Extinction Risk

Related terms:

Dispersal, Inbreeding, Population Size, Species-Area Relationship, Biodiversity, Environmental Niche Modeling

Extinction Risk from Climate Change Solutions

Lee Hannah, in Climate Change Biology, 2011

Estimating Extinction Risk

Extinction risk from climate change solutions may be calculated using the same species–area relationships used to calculate extinction risk from climate change. Smaller areas hold fewer species, in a nonlinear relationship. Because the shape of the curve is known (see Chapter 12), declining area may be equated to lesser numbers of species. The difference between the number of species and future number of species equates to species' extinctions.

Extinction risk may be calculated using this method for biomes on a global scale. It is important to subdivide the calculation by biome because some biomes harbor more species than others, and the distribution of climate change impacts or land use for renewable energy production will vary across biomes as well.

Using this method, the extinction risk of 1000 possible alternative energy mixes, each in 1000 different spatial variants, has been calculated (Figure 17.5). This assessment produces a "landscape" of extinction risks associated with different energy pathways. Pathways with large proportions of biofuels in the mix and large tropical spatial components are associated with exceptionally high extinction risks. On average, the global extinction risk associated with biofuels is two orders of magnitude (200 times) greater than any other energy option. Options with high proportions of solar, nuclear, and carbon sequestration have generally low associated extinction risks. Some high extinction risk variants are found with most pathways, indicating that only a limited number of climate-friendly, biodiversity-friendly energy pathways exist. These must be pursued selectively and early to avoid major impacts on biodiversity.
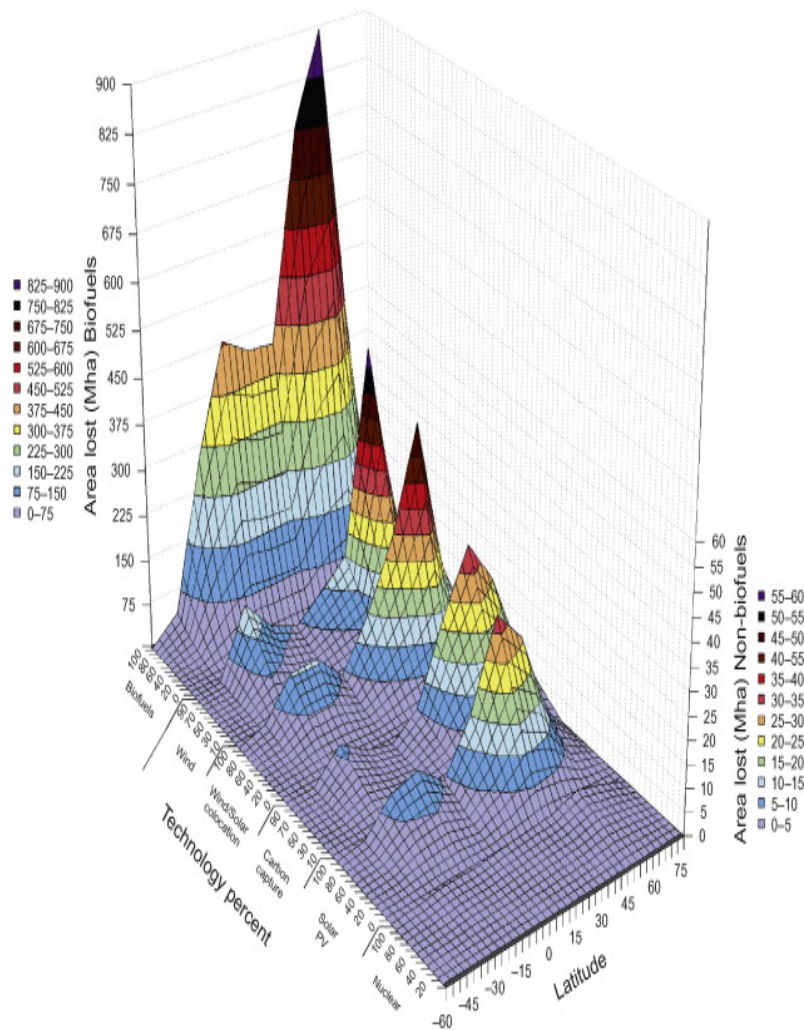
Figure 17.5. Energy Pathways and Extinction Risk.

