A Perspective on Modern Pesticides, Pelagic Fish Declines, and Unknown Ecological Resilience in Highly Managed Ecosystems

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Pesticides applied on land are commonly transported by runoff or spray drift to aquatic ecosystems, where they are potentially toxic to fishes and other nontarget organisms. Pesticides add to and interact with other stressors of ecosystem processes, including surface-water diversions, losses of spawning and rearing habitats, nonnative species, and harmful algal blooms. Assessing the cumulative effects of pesticides on species or ecological functions has been difficult for historical, legal, conceptual, and practical reasons. To explore these challenges, we examine current-use (modern) pesticides and their potential connections to the abundances of fishes in the San Francisco Estuary (California). Declines in delta smelt (Hypomesus transpacificus), *Chinook salmon* (Oncorhynchus tshawytscha), *and other species have triggered mandatory and expensive management actions in the urbanizing estuary and agriculturally productive Central Valley. Our inferences are transferable to other situations in which toxics may drive changes in ecological status and trends.*

Keywords: endangered species, toxic runoff, aquatic habitat, ecosystem, delta smelt

Pollution poses complex threats to the biological diversity of aquatic systems (Dudgeon et al. 2006). Pollution encompasses land-based sources of nutrients, sediments, pathogens, and trash. In addition, tens of thousands of pesticides, metals, petroleum hydrocarbons, pharmaceuticals, personal-care products, plasticizers, and emerging industrial agents (e.g., engineered nanoparticles) regularly enter lakes, rivers, estuaries, and nearshore marine environments. Each of these chemical contaminants can be toxic to individual organisms, with effects that can aggregate to the level of populations, species, communities, and ecosystems. Pollutants also interact with and exacerbate other chemical (Monosson 2005) and nonchemical (Crain et al. 2008) stressors. In North America, toxic chemicals probably were an important contributing factor in the decline of fishes in the twentieth century (Miller RR et al. 1989), and they continue to reduce the probability of the persistence of numerous taxa (Ricciardi and Rasmussen 1999).

Concern over water pollution has expanded in recent decades from focal end-of-pipe discharges (so-called *point sources*) to diffuse pathways of deposition and terrestrial runoff (so-called *nonpoint sources*). In the United States, for example, the enactment of the Clean Water Act in 1977 effectively reduced the quantity of toxic substances released from point sources. Subsequently, however, nonpoint sources, such as the atmospheric deposition of persistent organic pollutants and storm-water runoff, have emerged as the most important sources of waterway and coastline pollution (USCOP 2004).

Two ecosystem-level factors are compounding the effects of nonpoint-source pollution. The first is the increase in toxic runoff that accompanies human population growth and urbanization in coastal watersheds (Beach 2002). The second is climate change, which is projected to increase storm frequency and intensity (i.e., more runoff) and decrease surface-water quantity and quality (i.e., less dilution of pollutants) in some regions (e.g., Franczyk and Chang 2009). Therefore, future regional changes in land use and precipitation are likely to have a disproportionately larger effect on land-based runoff than on point-source discharges.

There has been substantial effort in recent years to develop ecosystem-based approaches to managing coastal and estuarine systems and their connected watersheds (McLeod et al. 2005, Levin PS et al. 2009). Land-based sources of pollution are linked closely to everyday human activities (e.g., building a house, using a pesticide, fertilizing a lawn or a crop, paving

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a road, driving a vehicle). Therefore, it will be virtually impossible to successfully manage for resilience (the ability of a species or other ecosystem element to withstand or recover from a disturbance; Levin SA and Lubchenco 2008) in coastal ecosystems without engaging the most basic aspects of human behavior.

Efforts to control the runoff of nutrients, sediments, and trash have recently focused on nitrogen fluxes that drive the growth of hypoxic or anoxic zones in the Gulf of Mexico and elsewhere, nutrients and sediments that affect coral reefs, and plastic debris accumulating along coastlines and in open-ocean convergence zones such as the North Pacific Gyre. Monitoring nutrients, sediments, and trash in aquatic systems remains challenging, particularly over large spatial and temporal extents. In many cases, however, the impacts on biota are visually evident, from dead zones to sediment-coated reefs and starving seabirds.

