

Health & Ecological Risk Assessment

Guidance for Developing Amphibian Population Models for Ecological Risk Assessment

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ABSTRACT

Despite widespread acceptance of the utility of population modeling and advocacy of this approach for a more ecologically relevant perspective, it is not routinely incorporated in ecological risk assessments (ERA). A systematic framework for situation-specific model development is one of the major challenges to broadly adopting population models in ERA. As risk assessors confront the multitude of species and chemicals requiring evaluation, an adaptable stepwise guide for model parameterization would facilitate this process. Additional guidance on interpretation of model output and evaluating uncertainty would further contribute to establishing consensus on good modeling practices. We build on previous work that created a framework and decision guide for developing population models for ERA by focusing on data types, model structure, and extrinsic stressors relevant to anuran amphibians. Anurans have a unique life cycle with varying habitat requirements and high phenotypic plasticity. These species belong to the amphibian class, which is facing global population decline in large part due to anthropogenic stressors, including chemicals. We synthesize information from databases and literature relevant to amphibian risks to identify traits that influence exposure likelihood, inherent sensitivity, population vulnerability, and environmental constraints. We link these concerns with relevant population modeling methods and structure in order to evaluate pesticide effects with appropriate scale and parameterization. A standardized population modeling approach, with additional guidance for anuran ERA, offers an example method for quantifying population risks and evaluating long-term impacts of chemical stressors to populations. *Integr Environ Assess Manag* 2019;00:1–11. © 2019 SETAC

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INTRODUCTION

Population modeling has been identified as a priority for ecological risk assessment (ERA) (NRC 2013; Forbes et al. 2016), recognizing that in most cases, organism-level endpoints collected in a laboratory do not adequately represent protection goals in the appropriate ecological context (Cairns 1988; Chapman 2002; De Lange et al. 2010; Forbes and Calow 2013). Modeled estimates of population growth rate and species abundance are practical population-level endpoints that can be used to determine the long-term effects of stressors such as chemical contaminants. Population models have been used in wildlife management for species of concern for decades (Caswell 2001; Morris and Doak 2002), yet their application in ERAs has been limited. In ecotoxicology, models can depict detrimental exposures to chemicals within the context of population decline and recovery or translate sublethal effects into long-term impacts.

Despite global population concerns, amphibians are underrepresented in ecological models (Schmolke et al. 2010), and standardized toxicological methods for amphibians are relatively recent (OECD 2014, 2015); therefore, significant advancement in understanding critical data needs and uncertainties of amphibian population models is needed (EFSA 2018). Here we provide a review of constraints important to amphibian population dynamics as a first step in the necessary translational ecology, interpreting ecological knowledge in terms amenable to a decision-making process (Enquist et al. 2017).

Because of their varied habitat, life-histories, and vulnerability to multiple environmental stressors, assessment of chemical risk to amphibians should be placed in a larger, ecologically relevant context (Johnson et al. 2017). Amphibian life cycle and ontogeny are distinct from other taxonomic classes and vary widely between orders. Most notably, the aquatic larval and terrestrial adult life stages of anurans present challenges to risk assessments compared with other taxa. For example, 2 surrogate groups are typically used to represent amphibians in risk assessment; fish

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are used as surrogates for an aquatic tadpole stage, and small birds or mammals are used to represent terrestrial juvenile and adult stages. This disjointed surrogacy introduces uncertainty in the assessment of chemical risk to amphibians. Furthermore, the exposure routes and physiological development of ectothermic amphibians differ considerably from these surrogate groups (Quaranta et al. 2009). Therefore, the inherent sensitivity of amphibians to long-term exposures is likely to be mischaracterized by these surrogates, creating additional challenges for population-level risk assessment.

In addition to differences in species sensitivity and demographic traits, characteristics that influence population dynamics across space and time can also be associated with vulnerability (Van Allen et al. 2012). Existing studies identify ecological and life-history traits that are drivers of amphibian threats and decline (Lips et al. 2003; Sodhi et al. 2008). These provide a preliminary means of evaluating the anticipated risk for a population, which could determine the extent to which species life-history and habitat should be represented in the assessment. For example, species with variable habitat availability might require more temporal detail to capture representative population dynamics. Species with greater dispersal could suggest the need for additional spatial context for inclusion of animal movement patterns. Consideration of mechanistic or functional traits (e.g., physiology or diet) that contribute to variability of exposure impact can also improve taxa-specific risk assessments (Baird and Van den Brink 2007; Rubach et al. 2012; Rico and Van den Brink 2015). Likewise, life-history strategies and associated demographic rates can be indicative of population resilience to extrinsic stressors (Stark et al. 2004). Defining a species' trait-based vulnerability also facilitates determination of focal species (representative of other species in a defined group) which could be modeled further (EFSA 2018). By identifying traits associated with species vulnerability, we can ensure that they inform population models so that estimations of risk are adequately represented.