Global area use of energy technologies by latitude. Average land use by latitude associated with global energy development to 2100. Land use is shown rather than biodiversity loss because at 200 times greater impact on biodiversity for biofuels, no other technology shows on a graph of the same scale. The high area losses for biofuels shown here combine with high concentration of biofuel potential in the species-rich tropics to make species extinction risk from biofuels the highest of all sources by orders of magnitude.

Source: N. Snider and L. Hannah.

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Mitigation

Lee Hannah, in Climate Change Biology (Second Edition), 2015

Estimating Extinction Risk

Extinction risk from climate change solutions may be calculated using the same species–area relationships used to calculate extinction risk from climate change. Smaller areas hold fewer species, in a nonlinear relationship. Because the shape of

the curve is known (see Chapter 12), declining area may be equated to losses of species. The difference between the current number of species and the future number of species equates to species' extinctions.

The area needed by different energy mixes to meet global demand is a first step towards calculating extinction risk (Figure 18.5). Generally, higher land use demand and more tropical land use will equate to higher extinction risk. The area demand of 1000 possible alternative energy mixes to meet global demand, each in 1000 different spatial variants and assigned at random according to energy potential, has been calculated. This assessment produces a landscape of extinction risks associated with different energy pathways and spatial configurations.
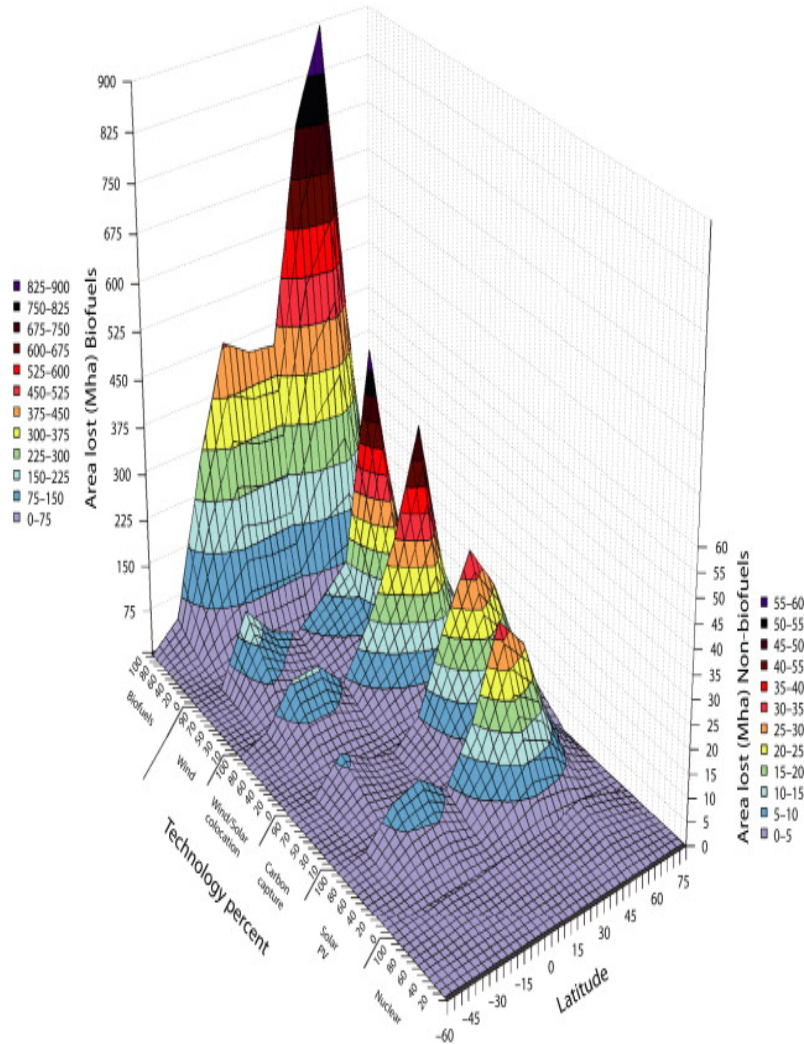

Figure 18.5. Energy pathways and extinction risk.

Global area use of energy technologies by latitude. Average land use by latitude associated with global energy development to 2100. Land use is shown rather than biodiversity loss, because at 200 times greater impact on biodiversity for biofuels, no other technology shows on a graph of the same scale. The high-area losses for biofuels shown here combine with a high concentration of biofuel potential in the species-rich tropics to make species extinction risk from biofuels the highest of all sources by orders of magnitude.