Toxic chemical contaminants, by contrast, are much more numerous, varied, difficult to detect, and expensive to monitor. Their movement through aquatic ecosystems is complex and often challenging to predict via fate-and-exposure modeling. Recent examples of spatially extensive, intensive efforts in the United States include the US Geological Survey's National Water Quality Assessment Program (*http://water.usgs.gov/nawqa*) and the US Environmental Protection Agency's Environmental Monitoring and Assessment Program (*www.epa.gov/emap2*). Furthermore, evaluating the ability of a species or other ecosystem element to tolerate or recover from environmental toxicity traditionally falls within the disciplines of ecotoxicology (for effects on species and communities) or medicine (for effects on human health). The integration of toxics into an ecosystem-based framework therefore requires a basic grasp of toxicology. Despite past calls to integrate conservation and toxicology (e.g., Hansen and Johnson 1999), this fusion has been slow to evolve.

In the present article, we examine the connections and gaps between conservation science and toxicology in the context of current-use (modern) pesticides and the decline of pelagic fishes in the San Francisco Estuary (California). The ecological status of this highly managed estuary, formed by the confluence of the Sacramento and San Joaquin Rivers in California's Central Valley, is declining (Strange 2008) in response to interacting anthropogenic factors, including point and nonpoint sources of pollution, water withdrawals for agricultural and domestic use, land-cover change, altered flow regimes, and colonization by nonnative species. The abundances of delta smelt (*Hypomesus transpacificus*), longfin smelt (*Spirinchus thaleichthys*), striped bass (*Morone saxatilis*, measured by surveys of young-of-the-year), and threadfin shad (*Dorosoma petenense*) have sharply decreased since the early 2000s (Thomson et al. 2010). The abundances of other estuary-dependent fishes, including Chinook salmon (*Oncorhynchus tshawytscha*), steelhead (*Oncorhynchus mykiss*), and green sturgeon (*Acipenser medirostris*), have also decreased. Several of these species are protected under both the US Endangered Species Act and the California Endangered Species Act. The Central Valley is one of the most productive agricultural regions in the United States, and this productivity depends in part on the use of more than 800 different pesticides, with hundreds of thousands of kilograms of chemicals (active ingredients) applied each year (2005–2008 data are from the California Department of Pesticide Regulation's Pesticide Use Database, available online at *www.cdpr.ca.gov/docs/pur/purmain.htm*). The uncertain linkages between pesticide use and decreased abundances of fishes in the San Francisco Estuary illustrate how nonpoint-source pollution can pose challenges for the conservation of aquatic systems.

Fewer fish, more conflict

The abundances of many aquatic species in the San Francisco Estuary have declined since extensive human activities in the region began in the mid-1800s (Brown and Moyle 2005). Conflicts over water management have recently intensified, in part because of relatively recent pelagic fish declines. Collapses of salmonid populations have led to the complete closure of some commercial and recreational fisheries in recent years (e.g., Chinook in waters off California and Oregon). The situation has spurred both litigation and an unprecedented level of regulatory review (NRC 2010). Apportioning causation among the drivers of these declines has proven scientifically difficult and socially and politically contentious. Estimates of the potential effectiveness of different interventions, including major new investments in infrastructure, water management, ecological restoration, control of nonnative invasive species, and pollution control, are highly uncertain. These collective investments are projected to cost billions of US dollars (Lund et al. 2007).

Pesticides are a possible contributing factor in the decline of delta smelt and other imperiled species (NRC 2010), and there is more information on pesticide use, transport, aquatic fate, and toxicity (from both laboratory and field studies) for the San Francisco Estuary than for almost any other large estuary in the world (San Francisco Estuary Institute, Regional Monitoring Program, *www.sfei.org/rmp*; see also Kuivila and Hladik 2008). Nevertheless, the ecosystem-level effects of this diverse group of chemicals have not been evaluated. The San Francisco Estuary illustrates the historical, legal, conceptual, and practical reasons for the limited integration of pesticide science and the management of complex systems over large spatial scales and decades. Examination of the system also reveals opportunities to more effectively align future toxicology research with the information needs of ecosystem-based management. Sustainable practices on land, water quality, and ecosystem-based approaches are among the core priorities set forth in the new US National Ocean Policy (*www.whitehouse.gov/administration/eop/oceans/policy*).