As simplifications of ecological systems, models contain uncertainty, both quantitative (e.g., data variance) and qualitative (e.g., lack of data), that must be considered. The latter includes the breadth of factors that could influence population susceptibility but are omitted from ERAs because of the lack of information. Indeed, if ecological factors or life-history traits known to influence risk are overlooked in model development, this can result in considerably more uncertainty (Forbes and Calow 2013); however, the use of qualitative information can provide some initial guidance in decision making when categorical traits are linked to highly vulnerable populations (Pacifi et al. 2015). Despite available guidance on development of population models for ERA (Pastorok et al. 2002; Munns and Mitro 2006; Barnhouse et al. 2008; Schmolke et al. 2010; Forbes et al. 2016), they are still not being used regularly in decision making. Recent studies on the application of population models in ERA present more systematic processes to

develop and implement models (Schmolke et al. 2017; Raimondo et al. 2018). Raimondo et al. (2018) presented a framework in which a range of model realism and precision associated with an assessment is used to guide the investment level commensurate with needs of the protection goal. A decision guide by Schmolke et al. (2017) moves risk assessors through a stepwise process to assist them in developing a model that is appropriate for the assessment objective and available data, demonstrated using an endangered plant species. Consensus on standardized model development procedures is important for applying population modeling in risk assessment consistently and reproducibly, and additional work is needed to expand and demonstrate guidance across different taxa and exposure scenarios. Additionally, interpretation of model outputs and associated uncertainty in the context of decision making and ERA objectives should be derived from consensus approaches. The current study expands on model development guidance for risk assessment by outlining modeling endpoints and options specific to anuran amphibians and incorporating critical traits into model development. We link amphibian traits associated with population vulnerability to decision steps, sensu Schmolke et al. (2017), to guide model development and use of spatial and temporal representations that are relevant to anuran species.

Amphibian vulnerability traits

Database compilations of amphibian demography and habits (Oliveira et al. 2017; Tacutu et al. 2018; AmphibiaWeb 2019) and previously published reviews and meta-analyses (Lips et al. 2003; Sodhi et al. 2008; Egea-Serrano et al. 2012; EFSA 2018) were used to construct a list of traits related to amphibian vulnerability (Table 1). Traits are classified by the way in which they might be indicative of population risk and how they would influence model development. Vulnerability is dependent on the exposure regime, the sensitivity of the organism, the potential for the population to recover based on its life-history, and indirect or environmental effects that might exacerbate impacts. This identification of traits allows their potential indication of increased toxicant exposure risk to guide decisions in model development pertaining to surrogacy, parameterization, or spatio-temporal model complexity.

Exposure-related traits can be linked to species' habits or chemical use and properties. The spatial overlap of species and chemical distribution as well as the likelihood of contact or ingestion comprise this group of traits. For example, species with specialist diets could be more vulnerable to effects of chemical use if their primary prey item is impacted, an effect that will vary between larvae and adults of the same species. Spatial and temporal aspects of chemical use might also affect the impact on a population, relative to the life-cycle stages that are exposed and the duration of exposure.

Organismal traits indicative of inherent sensitivity offer a means of evaluating surrogate species data. For example, if dietary habits result in different exposure between the surrogate and the target species, observed effects in the

Table 1. Amphibian population vulnerability traits with the modeling decision steps^a potentially affected by the trait

