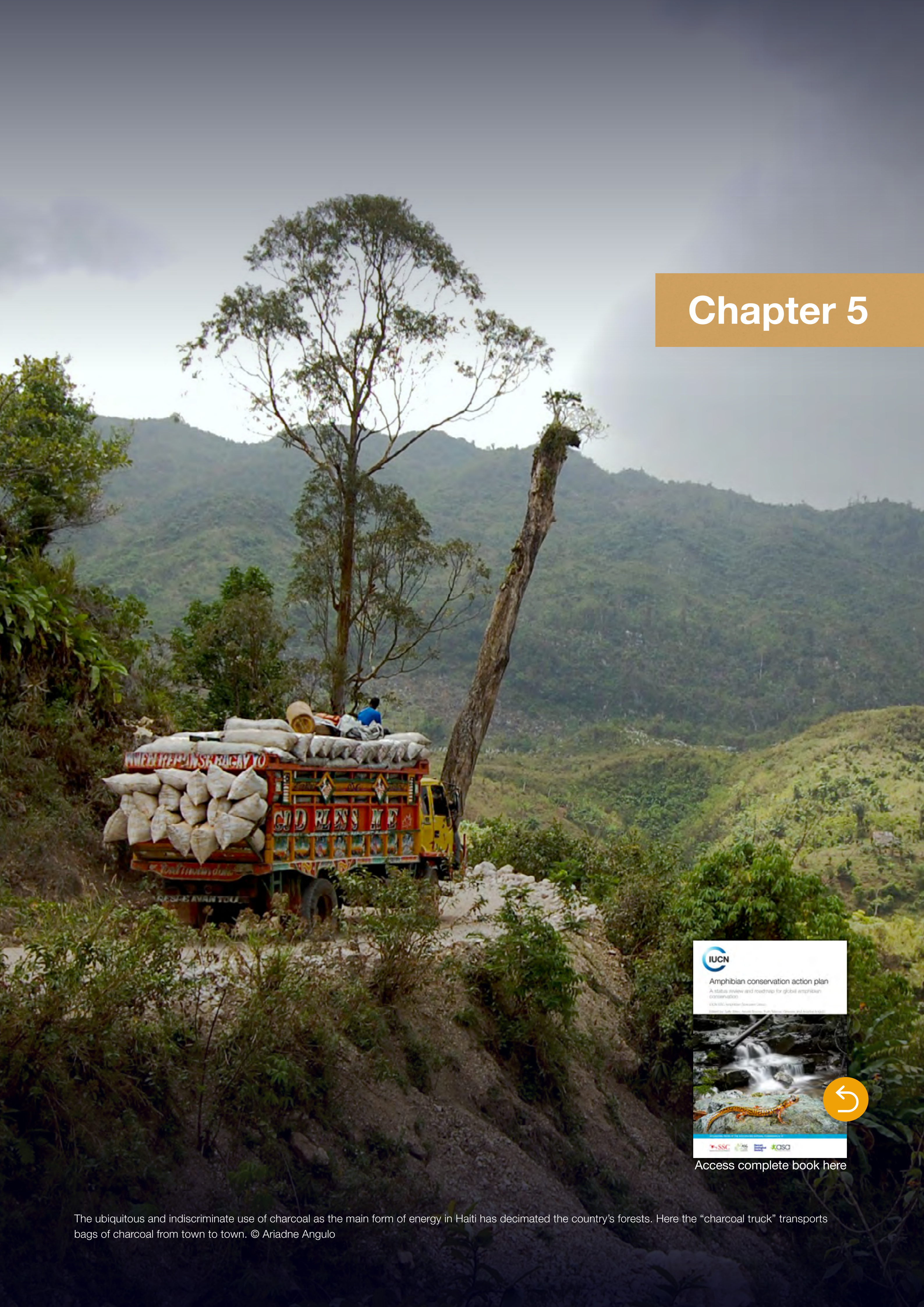
Chapter 5



Access complete book here

The ubiquitous and indiscriminate use of charcoal as the main form of energy in Haiti has decimated the country's forests. Here the "charcoal truck" transports bags of charcoal from town to town. © Ariadne Angulo

Chapter 5

Habitat loss: protection and management

Nicolás Urbina-Cardona[1], Ariadne Angulo[2], Andrew Turner[3,4], Dan Cogălniceanu[5], Deepthi Wickramasinghe[6], Ireri Suazo-Ortuño[7], James Watling[8], Luz Dary Acevedo[9], Omar Hernandez-Ordoñez[10], Thomas Doherty-Bone[11], Jeanne Tarrant[12]

[1] Pontificia Universidad Javeriana, Facultad de Estudios Ambientales y Rurales, Departamento de Ecología y Territorio, Bogotá, Colombia
[2] IUCN SSC Amphibian Specialist Group, Toronto, Ontario, Canada
[3] CapeNature Biodiversity Capabilities Directorate, Cape Town, South Africa
[4] Department of Biodiversity and Conservation Biology, University of the Western Cape, Cape Town, South Africa
[5] Ovidius University Constanta, Faculty of Natural Sciences and Agricultural Sciences, Constanta, Romania
[6] University of Colombo, Colombo, Sri Lanka
[7] Universidad Michoacana de San Nicolás de Hidalgo, Morelia, México
[8] Department of Biology, John Carroll University, University Heights, USA
[9] World Conservation Society, Barrio Versalles, Colombia
[10] Departamento de Zoología, Universidad Nacional Autónoma de México, México City, México
[11] Conservation Programs, Royal Zoological Society of Scotland, Edinburgh, United Kingdom
[12] Threatened Amphibian Programme, Endangered Wildlife Trust, Gauteng, South Africa

Abstract

Habitat loss is the primary driver of amphibian declines. The protection and management of habitats are thus the most critical conservation actions needed for at least 60% of amphibians, with habitat loss accounting for population declines and extinctions at local and regional levels. Habitat loss is directly related to pollution, but it also exacerbates other major threats to amphibians, such as disease, illegal trade, and invasive species. Habitat loss also reduces the ability of amphibian species to disperse and alter their distribution within their ecophysiological tolerance ranges in order to adapt to climate change. Currently, less than 30% of amphibian species are represented in the global protected-area system. The restricted geographic distribution, high habitat-specificity, and dependence on narrow climatic envelopes of many amphibian species mean that amphibians are particularly prone to local extinctions. Of the 37 amphibian species reported as extinct as of 2021, 48.6% were distributed in South and Southeast Asia, and 21% in Mesoamerica. These species mainly inhabited inland wetlands and forests. Considerable research into understanding the effects of habitat loss, fragmentation and degradation on amphibians have been undertaken over the past 15 years, including a review on the effectiveness of amphibian-targeted conservation interventions.

Habitat protection and management priorities must include the urgent preservation of remnant native forest habitats, given that over 85% of amphibian species occur in these systems. Conservation actions must also include the protection and rehabilitation of other aquatic and terrestrial breeding habitats critical for supporting viable amphibian populations. Given the limited resources for conservation, protection of globally important sites for amphibians (such as Alliance for Zero Extinction- AZE, and Key Biodiversity Areas - KBA), and their integration with protected areas into a network of conservation areas, is a key priority. The creation, rehabilitation and restoration of amphibian habitats, including in urban and agricultural landscapes, must not be excluded from the toolkit of interventions

needed to avoid declines of more generalist species. Beyond implementing direct habitat protection mechanisms, it is essential to ensure targeted management of newly created protected areas and improve that of existing protected areas, inclusive of amphibians. For these actions to be sustainable, it is critical to facilitate the participation, communication, and involvement of a broad range of stakeholders, including government entities, productive-extractive sectors, NGOs, academia, local communities, and civil society.

Introduction

Through their 350-million-year presence on Earth, amphibians have come to inhabit all continents, except Antarctica, and have adapted to a vast array of habitats. From montane grasslands to coastal wetlands, tropical forests, and savannahs, amphibians make up a large proportion of the vertebrate biomass in most temperate and tropical ecosystems (Burton & Likens, 1975; Duellman, 1999, *see also* Chapter 1) and provide important ecosystem services (Hocking, Babbitt & Hocking, 2014; Valencia-Aguilar, Cortés-Gómez & Ruiz-Agudelo, 2013). Only 5% of the earth's surface remains unmodified by anthropogenic transformations (Kennedy et al., 2019); with natural ecosystems currently under severe pressure from human presence and activity, amphibians are the animal class most negatively impacted by the current extinction crisis (Catenazzi, 2015; Houlahan et al., 2000; IUCN, 2021; Kiesecker, Blaustein & Belden, 2001), experiencing extinction rates as much as 200 times that of the background rate (Roelants et al., 2007). Habitat loss is the primary driver of amphibian declines (Green et al., 2020; Nori et al., 2015; Stuart et al., 2004). Loss, transformation, fragmentation, modification and degradation of habitat affect the highest proportion of assessed amphibians, followed by the threat of invasive species and disease (IUCN, 2021; *see also* Chapters 1 and 4). The world's forests harbour 85% of amphibian diversity (IUCN, 2021); yet half of these habitats have been lost (Crowther et al., 2015). At an even larger scale, half of the world's habitable land has been converted for agricultural use (Ritchie & Roser, 2019) and freshwater systems are particularly impacted (WWF, 2020). Only a third of the world's longest rivers remain free-flowing (Grill et al., 2019); those that are dammed result in flooding of important amphibian habitat (Dare et al., 2020; Dayrell et al., 2021; Jenkins et al., 2015). Water disruption and diversion could lead to local extinctions of amphibians that utilise lotic freshwater habitats as breeding sites (Crnobrnja-Isailović et al., 2021). Alarmingly, 87% of all wetlands have been lost globally since the year 1700 (Ramsar Convention on Wetlands, 2018), with the rate of wetland destruction three times faster than that of rainforests (Pearce & Madgwick, 2020). In addition to habitat fragmentation and loss, degradation of remaining wetlands and other amphibian habitats involves stressors such as pollution, loss of connectivity, biological invasions and emerging diseases (Buck et al., 2012; Lehtinen, Galatowitsch & Tester, 1999).

Underpinning this loss of habitat is historic and, in most global regions, continuing human population growth, combined with unsustainable resource use, and unsustainable consumption (Foley et al., 2005). To address this, conservation efforts must include addressing societal needs across local, regional, national and global scales. Conserving habitats critical to amphibians must bridge the spheres of policy, human health and wellbeing, governance, and education (Tarrant, Kruger & du Preez, 2016; Vergara-Ríos et al., 2021). Perhaps more than ever, there is a growing awareness of environmental issues and willingness by the public to demand governments and corporations to drive necessary changes (Li et al., 2022; Pawaskar, Raut & Gardas, 2018; Varumo et al., 2020). Without fundamental changes, further biodiversity loss will be inevitable and environmental sustainability undermined (Mace et al., 2018). The amphibian conservation community must play an active role in driving behaviour change at all levels to reduce, halt and ultimately reverse amphibian species loss.

The ASG Habitat Protection & Management Working Group was established to consolidate the habitat-related themes covered in the 2007 *ACAP*, namely the '*Key Biodiversity Areas*' and '*Freshwater Resources and Terrestrial Landscapes'* chapters. In this iteration of the ACAP we provide a synopsis of knowledge, achievements, and challenges to address the threat of habitat loss over the last 15 years, and identify a clear set of targets and actions for the next ten years.