Source: N. Snider and L. Hannah.

Pathways with large proportions of biofuels in the mix use far more area than other energy mixes and have greater impacts in the tropics, where biodiversity is high. On average, the global area used for biofuels is 200 times greater than any other

energy option. Options with high proportions of solar, nuclear, and <u>carbon sequestration</u> have generally low-associated extinction risks. Some high-extinction risk variants are found with most pathways, indicating that only a limited number of climate-friendly, biodiversity-friendly energy pathways exist. These must be pursued selectively and early to avoid major impacts on biodiversity.

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The Role of Body Size in Multispecies Systems

Ute Jacob, ... Jennifer A. Dunne, in <u>Advances in Ecological Research</u>, 2011

C Extinction Scenarios the Robustness of the Weddell Sea Food Web

The influence of extinction risk on trophic vulnerability and species life histories are both complex and specific to the source of the threat (Purvis *et al.*, 2000). Further, such relationships depend on the species sensitivity to a stress that intensifies through time, such as a gradual change in <u>salinity</u> or temperature (Ives and Cardinale, 2004). While the relationship between functional traits and the susceptibility of extinction risk is complex, recent research does suggest some clear and consistent patterns, namely, that smaller species are apparently less vulnerable to extinction than larger species (Cardillo, 2003; McKinney, 1997). In the light of this, it is perhaps heartening that in our topological simulations it was the sequences of deletions based on the removal of small, and not large, organisms that caused a cascade of secondary extinctions. It appears that within the Weddell Sea network, larger bodied species can be lost without causing a direct collapse of the network topology, which is the opposite case in other marine systems, where the loss of large apex predators has lead to multiple <u>trophic cascades</u> (i.e. Myers *et al.*, 2007; Raffaelli, 2007).

We should of course interpret these findings with a great deal of care. A major caveat, which needs to be borne in mind with such topological analysis, is that population dynamics are ignored and therefore no top-down extinctions, or other indirect effects, can occur (Montoya *et al.*, 2009). Further, the strength of the bottom-up extinctions will be reduced in the absence of population dynamics, as species need to lose all their prey items before going extinct and not just part of their diet, this need not be the case in real ecosystems (Ebenman and Jonsson, 2005). There are a range of studies showing the strong effect of top-down control (e.g. Berger *et al.*, 2001; Borrvall and Ebenman, 2006; Estes and Palmisano, 1974; Reisewitz *et al.*, 2006; Terborgh *et al.*, 2006) and the importance of considering interaction strength when analyzing the response of perturbations in ecological communities (e.g. McCann *et al.*, 1998; Montoya *et al.*, 2009; Novak *et al.*, 2011) and hence the analysis of topological robustness should be considered only as a best-case scenario.

Despite these considerations, there are several studies that have used mass-balanced models of population dynamics to examine the role of large whales and the consequences of their loss within the Antarctic food web. These studies indicate weak top-down control in these systems (Bredesen, 2003; Trites *et al.*, 2004) and suggest that the removal of large whales might have little measurable effect on lower <u>trophic levels</u> or on the population dynamics of other species in the food web. It is also worth noting that, in these models, once the populations of large whales were reduced to small numbers, they take a long time to recover (Bredesen, 2003; Trites *et al.*, 2004).

Secondary extinction cascades can occur from a body mass-based sequence of primary extinctions suggesting the importance of trophic size–structure for this system. It is also intriguing that this is despite the lack of any clear relationships between body mass and vulnerability, generality, or trophic level when examined across all species in the network. That is to say, the robustness analysis still makes clear the importance of the smaller species near the base of the food web that provide and channel energy for the many larger species of this system.

Our results also reinforce the findings of other studies regarding the importance of the highly connected species for robustness (Dunne *et al.*, 2002; Eklöf and Ebenman, 2006; Petchey *et al.*, 2008), with both in- and out-links being important. The suddenness of the collapse when removing the most vulnerable species (only approximately 25 primary removals within this sequence were required for the system to collapse to half of its size) reiterates the importance of detritus and planktonic copepods (the two most vulnerable species) for the Antarctic food web. It is also interesting that there is no positive relationship between generality and body mass, unlike as has often been observed in other systems (e.g. Woodward *et al.*, 2005). As has been found in some other studies (e.g. Digel *et al.*, 2011), the Weddell Sea web is perhaps rendered more susceptible to the loss of generalist predators, at least when undergoing a topological-based extinction simulation, and then would be expected in webs with a stronger generality allometry, where the loss of larger more specialized species causes more trophic cascades (Myers *et al.*, 2007).