Model decision step	Trait description	Relationship of trait to risk of adverse effects	Relationship	Source
Organism-level processes	Molecular receptors	+/-	Similarity to receptor of surrogate	Egea-Serrano et al. 2012
Organism-level processes	Food retention	+	Greater internal dose	EFSA 2018
Organism-level processes	Metabolism	+/-	Higher metabolism, lower internal dose; higher metabolic rate, increased aquatic species sensitivity	Baas and Kooijman 2015; EFSA 2018
Organism-level processes	Excretion	-	Higher excretion, lower internal dose, lower risk	EFSA 2018
Organism-level processes, Indirect effects	Diet (larvae and juvenile/adult)	+	Specialist diets more sensitive than generalists	EFSA 2018
Organism-level processes, temporal representation	Tolerance (repeated exposure)	+/-	Species and location dependent	Hua et al. 2014
Organism-level processes	Body size (average)	-	Relative body burden reduced for larger species in same environment; also linked to life-history characteristics	Sodhi et al. 2008
Life-history representation	Life span	-	The longer the life span, the more vulnerable a population is	Rowe 2008
Life-history representation	Larval duration	+	The longer the development time, the more vulnerable the population is; determinant of annual breeding success	EFSA 2018; Sodhi et al. 2008
Life-history representation	Time to maturation	+	The longer the maturation time, the more vulnerable the population	Rowe 2008; Relyea and Diecks 2008
Life-history representation, Temporal representation	Frequency of breeding	-	The more breeding opportunities, the less vulnerable the population	Stearns 1992
Life-history representation	Mode of reproduction	+/-	More terrestrial breeding modes associated with smaller clutches	Duellman and Trueb 1986
Life-history representation, Organism-level processes, Density dependence	Survival	+	The higher the baseline survival rate, the greater the risk to extrinsic sources of mortality	Rowe 2008

(Continued)

Table 1. (Continued)

Model decision step	Trait description	Relationship of trait to risk of adverse effects	Relationship	Source
Life-history representation, Organism-level processes, Density dependence	Reproduction rate	–	The higher the reproductive potential, the more resilient the population	EFSA 2018
Life-history representation, Spatial representation, Density dependence	Dispersal	–	The greater the dispersal rate, the lower the chance the population will be impacted by local exposure	EFSA 2018
Spatial representation, Density dependence	Geographic range	–	The larger the geographic range, the lower the chance that the population will go extinct from local exposure	Sodhi et al. 2008
Spatial representation, Density dependence	Population size	–	Smaller populations more at risk	Dickinson et al. 2014; Pearson et al. 2014
Organism-level processes, spatial representation	Pathogens	+	Synergistic effects of pathogens and increased chemical risks, unless fungicide reduces pathogen prevalence	Kiesecker 2002; Rohr et al. 2008; Hua et al. 2016; Stutz et al. 2017
Spatial-temporal representation	Habitat loss or degradation/availability	–	Reduced habitat availability/increased fragmentation will limit population resilience	Stuart et al. 2004; Fonseca et al. 2013
Temporal representation	Hydroperiod variability	+/–	Variability in annual breeding success, depending on larval duration	Sodhi et al. 2008; Lambert et al. 2016
Spatial-temporal representation	Environmental variability	+	Variability in habitat availability/quality affects breeding success/survival, might dictate metapopulation dynamics	Lambert et al. 2016

+ = positive relationship; – = inverse relationship; +/- = trait could make species more or less susceptible, depending on specifics.
^asensu Schmolke et al. 2017.

surrogate might not appropriately represent the expected effects in the species of interest. Proximate, or mechanistic, distinctions in toxicological effects are partially dependent on evolutionary relatedness, although differences in species sensitivity are only weakly related to phylogeny (Egea-Serrano et al. 2012). Physiological characteristics, such as food retention time in conjunction with metabolic and excretion rates, determine internal exposure (Baas and Kooijman 2015). Additionally, processes of elimination via egg production or shedding could reduce body burdens, as could physiological transitions such as estivation or metamorphosis. Amphibians have a variety of exposure routes throughout their life cycle in 2 different ecosystems, with permeable dermal exposure leading to higher uptake than mammals in the same terrestrial environment (Brühl et al. 2013). Similarities in organismal-level traits of surrogates can offer some initial context of observed effects and their representativeness.