Status update

Drivers of land-use change: Habitat loss and fragmentation

The growth of the human population in the past two hundred years has led to an unprecedented increase in the demand for natural resources (Ellis, 2015), especially from more industrialised countries. To meet the food, fibre, water, energy, and shelter needs of almost 8 billion people - as of 2020 (Kaneda, Greenbaum & Kline, 2020) - natural ecosystems have been transformed into farmlands, pastures, plantations, urban areas, and infrastructure networks (Foley et al., 2005; Sutherland et al., 2021; Figure 5.1). Habitat conversion for food production is a major driver of biodiversity loss (Newbold et al., 2016; Tscharntke et al., 2005) and climate change (Godfray et al., 2018; Poore & Nemecek, 2018), reducing species richness in amphibian communities (Dudley & Alexander, 2017; Gardner, Barlow & Peres, 2007) and decreasing the spatial and temporal distribution of species (Collins & Fahrig, 2017; de Oliveira et al., 2015). On the other hand, urbanization reduces the number of amphibian species that can survive and disperse in urban and suburban landscapes due to the alteration of key processes related to habitat availability and quality (Hamer & McDonnell, 2008). While multiple drivers modify natural systems including urbanisation, energy production, and mining, we focus here on food production as the primary driver. Specifically, livestock production is the largest anthropogenic land-use type, accounting for 75% of agricultural land (Machovina, Feeley & Ripple, 2015; Steinfeld et al., 2006). Meat production is directly responsible for 89% of rainforest conversion in South America (De Sy et al., 2015) and impacts freshwater availability and quality (Albert et al., 2020; Aritola et al., 2019). By 2050, agriculture is estimated to occupy one billion hectares of land (roughly the size of China), and will be coupled with increased use of fertilisers and pesticides (Tilman et al., 2001). The agricultural expansion will continue to transform biodiverse ecosystems in South America and sub-Saharan Africa, where large tracts of land still have unexploited agricultural potential (Laurance, Sayer & Cassman, 2014). Although some agricultural practices such as rice paddies generate wetlands, they do not provide high-quality habitat for all amphibians in the region (Borzée, Heo & Jang, 2018; Fujioka & Lane, 1997; Holzer et al., 2017; Naito et al., 2013). Additionally, climate change may affect regional seasonality and increase extreme weather events (Cochrane & Barber, 2009), which in turn could affect land occupation, use, and intensity patterns (Laurance et al., 2014; Figure 5.1, *also see* Chapter 3)

A collateral driver of landscape transformation is the associated expansion of linear infrastructure, including road networks into previously inaccessible areas (Gallice, Larrea-Gallegos & Vázquez-Rowe, 2019). Globally, the road network is expected to continue to expand, especially in megadiverse countries in Latin America and Africa (van der Ree et al., 2011). Roads often decrease landscape connectivity (D'Amico et al., 2016) and increase animal-vehicle collisions with severe ecological, social, and economic consequences (Oddone Aquino & Nkomo, 2021). Road infrastructure has both a direct impact on amphibians, and indirect impacts on biological processes (Andrews et al., 2008). Examples include habitat loss and increase in habitat degradation and fragmentation, increase in edge effects, limited circulation of individuals, increase in genetic isolation of populations residing on each side of the road, higher mortality rate and consequent numerical impoverishment of the populations living on the side of the road, and increased human access to natural habitats (see Schmidt & Zumbach, 2008).

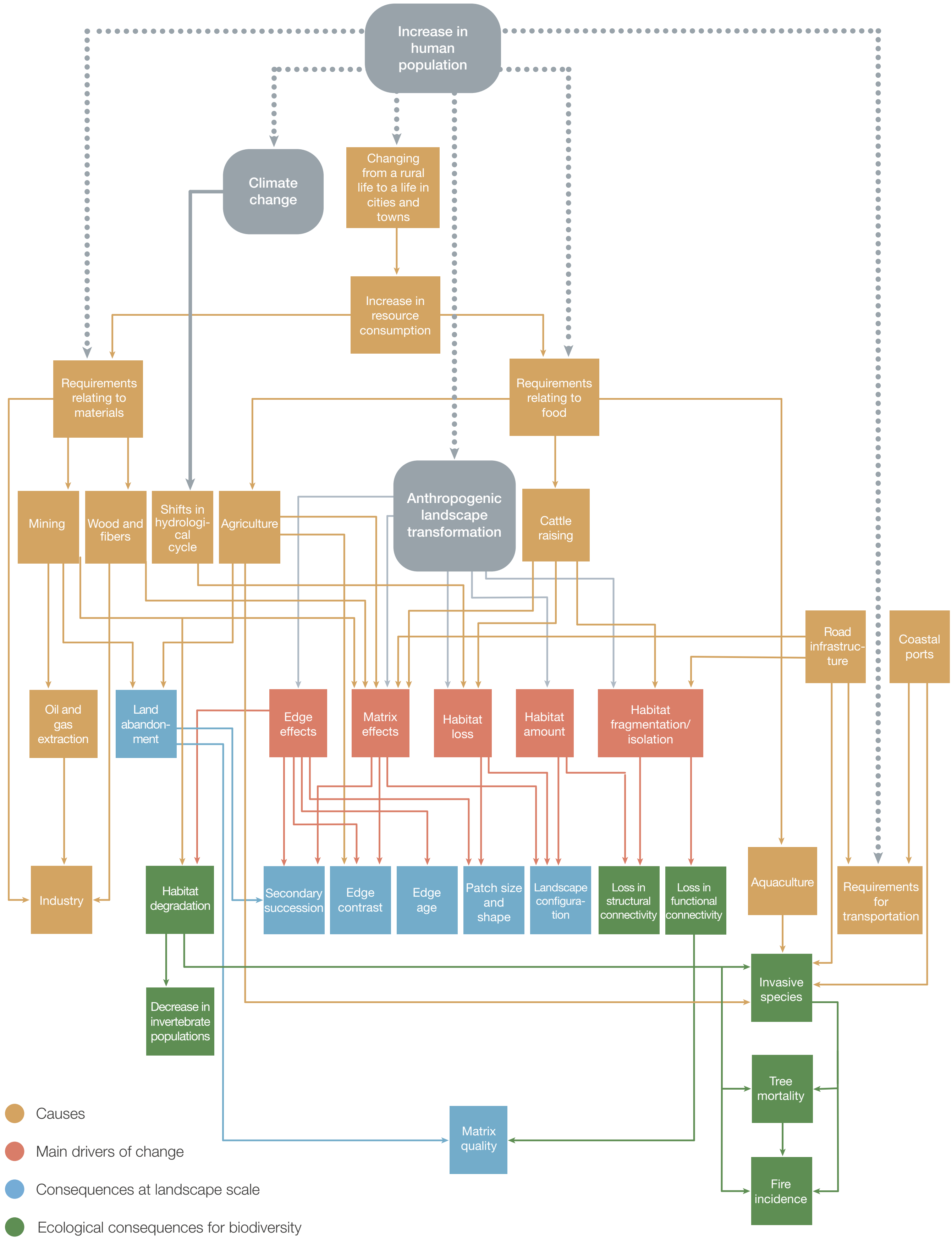


Figure 5.1: Causes and consequences of the anthropogenic transformation of the landscape. The causes are shown in blue; the main drivers of change are shown in orange; the consequences at landscape scale are shown in yellow; the ecological consequences for biodiversity are shown in pink. Note that although human population is a main driver, it is heavily influenced by geographic, cultural and socioeconomic factors, including barriers to family planning. Source: Figure developed by the authors based on the literature cited in this chapter.

Many amphibian species rely on different habitats for foraging, refuge, and reproduction, making landscape connectivity critical to the processes of dispersal and migration that maintains genetic and species diversity (Gilbert-Norton et al., 2010; Resasco, 2019).

Another insidious form of habitat degradation that is often exacerbated by transportation networks is the introduction, intentional or accidental, of invasive alien species (Bucciarelli et al., 2014; Kats & Ferrer, 2003; Nunes et al., 2019). Introduction of invasive alien species to a habitat can threaten native amphibians through direct effects such as predation (Bosch et al., 2006; Ficetola et al., 2011; Maerz, Blossey & Nuzzo, 2005; Martín-Torrijos et al., 2016; Vannini et al., 2018) and indirect effects such as altered water quality (Cotton et al., 2012; Maerz et al., 2005; Pinero-Rodríguez et al., 2021), water availability (Cordero-Rivera, Velo-Antón & Galán, 2007), and fire dynamics (Measey, 2011; van Wilgen, 2009). Likewise, some invasive and highly traded species such as the bullfrog *Lithobates catesbeianus* are vectors of emerging diseases such as ranavirus and chytrid fungus (Schloegel et al., 2009). Managing habitats and the invasion pathways that lead to them helps control existing invasions and minimise the risk of new invasions and are thus essential for safeguarding amphibian populations (Falaschi et al., 2020). Furthermore, it is critical to maintain continuity of invasive alien species control operations, particularly steady and reliable funding, to achieve success (Davies et al., 2020).

Effects of landscape transformation on amphibians

Landscape transformation resulting from habitat loss and fragmentation has led, directly and indirectly, to the decline of amphibian populations globally (Cushman, 2006; Gardner et al., 2007; Hamer & McDonnell, 2008; Sutherland et al., 2021; Urbina-Cardona, 2008). The loss of natural areas limits habitat for species not able to adapt to anthropogenic landscapes (Ribeiro, Colli & Soares, 2019) and leads to the homogenisation of biotic communities (Echeverría-Londoño et al., 2016; Ernst, Linsenmair & Rödel, 2006). Generalist species can inhabit modified environments, depending on their habitat requirements, movement capacity, and reproductive mode (Crump, 2015; Dale et al., 1994; Dixo & Metzger, 2010; Figure 5.2). However, for many species, high habitat specificity and endemicity preclude them from surviving in altered habitats (Roach, Urbina-Cardona & Lacher, 2020; Santos-Barrera & Urbina-Cardona, 2011). Most amphibian species occupy forest habitats (~85%), followed by wetlands (~ 66%), artificial terrestrial environments (~26%), grasslands (~17%), and to a lesser extent other habitat types (IUCN, 2021; numbers do not add up to 100% because a species may occupy more than one habitat; Figure 5.3).

Generalist species tend to have a wide geographic distribution in which they occur in a wide diversity of habitats with high abundance (Rabinowitz, Cairns & Dillon, 1986). Many generalist species can adapt to modified habitats, so habitat management actions must address the creation and enhancement of such environments. Such actions can also encourage public involvement, for example, the creation of ponds, ditches, and rice fields (Hartel et al., 2020; Magnus & Rannap, 2019; Mendenhall et al., 2014). This has the added advantage of giving people access to nature, instilling empathy and an appreciation of conservation efforts that can be leveraged to promote more effective policies (Balázsi et al., 2019; Oscarson & Calhoun, 2007). In contrast, rare amphibian species tend to have a higher degree of threat given their high level of habitat specificity (Toledo et al., 2014). Creation and rehabilitation of habitats for specialist or threatened species is also being increasingly explored and found to be effective (Fog, 1997; Ruhí et al., 2012; Valdez et al., 2019).