Changing pesticide-use patterns over time

In the mid-twentieth century, concerns over pesticide effects on the health of humans and the environment were largely focused on the organochlorine class of insecticides, including DDT (dichlorodiphenyltrichloroethane), aldrin, chlordane, dieldrin, endrin, heptachlor, and toxaphene. The number of chemicals under consideration was relatively low, and the organochlorine pesticides were overtly toxic to aquatic life, with mass fish kills commonly reported following their application (e.g., Cottam and Higgins 1946). The pesticides were and are measurable in sediments and tissues (and, to a lesser extent, water), making it possible to track their movements to the present day. Moreover, they are highly persistent and lipophilic and accumulate in species at upper trophic levels. Most uses of DDT and the other organochlorines were banned in the United States in the 1970s, in part because pesticide biomagnification was linked to eggshell thinning, clutch failure, and the decline of eagles, osprey, pelicans, and other piscivorous birds (e.g., Porter and Wiemeyer 1969).

The banning of organochlorines reduced the inputs of a few high-profile insecticides into aquatic ecosystems, including the San Francisco Estuary. Nevertheless, the number and diversity of pesticides (e.g., insecticides, herbicides, fungicides) in current use have greatly expanded during the past 50 years. These chemicals are by design less persistent than the legacy pesticides listed above, and many are more difficult to measure in aquatic systems such as the San Francisco Estuary (see Kuivila and Hladik 2008).

Fish kills, now rare, do not necessarily reflect the comprehensive effects of toxics on ecosystems. The leading edge of toxicological research is focused instead on sublethal health effects, including endocrine disruption, impaired immune function, abnormal development, altered behaviors, reduced growth, and reproductive impairment. Sublethal toxicity can influence both individual fitness and interspecific interactions (e.g., predator–prey dynamics, disease transmission). However, measuring the sublethal physiological impacts of pesticides—particularly in the field—requires a more sophisticated experimental design than measuring acute mortality in a controlled laboratory setting. This includes the development and implementation of metrics that are reliably associated with both pesticide exposure and declines in fish health that can be explicitly linked to individual fitness. Furthermore, because herbicides can be toxic to primary producers and insecticides to freshwater and estuarine invertebrates, including insects and crustaceans, pesticides can affect fishes indirectly through bottom-up food-web effects (Macneale et al. 2010) in addition to top-down effects (e.g., losses of osprey and other piscivorous birds). Advances in environmental chemistry, molecular and cellular toxicology, organismal physiology (including behavior), and population and community ecology have improved the understanding of the effects of pesticides on aquatic ecosystems (Relyea and Hoverman 2006, Clements and Rohr 2009).

The costs and benefits of pesticide use

Modern pesticides are intended to disrupt the physiology of specific taxonomic groups, including microbes, fungi, plants, insects, mollusks, fishes, birds, and rodents. It has been nearly impossible to manufacture a pesticide that is selective for the target species yet nontoxic to other species, particularly to closely related taxa. Therefore, there is an enduring tradeoff between the societal benefits of applying pesticides (e.g., increased agricultural production, reductions in vector-borne diseases) and minimizing unintended impacts on aquatic ecosystems and human health. In the United States, the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) requires that any adverse ecological outcomes be balanced against the costs or consequences of regulating pesticide use. In the past, it was usually easier to estimate the economic costs than to estimate the ecological costs, particularly when relevant data for nontarget species are lacking.

Many pesticides applied on land eventually enter aquatic systems. In the United States, a recent national study detected pesticides in more than 90% of all stream samples from agricultural, urban, and mixed-use areas (Gilliom 2007). Moreover, more than 50% of the samples contained six or more pesticides. In general, for aquatic species in the San Francisco Estuary and watershed, pesticide exposures are chemically complex, pulsatile, seasonal, and geographically widespread (see the review by Kuivila and Hladik 2008 for use rates, transport and loading patterns, chemical detections, and concentrations). For fishes and their habitats, knowledge of pesticide exposure is often incomplete because analytical methods have not been developed for some chemicals or because the available methods are unable to detect pesticides at low concentrations that may nevertheless adversely affect aquatic organisms (Kuivila and Hladik 2008). Moreover, pesticides in aquatic systems are one of many factors that may reduce the probability of persistence of fishes, and data on the effects of pesticides on other ecological elements and processes are limited.

