California, NOAA Technical Memorandum, NOAANMFS-SWFSC-539, National Marine Fisheries Service 571 pp. (see *Supplemental Material*, Reference 5,); also available: http://www.cio.noaa.gov/services_programs/prplans/pdfs/ID232_Final_Product_Green_Turtle_Technical_Memorandum_SWFSC_No_539.pdf (October 2017).

Shaffer ML. 1981. Minimum population sizes for species conservation. BioScience 31:131-134.

Shaffer ML. 1987. Minimum viable populations: coping with uncertainty. Pages 69-86 in Soule ME, editor. Viable populations for conservation. Cambridge, England: Cambridge University Press.

Shaffer ML, Stein MA. 2000. Safeguarding our precious heritage. Page 301-321 in Stein BA, Kutner LS, Adams JS,e editors. Precious heritage: the status of biodiversity in the United States. New York: Oxford University Press.

Starfield AM. 1997. A pragmatic approach to modeling for wildlife management. Journal of Wildlife Management 61:261-270.

Steiger J. 1994. The consultation provision of Section 7(a)(2) of the Endangered Species Act and its application to delegable federal programs. Ecology Law Quarterly 21(2):243-328.

Sullivan BK, Averill-Murray R, Sullivan KO, Sullivan JR, Sullivan EA, Riedle JD. 2014. Winter activity of the Sonoran desert tortoise (*Gopherus morafkai*) in central Arizona. Chelonian Conservation Biology 13:114–119.

Taylor BL, Wade PR, Ramakrishnan U, Gilpin M, Akçakaya HR. 2002. Incorporating uncertainty in population viability analyses for the purpose of classifying species by risk. Pages 239-252 in Beissinger SR, McCullough DR, editors. Population viability analysis. Chicago, Illinois: University of Chicago Press.

[ESA] US Endangered Species Act of 1973, as amended, Pub. L. No. 93-205, 87 Stat. 884 (Dec. 28, 1973). Available: http://www.fws.gov/endangered/esa-library/pdf/ESAall.pdf (Jan 2018)

U.S. Department of the Interior. 2009. The meaning of "Foreseeable Future" in Section 3(20) of the Endangered Species Act. Solicitor memorandum M-3702, Office of the Solicitor, Washington, D.C. (see *Supplemental Material*, Reference 6,)

U.S. Environmental Protection Agency. 2011. Expert elicitation task force white paper. Washington, DC: U.S. Environmental Protection Agency Science Policy Council. (see *Supplemental Material*, Reference 7,); also available: https://yosemite.epa.gov/sab/sabproduct.nsf/0/F4ACE05D0975F8C68525719200598BC7/$File/Expert_Elicitation_White_Paper-January_06_2009.pdf (October 2017).

[USFWS and NMFS] U.S. Fish and Wildlife Service and National Marine Fisheries Service. 1994. Interagency policy on information standards under the ESA. Federal Register 59(126): 34271.

[USFWS and NMFS] U.S. Fish and Wildlife Service and National Marine Fisheries Service. 1996. Policy regarding the recognition of distinct vertebrate population segments under the Endangered Species Act. Federal Register 61(26):4722-4725.

[USFWS and NMFS] U.S. Fish and Wildlife Service and National Marine Fisheries Service. 2014. Final policy on interpretation of the phrase ''Significant Portion of Its Range'' in the

Endangered Species Act's definitions of ''Endangered Species'' and ''Threatened Species''; final rule. Federal Register 79(126):37578-37612.

[USFWS] U.S. Fish and Wildlife Service. 2010. Endangered and threatened wildlife and plants; 12-month finding on a petition to list the Sonoran population of the desert tortoise as endangered or threatened. Federal Register 75(239):78094-78146.

[USFWS] U.S. Fish and Wildlife Service. 2015a. Species status assessment for the Sonoran Desert tortoise. Version 1.0, September 2015. Albuquerque, New Mexico: Southwest Region U.S. Fish and Wildlife Service. 35 pp. (see *Supplemental Material*, Reference 8,) ; also available: https://www.fws.gov/southwest/es/arizona/Documents/SpeciesDocs/SonoranTort/SDT_SSA_Report_September2015_Chp1-3.pdf (October 2017).

[USFWS] U.S. Fish and Wildlife Service. 2015b. Endangered and threatened wildlife and plants; 12-month finding on a petition to list Sonoran Desert tortoise as an endangered or threatened species. Federal Register 80(193):60321-60335.