Forests contain diverse microhabitats that are used for shelter, foraging, and reproduction (Bowen et al., 2007; Rios-López & Aide, 2007; Wells, 2007), making them home to more species of amphibians than any other habitat. Most rare species are particularly abundant in forest interiors (Schneider-Maunoury et al., 2016), where heterogeneous environments have greater stability in temperature and relative humidity

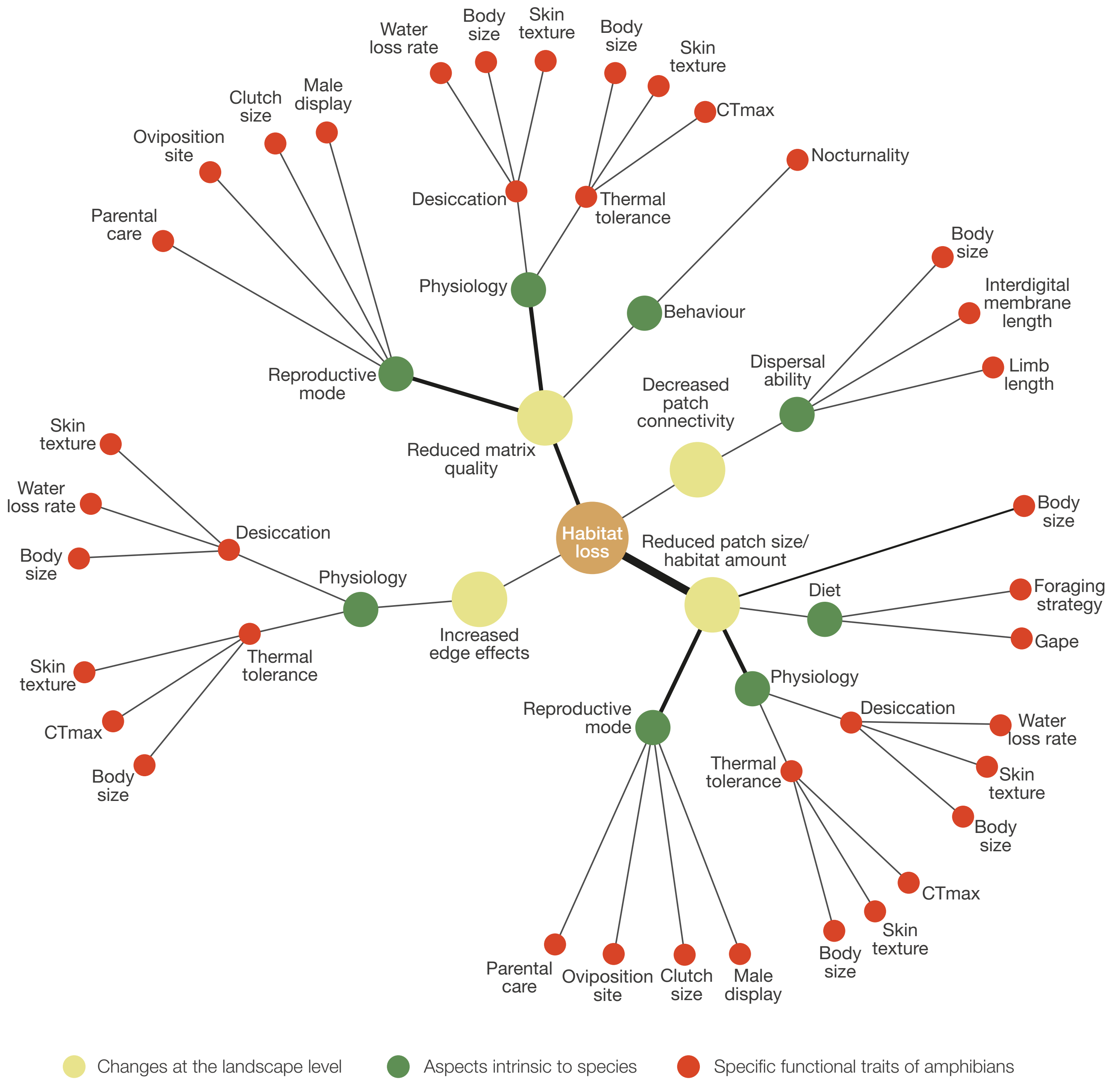


Figure 5.2: Effects of habitat transformation on amphibian species. Changes at the landscape level are shown in orange; aspects intrinsic to species are shown in green, and specific functional traits of amphibians are shown in red. Figure developed by the authors based on the literature cited in this chapter.

(Brüning et al., 2018; Soto-Sandoval et al., 2017). Management and protection of primary forest cores are thus a priority for amphibian conservation (Pfeifer et al., 2017). Environmental changes affect the physiological and biological processes of amphibians, so their occurrence depends on factors such as temperature and humidity (McDiarmid & Altig, 1999). Life-history traits and habitat preferences can predict a species' ability to tolerate environmental change (Álvarez-Grzybowska et al., 2020; Cortés-Gómez, Ramirez & Urbina-Cardona, 2015; Figure 5.2). For example, small-bodied species often avoid forest edges and the anthropogenic matrix where increased wind, light, heat (Pfeifer et al., 2017; Watling & Braga, 2015), and reduced canopy cover, leaf-litter and refugia (Demaynadier & Hunter, 1998) cause individuals to rapidly dehydrate (Figure 5.2). In contrast, large-bodied species with high dispersal capacity and aquatic larvae tend to inhabit pastures and food production systems (de Melo et al., 2017;

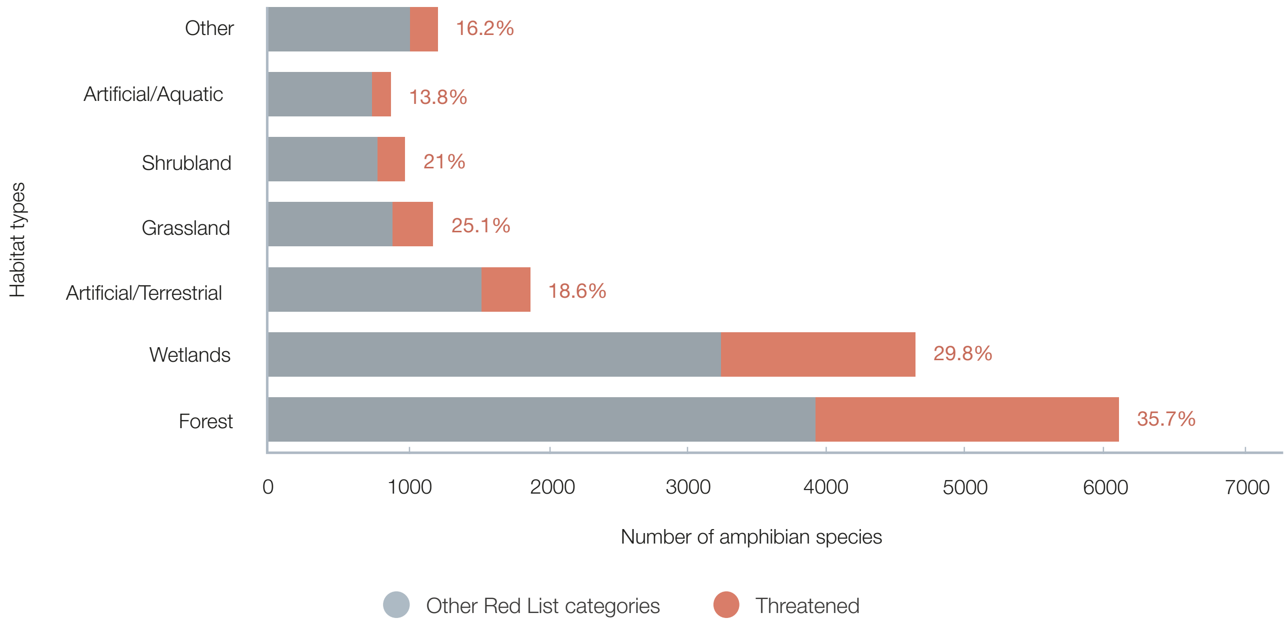


Figure 5.3: The top six habitat types for amphibians as reported on The IUCN Red List of Threatened Species™ (IUCN, 2021). The habitats are arranged according to the number of amphibian species occupying the habitat. The "Other" category in this figure includes marine intertidal, coastal, neritic, and supratidal, as well as introduced vegetation, savanna, desert, rocky areas, caves, and subterranean habitats. The percentage of threatened species that occupies each habitat is reported at the front of the bar; it should be noted that the total percentage does not correspond to 100% as a species may occupy more than one habitat. Source: IUCN, 2021.

Galindo-Uribe et al., 2022; Haddad et al., 2015; Mendenhall et al., 2014; Pineda et al., 2005; Queiroz, da Silva & Rossa-Feres, 2015; Trimble & van Aarde, 2014; Vasconcelos et al., 2009).

Edge effects and habitat degradation

Habitat loss and fragmentation often worsen due to edge effects (Fahrig et al., 2019; Fletcher et al., 2018). The edge effect is defined as the interaction that occurs between adjacent natural and anthropogenic vegetation covers creating an ecotone (Murcia, 1995). Globally, 70% of native forests are less than 1 km from an edge, so understanding edge effects is crucial for assessing the impact on biotic communities after deforestation (Alignier & Deconchat, 2011; Broadbent et al., 2008; Haddad et al., 2015). The diversity and structure of amphibian assemblages inhabiting forest fragments may be influenced by distance to disturbed areas (Pearman, 1997; Suazo-Ortuño, Alvarado-Díaz & Martínez-Ramos, 2008). In the Neotropics, most amphibian species are sensitive to edge effects, even at distances of 400 m, due to their responses to microclimatic changes in temperature, wind, and relative humidity (Schneider-Maunoury et al., 2016). Species most vulnerable to habitat loss and fragmentation are those inhabiting forest cores since they depend on high-quality habitat, and mostly avoid edges and the anthropogenic matrix (Lehtinen, Ramanamanjato & Raveloarison, 2003; Urbina-Cardona, Olivares-Pérez & Reynoso, 2006). Consequently, species adapted to mature forest interiors may disappear from small and irregularly shaped remaining patches in the absence of suitable breeding sites (Cabrera-Guzmán & Reynoso, 2012; Riemann et al., 2015; Tocher, Gascon & Zimmerman, 1997) or structural connectivity (Gillespie et al., 2015). In West Africa, degradation of vegetation structure had a stronger deleterious effect on forest amphibian species richness than habitat fragmentation (Hillers, Veith & Rödel, 2008). Likewise, it is important to consider that in highly fragmented landscapes, each forest patch may have a unique biotic community, so the loss of a single small fragment could lead to a global or regional loss of species (Fahrig, 2017; Fletcher et al., 2018). Linear remnants of native vegetation also constitute dispersal corridors for some amphibian species (De Lima & Gascon, 1999; Hansen et al., 2019).

Matrix effects and substitutable resources at a landscape level

In transformed landscapes, the dynamics between natural patches and other landscape elements are highly influenced by the anthropogenic matrix (Dixo & Metzger, 2010; Ferrante et al., 2017; Van Buskirk, 2012; Watling et al., 2011). Matrix effects on population abundance and survival are associated with resource availability, the abiotic environment and the dispersal capacity of the study species (Driscoll et al., 2013). In areas with intense agricultural practices (monocultures; burning, slashing, and logging; low temporal rotation; high use of pesticides-herbicides; and soil mismanagement; Ellis, 2015; Kremen, Williams & Thorp, 2002), amphibian assemblages show low species richness and high abundance of generalist species (Cáceres-Andrade & Urbina-Cardona, 2009; Gascon et al., 1999; Vasconcelos et al., 2009). In contrast, small-scale rural and family agricultural practices, with agro-ecological, multifunctional, or sustainable approaches, promote greater permeability of the matrix (Brüning et al., 2018). Permeable landscapes reduce the negative consequences of fragmentation (Foley et al., 2005; Oteros-Rozas et al., 2019; Perfecto & Vandermeer, 2010) and facilitate the dispersal of amphibian species (Kehoe et al., 2015; Perfecto & Vandermeer, 2008, 2010), although this depends on the landscape elements that are used by species (Tarrant & Armstrong, 2013; Van Buskirk, 2012). Likewise, land cover type, structural complexity and the size of the matrix surrounding remaining natural patches play an important role in retaining connectivity and species richness (Cline & Hunter, 2016; Phillips et al., 2018; Watling et al., 2011).