Life-history traits include characteristics such as maturation age, fecundity, and survival rate. These vital rates are commonly needed for population model parameterization, and they represent evolutionary tradeoffs related to a species' longevity, movement patterns, and resilience. A global analysis revealed that life-history strategy and limited geographic range were most closely linked to amphibian species risk (Sodhi et al. 2008), and patterns in life-history traits correspond to geographic differences in altitude or latitude (Morrison and Hero 2003). Restricted elevational range and large body size, along with aquatic habitat use were also linked to population declines in Central American species (Lips et al. 2003). Inherent properties of larger amphibians include a "slower" life-history; larger species tend to mature later and reproduce more slowly (Stearns 1992) and are thus more vulnerable to population decline by anthropogenic stressors (Rowe 2008). "Fast" life-history traits are often associated with invasive species (Allen et al. 2017); however, "faster" life-histories are often more vulnerable to infection and disease (Johnson et al. 2012). Life-cycle variation (e.g., duration of larval period) influences external exposure as a result of time spent in a contaminated habitat or internal dose as a function of size at metamorphosis, which may vary between species to the same contaminant. Duration of the larval phase can also contribute to interannual variance in breeding success. Rapid development is frequently an adaptation to variability in environmental conditions, such that a prolonged developmental stage is often less resilient to environmental stochasticity, as well as a limited distribution (Sodhi et al. 2008). Survival can be dependent on body size, site, season duration, and freeze transitions, whereas recruitment can be dependent on environmental conditions such as pond depth, the previous year's precipitation, season duration, annual number of egg masses, body size, and site (Lambert et al. 2016). The capacity for any population to recover from detrimental impacts is often dependent on the species' reproductive rate, ability to recolonize based on movement patterns and home range, and co-occurrence with multiple stressors (EFSA 2018).

Indirect or environmentally linked traits are important to consider because physiology and life-history strategy alone are not representative of extinction risk in amphibian species (Green 2003; Fonseca et al. 2013). Environmental traits include extrinsic factors that influence a population, and they vary spatially or temporally. These landscape-scale threats are mostly informative in determining the spatial and temporal extent of model representation or the population's interactions with the environment. Factors frequently linked to amphibian decline are pathogens, pollution, habitat loss or degradation, and changing environmental conditions (Blaustein and Kiesecker 2002; Sodhi et al. 2008; Mann et al. 2009), although stressors contributing to population declines are many and vary geographically (Grant et al. 2016). Sensitivity to environmental stressors has been linked to smaller population size and geographic range (Dickinson et al. 2014; Pearson et al. 2014). Extreme seasonality associated with hydroperiod was also linked to population declines (Sodhi et al. 2008). The risk of exposure to pesticides in the United States is greatest in Midwest states, where susceptibility to *Batrachochytrium dendrobatidis* (the amphibian chytrid fungus) and habitat degradation due to human activity are also high (Grant et al. 2016). Indirect effects of reduced resources potentially impact species with longer larval periods (Relyea and Diecks 2008).

Anuran population modeling decision steps

The decision guide of Schmolke et al. (2017) comprises 4 primary phases: 1) definition of model objectives, 2) compilation of available data, 3) decision steps, and 4) development of a representative model to address the model objectives. Here, we focus on the decision steps specifically for amphibians, identifying ways in which this information might guide adaptations to a minimum anuran conceptual model (Awkerman and Raimondo 2018) (Figure 1). The initial 2 phases of the decision guide pertaining to defining model objectives and compilation of available data are ERA-specific and not discussed here. Through the process of describing decision steps, we identify where critical traits should be considered. While data availability might preclude incorporation of empirical functions for processes

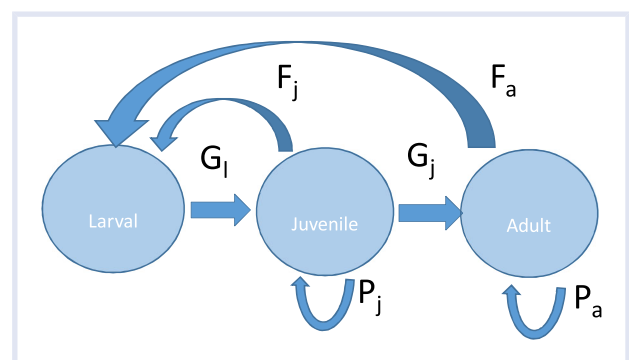


Figure 1. A 3-stage life-cycle model for anurans. Adapted from Awkerman and Raimondo (2018). a = adult; F = fecundity; G = growth; j = juvenile; l = larval; P = survival.

such as density dependence or indirect effects, determining whether their inclusion could greatly influence model outcomes is an important step in model development.