[USFWS] U.S. Fish and Wildlife Service. 2016a. Long-term listing transformation and five guiding principles of the Unified Listing Team. Director's memorandum 16 March 2016, Washington, D.C. (see *Supplemental Material*, Reference 9,).

[USFWS] U.S. Fish and Wildlife Service. 2016b. USFWS species status assessment framework. Falls Church, Virginia: U.S. Fish and Wildlife Service. 21 pp. (see *Supplemental Material*, Reference 10,) ; also available: https://www.fws.gov/endangered/improving_ESA/pdf/SSA%20Framework%20v3.4-8_10_2016.pdf (October 2017).

[USFWS] U.S. Fish and Wildlife Service. 2016c. Species status assessment for the eastern massasauga rattlesnake (*Sistrurus catenatus*). Falls Church, Virginia: U.S. Fish and Wildlife Service 117 pp. (see *Supplemental Material*, Reference 11,) ; also available: https://www.fws.gov/midwest/endangered/reptiles/eama/pdf/SSAFinalV2July2016EMR.pdf (October 2017).

[USFWS] U.S. Fish and Wildlife Service. 2016c. Endangered and threatened wildlife and plants; threatened species status for the eastern massasauga rattlesnake. Federal Register 81(190):67193-67214.

Van Devender TR. 2002. Natural history of the Sonoran tortoise in Arizona: life in a rock pile. Pages 3-28 in Van Devender TR, editor. The Sonoran Desert tortoise: natural history, biology, and conservation. Tucson, Arizona: The University of Arizona Press.

Vucetich J A, Nelson MP, Phillips MK. 2006. The normative dimension and legal meaning of endangered and recovery in the U.S. Endangered Species Act. Conservation Biology 20:1383-1390.

Waples RS, Nammack M, Cochrane JF, Hutchings JA. 2013. A tale of two acts: endangered species listing practices in Canada and the United States. BioScience 63:723–734.

Wilhere GF. 2011. Inadvertent advocacy. Conservation Biology 26:39-46.

Williams BK, Johnson FA. 2015. Value of information and natural resource decision making. Wildlife Society Bulletin 39:488-496.

Wolf S, Hartl B, Carroll C, Neel MC, Greenwald DN. 2015. Beyond PVA: why recovery under the Endangered Species Act is more than population viability. BioScience 65:200-207.

Woods T, Morey S. 2008. Uncertainty and the Endangered Species Act. Indiana Law Journal 83:529-536.

Zylstra ER, Steidl RJ, Jones CA, Averill-Murray RC. 2013. Spatial and temporal variation in survival of a rare reptile: a 22-year study of Sonoran Desert tortoises. Oecologia 173:107–116.

Figure 1. Diagram of the relationship between species status assessment (SSA) and various *endangered species* decisions. For a *listing* decision (Table 1), an SSA is used to assess the current and future condition of the species under scenarios of continuing, increasing or decreasing stressors and conservation efforts. If the species is warranted for *listing* under the ESA, then a series of decision will be made including *candidate conservation*, *recovery planning*, *consultation* and permitting, *five-year review*, and *reclassification*. Each decision can be informed by the SSA, which has been adapted to account for decision context and updated to include new data and information. The box below the decision indicates some of the aspects that need to be incorporated into an SSA for a specific decision.

Figure 2. The current distribution map for the eastern massasauga rattlesnake (*Sistrurus catenatus*) including parts of southern Ontario, Canada and midwest and northeast US. Three analysis units, which were identified to represent adaptive capacity, are shown in the western, central, and eastern portion of the range. Dots represent counties with at least one population remaining as of 2014; X represents a county that no longer supports a population. A contraction is evident from historical to current times.

Figure 3. The projected number of eastern massasauga rattlesnake (*Sistrurus catenatus*) populations before 2014 (H), 2014-2016 (Current), and into the future (10, 25, and 50 years from 2016) based on population model by Faust et al. (2011 S9). Projected numbers are shown for the three analysis units representing the western (WAU) central (CAU), and eastern (EAU) portions of the species range (cf Figure 2).

Figure 4. Conceptual models used for Sonoran Desert tortoise (*Gopherus morafkai*) assessment to A) illustrate factors used to measure habitat quality and quantity, and B) diagram the population model presented in McGowan et al. (2017).