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The Samoan Archipelago

Douglas Fenner, in World Seas: an Environmental Evaluation (Second Edition), 2019

28.5.1 Species Extinction Risks

A study of the extinction risk of reef corals worldwide (Carpenter et al., 2008) reported that 75 coral species in the US Pacific waters were threatened to some degree. About 54 of the 75 species occur in American Samoa (Brainard et al., 2011). Kenyon, Maragos, and Fenner (2010) compiled records of which species were found in federally protected waters in American Samoa. Subsequently, NOAA listed 15 species as Threatened under the US Endangered Species Act, of which six have been found in American Samoa. Zgliczynski et al. (2013) reported 21 species of reef fish on Tutuila which have been listed as endangered or vulnerable on the IUCN Red List. American Samoa had the lowest abundance of large-bodied species listed as endangered or vulnerable of any US Pacific region, with 2 individuals km^{-2}. Among the small-bodied species listed, 5 of the 22 grouper species recorded in American Samoa were significantly more abundant on the uninhabited islands than the inhabited islands. The coral hind and blacktip grouper were rare or not recorded on the inhabited islands.

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Estimating Extinction Risk from Climate Change

Lee Hannah, in Climate Change Biology (Second Edition), 2015

Checking the Estimates

The species-based estimates of extinction risk have been confirmed by two independent methods. The second approach used dynamic global vegetation models (DGVMs) and EAR to estimate extinction risk for biodiversity hot spots. The global biodiversity hot spots have been defined based on areas of high species endemism. Because endemism rates are known for hot spots, it can be assumed that loss of habitat type for a hot-spot endemic is a global loss of habitat for that species. Based on this assumption, projection of future extent of vegetation types for a hot spot from a DGVM can be used to estimate extinction risk for hot-spot endemics. If a habitat type decreases within a hot spot, it is as if a small habitat island for the endemics occupying that habitat has just gotten smaller.

The DGVM hot-spot approach applied EAR to areas of lost habitat within hot spots to estimate extinction risk. This approach therefore had the double advantage of taking an independent approach and applying the correct SAR for endemics (the EAR). The results obtained using this approach are presented in Table 12.2. They are lower than the species-based estimates, as is expected using the slower-accumulating EAR. However, they are not negligible: they give strong support to the notion of significant extinction risk associated with climate change.

Table 12.2. Projected Species-Area-Based Percentage of Extinctions of Endemic Species in 12 Hot Spots Judged to be Especially Vulnerable to Global Warming[a]

Hot Spot[b]	Global Vegetation Model	Perfect Migration				Zero
		Broad Biome Definition		Narrow Biome Definition		Broad Defini
		Broad Specificity	Narrow Specificity	Broad Specificity	Narrow Specificity	Broad Specit
California Floristic Province	BIOME3	2.4	30.9	2.5	27.8	4.5
(2125; 71)	MAPSS	3.0	4.0	7.8	40.9	5.3
Cape Floristic Region	BIOME3	2.4	2.3	4.5	5.8	2.7
(5682; 53)	MAPSS	11.8	28.6	17.4	52.4	15.4
Caribbean	BIOME3	3.1	2.7	3.1	2.8	4.0
(7000; 779)	MAPSS	7.2	12.1	10.0	15.5	9.0
Indo-Burma	BIOME3	1.9	17.8	5.1	18.	2.7
(7000; 528)	MAPSS	5.5	23.8	6.7	29.6	6.2
Mediterranean Basin	BIOME3	1.9	10.6	2.9	9.7	3.9
(13,000; 235)	MAPSS	3.7	4.4	5.6	26.6	8.1