The anuran guidance for trait identification (Table 1) is based on parameters and characteristics identified by Schmolke et al. (2017). A simple, 3-stage demographic matrix model (Figure 1) is provided as an initial minimal conceptual model of anuran species and provides a starting point for potential iterative expansion of the model as appropriate. In brief, the model has an annual time step and is represented by 3 life stages: the larval stage, including the 1st year of life; the juvenile stage, following the 1st year and preceding reproductive maturity; and the adult stage, when individuals are reproductively active. The model includes fecundity, estimated for adults and juveniles transitioning to adults; survival, as the proportion of individuals remaining in a life stage during the time step; and growth, as a probability of transitioning to the next life stage. Below, we provide recommendations of modifications to this initial minimal conceptual model based on identified anuran traits that may indicate increased pesticide risk. Recommendations are given for each of the decision steps below (originally described by Schmolke et al. 2017), pertaining to life-history representation, organism-level processes, temporal and spatial extent, density dependence, indirect effects, population size, and environmental conditions. Consideration of the identified anuran traits and their potential interactions with chemical risk apply beyond the initial minimal conceptual model, that is, they should be addressed in the development of anuran population models for chemical risk assessment in general.

Life-history representation. A basic life-history representation for anurans (e.g., Awkerman and Raimondo 2018) (Figure 1) uses an annual time step that can be applied or adapted for most modeling scenarios. When 1st-year survival is not known, estimates could combine early life-stage transitions that occur in different circumstances or in different habitats (i.e., separate estimates of hatching rate, metamorphosis rate, and postmetamorphosis survival). Alternatively, additional stages can be added to the model (e.g., embryo), adjusting the time step accordingly. Although growth functions can define the thresholds of some stage-based models (Caswell 2001), developmental plasticity in amphibians suggests that using size to determine life stage could inaccurately represent development and include false assumptions about successful metamorphosis (Richter-Boix et al. 2011). Therefore, growth in this context refers to size-independent transitions to discrete life stages. The focal taxonomic group, anurans, typically displays an aquatic larval development phase followed by terrestrial juvenile and adult phases (Duellman and Trueb 1994), such that movement and habitat constraints should be considered for 2 distinct environments. Although modes of reproduction and life cycles vary greatly among amphibians, it is with this common pattern of habitat use in mind that these decision steps are discussed. Traits relevant to this decision

step are primarily demographic rates, such as survival, reproduction rate, frequency of breeding, larval duration, time to maturation, and life span.

Organism-level processes. Organism-level processes, especially if they are relevant to sublethal effects, might be included in population models as appropriate. Sublethal growth effects are most likely to impact populations when they are in conjunction with developmental constraints of metamorphosing in ephemeral breeding ponds. Growth effects in amphibians should be presented in appropriate ecological context, considering the transition from tadpole to juvenile stage and the habitat available. In life cycles where larval habitat is constrained, developmentally delayed individuals could be unable to metamorphose and effectively reduce larval survival rates in temporarily available habitat. Reproductive effects could include reduced number of clutches, smaller clutch size, or both.

Toxicity effects that have been documented in other anuran species might provide a range of possible effects if data for the species of interest are not available. When considering the representativeness of surrogate species data, comparison of energetic functions (metabolic and excretion rates, food retention time) and effect likelihood (based on diet and body size) between the surrogate and target species offers some perspective. Differences in exposure history, habitat, or presence of multiple stressors could also affect the degree to which an organism responds to exposure. To summarize, life history and environmental traits should guide model structure and parameterization with organismal effects incorporated within the appropriate vital rate. In addition to considering the physiological similarities of the surrogate, biological and ecological relevance of the observed surrogate effects should also be considered.

Temporal and spatial extent. Heterogeneity in habitat availability or contaminant exposure will motivate broader spatial or temporal perspective. Longer temporal scales could be incorporated for multiyear projections. The interannual variability in environmental conditions or breeding status requires a longer temporal extent. One such adaptation to a deterministic model is inclusion of an occupancy or breeding state that can change from year to year for each site based on hydroregime and habitat conditions. Alternatively, environmental variability between years can be represented by stochastic assignment of reproductive success or other life-history parameters on a yearly basis. Finer resolution of temporal representation of chemical exposure could be considered (e.g., weekly or monthly estimations of exposure likelihood), particularly in scenarios where habitat use and chemical application rates differ throughout the year, such that exposure is variable (Swanson et al. 2018). Shorter time steps could also accommodate differences in dispersal based on habitat and life-cycle transitions associated with metamorphosis (Pittman et al. 2014). Likewise, finer spatial resolution could increase realism with more detail about heterogeneous

chemical overlap with species' distribution or habitat-specific movement (e.g., between breeding ponds) to simulate metapopulation dynamics. Spatially explicit models can accommodate varying exposure likelihoods by altering model parameters in accordance with animal migration or differences in contaminant distribution throughout the animal's range. Parameters can function as dose–response curves linking local chemical concentration to a vital rate or categorically defining distributions for exposed and unexposed individuals. Spatial dynamics and contaminant fate and transport can be particularly relevant to species with stage-specific habitat use and that often breed in ponds composed of accumulated water runoff. Movement between terrestrial and aquatic environments and stage duration is influential in exposure regime.