Pesticides represent an unusually complex and dynamic category of stressors for fishes and their associated communities. First, hundreds of chemicals are in current use; the application patterns for each can change seasonally and annually, and new pesticides are brought to market every year. Second, pesticides almost always occur in mixtures, and their toxicity may be additive or interactive (e.g., the cumulative effects of some organophosphate insecticides on juvenile salmon are greater than the additive effects and can be considered synergistic; Laetz et al. 2009). Third, pesticide products usually contain additional chemicals, such as adjuvants, surfactants, wetting agents, and emulsifiers, that increase the effectiveness of the biologically active ingredient. The fate and toxicity of these so-called inert ingredients and their effects on aquatic species are poorly understood. Fourth, pesticides may interact with other classes of toxic chemicals, abiotic habitat variables (e.g., surface-water temperatures, ultraviolet light), and bacterial and viral pathogens (Clifford et al. 2005). Pesticides are, therefore, a distinctive example of why regulation of land-based sources of nonpoint-source pollution has proven very challenging for federal, state, and local governments (USCOP 2004).

Toxicity testing as a source of data for ecological forecasting

In the United States in the 1970s, regulations for registering or reregistering pesticides under FIFRA or for developing pesticide-specific criteria for the protection of aquatic life under the Clean Water Act were structured around a limited set of biological responses to a chemical exposure (endpoints). To classify pesticides from *practically nontoxic* to *highly toxic* under the act, it was necessary to measure the same standard endpoints—growth, reproduction, and death—in a small number of representative species. Accordingly, the *median lethal concentration* (LC_{50}; the concentration at which a pesticide kills half of a test population) has been a cornerstone of pesticide toxicology for decades.

Because the vast majority of pesticide exposures are sublethal and intermittent, the ecological relevance of the LC_{50} and similar metrics is limited. Minimum reporting for acute tests is also required for only a few species: rainbow trout (*O. mykiss*), bluegill (*Lepomis macrochirus*), bobwhite quail (*Colinus virginianus*), mallard duck (*Anas platyrhynchos*), and daphnids (*Daphnia* spp.). Therefore, the first-tier estimates of pesticide effects on aquatic ecosystems under FIFRA are based on toxicity data for one cold-water fish, one warm-water fish, and a cladoceran (crustacean), all in freshwater. The US Environmental Protection Agency may require additional toxicity tests or tests on other species, but the basic federal process for generating pesticide-toxicity data has remained largely unchanged since 1972. Similarly, under the Clean Water Act, ambient water-quality criteria have the goal of protecting 95% of aquatic species. The toxicity data for different species are therefore aggregated in the process of developing the criteria (e.g., Dyer et al. 2008). However, these species-sensitivity distributions are typically lists of LC_{50} values for a variety of species that may or may not interact in real systems. The species with the greatest sensitivity to pesticides—commonly daphnids—have proven useful for monitoring ambient pesticide toxicity in *in situ* or field-collected water samples (e.g., Werner et al. 2000). However, daphnids are not among the species of regulatory concern in the San Francisco Estuary.

Much of the available data on pesticide toxicity still come from FIFRA-mandated standardized tests on a few species presumed to serve as surrogates for many other species. However, targeted mechanistic research (e.g., genomics, proteomics, metabolomics) increasingly yields detailed information about sublethal toxicological processes, such as endocrine disruption (Van Aggelen et al. 2010). Community-level research (e.g., Relyea and Diecks 2008) is yielding new insights into the effects of pesticides on interspecific interactions in aquatic systems. Nevertheless, the ecological risk-evaluation process for registering new pesticides and reregistering older pesticides has been slow to incorporate these sources of scientific information.

Population-level effects of pesticides

The goal of linking the effects of pesticides on individuals to their effects on populations underpinned the early toxicological emphasis on acute lethality, reduced growth, and reduced reproduction. Most standardized toxicity testing generates data on individuals. At the same time, FIFRA and the Clean Water Act are intended to protect nontarget populations, and not necessarily individuals within those populations. Whereas the LC_{50} is defined as the calculated median lethal concentration for a population, the population in question is almost always one of test organisms in a laboratory (often cultured for many generations) rather than a wild population.

Computational methods for translating toxicity data derived from individual organisms to the level of wild populations are increasingly being developed and refined. Nevertheless, in ecotoxicology, population modeling is still considered a relatively new subdiscipline (Forbes et al. 2008). Recent studies have estimated sublethal effects on fish populations (e.g., Miller et al. 2007, Baldwin et al. 2009). However, there has generally been more progress at lower trophic levels. This includes daphnids, chironomids, copepods, and other invertebrates with life histories and abundances that make their populations more amenable to study, particularly across multiple generations (Stark et al. 2004).