Figure 5. Plots of predicted future median abundance of Sonoran Desert tortoise (*Gopherus morafkai*) over years from present (solid line, primary (left) axis) with 95% confidence interval (dashed lines, primary axis) and the probability of quasi extinction (shaded area, secondary (right) axis) for the worst (A, B) and best case (C, D) future scenarios in Arizona, US (A, C) and Sonora, Mexico (B, D). The population model presented in McGowan et al. (2017) was used to

project population abundance. Two quasi-extinction thresholds were evaluated – 2% or 4% of the initial population abundance. The worst case scenario (A, B) combines high stressors and low conservation efforts with the 4% quasi-extinction threshold. The best case scenario (C, D) combine low stressors and high conservation efforts with the 2% quasi-extinction threshold.

Table 1. Glossary of selected terms related to the Endangered Species Act (ESA) and species status assessments.

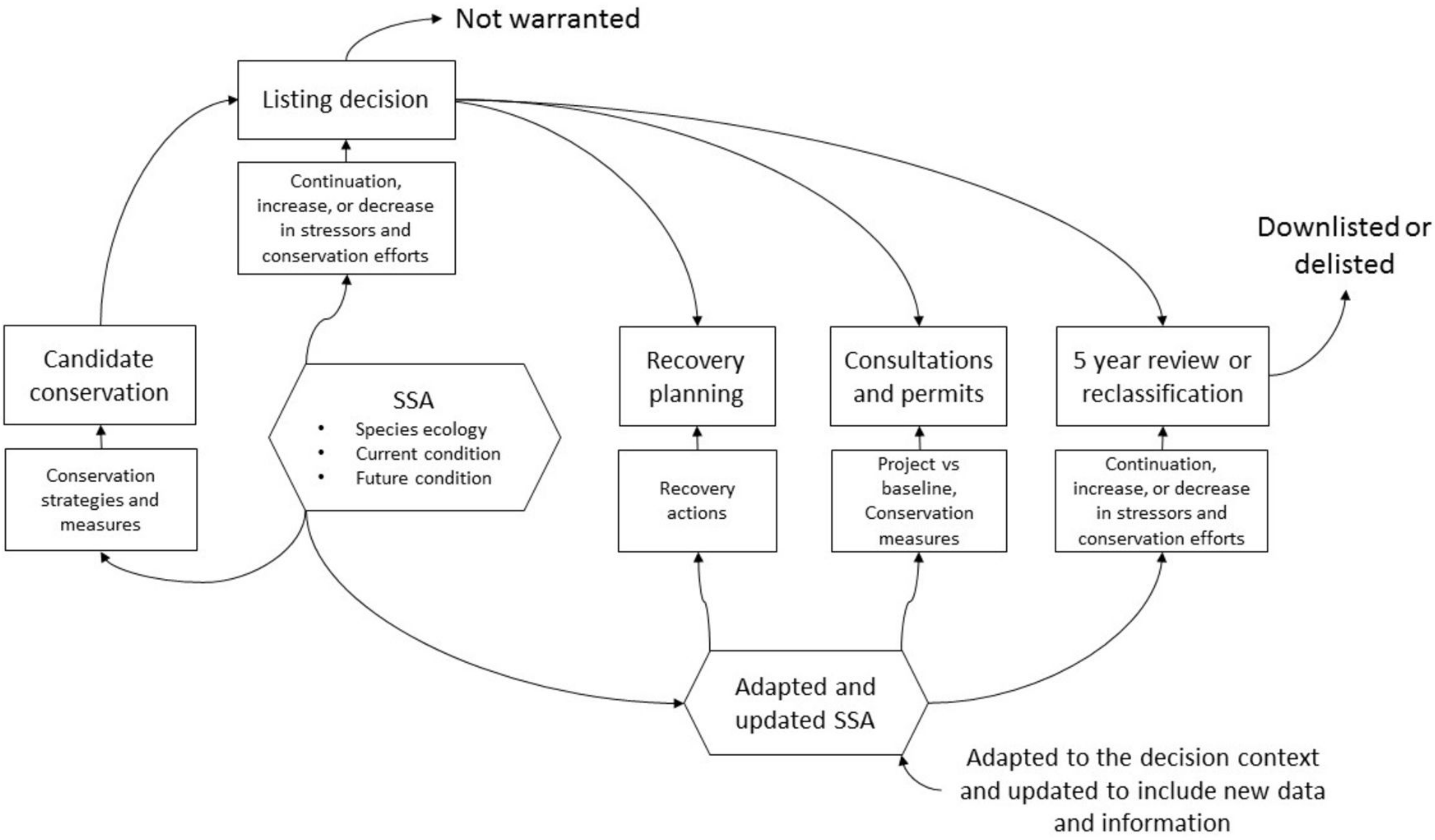
Terms	Definition
Best Available Science	A phrase used to reference an ESA provision that the Secretaries of the Interior and Commerce make listing determinations "based on the best scientific and commercial data available" (U.S. Fish and Wildlife Service and National Marine Fisheries Service 1994).
Candidate Conservation	Voluntary conservation efforts focused on species that are candidates for listing, or species that may be considered for listing, under section 4 of the ESA to improve the overall status of the species.
Consultations and Permits	The process by which the NOAA and U.S. Fish and Wildlife Service consult with Federal agencies proposing actions that may affect a listed species (consultations under section 7 of the ESA) or permit nonfederal entities to legally take a listed species under section 10 of the ESA.
Endangered Species	As defined in section 3 of the ESA, an endangered species is any species which is in danger of extinction throughout all or a significant portion of its range.
Five Factors	The factors listed in section 4 of the ESA that identify broad categories of natural or manmade actions that are used to determine the causes for any species listed as endangered or a threatened. The five factors are (A) destruction, modification, or curtailment of habitat or range; (B) overutilization; (C) disease or predation; (D) inadequate regulatory mechanisms; and (E) a catch-all category of other natural or man-made factors.
Five-year Review	A review of the status of species listed under section 4 of the ESA that is conducted once every five years to ensure that the species has the appropriate ESA status and level of protection.
Foreseeable Future	A timeframe which a decision maker can reasonably rely on predictions about the future in making determinations about the future status of the species (U.S. Department of the Interior