In some tropical ecosystems, matrix effects may impact amphibians more than edge effects (De Lima & Gascon, 1999; Isaacs Cubides & Urbina Cardona, 2011; Mendenhall et al., 2014). For example, an intensively managed matrix with sparse, homogeneous vegetation such as a cornfield may increase edge effects on amphibian populations up to 150m into the forest (Santos-Barrera & Urbina-Cardona, 2011). In contrast, crops with a complex structure that maintain elements of the original native vegetation (e.g. shaded coffee or cocoa plantations) can buffer edge effects in native habitat by increasing amphibian species richness in the ecotone (Mendenhall et al., 2014; Rice & Greenberg, 2000; Roach et al., 2020; Santos-Barrera & Urbina-Cardona, 2011). These kinds of agroforestry systems could harbour an important percentage of amphibian species in montane cloud forests and tropical rainforests (Murrieta-Galindo et al., 2013; Murrieta-Galindo et al., 2013; Pineda & Halffter, 2004). Due to its use for biofuel, oil palm monocultures (of exotic invasive species *Elaeis guineensis*) have increased globally (Danielsen et al., 2009), reducing the richness of amphibian assemblages when compared to surrounding native forests (Faruk et al., 2013; Gallmetzer & Schulze, 2015; Gilroy et al., 2015; Konopik, Steffan-Dewenter & Grafe, 2015; Scriven et al., 2018). Tropical amphibians are particularly affected by the intensive implementation of large-scale (more than 100 ha) monoculture tree plantations of exotic species at an early age (less than 10 years) and in which, due to high rotation, vegetation growth in the understorey is prevented (López-Bedoya et al., 2022). This is alarming since 40% of forest plantations are located in the tropics, and for the Neotropics 85% of these plantations are composed of monocultures of exotic species (Payn et al., 2015). We recommend that the effects of forest edges and anthropogenic matrices be incorporated into systematic conservation planning protocols to identify corridors that may allow animal movement in response to global change (Baldwin, Calhoun & deMaynadier, 2006; Muths et al., 2017; Nori et al., 2015; Pence, 2017).

In the larval or juvenile stage, amphibians are more vulnerable to dehydration, predation, and the effect of contaminants (Crump, 2015; *also see* Chapter 4). Anthropogenic systems thus affect the quality and quantity of habitat found at the edges of remaining fragments (Didham, Kapos & Ewers, 2012; Harper et al., 2005; Murcia, 1995; Saunders, Hobbs & Margules, 1991). It is important to consider that species use different habitats that allow them to maintain populations over time, and habitats within the matrix could be relevant to different life stages and activities of species (Pope, Fahrig & Merriam, 2000; Van Buskirk, 2012). For example, some native forest-dwelling amphibian species may pass through anthropogenic matrices or use them for reproduction (Gascon et al., 1999). Neckel-Oliveira and Gascon (2006) found that the tarsier tree frog (*Phyllomedusa tarsius*) was more abundant in the anthropogenic matrix due to the presence of large and permanent ponds, but also reported low reproductive success and survival of eggs and embryos due to predation and desiccation. In contrast, Van Dyke et al. (2017) found that amphibian species richness was positively linked to clustered pools in forests compared to isolated ones. Finally, Camacho-Rozo and Urbina-Cardona (2021) found that temporary water bodies created in pastures by anthropogenic activities (e.g. cattle or tractor tracks) harbour less than 15% of larval anuran species than natural temporary ponds. Thorough knowledge of the life history, behaviour, and dispersal of target amphibian species is key to ecological restoration and species reintroductions (Tarrant & Armstrong, 2013; *also see* Chapter 14).

Heterogeneity in vegetation structure has a strong impact on amphibian assemblages (Cortés-Gómez, Castro-Herrera & Urbina-Cardona, 2013; Gardner et al., 2007) across spatial scales from microhabitats to landscape level (Duarte-Ballesteros, Urbina-Cardona & Saboyá-Acosta, 2021). For instance, matrices with high structural complexity can reduce temperature extremes (Scheffers et al., 2014) and buffer edge effects on forest fragments (e.g. coffee plantations; Santos-Barrera & Urbina-Cardona, 2011). In heterogeneous agricultural landscapes, vegetation buffers environmental extremes by reducing exposure of amphibians to unfavourable conditions such as dehydration and elevated temperatures (Farallo & Miles, 2016; Watling & Braga, 2015; Whitfield & Pierce, 2005). The rate of temperature increase may be 60% lower in microhabitats located in forested areas compared to more exposed microhabitats (Scheffers et al., 2013, 2014). In this sense, forest plantations can be an opportunity to complement amphibian conservation if they are implemented on small areas (less than 85 ha) on disused pastures, maintaining a large variety of forest species (mixed plantations including native species) over the long term (more than 26 years), and allowing species to grow in the shrub layer (which increases the diversity of arthropod prey and oviposition sites; López-Bedoya et al., 2022). It is therefore important to maintain heterogeneity in vegetation cover and aquatic resources within the matrix, and to promote environmentally friendly management practices (e.g. low use of agrochemicals, fire management, maintenance of hedgerows and native vegetation, control of invasive species, and maintenance of leaf litter on the ground; Arroyo-Rodríguez et al., 2020; Melo et al., 2013; Urbina-Cardona et al., 2015; Zabala-Forero & Urbina-Cardona, 2021).

Colonisation and persistence of amphibian diversity in secondary forest

Secondary forests are forests regenerating largely through natural processes after significant human and/or natural disturbance of the original forest vegetation (in which floristic composition and structure have been modified) at a single point in time or over an extended period (Brown & Lugo, 1990; Chokkalingam & de Jong, 2001). Anthropic secondary forests can be classified based on the original type of disturbance: **i)** abandoned open areas with intense agricultural practices (monocultures); **ii)** burned forests; **iii)** abandoned selective logging sites; and **iv)** agroforestry. Those forests have become a frequent or even dominant vegetation type in human-modified landscapes (Arroyo-Rodríguez et al., 2017) and there is a continuous increase in this type of forest, mainly in tropical regions (Hansen et al., 2019). Despite increasing agricultural intensification globally, about 1.47 million km^2 of agricultural systems have been abandoned due to loss of soil productivity or socioeconomic and political factors (Bowen et al., 2007; Guariguata & Ostertag, 2001). Secondary forests are important biodiversity repositories and may provide complementary and supplementary

resources to fauna (Arroyo-Rodríguez et al., 2017), and the abandonment and recovery through time of biodiversity can allow other species to colonise these forests (Laurance et al., 2011).
Secondary succession pathways depend on multiple factors and processes at different scales, driving direct or indirect changes at different levels:

- On previous land use and landscape composition (e.g. type, duration, intensity, and frequency of disturbance regime; Chazdon, 2003; Thompson & Donnelly, 2018; Walker et al., 2010).

- Landscape configuration (e.g. proximity to remaining forest patches and anthropogenic matrix structure; Brüning et al., 2018; Laurance et al., 2002; Tscharntke et al., 2012) and composition (Tscharntke et al., 2012).

- Patch characteristics (e.g. soil properties, size, shape, isolation, and microclimate; Chazdon, 2003; Guariguata & Ostertag, 2001).

With increasing time since agricultural abandonment and structural complexity of vegetation, some amphibian assemblages can increase their richness and number of individuals (Acevedo-Charry & Aide, 2019; Thompson & Donnelly, 2018). There is mainly an increase in the abundance of generalist forest species, given the colonisation of species from the matrix (*sensu* spillover edge effects: Riest et al., 2004; Bowen et al., 2007). However, changes in the structure and composition of assemblages in secondary forests are dynamic given the increase in abundance of generalist forest species, colonisation of species from the matrix, and the possible arrival of specialists from the mature forest (Acevedo-Charry & Aide, 2019; Bowen et al., 2007). Vegetation succession interacts with species traits (e.g. tolerance to extremes in temperature and relative humidity, diet specialisation, preference for oviposition sites and breeding seasons; Gottsberger & Gruber, 2004; Suazo-Ortuño et al., 2018; Thompson & Donnelly, 2018) and natural disturbance regimes (e.g. hurricanes: Marroquín-Páramo et al., 2021; fires: Dunn, 2004; Mora et al., 2015), making the recovery process complex at the landscape, community, and population levels (Russildi et al., 2016; Walker et al., 2010). For example, a study found that the increase in frequency and intensity of hurricanes created a homogenisation of amphibian assemblages inhabiting tropical dry mature forests, but amphibian assemblages inhabiting pastures were highly resilient to change (Marroquín-Páramo et al., 2021).

There is a trend towards increasing functional diversity (Ernst et al., 2006; Hernández-Ordóñez et al., 2019) and amphibian species richness in mature forests (Basham et al., 2016; Pawar, Rawat & Choudhury, 2004) in late-successional stages (Herrera-Montes & Brokaw, 2010; Hilje & Aide, 2012) and in the interior of native forest fragments (Zabala-Forero & Urbina-Cardona, 2021). Attention is drawn to the importance of appropriately screening amphibian functional traits and functional diversity indices as tools to inform on the loss of functional attributes in assemblages (and the functional role of species in ecosystem processes) that may jeopardise ecosystem stability under scenarios of anthropogenic landscape transformation (Galindo-Uribe et al., 2022; Tsianou & Kallimanis, 2016). This is because small changes in plant structure, the number of available microhabitats, and the presence of water bodies generate drastic changes in species composition in forests with different successional stages (Cortés-Gómez et al., 2013; Hernández-Ordóñez, Urbina-Cardona & Martínez-Ramos, 2015; Magnus & Rannap, 2019; Urbina-Cardona & Londoño-M, 2003). Once food-production systems were abandoned and rainforest regeneration began, amphibian species richness was the first parameter to recover (after 23 years), followed by species density (28 years for amphibians; Hernández-Ordóñez et al., 2015). In contrast, other parameters such as species composition are estimated to take between 80 and 150 years to recover (Bowen et al., 2007; Thompson & Donnelly, 2018). Management of secondary forests is thus crucial for biodiversity conservation because of their role in maintaining connectivity between older forest patches, facilitating dispersal of species with low matrix tolerance, as well as the mitigation of edge effects in remaining forest fragments (Goldspiel et al., 2019; Suazo-Ortuño et al., 2015; Thompson & Donnelly, 2018).

Amphibian representation in the protected area system

The IUCN defines protected areas (PAs) as “a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values”. PAs are a fundamental cornerstone in the conservation of biodiversity, including amphibians (Le Saout et al., 2013; Venter et al., 2014). The Convention on Biological Diversity (CBD)’s Kunming-Montreal Global Biodiversity Framework (GBF) adopted in late 2022 aims to halt and reverse global biodiversity loss by remedying the multifaceted drivers behind biodiversity declines around the planet. It follows on from the Aichi targets (2010–2020) and will guide interventions to conserve biodiversity and ecosystem services for the next three decades. It includes four *goals* and 23 *targets* to be achieved by 2030. Urgent action is therefore required in less than a decade to initiate and complete these targets toward the achievement of the outcome-oriented goals for 2050. Almost all of these 23 targets have some relevance to habitat protection and management, with the most pertinent being: Target 1- Spatial planning to prioritise conservation; Target 2 - Effective restoration of PAs; and Target 3 - 30 percent of terrestrial areas protected by 2030; Target 4 - Management to halt extinction risks to threatened species; Target 6 - Reduce and mitigate the impacts of invasive alien species; and Target 11 - Restore & maintain ecosystem functions and services. The full descriptions can be found at https://www.cbd.int/gbf/ (CDB, 2022). Notably, Target 3 (the “flagship” target of the Framework, commonly known as 30×30), calls for at least 30% of terrestrial, inland water, and coastal and marine areas to be effectively conserved and managed by 2030. This framework will require the inclusion of all sectors, clear communication, and effective funding mechanisms to be enacted effectively if targets are to be achieved. In terms of the previously proposed CBD Aichi Biodiversity targets, by 2015, it was clear that while existing terrestrial PA proportions were relatively close to being achieved (14.6% of terrestrial and 2.8% of marine environments), >59% of ecoregions, >77% of important sites for biodiversity, and 57% of 25,380 species were not well represented in the PA network (Butchart et al., 2015). Within the existing PA system, 137 sites represent high irreplaceability for the conservation of amphibians, birds, and mammals, with the potential to conserve 385 amphibian species of which 179 species are threatened (Le Saout et al., 2013). Recently, Button and Borzée (2021) proposed a method to identify geographic priorities for amphibian habitat protection globally.