In summary, traits such as mode of reproduction, geographic range, dispersal, and population size might influence spatial context of the model related to habitat availability and movement patterns. Habitat loss or degradation in parts of the species distribution could also be included with additional spatial detail. Variable likelihood of exposure might lead to inclusion of spatially or temporally explicit dynamics. Environmental stochasticity and hydroperiod variability suggest longer temporal extent is required.

Density dependence. Similar to the preceding decision steps, density dependence must be considered in separate stage-specific environments because population regulation mechanisms can vary between terrestrial and aquatic habitats (Wilbur 1980; Berven 1990), with potential degrees of density dependence within a stage (Leips et al. 2000). Anuran populations have the greatest density during the larval stage, potentially regulating growth more strongly than at other stages via resource limitation, cannibalism, or interactions therein (Skelly and Kiesecker 2001; Altwegg 2003), although terrestrial stages can be affected by high densities of individuals (Harper and Semlitsch 2007). Intra-specific population regulation can also be a function of an age-structured population (Wissinger 2010) for which interactions between stages would need to be considered in model development. Lower survival at earlier life stages often has relatively less dramatic consequences on populations (Vonesh and De la Cruz 2002). Long-term data sets suggest that density-dependent processes are not always evident in many herptile populations (Leão et al. 2018). Density dependence is rarely estimated for specific species and scenarios, but commonly used functions are available for exploratory inclusion of such processes in population models (e.g., Getz 1996).

Mode of reproduction and larval duration provide some indication of whether tadpoles are likely to be spatially constrained during development; however, density dependence could be situation-specific, depending on the breeding habitat and the environmental conditions. Habitat loss or degradation could limit adult density, in addition to top-down population regulation through predation and parasitism. Long-term data sets of population sizes can be

used to further investigate density dependence (Wissinger 2010; Leão et al. 2018).

Population size and environmental conditions. Smaller populations are more vulnerable, such that endpoints focused on abundance, in addition to population growth rates, might be more relevant for risk assessment. Environmental conditions can be very stochastic, and variations in reproductive success and survival should be included or simulated in population evaluations. Such modifications will impact the temporal and spatial scale of population models. Effects of multiple stressors on survival and reproduction could be additive or synergistic, making empirical estimates difficult to obtain. Changing climate is anticipated to have compounding impacts on environmental conditions of some amphibian populations (Kissel et al. 2019). Traits related to other stressors, such as pathogens or habitat loss or degradation, can be related to population size and environmental conditions. Mode of reproduction and geographic range can also influence inclusion of context relevant to environmental conditions.

Indirect effects. Indirect effects of resource reduction are most severe on species with specialist diets, and they can include impacts on prey items of terrestrial life stages or resources of aquatic stages. Higher predation rates might result from indirect effects when vegetation used for camouflage or refugia is reduced via fungicide or herbicide exposure. Some pesticides might also alter amphibian microbiomes and reduce resistance to pathogens (McCoy and Peralta 2018). Disease-causing pathogens have been linked to widespread amphibian mortality; however, susceptibility can vary by species and by life stage, and pathogen virulence also varies (Daszak et al. 2003). Simulations of pathogen-induced mortality in amphibian population models should include estimates relevant to the species and habitat being modeled. Traits to consider are diet and habitat that could be affected by chemical use or pathogen exposure.

DISCUSSION

Amphibians are in peril due to their unique exposure routes, their complex life cycles, and multiple stressors that many encounter (Lips et al. 2003, 2006; Sodhi et al. 2008). Occupancy declines are widespread among species in the United States (Adams et al. 2013), although impact varies according to geographic region and inherent species sensitivity (Sodhi et al. 2008; Grant et al. 2016). Amphibians are among the most ecologically vulnerable taxonomic groups to a broad range of stressors based on physiology, habitat, and life-history traits (Green 2003). Traits to consider when developing a population model are those that have been linked to population vulnerability and those that influence exposure regime, inherent species sensitivity, life-history strategy, or extrinsic environmental factors (Table 1). We reference a systematic guide to population modeling (Schmolke et al. 2017) to determine where existing

knowledge about amphibian vulnerability could inform the decision process. The incorporation of amphibian traits and population model parameters as demonstrated here provides guidance for understanding how the inclusion of data or the complexity of the model may influence model outcomes, estimation of population risk, and associated uncertainty.