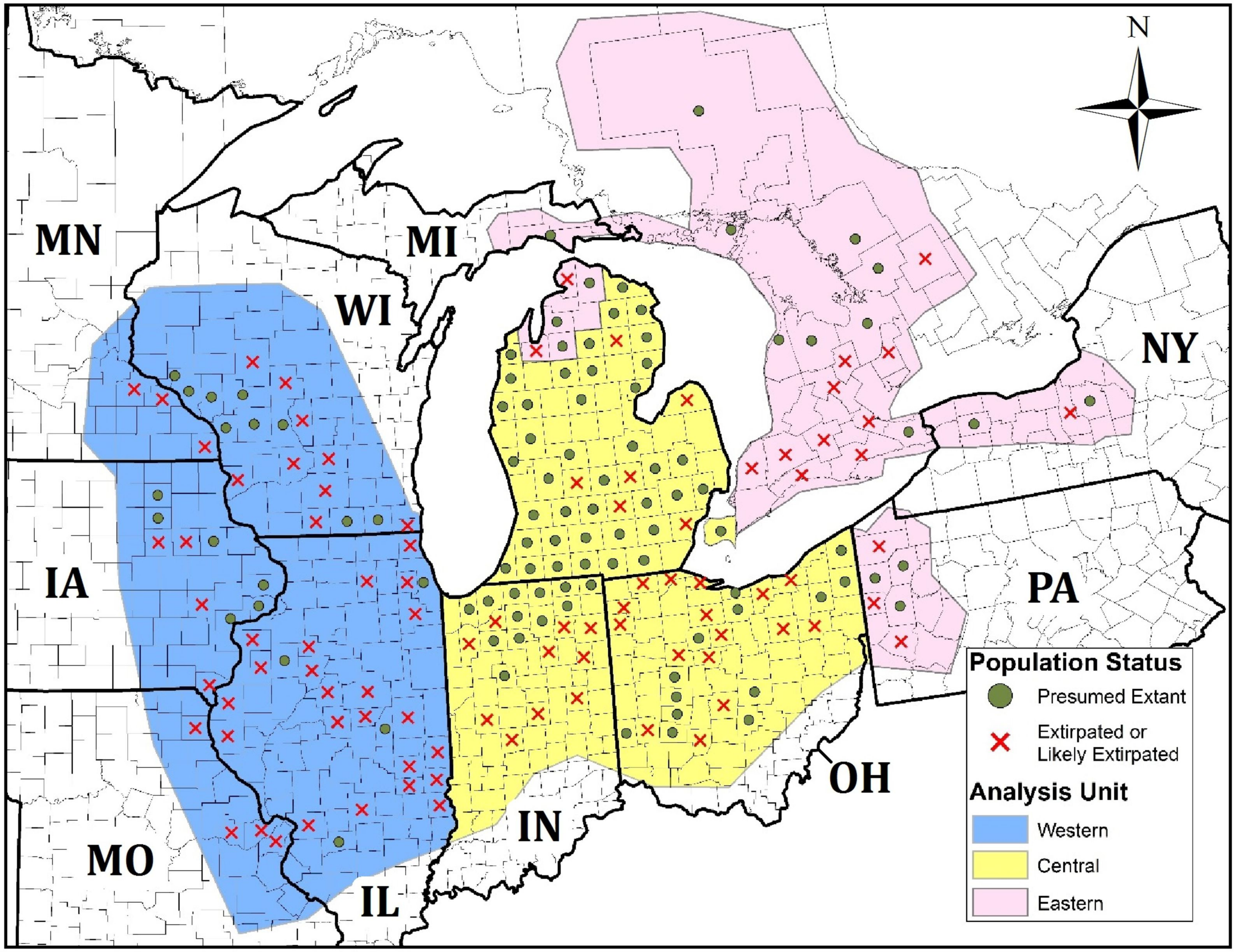
	2009).
Listing	The process by which species are added to the lists of endangered and threatened wildlife and plants under section 4 of the ESA.
Reclassification	The process by which the classification of listed species (threatened species or endangered species) are changed under section 4 of the ESA. Endangered species may be downlisted to threatened species or delisted and removed from the lists. Threatened species may be uplisted to an endangered species or delisted and removed from the lists.
Recovery Planning	The process of developing a recovery plan for a listed species including the recovery vision and strategy, recovery criteria, recovery actions, and estimates of the time and costs to achieve the plan's goals under section 4 of the ESA.
Redundancy	Redundancy describes the ability of a species to withstand catastrophic events by spreading risk among multiple populations to minimize the potential loss of the species.
Representation	Representation describes the ability of a species to adapt to changing environmental conditions over time as characterized by the breadth of genetic and environmental diversity within and among populations.
Resiliency	Resiliency describes the ability of a species to withstand stochastic disturbance and is positively related to population size and growth rate and may be influenced by connectivity among populations.
Species	As defined in section 3 of the ESA, the term species includes any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature.
Threat	Any action or condition that is known to, or is reasonably likely to, negatively affect individuals of a species, including direct impact on individuals and alterations of their habitat or required resources.