The global PA network is fragile because many PAs do not guarantee the persistence of representative species and ecosystem processes (Kukkala & Moilanen, 2013; Margules & Sarkar, 2007). Globally, 25% of amphibian species have distributions totally outside PAs, and 18% have less than 5% of their distribution represented in PAs (Butchart et al., 2015; Nori et al., 2015). Regionally, for example, only 32% of the range of South Africa’s threatened amphibians occurs within PAs (Skowno et al., 2019). We need to ensure that priority amphibian habitats are included within formally declared PAs as well as other types of conservation areas, and that management of these is improved with amphibians and their habitats as conservation targets (Nori et al., 2015). Historically, amphibians have often not been prioritised in conservation planning, both in establishing PAs and in the development of management plans (Burbano-Girón et al. 2022; González-Fernández et al., 2022; Kueh et al., 2004; Rodrigues et al., 2004; Rodrigues et al., 2004; Urbina-Cardona & Loyola, 2008; Venter et al., 2014). For amphibians with restricted geographic distribution, it is necessary to protect all remaining habitats, including Alliance for Zero Extinction (AZE) sites and other Key Biodiversity Areas (KBAs), as these are often irreplaceable (see Ochoa-Ochoa et al., 2011; Ochoa-Ochoa, Urbina-Cardona & Flores-Villela, 2011). For example, South Asia is rich in amphibian species richness and endemism, representing three amphibian hotspots - Eastern Himalayas, Indo-Burma, and Sri Lanka (including Western Ghats) - that are underrepresented in PAs (Meegaskumbura et al., 2002; Pratihar et al., 2014). Asia and Latin America are the regions that harbour the greatest number of species worldwide without any representation in the PA system (115 gap species; Nori et al., 2015). Yet, the declaration and establishment of Important

Amphibian Areas (IAAs) and related regulations are lagging (Rowley et al., 2010).

However, amphibians are increasingly being recognised in PA planning (Ford et al., 2020). For example, the WWF Oasis network of Italy was specifically assessed for contributions to amphibian conservation (Bombi et al., 2012). Various NGOs have been actively working to facilitate the creation of PAs specifically to protect amphibians (Moore, 2011; Smith, Meredith & Sutherland, 2019; *see also* Table 5.1). Although private and community-managed PAs are usually small in area, they play an important role in amphibian conservation. For example, in Mexico, 73% of endemic species are represented in small private reserves (Ochoa-Ochoa et al., 2009). Nevertheless, achieving representation of amphibian species in a single PA is insufficient, because it can lead to small, isolated subpopulations. Rather, it is critical to ensure that species' core distributions are within PAs (Urbina-Cardona & Loyola, 2008). Some of the regions with the greatest amphibian species richness, including the tropical Andes in Peru, Ecuador and Colombia, southern Mexico, eastern Brazil, Papua New Guinea, and Indonesia, parts of Madagascar, Cameroon, and southwest India, are also areas with the highest rates of deforestation and least representation within the PA system (Nori et al., 2015); this underscores their great importance as priority areas for conservation (Button & Borzée, 2021). Thus, it is crucial to have clear spatial priorities that enable coordinated local planning of conservation area networks involving both government PAs and private initiatives (Ochoa-Ochoa et al., 2009).

Site prioritisation and management effectiveness

The creation and designation of PAs does not, by itself, ensure adequate species protection. Disturbance, hunting, and forest-product exploitation threaten the integrity of reserves worldwide (Laurance et al., 2012; Pouzols et al., 2014). The effectiveness of PAs to resist anthropogenic pressures is influenced by multiple factors, including a country's socio-economic and governance conditions (Barnes et al., 2016; Schleicher et al., 2017). PAs are not just under the jurisdiction of governments, but also

Table 5.1: Examples of different types of protected areas established to protect amphibian species

Site name	Date established	Target amphibian species	Site size (ha)	Country	Significance	Type of protection
Jorepokhri Wildlife Sancturary	1985	*Tylototriton himalayanus*	4	India	It has a small breeding population of the Himalayan newt. It is in danger because of the constructions made in the sanctuary.	Strict Protection, West Bengal State Forest Department
Natural Reserve "Monticchie"	1985	*Rana latastei*	230	Italy	One of the remaining large populations of this Italian endemic Ranidae	Special Area of Conservation – Europe Natura2000 site code IT2090001
"Paludi di Arsago" Area of Herpetological National Relevance	1995	*Pelobates fuscus insubricus*	543	Italy	Last remaining large population of this very rare Italian Pelobatidae	Special Area of Conservation – Europe Natura2000 site code IT2010011

Site name	Date established	Target amphibian species	Site size (ha)	Country	Significance	Type of protection
Guayacán Rainforest Reserve	2003	*Agalychnis lemur*	49	Costa Rica	Reserve is home to one of two known metapolulations of *A. lemur*, and has more species of amphibians (70+) than any other site in Costa Rica (https://cramphibian.com/guayacan-rainforest-reserve/)	Private Reserve
Ranita Dorada Reserve	2008	11 species	120	Colombia	Formerly an AZE site, trigger species *Andinobates dorisswansonae* and *A. tolimensis* now improved in status causing the site to be de-listed	Private Reserve
Ranita Terribilis Reserve	2012	*Phyllobates terribilis*	66.4	Colombia	KBA site. In 2020 the Eperãra Siaapidarã people incorporated their K´õk´õi Eujã Natural Reserve into the National Protected Area System, expanding the species' protection to 11,641 ha	Private Reserve
Sierra Caral Reserve	2012	10 threatened species; 7 endemic species	1901	Guatemala	The new reserve stimulated the declaration of the Sierra Caral National Protected Area in 2014	Private Reserve followed by National Protected Area
Yal Unin Yul Witz Reserve	2015	11 species	845	Guatemala	Within the larger Cuchumatanes KBA/AZE	Private Reserve

Site name	Date established	Target amphibian species	Site size (ha)	Country	Significance	Type of protection
Elandsberg Nature Reserve	In progress	*Vandjikophrynus amatolicus*	4783	South Africa	First PA for this Critically Endangered species	Biodiversity Stewardship site (landowner agreements)
Sobonakhona Protected Environment Reserve	In progress	*Hyperolius pickersgilli* *Natalobatrachus bonebergi*	535	South Africa	First PA within a Traditional Authority area to be declared in the country with an amphibian as a target species	Biodiversity Stewardship site (landowner agreement)
Mount David Nature Reserve	In progress	*Capensibufo selenophos*	821	South Africa	Also, the only remaining population of *Erica jasminiflora* occurs on the property	Biodiversity Stewardship site (landowner agreement)
Gingingdlovu Protected Environment Reserve	In progress	*Hyperolius pickersgilli*	125	South Africa	Linking coastal wetland across three private properties	Biodiversity Stewardship site (landowner agreement)
Hampton Nature Reserve	1998	*Triturus cristatus*	145.8	United Kingdom	Largest population of great crested newt in Europe	Special Area of Conservation - Europe Natura 2000 UK0030053; Site of Special Scientific Interest (UK); owned by private company managed by conservation NGO (Froglife).
Hyla Park Nature Preserve	1995	*Hyla versicolor*	8	Canada	Protecting most northeasterly population of *Hyla versicolor*	Public land leased by conservation organisation

local communities, private enterprises, and NGOs, as well as co-management between partners (Dudley, 2008; Roach et al., 2020). Examples of differing management structures include state protection, landowner agreements that provide formal protection of important biodiverse areas in the long term (Barendse et al., 2016), conservation agreements with local community zoning for land and resource use (e.g. areas for timber extraction), and indigenous conservation areas (Aguilar-López et al., 2020; Berkes, 2009; Ochoa-Ochoa et al., 2009). It is essential to align the objectives and goals of the PAs with the visions of the people living around them to ensure that human pressure is not increased due to cropland conversion and instead allows for increases in human development indices (Geldmann et al., 2019; Laurance et al., 2012). Community-based conservation initiatives (Meine, Soulé & Noss, 2006) allow for the integrated management of transformed landscapes that support biodiversity conservation (Arroyo-Rodríguez et al., 2020; Garibaldi et al., 2021; Melo et al., 2013; Palomo et al., 2014). Megadiverse countries often have a low socioeconomic status (i.e. those with the highest amphibian species richness are highly impacted by human activities; Nori et al., 2015). Effective habitat protection in these amphibian species-rich but often resource-poor nations must therefore be supported by adequate management actions (Smith & Sutherland, 2014) and integrated with development activities that improves the socio-economic well-being of the local communities, who are often directly dependent on nature for their resources, in order to increase their resilience to future challenges and reduce negative environmental impacts (Adger, 2000; Bennett, Radford & Haslem, 2006; Perfecto & Vandermeer, 2008, 2010).

Since some of the functional traits of amphibian species determine their degree of threat it is necessary to consider their monitoring in PA (González-del-Pliego et al., 2019; Loyola et al., 2018). An understanding of critical sites for the survival of amphibian species is essential in PA designation, inclusive of the functional traits and degree of endemism of species (Loyola et al., 2008; Menéndez-Guerrero, Davies & Green, 2020; Tsianou & Kallimanis, 2016). It is key to understand the distribution of amphibian species within each PA to inform management plans (Nori et al., 2015) and monitor not only their presence, but other aspects such as biomass, body condition, demography, trophic structure, and functional diversity (Álvarez-Grzybowska et al., 2020; Riemann et al., 2017; Trimble & van Aarde, 2014; Urbina-Cardona et al., 2015). To fulfil these tasks, PA management requires strengthening through improving facilities, ranger training, reinforcing compliance, and supporting research.

For PAs associated with low socio-economic communities, improving general land-use practices as well as including development activities to reduce the negative environmental impacts of nature-dependent local communities is critical.

Given their often-limited distributions and habitat specificity, amphibian protection needs to be more species-focused and allow for the creation of smaller PAs or the implementation of other in situ conservation methods. Several approaches allow for this: KBAs are sites that contribute significantly to the global persistence of biodiversity and provide a standardised approach to identifying sites of particular importance for biodiversity, allowing to guide implementation and monitoring of the post-2020 global biodiversity framework Target 3, focusing on conservation and management of at least 30 per cent of the world’s land, coastal areas and oceans (Smith et al., 2019). Sites qualify as global KBAs if they meet one or more of 11 criteria in “A Global Standard for the Identification of Key Biodiversity Areas” (IUCN, 2016), which harmonises existing approaches to the identification of important sites for biodiversity and has received considerable support from the conservation community. The Key Biodiversity Area Partnership—a coalition of 13 international conservation organisations—was formed to address the rapid loss of biodiversity by supporting the identification, monitoring, and safeguard of sites that are critical for the survival of species and ecosystems.