Hot Spot[b]	Global Vegetation Model	Perfect Migration				Zero Migration	
		Broad Biome Definition		Narrow Biome Definition		Broad Definition	
		Broad Specificity	Narrow Specificity	Broad Specificity	Narrow Specificity	Broad Specificity	
New Caledonia	BIOME3	0.0	0.0	0.0	0.0	0.0	
(2551; 84)	MAPSS	0.0	0.0	18.8	75.0	0.0	
New Zealand	BIOME3	2.5	5.3	2.5	5.3	2.8	
(1865; 136)	MAPS	4.6	24.8	4.6	29.1	6.1	
Polynesia & Micronesia	BIOME3	2.2	16.6	3.0	17.7	2.2	
(3334; 223)	MAPSS	3.8	42.9	8.1	55.3	5.1	
Mountains of South Central China	BIOME3	4.3	3.1	4.3	12.1	8.0	
(3500; 178)	MAPSS	3.5	27.3	8.8	21.6	9.5	
Succulent Karoo	BIOME3	2.4	19.1	3.0	22.5	3.2	
(1940; 45)	MAPSS	7.0	30.1	10.1	46.7	8.8	
Southwest Australia	BIOME3	2.3	9.8	5.3	18.4	2.8	
(4331; 100)	MAPSS	15.2	32.2	17.2	38.7	18.1	
Tropical Andes	BIOME3	2.7	10.6	4.0	13.9	6.4	
(20,000; 1567)	MAPSS	3.5	13.0	3.7	13.9	9.8	

a

Percentages are shown for two migration scenarios, two biome breadth definitions, two levels of biome specificity, and two global vegetation models (BIOME3 and MAPSS).

b

Numbers of endemic plant and vertebrate species, respectively, are shown in parentheses below hot-spot names.

The second test came in a study of extinction risk in lizards. Lizard populations in Mexico have become extinct in areas that have warmed significantly. A model of these population extinctions then predicted population extinctions accurately on other continents. The population extinction projections were used to estimate species-level extinction risk. This ecophysiological method projected lizard extinctions of 6% in 2050 scenarios and 20% in 2080 scenarios. The hot-spots analysis and the lizard analysis suggest that the low end of the range of original estimates may be appropriate for 2050, whereas the lizard analysis suggests that the high end of the original estimates may be more similar to end-century risks.

The Future Species of Anthropocene Seas

Nicholas K. Dulvy, Holly K. Kindsvater, in Conservation for the Anthropocene Ocean, 2017

Observation Error and Lazarus Species

A false positive (type II error) in extinction risk can also occur if insufficient effort has been expended to find the species presumed extinct (Diamond, 1987). In the oceans, the broad scale and depth range of species' ranges, which may encompass several political jurisdictions, make this a persistent concern. Hence, the classification of extinctions requires a balance of two risks: (1) that a species is extinct and has gone undetected and unreported, and (2) that a Lazarus species is categorized as extinct at some scale when it is still present and, embarrassingly, is sighted at a later date (Keith and Burgman, 2004). For example, the Barndoor Skate (*Dipturus laevis*) was declared near extinct based on its absence in Atlantic shelf trawl surveys (Casey and Myers, 1998), yet significant numbers were subsequently discovered on the continental slope, preventing its listing under the US Endangered Species Act (Kulka et al., 2002).

In the tropics, observation and monitoring is a persistent challenge, even in nearshore waters. One paper suggested sawfishes may be extinct in Mexico (del Monte-Luna et al., 2009). This seemed plausible, as the last Largetooth Sawfish was landed in 1997 at Mujeres Island, Quintana Roo, Mexico. However, in 2016 the scientific world was stunned and relieved when a Largetooth Sawfish was reported in Veracruz, rewarding recent efforts to raise awareness, and providing hope that all is not lost for this species in the Gulf of Mexico (R. Bonfil, Personal Communication).