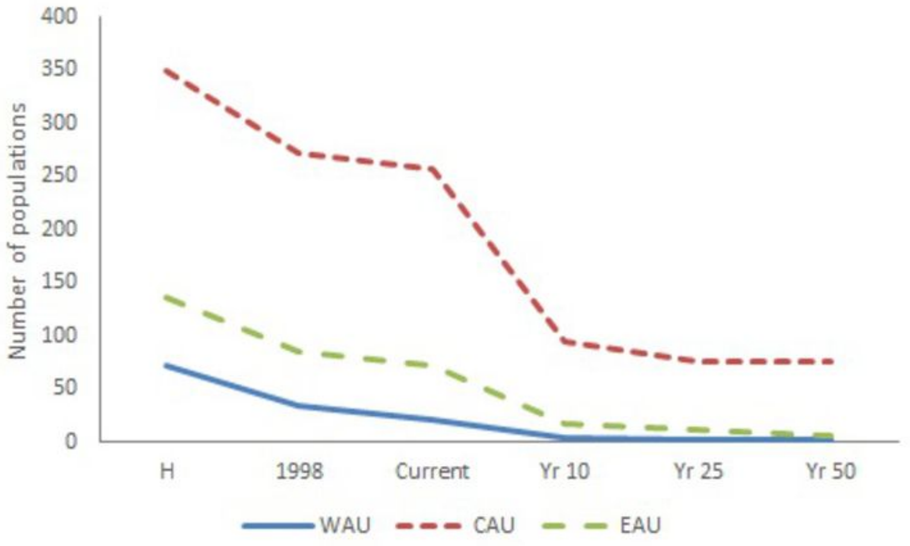
Threatened Species	As defined in section 3 of the ESA, a threatened species is any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.

Table 2. Identification of the decision elements included in Endangered Species Act (ESA) decisions. ("Policy" includes legislation, agency regulations and policies, and court decisions.)