AZE sites comprise the most irreplaceable subset of KBAs, holding Critically Endangered or Endangered species restricted to a single site globally. Unless AZEs are properly conserved, they are sites where species extinctions are imminent (Ricketts et al., 2005). Nearly 40% of current AZEs are triggered by amphibians (334 out of 865 sites), the largest of any taxonomic group; yet, fewer than half are currently protected. By identifying and mapping AZE sites and other KBAs, information about the global importance of these areas for the survival of globally threatened and range-restricted amphibians can be provided to key stakeholders to make the best decisions about how to manage that land (or water), where to avoid development, and how to best protect the biodiversity for which the sites are so important. Given limited resources for conservation, this information is vital for conservation efforts centred on habitat protection to prioritise sites of global significance for threatened and restricted range amphibians.

Although not a prioritisation tool, it is important to consider Other Effective Area-based Conservation Measures (OECM) in the amphibian habitat management toolkit. OECMs are geographically-defined areas that are managed in a manner that sustains biodiversity (Gurney et al., 2021). OECMs can include indigenous or community-conserved areas, watershed protection areas, and other initiatives that deliver effective in situ conservation regardless of their primary objective, and this could work in tandem with prioritisation tools such as KBAs and AZEs. OECMs can also bring a more equitable approach to the decision-making process (Gurney et al., 2021).

If amphibian species are not considered within systematic conservation planning, the resulting network of conservation areas may not be congruent with the geographic distribution of this taxonomic group, even where "umbrella" species of groups such as mammals have been used (as demonstrated by Urbina-Cardona & Flores-Villela, 2010). Due to the high habitat specificity of some rare amphibian species, umbrella species are not a good tool for their conservation (Branton & Richardson, 2014; Caro et al., 2004; Roni, 2003). Likewise, amphibians have rarely been used as umbrella, flagship or keystone species to understand the consequences of landscape change (Lindenmayer & Westgate, 2020) or to evaluate the effect of invasive species management on the microenvironmental conditions of the habitat (Cox et al., 2022). In this sense, the danger of using a few species (keystone, umbrella or landscape) for the definition of spatial priorities for biodiversity conservation is highlighted, given their low capacity to represent other attributes of biodiversity. Thus, when prioritising networks of areas for biodiversity conservation, it is essential to include diverse biotic groups and different attributes (such as composition, structure and function; Burbano-Girón, et al., 2022). Additionally, spatial conservation priorities must be re-evaluated in the context of climate change scenarios and land use to ensure the persistence of species, populations and assemblages (Agudelo-Hz, Urbina-Cardona & Armenteras-Pascual, 2019; Grant, Miller & Muths, 2020; Urbina-Cardona, 2008). For example, in Australia 10–15% of land cover has been determined to be the target for the national reserve system; however, the representation of amphibians is highly variable, and this management approach ignores species` requirements for connectivity (Lemckert, Rosauer & Slatyer, 2009). Protecting important sites for amphibians is critical, but so is promoting connectivity between different planning and prioritisation initiatives to ensure a network of conservation areas and not just isolated points that will not allow the dispersal of species under global change scenarios (Carvalho et al., 2010).

Actions and opportunities for habitat protection and management

Actions and opportunities

Conservation actions should be informed by the best available evidence. However, evidence is often scarce and dispersed, and practitioners may not always use it to guide decisions (Fabian et al., 2019; Knight et al., 2008), instead relying on experience (Cook, Hockings & Carter, 2010) or even anecdotes and myths (Sutherland et al., 2004). Smith, Meredith and Sutherland (2021) compiled 129 actions for amphibian conservation based upon 430 studies worldwide (www.conservationevidence.com), of which 42 have proven some conservation benefit, eight demonstrate to be ineffective or harmful, 18 show a trade-off between benefit and harms, and in 61 the effectiveness is still unknown or there is no evidence found of assessed. Fifty-four actions focused on reducing the impact of anthropogenic landscape transformation, 20 focused on species management, and 35 focused on ecosystem protection and management. Three actions focused on education and awareness, while others focused on the legal protection of species, or livelihood and economic incentives such as engaging landowners and other volunteers to manage land for amphibian protection or pay farmers

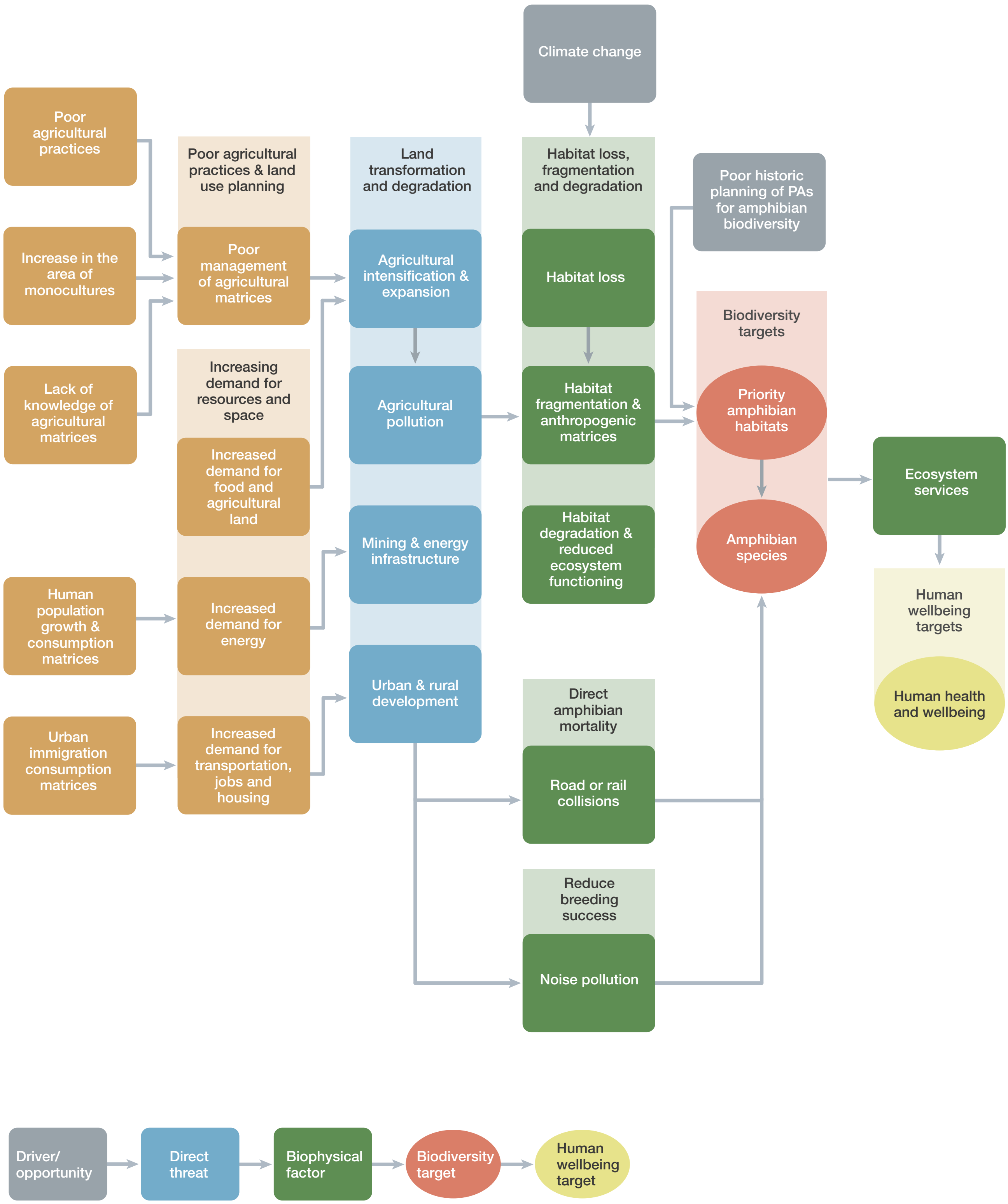


Figure 5.4: This Situation Model shows the key issues relevant to integrating habitat protection and management for amphibians into strategic planning. The model is a visual map of the observed and presumed causal relationships in the context of habitat protection and management and the factors influencing direct and indirect threats and those affecting conservation targets. Such planning allows for identification of key points for interventions to address threats and develop well-informed strategies. It was developed using the Conservation Standards approach to guide strategic planning to address contributing factors influencing direct and indirect threats to amphibian conservation targets. Source: Developed by Claire Relton (Endangered Wildlife Trust, South Africa) using Miradi software.

to cover costs of conservation measures (Smith et al., 2021). Interventions that have been reported in the literature are not always comparable for various reasons: lack of standardisation in the metrics, lack of robust experimental designs such as BACI (Before-After; Control-Impact), or a bias towards better-known biomes and regions (Christie et al., 2020).

This chapter presents suggestions for habitat management and research needed to maintain and improve habitat quality for amphibians. Below we highlight these recommendations (in no particular order, as a prioritisation exercise was not carried out), which will also inform a targeted implementer document:

- **Monitoring and evaluation**: to determine the benefits and limitations of conservation interventions it is key to monitor and assess their impact (Darrah et al., 2019; Schmidt, Brenneisen & Zumbach, 2020). Habitat interventions need to consider the requirements of each species (Urbina-Cardona et al., 2015), tolerance to environmental and structural changes in the habitat (Navas & Otani, 2007; Watling & Braga, 2015), historical landscape disturbance (Betts et al., 2019; Marroquín-Páramo et al., 2021), and spatial-temporal scale (Tscharntke et al., 2012).

- **Connectivity**: amphibians benefit from matrices with remnant corridors, water sources (natural and artificial; Mendenhall et al., 2014), and reduced use of agrochemicals. Vegetated riparian areas, as well as agricultural wetlands, are key to facilitating the dispersal of amphibian species and increasing landscape connectivity (Borzée et al., 2018; Ficetola, Padoa-Schioppa & De Bernardi, 2009; Holzer et al., 2017; Luke et al., 2019; Semlitsch & Bodie, 2003). Some countries (e.g. Colombia and Costa Rica) have considered the conservation of riparian vegetation in their public policy. Connectivity, however, is not limited to riparian corridors. There are interventions to mitigate the impact of infrastructure development on amphibians and their habitats that focus on habitat connectivity, such as the installation of wildlife underpasses and culverts (Beier et al., 2008), rows of stumps or branches to reduce erosion and manage sediments (Goosem et al., 2010) and through the protection and restoration of sensitive habitats (Mitchell, Breisch & Buhlmann, 2006).

- **Globally important sites for amphibians**: Identification and mapping of AZE and KBA amphibian sites in all countries, and their spatial integration with other strategies such as PAs or conservation agreements (*see also* Box 5.2) as a network of conservation areas, are critical steps in the process of safeguarding the amphibian species occurring within these sites as they in turn allow for more informed decision-making and better engagement with key stakeholders in or near these sites (*see example in* Box 5.3).

- **Sustainable and regenerative agricultural practices**: agroecology provides the ecological basis for biodiversity conservation from agriculture, promoting, from the self-sufficiency principle, natural resource renewal, natural biological control, provision of ecosystem services, and crop rotation (Altieri & Nicholls, 2000; Melo et al., 2013). Embracing beneficial land-use practices, such as mixed forest plantations (López-Bedoya et al., 2022), traditional farming, sacred forest sites, and incorporating indigenous knowledge into collaborative approaches is key to strengthening conservation effectiveness (Cocks, 2006; Oscarson & Calhoun, 2007).