Stage-specific habitat use in amphibians requires consideration of exposure regime in aquatic environments, usually embryo and larval stages, as well as terrestrial environments, usually juvenile and adult stages. Pesticide application and timing, fate and transport, exposure routes, and other details relevant to exposure should be considered when including data from surrogate species or field studies in a population model. To evaluate the potential for chemical contaminants to affect amphibian populations, the current ERA practice relies on fish, bird, and mammalian surrogates, which introduces significant uncertainty in the evaluation of their population-level risks. Acute toxicity or no-effect endpoints for fish often are sufficiently protective of amphibians for many chemicals (Weltje et al. 2013); however, population-level effects in amphibians include potential exposure in different environments at different life stages and dependence on unimpeded physiological transformation between the 2. Developmental impacts, for example, are unlikely to result in the same population-level impact for fish and amphibians. Body size is often used in toxicity tests of aquatic organisms as a measure of chemical impacts on growth. However, this test endpoint alone does not adequately capture the effects on amphibian development in relation to subsequent survival and reproduction.

While risk assessments typically aim to protect populations, organism-level metrics are used as assessment endpoints due to both the practicality of testing and availability of data. Physiological effects are likely to differ between ectothermic amphibians and endothermic birds and mammals used as surrogates. Mechanistic traits indicative of species sensitivity can improve extrapolation based on taxonomic similarity (Rubach et al. 2012), and physiological similarity should be considered when selecting a representative surrogate. Various impacts of pesticide exposure have been documented in amphibians, including reduced survival, lower fecundity, developmental abnormalities, delayed growth, and increased disease susceptibility (Boone and James 2003; Relyea 2004). A meta-analysis performed by Baker et al. (2013) found amphibian survival was significantly reduced by phosphonoglycines (with polyethoxylated tallow amine [POEA] surfactant), carbamates, chloropyridinyls, inorganic chemicals, organophosphates, and triazines; growth was negatively impacted by organophosphates and phosphonoglycines (with POEA surfactant). Representative of many amphibian studies, those included in this analysis focused on aquatic larval stage effects, and not all chemical classes were included (Baker et al. 2013), showing significant information gaps in the understanding of pesticide exposure on amphibians, although inference using chemical traits could also be informative (Hua et al.

2014). Trait-based approaches are emerging in ERA to compensate for widespread data deficiencies by including basic inference of species susceptibility based on patterns and processes observed in existing toxicity data (Baird and Van den Brink 2007; Baird et al. 2008).

Life-history strategies have long offered insights on how similar organismal effects can impact species differently, depending on population dynamics (Heppell et al. 2000). The incorporation of functional traits and vital rates demonstrates selective gradients and tradeoffs that can help explain population trends and species occurrence and further anticipate likely outcomes of environmental stressors (Salguero-Gomez et al. 2018). In general, the patterns in vital rate tradeoffs are not unique to amphibians; however, most tend toward the “faster” end of the spectrum with high fecundity, moderate longevity, and relatively early maturation. Environmental stochasticity is important in the population dynamics of many amphibian species, such that spatial and temporal scales greatly affect the population-level interpretation of potential impacts and should incorporate a representative range of conditions.

Other landscape-level influences on amphibian population dynamics include habitat degradation, pathogen presence, and chemical exposure. Interactions between parasite occurrence, habitat degradation, and chemical exposure introduce further complications in assessment of vulnerability (Koprivinkar and Redfern 2012; Stutz et al. 2017) with the potential for agrochemicals to reduce immunity to parasite infection (Kiesecker 2002; Rohr et al. 2008) and decrease parasite survival (Hua et al. 2016). Multiple parasites do not always produce synergistic effects; compensatory effects of coinfection are also possible (Wuerthner et al. 2017). Positive identification of *B. dendrobatidis* in national field samples has been associated with dissolved organic content, total N, and P as well as cooler water temperatures (Battaglin et al. 2016). Although sensitivity to *B. dendrobatidis* varies among populations (Bradley et al. 2015), it has caused population declines and biodiversity loss globally (Lips et al. 2006). Pesticide concentrations in amphibian tissue were also correlated with land cover, with urban and agriculture areas containing higher concentrations and forested areas having lower concentrations (Battaglin et al. 2016). Identifying the likelihood of additional extrinsic stressors, based on spatial distribution of an amphibian population, is useful in accounting for possible threats during ERA.