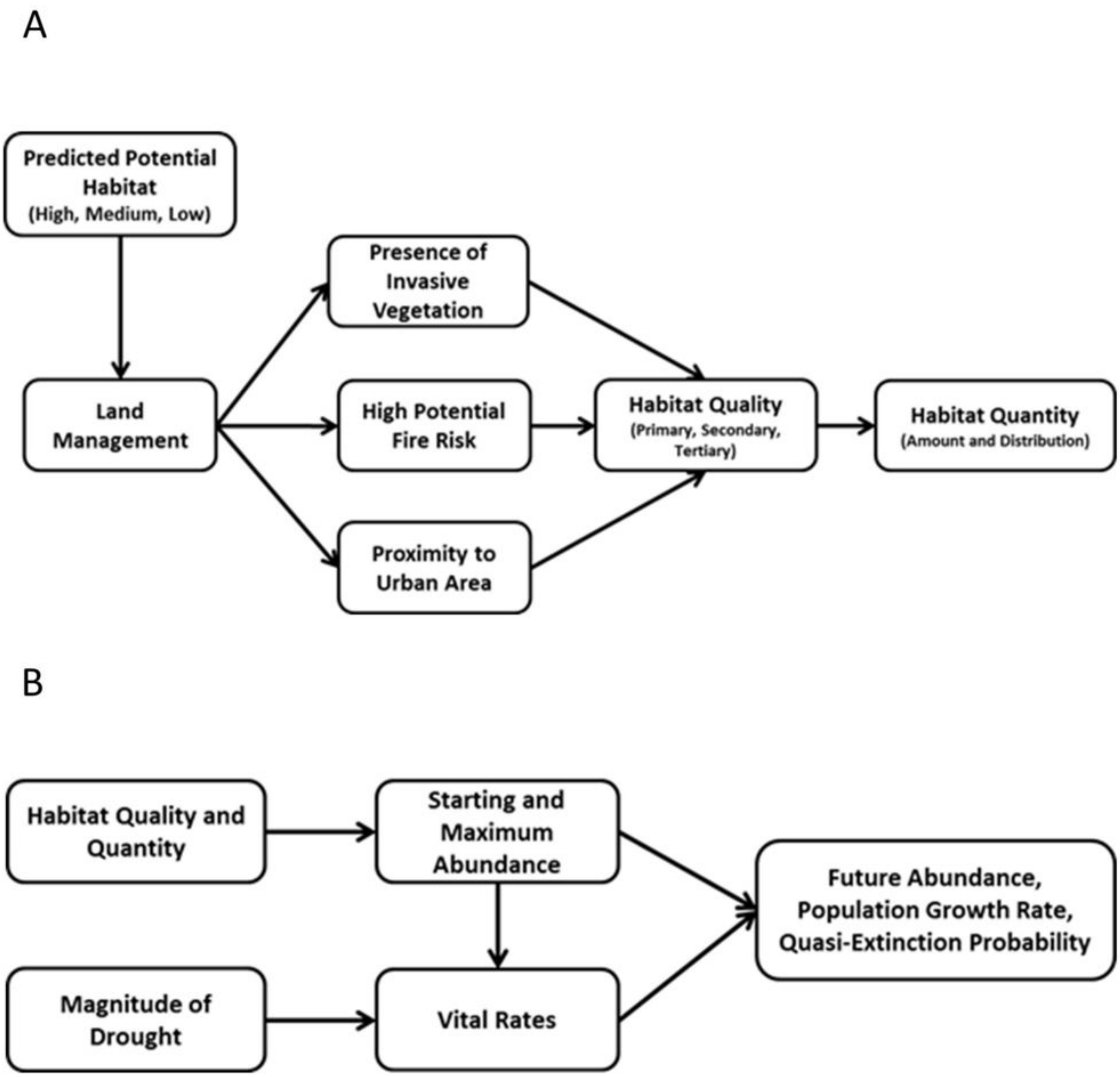
Decision elements	Science – species risk	Policy – ESA standards
Process	SSA Framework	ESA Decision Making
Who	Biologists and other scientists	Decision Makers
How	Scientific Analysis	Policy Analysis
	(Biological and environmental data)	(Legal interpretation)
Outcome	Viability Characterization	Policy Judgment