- **Stakeholder agreements**: habitat protection based on collaboration between landowners and communities, while allowing productive land use with regular monitoring, is effective in both conserving habitat and restoring degraded ecosystems (Charles, 2021; South African National Biodiversity Institute (SANBI) and Wildlands Conservation Trust, 2015). Such approaches are cost-effective and rely on landowner engagement, often resulting in landscape-level protection and improved habitat management (South African National Biodiversity Institute (SANBI), 2015).

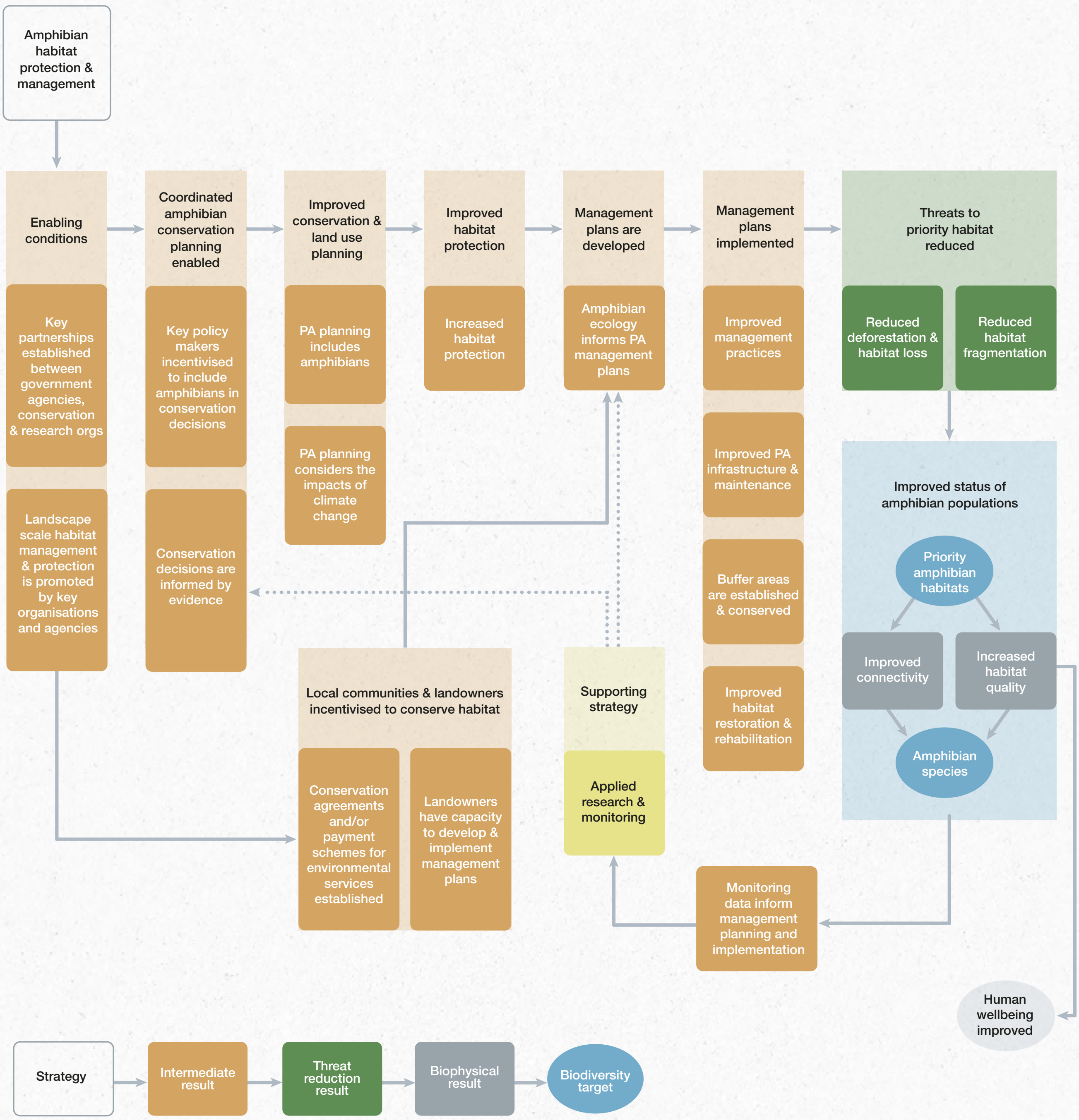
- **Voluntary biodiversity offsets**: “Biodiversity offsets are measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development and persisting after appropriate avoidance, minimisation and restoration measures have been taken" (IFC, 2012). Biodiversity offsets are being adopted across international lending, corporate business, national policy, and voluntary programmes (Gelcich et al., 2017). The IFC determines the need for critical habitat conservation through evaluating specific habitat attributes to conserve a prioritised restricted-range species, and then demonstrating a positive net gain from a monitoring system. Recently, offsets projects are prioritising amphibian species to assess, conserve and monitor their habitat (Sangermano et al., 2015; World Bank, 2019); so there are still no robust results on the effect of conservation actions on the populations of prioritised amphibian species. There are, however, important ethical considerations (Karlsson & Edvardsson Björnberg, 2021), risks (Carreras Gamarra, Lassoie & Milder, 2018), limitations, and evidence gaps (Gardner et al., 2013; zu Ermgassen et al., 2019) associated with biodiversity offsets, so thought needs to be given to these aspects in any proposed offset project.

» **Higher-level interventions**: certain interventions to support the protection of remaining natural habitats need to be at the policy level, although many can be integrated locally. These can include safeguarding KBAs and AZEs, ending subsidies for damaging agricultural practices, reducing monoculture expansion (e.g. soy, rice, oil palm, etc.3), allocating resources to less environmentally damaging alternative land-uses, halting rainforest conversion (McAlpine et al., 2009), and demand-side mitigation measures (Bajželj et al., 2014), such as promoting dietary shifts, waste reduction (Foley et al., 2011) and ecological restoration of land illegally appropriated from fires (Driscoll et al., 2021). Reproductive health and empowering women is a cross-sectoral approach that can be both national policy-level and locally scaled, led by diverse agents, and linking reproductive health, education, sustainable development, community organisation, and habitat conservation. Although still relatively few in number, cross-sectoral initiatives are key in the context of the SDGs given their aim to improve both planetary and human well-being (Mayhew et al., 2020). The Conservations Measures Partnership (www.conservationmeasures.org) has developed theories of change setting out how integrating reproductive health interventions can lead to greater conservation outcomes. The IUCN SSC CEESP Biodiversity & Family Planning Task Force is developing training for IUCN members explaining why removing barriers to family planning is relevant, including for species conservation, and how cross-sectoral projects can be designed and implemented furthering human and environmental health outcomes. These and other developments highlight a current rapid increase in the focus on more holistic cross-sectoral programmes benefiting human and environmental health, which could provide opportunities for amphibian conservation.

» **Rehabilitation of degraded habitat and creation of artificial habitat**: with over 3000 species, including a significant number of threatened species, benefiting from artificial habitats (Figure 5.3), the creation or restoration of habitats, such as ponds and seasonal wetlands, is an important tool for enhancing amphibian biodiversity (Ruhí et al., 2012; Scott, Metts & Whitfield Gibbons, 2008; Simon et al., 2009) as well as protecting threatened species (Beranek, Clulow & Mahoney, 2020). For example, an evaluation of 20 years of monitoring data using occupancy modelling by Moor et al. (2022) found that a landscape-level pond-construction programme in the Swiss lowlands halted and even reversed declines of amphibian species, including threatened species across five regions, demonstrating that relatively simple but landscape-scale and persistent conservation action can benefit amphibians despite pressures from other stressors in human-dominated landscapes. Such interventions need to consider characteristics including age, vegetation cover, water quality of the created habitats (Briggs, 2010; Stumpel & van der Voet, 1998), as well as the habitat requirements for target species, ecological connectivity and ideally be implemented at the landscape level to ensure viable populations (Petranka & Holbrook, 2006; Rannap, Lõhmus & Briggs, 2009).

Box 5.1: Theory of Change models

Using a Theory of Change model (also known as a results chain) (Box Figure 5.1) can be useful to illustrate how interventions linked to habitat protection and management lead to improved status for amphibians and their habitats. This approach supports project planning and monitoring, mapping the pathways to achieving conservation goals, identification of activities and development of indicators to measure outcomes in response to interventions. This results chain was developed using the Conservation Standards approach illustrating the theory of change for habitat protection and management as a strategy for reducing threats in response to actions for achieving biodiversity targets (in this case, improved status of amphibian populations).



Box Figure 5.1: Example Theory of Change model. Source: Figure developed by Claire Relton (Endangered Wildlife Trust, South Africa) using Miradi software.

Box 5.2: Case study – conservation agreements

The Wildlife Conservation Society has developed conservation agreements with private landowners and ethnic communities in areas surrounding four PAs (Farallones NP, Florencia Forest NP, Chingaza NP, and Tatama NP) with a high diversity of threatened species in Colombia (Saboyá-Acosta & Urbina-Cardona, 2022). Under these conservation agreements, the owner of each property or community defines the area that will be left for preservation and implementation of management actions (exclusion of livestock or crop areas, maintenance of riparian vegetation, ecological restoration, trafficking reduction, participatory greenhouses, technical advice for the implementation of silvopastoral systems, the establishment of trails for ecotourism and eradication of illicit crops; World Conservation Society, 2020).

Successful agreements have been measured in habitat recovery through freeing up areas for active restoration and reducing intervention for agricultural or livestock uses. To date, 10 agreements are covering 630.96 hectares in conservation agreements in three protected areas and their surroundings: Five agreements in Farallones NP (237.26 hectares and 16 threatened species), three in Selva de Florencia NP (268, 6 hectares and 13 threatened species), and two in Chingaza NP (125.1 hectares and four threatened species). Conservation agreements are being developed with ethnic communities for species in a critical state of threat such as *Oophaga histrionica*, which is being worked with Embera chami reservation, area of influence of Tatama NP. Some of the threatened species benefiting from these agreements are *Oophaga histrionica, Oophaga anchicayensis, Atelopus lozanoi,* and *Andinobates daleswansoni.*

Box 5.3: Case study – KBAs and local human communities

Key Biodiversity Areas (KBAs) are often situated near impoverished communities that depend on the natural resources from within the site for their livelihoods. The Mount Nimba Strict Nature Reserve on the borders of Guinea, Liberia, and Côte d'Ivoire offers an important case study for conservation prioritisation. Covering 17,540 ha, the site is an AZE that contains the entire known populations of Hyperolius nimbae and Nimbaphrynoides occidentalis. In addition to a wealth of other biodiversity, the Mount Nimba range contains valuable minerals and dense forests. These resources have attracted mining and logging companies but are also vital to the livelihoods of local communities. Recognising the increased pressure on Mount Nimba from unsustainable resource extraction, the Critical Ecosystem Partnership Fund funded a project Strengthening capacity of local communities to sustainably manage Mount Nimba's natural resources, which was completed in 2018. Local communities around Mount Nimba received training in improved gardening and livestock farming practices, sustainable resource use, as well as project and financial management, improving their farming yields and subsequently, their income. As a result, the local communities are less reliant on Mount Nimba's natural resources. Through community empowerment focused on sustainable conservation, this project has improved the likelihood that these forests will persist and improve into the future and support the long-term survival of these amphibians (Birdlife International, 2018; UNESCO, 2018).