In South Africa, by contrast, extensive long-term sampling bolsters our confidence that sawfishes are regionally extinct. South Africa has long time series of elasmobranch abundance from netting programs designed to protect bathers from sharks. These data show that sawfishes were formerly common in KwaZulu-Natal, but that numbers declined (likely due to incidental mortality in trawl fisheries and degradation of juvenile habitat) (Everett et al., 2015). The last reported observation of any sawfish species in South Africa was in 1999 (Fig. 3.1). Although this fish was released alive, no sawfish has been recorded since, despite the presence of survey gear through 2012. Both sawfish case studies illustrate that our confidence in species' disappearance depends on "observation error," a rather prosaic term that encompasses awareness, search effort, and continued monitoring using appropriate methods and gears.
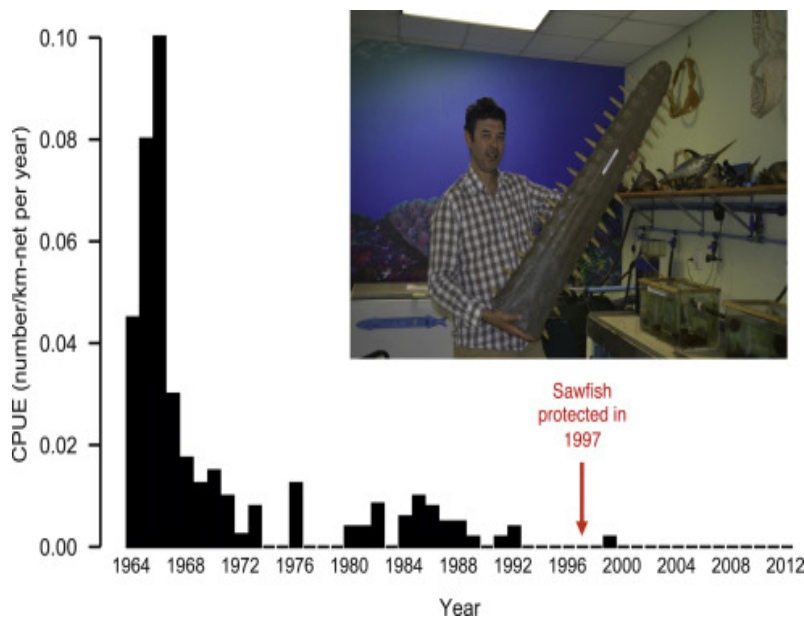
Figure 3.1. Sawfish catch-per-unit-effort from South African bather nets from 1964 to 2012. *(Redrawn from Everett, B.I., Cliff, G., Dudley, S.F.J., Wintner, S.P., van der Elst, R.P., 2015. Do sawfish Pristis spp. represent South Africa's first local extirpation of marine elasmobranchs in the modern era? African Journal of Marine Science 37, 275–284.)* Inset is the rostrum, probably of the Largetooth Sawfish *Pristis pristis*, of the last known captive South African sawfish that was housed in uShaka marine World, Durban. *(Photo credit Colin A. Simpfendorfer.)*

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Sexual Selection and Sexual Conflict

Ulrika Candolin, in Encyclopedia of Ecology (Second Edition), 2019

Sexual Selection and Speciation

While the influence of sexual selection on population viability and extinction risk is poorly known, its impact on hybridization and speciation is better understood. Sexual selection can be an important player in species divergence and convergence, given its power to drive changes in mate recognition traits. For example, the evolution of reproductive isolation between two frog species—*Pseudacris* feriarum and *P. nigrita*—is driven by sexual selection; by reinforcing differences in their acoustic signals and preferences—through reproductive character displacement—sexual selection acts against hybrids and promotes divergence (Lemmon and Lemmon, 2010).

Sexual selection may frequently interact with natural selection in driving species divergence or convergence. An example is sympatric species pairs of stickleback, which have diverged in parallel in both sexual and ecological trait in many lakes. Species that inhabit the benthic zone of lakes have evolved to base their mate choice mainly on body size, while species that inhabit the limnetic zone mainly use male nuptial coloration as a mate choice cue (Boughman *et al.*, 2005).

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Protecting cave life

Matthew L. Niemiller, Steven J. Taylor, in Encyclopedia of Caves (Third Edition), 2019

Threats to subterranean fauna

Several threats have been identified that either have or are suspected to impact populations of troglobionts and stygobionts leading to increased extinction risk. Threats vary in scope, severity, and impact to individual species among karst regions globally. For example, limestone mining is a much greater threat to subterranean fauna in the South China Karst currently than in the Edwards Plateau of Texas where groundwater harvesting and contamination are greater concerns. Threats also vary temporally: impacts can be chronic, occurring over years to decades, or acute, on the order of just hours or days. For example, hundreds to thousands of dead and dying cavefish and cave crayfish washed out of Meramec Spring in Missouri in November 1981 within a week of a fertilizer pipeline rupture upstream resulted in catastrophic deoxygenation of groundwater (Vandike, 1982). In contrast, several decades worth of chronic pollution of creamery waste and heavy metal contamination resulted in the decline and extirpation of aquatic life at Hidden River Cave in central Kentucky (Lewis, 1996). Encouragingly, cavefish and other stygobionts have recolonized stream reaches previously impacted in Hidden River Cave presumably from areas upstream or tributaries to the main stream passage.