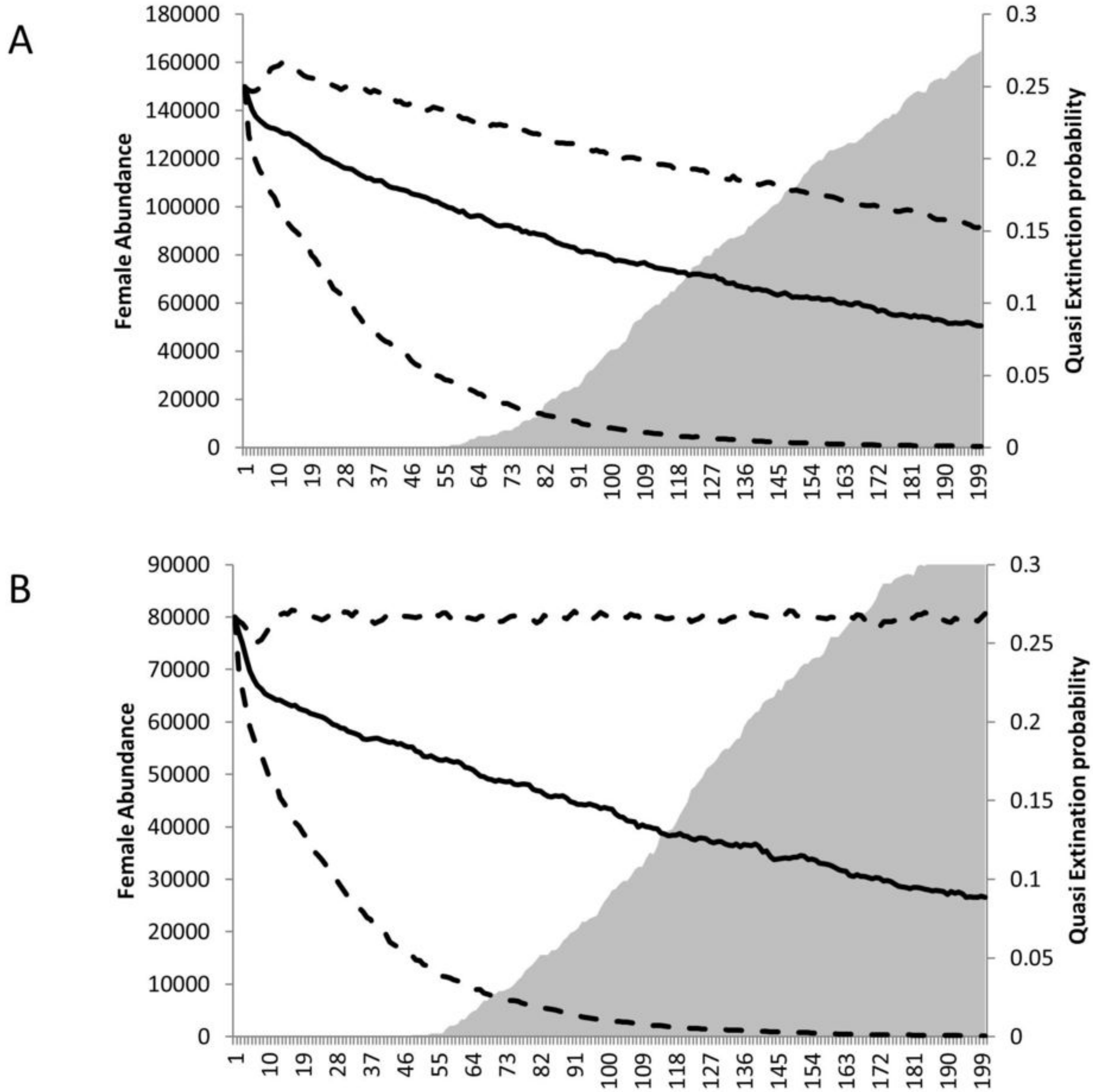

Not warranted
Listing decision
Continuation, increase, or decrease in stressors and conservation efforts
Downlisted or delisted
Candidate conservation
SSA
Species ecology
Current condition
Future condition
Recovery planning
Consultations and permits
5 year review or reclassification
Conservation strategies and measures
Recovery actions
Project vs baseline, Conservation measures
Continuation, increase, or decrease in stressors and conservation efforts
Adapted and updated SSA
Adapted to the decision context and updated to include new data and information


N
MN
MI
WI
NY
IA
PA
OH
IN
MO
IL
Population Status
Presumed Extant
Extirpated or
Likely Extirpated
Analysis Unit
Western
Central
Eastern