Population models are efficient tools for translating organism-level effects to population responses (Caswell 2001), and a growing body of work demonstrates their utility (Pastorok et al. 2002; Barnthouse et al. 2008; Forbes et al. 2016). Although the multiple years of data required to develop complex amphibian models could be prohibitive for cryptic species with fluctuating habitat availability, identifying qualitative indicators of vulnerability and patterns of similarity among species is valuable to address data gaps and understand model uncertainty in the context of different ERA objectives. Specific demographic rates will vary by

species and location, but similarities in population ecology among closely related taxa, for example, genus (Egea-Serrano et al. 2012) or functional (Rubach et al. 2012; Rico and Van den Brink 2015) groups could compensate for data shortcomings. By identifying common population vulnerabilities based on life-history traits and the spatial distribution of extrinsic stressors, these steps identify important model parameters and appropriate structure in addition to qualitative information on potential population threats. This information could motivate additional simulation of effects or sensitivity analyses where data are unavailable but expected to impact populations greatly, thereby representing data gaps and model uncertainties in the context of population dynamics.

Trait-based approaches like the one outlined here are being used to evaluate the vulnerability of species to climate change, demonstrating broad applicability of the approach to environmental issues on a global scale (Foden et al. 2013; Pearson et al. 2014). While inference-based extrapolation of effects based on traits may oversimplify the interactions of multiple environmental stressors that are known to impact amphibian populations, these approaches offer a starting point for risk assessment in the face of daunting data deficits. For ERAs that are required to have a widespread, general application (e.g., national-level pesticide registration), trait-based ERAs are transferrable across locations. Studies that focus on the implementation of trait-based approaches for specific contaminants and taxa provide a proof of concept for linking species traits to intrinsic sensitivity within taxonomic groups (Rubach et al. 2012). We add to this developing field by contributing preliminary guidance for anuran amphibians, linking potential vulnerabilities to representation within a common modeling structure.

While principles of parsimony dictate that models should be no more complex than they need to be, determining that level of complexity is a challenge in best modeling practices (Levins 1966; Rueda-Cediel et al. 2015). Model complexity can be further limited by available data, which must be considered in evaluating the trade-offs of generality, realism, and precision of a model. The framework presented by Raimondo et al. (2018) proposed that ERAs are bound by these trade-offs, which can then be used to guide the development of the population model. However, that framework did not account for data availability, address ecologically motivated inclusion of additional complexity, or discuss how various types of data may influence the output of a model. Here, we identify different types of data for amphibian model parameters that will facilitate the development of models with increasing realism and precision. We also discuss amphibian traits to guide model development and inclusion of data that could be influential in assessment of population-level effects, considering both species life-history and data availability. Used in conjunction with critical traits, this guidance will assist risk assessors in developing models of appropriate complexity and understanding principle sources of uncertainty.

CONCLUSIONS

The need to include appropriate ecological context into risk assessment was identified decades ago (Cairns 1988; Chapman 2002), and the inclusion of population models in this process has been advocated (Forbes and Calow 2013; NRC 2013; Forbes et al. 2016). While consensus on a systematic approach is currently lacking, recent efforts to refine guidance for developing population models for risk assessment provide advancement toward a standard of practice (Schmolke et al. 2017; Raimondo et al. 2018). This work contributes to that guidance by providing a trait-based approach to population modeling. Traits identified from literature that may interact with chemical exposure and their effects to anurans are brought together in the context of systematic stepwise model development. Hence, support is rendered for selection of the appropriate level of complexity in each model aspect as defined by the decision steps. Data deficiencies often prevent development of complex population models without spatial and temporal details to capture all stages, exposure regimes, and stressor effects for a given amphibian species. Despite this uncertainty, existing knowledge on data and modeling assumptions, ecological vulnerabilities, and trait-based risk assessment can compensate for some data gaps and help determine where additional precision is needed or where precision could be sacrificed for generality. As standardization of population-level risk assessment procedures continues to develop, identifying commonalities that might be inferred from existing information can provide some guidance on assessment for species potentially in peril. Approaching risk assessment systematically and identifying common threats and vulnerabilities by taxonomic group could facilitate first steps of inferring ecologically relevant impacts for ERAs that are needed for hundreds of species and chemicals, advancing translational ecology practices (Enquist et al. 2017).

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