Groundwater pollution is the primary threat for many stygobionts (Fig. 1), but the source and risk of groundwater pollution varies depending on land use within recharge basins of individual cave systems. For example, contamination associated with agriculture, such as pesticides, herbicides, and livestock waste, dominate where the prevailing land cover is cropland and pastureland, whereas chemicals associated with lawn care (pesticides, herbicides, fungicides), septic and sewer leachate, runoff from impervious surfaces (roads, parking lots, sidewalks, etc.), and industrial chemicals are the primary contaminants in residential and urban areas. Stygobionts living in karst terrains are particularly vulnerable to groundwater pollution, as karst aquifers often have low potential for self-purification and have a high probability of retention of contaminants. Termiticide treatment around Jordan Hall on the campus of Indiana University in the 1960s resulted in the extirpation of all groundwater life at the type locality of the groundwater amphipod *Bactrurus cellulanus* (Lewis, 2015), and this species is now thought to be extinct. Groundwater pollution also has the potential to impact terrestrial species, particularly if contaminants are deposited on mud banks and other riparian areas during flood events.

Fig. 1. A white formation in Meads Quarry Cave, Knox County, Tennessee, about 336 m upstream from the main entrance marks the primary location of lime input into the cave system and coincides with a large deposits of waste lime on the surface produced when a quarry was still in operation several decades ago. This has result in an increase in pH (mean 8.4 but as high as 12.5) downstream, and Berry Cave Salamanders (*Gyrinophilus gulolineatus*) have been observed with chemical burn-like lesions in the pool immediately downstream.

Changes in land use in recharge basins can lead to habitat loss and degradation for both troglobionts and stygobionts. Conversion of land for development, agriculture, and silviculture may result in increased sedimentation and hydrological changes that reduce available subterranean habitat and alter input of allochthonous organic inputs. Impervious surfaces, particularly in urban areas, increase the speed and amount of storm water runoff leading to degradation of habitat through more rapid transport of contaminants and increased sediment load. Hydrological changes may impact reproductive success in species that rely on hydrological cues (flow rates, temperature, etc.) to trigger reproduction. Habitat loss and degradation of cave systems can result from the direct destruction or alteration of habitat during quarrying, mining, and fracking operations, road construction, and urbanization, in general. Shockwaves resulting from blasting can travel underground for several kilometers and disturb populations of cave-roosting bats.

Impoundments on streams and rivers are another major threat impacting both terrestrial and aquatic subterranean species. While the direct loss of terrestrial habitat associated with flooding when dams are created is obvious, impacts associated with changes in local hydrology may be less apparent. Groundwater harvest for human use, irrigation in agricultural settings, and industry is a significant threat to groundwater life, particularly in arid regions globally.

Fungal diseases have emerged that have resulted in the extirpation of local populations of bats and amphibians. In North America, the introduced fungus *Pseudogymnoascus destructans* (Pd) has infected and killed millions of bats since it

was first detected in 2006. The fungus causes white-nose syndrome (WNS), with symptoms that include characteristic white fungal growth on the nose, wings, ears, and tail and changes in behavior of hibernating bats (Fig. 2). The fungus, which thrives in low temperatures and high humidity, causes bats to wake from hibernation more frequently causing them to consume fat reserves before spring. Bats either starve to death or succumb to the elements when trying to search for food during winter. To date, WNS has spread to 32 US states and 5 Canadian provinces and has caused massive population declines in some species, such as northern long-eared bats, little brown bats, and tri-colored bats. The fungus is native to Eurasia and has been confirmed in 13 bat species in Europe, although no mass mortality is known. Although humans can potentially spread the fungus from cave to cave inadvertently on their clothing and equipment, primary transmission is from bat to bat and bat to cave.

Fig. 2. White-nose syndrome caused by the introduced fungal pathogen *Pseudogymnoascus destructans* (Pd) that has infected and killed millions of bats since its introduction in 2006, such as this tri-colored bat (*Perimyotis subflavus*) from a cave in Jackson County, Alabama.