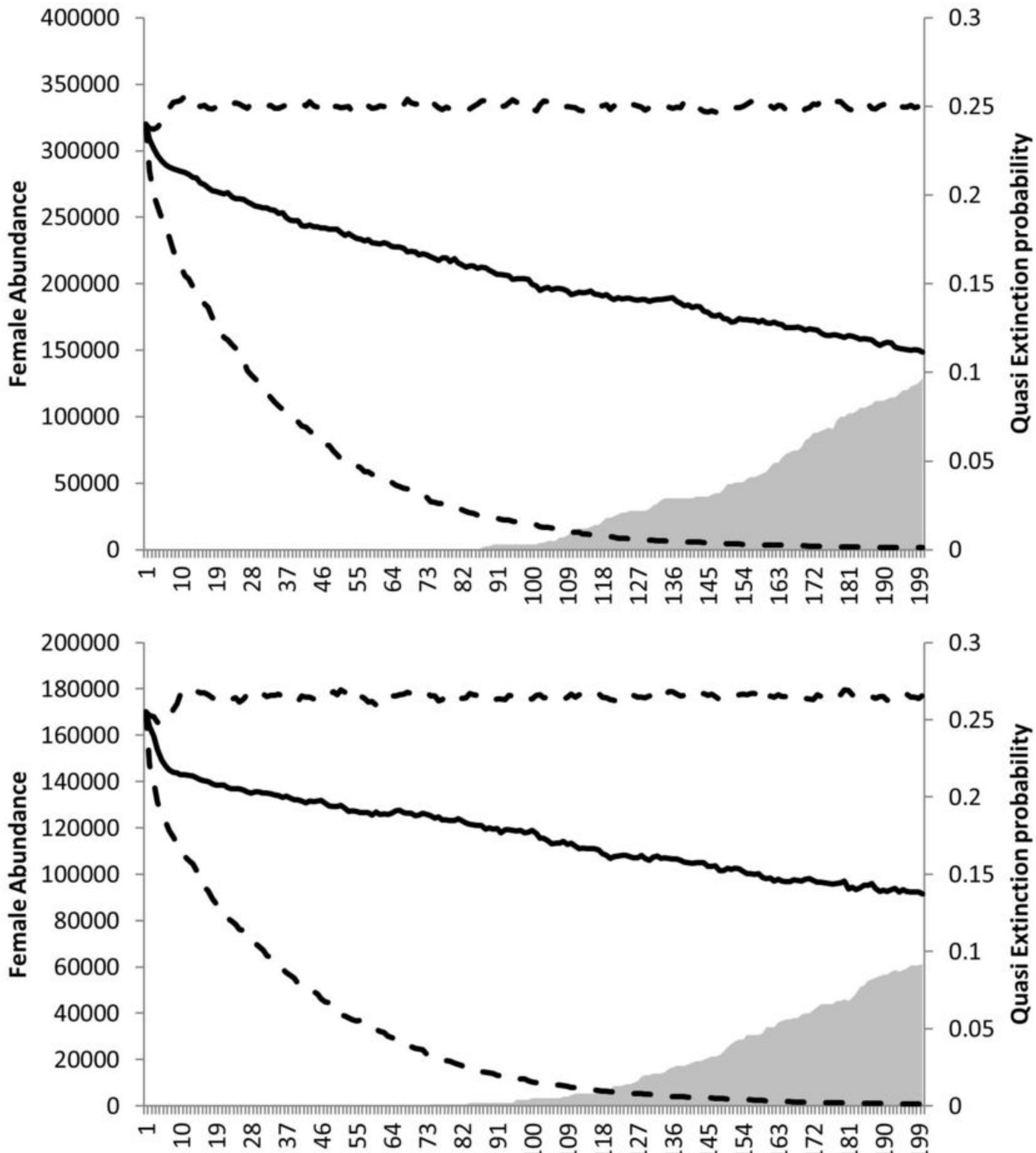
400
350
300
250
200
150
100
50
0
Number of populations
H
1998
Current
Yr 10
Yr 25
Yr 50
WAU
CAU
EAU


A
Predicted Potential Habitat
(High, Medium, Low)
Land Management
Presence of Invasive Vegetation
High Potential Fire Risk
Proximity to Urban Area
Habitat Quality
(Primary, Secondary, Tertiary)
Habitat Quantity
(Amount and Distribution)
B
Habitat Quality and Quantity
Starting and Maximum Abundance
Magnitude of Drought
Vital Rates
Future Abundance, Population Growth Rate, Quasi-Extinction Probability


A
Female Abundance
Quasi Extinction probability
180000
160000
140000
120000
100000
80000
60000
40000
20000
0
0.3
0.25
0.2
0.15
0.1
0.05
0
1 10 19 28 37 46 55 64 73 82 91 100 109 118 127 136 145 154 163 172 181 190 199
B
Female Abundance
Quasi Extination probability
90000
80000
70000
60000
50000
40000
30000
20000
10000
0
0.3
0.25
0.2
0.15
0.1
0.05
0
1 10 19 28 37 46 55 64 73 82 91 100 109 118 127 136 145 154 163 172 181 190 199

C

D



Female Abundance
Quasi Extinction probability
400000
350000
300000
250000
200000
150000
100000
50000
0
0.3
0.25
0.2
0.15
0.1
0.05
0
1
10
19
28
37
46
55
64
73
82
91
100
109
118
127
136
145
154
163
172
181
190
199
Female Abundance
Quasi Extinction probability
200000
180000
160000
140000
120000
100000
80000
60000
40000
20000
0