The standard LC_{50} measure may predict whether a pesticide exposure will kill fish over the course of 96 hours, but it does not encompass toxicity that is sublethal or delayed in time, nor species-specific variations in life history that affect both population dynamics and responses to stressors (e.g., Stark et al. 2004). Yet the regulatory process (e.g., pesticide registration review), although ostensibly focused on populations, has been slow to incorporate sources of scientific information other than the results of standardized toxicity tests when estimating the effects on populations of protected fishes.

It is unlikely that any single pesticide is driving the decline of any one species of fish in the San Francisco Estuary or elsewhere. It would probably be impossible to quantify the links between individual pesticides and individual fishes across an entire ecosystem. However, this does not imply that pesticides have no population-level effects on fishes. Rather, useful information in the form of relevant toxicological data is widely lacking.

Unknown resilience

Historically, toxic control efforts under the Clean Water Act were intended to achieve *no toxics in toxic amounts.* Resource agencies, private landowners, municipalities, industry, and environmental groups, as well as their scientists, lawyers, and consultants, have long debated the meaning of a *toxic amount.* Toxicologists often quote Paracelsus' maxim that the dose makes the poison, which implies that toxicity correlates positively with the degree of chemical exposure. However, the US Congress has passed overlapping laws that convey different levels of protection for aquatic

	Lead federal agency	Summary of aims	Regulatory focus	Data required for practical application
Federal Insecticide, Fungicide, and Rodenticide Act	Environmental Protection Agency	Prevent highly negative environmental effects, taking into account the economic, social, and environmental costs and benefits of pesticide use	Chemicals, pesticide registration and reregistration	Standardized toxicity tests (e.g., LC_{50}s) for a small number of aquatic species, including a freshwater fish (e.g., rainbow trout or bluegill) and an invertebrate (e.g., a daphnid)
Clean Water Act	Environmental Protection Agency	Restore and maintain the chemical, physical, and biological integrity of the nation's waters	Chemicals, including the development and implementation of aquatic life criteria for individual pesticides	Standardized toxicity data for species in at least eight families are used to define a species sensitivity distribution; 95% of the species in that distribution must be protected
Endangered Species Act	National Oceanic and Atmospheric Administration, Fish and Wildlife Service	Prevent extinctions of species and promote recovery of species that are threatened or endangered	Species; avoid, minimize, and mitigate adverse affects to listed species and their habitats	The best available science on probability that an action will cause adverse effects to listed species or their habitats

Figure 1. Key characteristics of and differences among federal laws with relevance to pesticides and their effects on species and ecosystems.

species (figure 1). Thus, a pesticide exposure that might be considered nontoxic under FIFRA may nevertheless be actionable under more-protective statutes, such as the Endangered Species Act. Statutory differences in the definition of a meaningful toxicological effect are forms of linguistic uncertainty (Regan et al. 2002).

No toxics in toxic amounts implies that aquatic communities can withstand some amount of pollutant loading and that regulators can effectively manage for ecological resilience in aquatic systems affected by toxics (Levin SA and Lubchenco 2008). This is, in part, the reasoning behind regulatory allowances of higher levels of pollution in "mixing zones" for end-of-pipe discharges under the Clean Water Act. However, laws such as FIFRA and the Clean Water Act were not necessarily intended to meet the information needs of the resource managers now charged with maintaining ecological resilience.

External to the regulatory process, there has been a trend toward greater ecological realism in ecotoxicology for many years. This includes the incorporation of ecological theory into the design of experimental studies (for reviews, see Relyea and Hoverman 2006, Clements and Rohr 2009). However, for aquatic communities, most of the key advances have been in relatively small freshwater systems rather than in estuarine or coastal systems. It has also generally been most practicable to study aquatic vertebrates that have relatively small dispersal distances and that do not move across free-flowing or tidally influenced waters. As a consequence, more is generally known about the direct and indirect effects of pesticides on frogs (e.g., Relyea and Dieckս 2008) than about those on estuarine fishes or food webs.