The collection of troglobionts and stygobionts for amateur or scientific purposes, illegal or otherwise, poses a threat to some species. Because of their uniqueness and the ease at which individuals can be captured, some cavefishes and cave salamanders can be easily exploited. Overcollection can potentially reduce or even extirpate local populations. For example, the low abundance currently of the cavefish *Amblyopsis spelaea* in the Echo River and River Styx sections of Mammoth Cave in Kentucky may be a consequence of when the species was sold as a novelty in the 1800s (Elliott, 2000). At Shelta Cave in Alabama, at least 111 specimens of the cavefish *Typhlichthys subterraneus* have been collected (Niemiller et al., 2013), most during 2-year period in the late 1950s when over 30 cavefish were regularly observed during a single survey. Such collection pressure has contributed to the decline of this population, as fewer than five cavefish have been observed during recent surveys. However, overcollection may be a minimal concern for certain taxa

that are difficult to capture and collect or have low appeal in hobbyist/amateur circles.

Although data are lacking for most species, habitat disturbance caused by recreational cavers may pose a threat, as the activities of even the most cautious caver could have significant impacts on stygobiotic populations that reside in shallow streams or troglobionts occurring in particular microhabitats, for example. Increased cave visitation may alter breeding activities, disturb prey populations, increase activity due to stress, or even result in death by trampling. However, potential impacts from recreational caving are likely minimal for most species. In contrast, "amateur" spelunkers that leave trash, vandalize formations, and paint graffiti on cave walls are of much greater concern.

Climate change is expected to have significant impacts on the quality, levels, and sustainability of groundwater through alteration of the hydrological cycle (e.g., Kløve et al., 2013). While research programs have begun to reveal how groundwater might respond to climate change in the past 10 years, knowledge of how stygobionts will be affected by climate change is still largely unknown. Similarly, the impacts of anthropogenic global warming on terrestrial cave ecosystems have not been well investigated. Species can shift their ranges, adapt to changing environmental conditions, or go extinct in response to climate change. Because of their unique habitat requirements and limited dispersal ability, many troglobionts and stygobionts may be quite vulnerable to the impacts of climate change (e.g., Mammola et al., 2017). Potential effects of climate change may directly alter abiotic conditions, such as water temperature, dissolved oxygen content, and rates of groundwater recharge, and exacerbate already existing threats, including groundwater pollution and extraction. In addition to altering environmental conditions, such as air and water temperature, humidity, and dissolved oxygen content, climate change may result in increased frequency of subterranean invasion and abundance of nonobligate species sensitive to changing surface conditions. Such a scenario would not be unprecedented, as past climatic change has been hypothesized to be the driving force behind subterranean colonization and evolution of several temperate North American troglobionts and stygobionts. These "invasive" species may prey on or outcompete cave-obligate species. Novel pathogens or parasites also may be introduced from other species that shift their ranges and niches into caves.

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Conservation Genetics

Richard Frankham, in Encyclopedia of Ecology (Second Edition), 2019

Do Genetic Factors Increase Extinction Risk?

Since inbreeding and loss of genetic diversity reduce reproduction and survival and the ability to evolve, they might be expected to increase extinction risk. However, it was hypothesized by Russell Lande in the late 1980s that demographic factors (human factors, plus demographic and environmental stochasticity and catastrophes) would often drive species to extinction before genetic factors could impact them, and this view was widely promoted. However, empirical evidence refuted this hypothesis for most populations. Further, genetic factors have been shown in increase extinction risk in wild habitats for outbreeding diploid species, based on direct observations and computer simulations parameterized with data

from real threatened populations. For example, inbreeding explained 26% of the variation in extinction risk in wild populations of the Glanville fritillary butterfly in Finland. Further, populations of deerhorn clarkia plant (*Clarkia pulchella*) with an inbreeding coefficient of 4% had a 25% extinction rate, while those with an 8% inbreeding coefficient had a 69% extinction rate over 3 years in the wild i.e., a small increase in inbreeding resulted in a large increase in extinction rate. Computer simulations indicate that inbreeding increases extinction risk over a wide range of scenarios in outbreeding species. For example, across 30 vertebrate species, inclusion versus exclusion of inbreeding depression at realistic levels resulted in a 37% reduction in median times to extinction in wild populations over a range of population carrying capacities. Loss of genetic diversity has also been shown to increase extinction risk under environmental change.

It is important to recognize that human impacts, genetic impacts, catastrophes and demographic and environmental stochasticity (natural fluctuations) all contribute to extinction risk, and that they typically interact in a downward feedback loop termed the extinction vortex.

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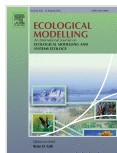## Recommended publications:

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