Ecosystem-based management versus ecological risk assessment

There has been an increasing emphasis on the ecosystem-based management (EBM) of complex systems, from large marine ecosystems to estuaries, river basins, and watersheds (USCOP 2004), and federal agencies are beginning to design and implement integrated ecosystem assessments for large water bodies (e.g., Puget Sound; Tallis et al. 2010). Nevertheless, the regulations governing pesticide use are still focused on ecological risk assessments (ERA) for individual chemicals. The initial step of an ERA is the construction of a conceptual model of the composition and expected use of a formulated pesticide product, its fate in the aquatic environment, and the potential for toxicity to nontarget organisms. Often, a screening-level hazard quotient is calculated as a deterministic ratio of predicted exposure to a toxicant (usually some proportion of an LC_{50} for a standard set of test species). If a pesticide is deemed hazardous, more in-depth, probabilistic exposure-response evaluations may be conducted if the necessary data are available.

The initial approaches to EBM (e.g., integrated ecosystem assessment; Levin PS et al. 2009) and the basic steps in ERA (USEPA 1998) are similar. Both involve the development of conceptual models, the identification of biological indicators of natural and anthropogenic stress, and risk analysis. The major difference between EBM and ERA is how science informs adaptive decisionmaking. In the context of pesticides, ERA under FIFRA is highly structured, limited in ecological scope, and designed to inform a decision process that culminates in a chemical registration (or reregistration) determination. Data from standard short-term toxicity tests can be useful for predicting how pesticides might alter species densities and thereby alter community structure (Relyea and Hoverman 2006). However, the specific types of toxicity information produced in support of pesticide ERA, with conventional testing procedures, are unlikely to provide the data needed to parameterize models and otherwise inform future strategies for EBM in aquatic systems.

The current state of the science leaves two important questions unanswered. The first is the relative contribution of pesticides to the decline of populations in specific systems, such as pelagic fishes in the San Francisco Estuary and watershed. The second is the reductions in pesticide inputs to the ecosystem necessary to increase the probability of species persistence or recovery.

In the future, it may be desirable to redirect some scientific resources from conventional testing and research on pesticide toxicity (i.e., iteratively defining the quantities at which individual chemicals are toxic) to research on the effectiveness of best-management practices (e.g., integrated pest management) and other pollution-reduction strategies at different temporal and spatial extents. Feasible source control and mitigation measures must be not only ecologically effective but also practical and affordable for local communities and landowners.

Regulation of pesticides and actions affecting threatened fishes and the resilience of aquatic ecosystems remains in flux. In recent years, federal agencies and others have disagreed over how to apply the best available science in order to estimate ecological effects and possible threats to species recovery. This has centered, in part, on national ESA consultations involving current-use pesticides and listed salmonids, including population segments in the San Francisco Estuary (e.g., NMFS 2008). To help resolve these differences and to provide scientific guidance, the National Academy of Sciences formed a committee that is critically examining these issues and developing guidance for identifying and implementing the best available science specific to the responses of species to indirect and sublethal pesticide effects, mixture toxicity, and the use of individual- and population-levels models in pesticide and endangered-species risk assessments. The committee's recommendations are anticipated in the summer of 2013.

Transferability

Uncertain linkages between pesticides and accelerated population declines are not restricted to the San Francisco Estuary and watershed. For example, pesticides are suspected as drivers of the recent colony-collapse disorder among honeybees (*Apis mellifera*) and other pollinators in North America and Europe. An extensive exposure survey recently showed complex combinations of pesticides and pesticide metabolites in bees and hives (wax and pollen), with an average of 6 and a maximum of 39 residue detections per sample (Mullin et al. 2010). Despite this relatively refined exposure information, pollination researchers confront many of the same uncertainties as the aquatic research community. For example, pesticides are one of many categories of chemical and nonchemical stressors whose interactions are poorly understood, the toxicity of pesticide mixtures is poorly understood, the effects of pesticides may be sublethal (e.g., memory loss and other forms of disorientation among foraging bees), and a single-chemical risk-assessment method does not support reliable predictions (Mullin et al. 2010). Pesticides have also been implicated in the worldwide decline of amphibians (e.g., Davidson et al. 2001), and research in recent years has been focused on pesticide mixtures, sublethal effects, and the impacts of interacting chemical and nonchemical stressors. To meet these common scientific and management challenges, we encourage an increased exchange of ideas and research tools among the disciplines of toxicology, ecology, and conservation science (Hansen and Johnson 1999, Macneale et al. 2010).

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