Identification of knowledge gaps and research

To improve habitat protection and management effectiveness for amphibians and provide cost-effective interventions in the field, we draw attention to the need to fill the following knowledge gaps (in no particular order of priority as a prioritisation exercise has not been carried out):

- **Integration**: the systematic conservation planning protocol (Margules & Sarkar, 2007), aims to represent all biodiversity (species, ecosystem processes and services and natural vegetation cover) in the smallest possible area and with the smallest possible budget in conservation area networks (Burbano-Girón, et al. 2022). In this sense, to ensure the long-term conservation of amphibians, it is necessary not only to consider their representation in PAs but also to connect them in a network of conservation areas that spatially integrates biodiversity conservation initiatives (e.g. strategies such as PAs, KBAs, AZE and conservation agreements led by some NGOs). However, given that anthropogenic transformation of the landscape and climate change imposes dynamics on amphibian habitat, it is necessary to project these networks of conservation areas into the future under different scenarios of climate change and land use/land cover (LULCC).

- **Modelling**: to refine conservation networks at the local scale, functional connectivity models for amphibian target species should be conducted at an appropriate resolution. Target species can be habitat specialists, ensuring that essential core habitats are conserved, or threatened flagship species that act as an 'umbrella' for protecting multiple species and important habitats.

- **Experimentation:** for these target species, physiological experiments should be carried out to understand their dehydration rates, locomotor performance curves, and critical temperatures, along different types of vegetation cover, to make inferences about their response to climate change and LULCC scenarios.

- **Intervention monitoring:** where interventions are carried out (e.g. ecological restoration, implementation of agrosilvopastoral systems, planting of live fences, creation of ponds, among others), monitoring should be conducted at the demographic level for the target species and at the assemblage level for the facets (taxonomic, functional and phylogenetic) of diversity. It is crucial that the results of this monitoring are compiled in a global database to be able to compare the effectiveness and success of interventions across regions, ecosystems and biotic groups.

- **Scaling:** likewise, at the level of amphibian assemblages, it is necessary to know the scale of effect at which the landscape configuration operates and what is the amount of habitat required to maintain the values of the (taxonomic, functional and phylogenetic) diversity facets within the ranges of a natural reference ecosystem (Watling et al., 2020).

- **Collaborations:** partnerships with social scientists and development agencies should be strengthened to improve the social development aspects that often underlie the success of amphibian conservation interventions and to ensure a holistic, integrated approach to achieving environmental objectives.

Acknowledgements

We thank Andrew Snyder, Diego Alejandro Gomez Hoyos, Frank Lemckert, Ian Little, James Lewis, Jeff Dawson, Luis Fernando Marin, Stefano Doglio and Vincenzo Ferri for giving freely of their time towards the development of this chapter and providing diverse and important insights, expertise and extremely helpful commentary. Claire Relton (Endangered Wildlife Trust, South Africa) created the Situation Model and Theory of Change diagram. We are grateful to Phil Bishop for showing us the way and inspiring us to never give up on amphibian conservation.

References

Acevedo-Charry, O. & Aide, T. M. (2019). Recovery of amphibian, reptile, bird and mammal diversity during secondary forest succession in the tropics. *Oikos*, *128*, 1065–1078. https://doi.org/10.1111/oik.06252

Adger, W. N. (2000). Social and ecological resilience: Are they related? *Progress in Human Geography*, *24*(3), 347–364. https://doi.org/10.1191/030913200701540465

Agudelo-Hz, W. J., Urbina-Cardona, N. & Armenteras-Pascual, D. (2019). Critical shifts on spatial traits and the risk of extinction of Andean anurans: An assessment of the combined effects of climate and land-use change in Colombia. *Perspectives in Ecology and Conservation*, *17*, 206–219. https://doi.org/10.1016/j.pecon.2019.11.002

Aguilar-López, J. L., Ortiz-Lozada, L., Pelayo-Martínez, J., Mota-Vargas, C., Alarcón-Villegas, L. E., & Demeneghi-Calatayud, A. P. (2020). Amphibian and reptile conservation in a privately protected area from a highly transformed region in southern Veracruz, Mexico. *Acta Zoológica Mexicana*, *36*, 1–14. https://doi.org/10.21829/azm.2020.3612164

Albert, J. S., Destouni, G., Duke-Sylvester, S. M., Magurran, A. E., Oberdorff, T., Reis, R. E., … Ripple, W. J. (2020). Scientists' warning to humanity on the freshwater biodiversity crisis. *Ambio*, *50*, 85–94. https://doi.org/10.1007/s13280-020-01318-8

Alignier, A. & Deconchat, M. (2011). Variability of forest edge effect on vegetation implies reconsideration of its assumed hypothetical pattern. *Applied Vegetation Science*, *14*, 67–74. https://doi.org/10.1111/j.1654-

109X.2010.01105.x
Altieri, M. & Nicholls, C. I. (2000). *Agroecología: Teoría y práctica para una agricultura sustentable*. México, DF. México: Programa de las Naciones Unidas para el Medio Ambiente.
Álvarez-Grzybowska, E., Urbina-Cardona, N., Córdova-Tapia, F., & García, A. (2020). Amphibian communities in two contrasting ecosystems: functional diversity and environmental filters. *Biodiversity and Conservation*, *29*, 2457–2485. https://doi.org/10.1007/s10531-020-01984-w
Andrews, K. M., Gibbons, J. W., Jochimsen, D. M., & Mitchell, J. (2008). Ecological effects of roads on amphibians and reptiles: a literature review. *Herpetological Conservation*, *3*, 121–143.
Aritola, J. F., Walworth, J. L., Musil, S. A., & Crimmins, M. A. (2019). Soil and land pollution. In M. L. Brusseau, I. L. Pepper, & C. P. Gerba (Eds.) *Environmental and pollution science* (pp. 219–235). Elsevier.
Arroyo-Rodríguez, V., Fahrig, L., Tabarelli, M., Watling, J. I., Tischendorf, L., Benchimol, M., … Tscharntke, T. (2020). Designing optimal human-modified landscapes for forest biodiversity conservation. *Ecology Letters*, *23*(9), 1404–1420. https://doi.org/10.1111/ele.13535
Arroyo-Rodríguez, V., Melo, F. P. L., Martínez-Ramos, M., Bongers, F., Chazdon, R. L., Meave, J. A., …, Tabarelli, M. (2017). Multiple successional pathways in human-modified tropical landscapes: New insights from forest succession, forest fragmentation and landscape ecology research. *Biological Reviews*, *92*(1), 326-340. https://doi.org/10.1111/brv.12231
Bajželj, B., Richards, K. S., Allwood, J. M., Smith, P., Dennis, J. S., Curmi, E., & Gilligan, C. A. (2014). Importance of food-demand management for climate mitigation. *Nature Climate Change*, *4*, 924–929. https://doi.org/10.1038/nclimate2353
Balázsi, Á., Riechers, M., Hartel, T., Leventon, J., & Fischer, J. (2019). The impacts of social-ecological system change on human-nature connectedness: A case study from Transylvania, Romania. *Land Use Policy*, *89*, 104232. https://doi.org/ 10.1016/j.landusepol.2019.104232
Baldwin, R. F., Calhoun, A. J. K. & deMaynadier, P. G. (2006). Conservation planning for amphibian species with complex habitat requirements: A case study using movements and habitat selection of the wood frog *Rana sylvatica*. *Journal of Herpetology*, *40*(4), 442–453. https://doi.org/10.1670/0022-1511(2006)40[442:CPFASW]2.0.CO;2
Barendse, J., Roux, D., Currie, B., Wilson, N., & Fabricius, C. (2016). A broader view of stewardship to achieve conservation and sustainability goals in South Africa. *South African Journal of Science*, *112*(5/6), 1–15. https://doi.org/10.17159/sajs.2016/20150359
Barnes, M. D., Craigie, I. D., Harrison, L. B., Geldmann, J., Collen, B., Whitmee, S., … Woodley, S. (2016). Wildlife population trends in protected areas predicted by national socio-economic metrics and body size. *Nature Communications*, *7*, 1–9. https://doi.org/10.1038/ncomms12747
Basham, E. W., González del Pliego, P., Acosta-Galvis, A. R., Woodcock, P., Medina Uribe, C. A., Haugaasen, T., … Edwards, D. P. (2016). Quantifying carbon and amphibian co-benefits from secondary forest regeneration in the Tropical Andes. *Animal Conservation*, *19*, 548–560. https://doi.org/10.1111/acv.12276
Beier, P., Majka, D., Newell, S., & Garding, E. (2008). *Best management practices for wildlife corridors*. Northern Arizona University, *1*(3), 1–14.
Bennett, A. F., Radford, J. Q. & Haslem, A. (2006). Properties of land mosaics: Implications for nature conservation in agricultural environments. *Biological Conservation*, *133*(2), 250–264. https://doi.org/10.1016/j.biocon.2006.06.008
Beranek, C. T., Clulow, J. & Mahoney, M. (2020). Wetland restoration for the threatened green and golden bell frog (*Litoria aurea*): development of a breeding habitat designed to passively manage chytrid-induced amphibian disease and exotic fish. *Natural Areas Journal*, *40*(4), 362–374. https://doi.org/10.3375/043.040.0409
Berkes, F. (2009). Community conserved areas: Policy issues in historic and contemporary context. *Conservation Letters*, *2*, 19–24. https://doi.org/10.1111/j.1755-263x.2008.00040.x
Betts, M. G., Wolf, C., Pfeifer, M., Banks-Leite, C., Arroyo-Rodríguez, V., Ribeiro, D. B., … Ewers, R. M. (2019). Extinction filters mediate the global effects of habitat fragmentation on animals. *Science*, *366*(6470), 1236–1239. https://doi.org/10.1126/science.aax9387
Birdlife International. (2018). *Fighting rural poverty: Community empowered to conserve Guinean mountain reserve*. https://zeroextinction.org Retrieved February 16, 2023 from https://zeroextinction.org/stories/fighting-rural-poverty-community-empowered-to-conserve-guinean-mountain-reserve/
Bombi, P., D'Amen, M., Salvi, D., Bologna, M. A., Marcone, F., Maggio, C., & Canu, A. (2012). Amphibians conservation in Italy: The contribution of the WWF Oases network. *Italian Journal of Zoology*, *79*(2), 287–295. https://doi.org/10.1080/11250003.2011.623722
Borzée, A., Heo, K. & Jang, Y. (2018). Relationship between agro-environmental variables and breeding Hylids in rice paddies. *Scientific Reports*, *8*, 1–13. https://doi.org/10.1038/s41598-018-26222-w
Bosch, J., Rincón, P. A., Boyero, L., & Martínez-Solano, I. (2006). Effects of introduced salmonids on a montane population of Iberian frogs. *Conservation Biology*, *20*(1), 180–189. https://doi.org/10.1111/j.1523-1739.2005.00296.x
Bowen, M. E., McAlpine, C. A., House, A. P. N., & Smith, G. C. (2007). Regrowth forests on abandoned agricultural land: A review of their habitat values for recovering forest fauna. *Biological Conservation*, 140(3–4), 273–296. https://doi.org/10.1016/j.biocon.2007.08.012
Branton, M. A. & Richardson, J. S. (2014). A test of the umbrella species approach in restored floodplain ponds. *Journal of Applied Ecology*, *51*, 776–785. https://doi.org/10.1111/1365-2664.12248
Briggs, L. (2010). Creation of temporary ponds for amphibians in northern and central Europe. *Freshwater Forum*, *17*, 63–70. http://hdl.handle.net/1834/22244
Broadbent, E. N., Asner, G. P., Keller, M., Knapp, D. E., Oliveira, P. J. C., & Silva, J. N. (2008). Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. *Biological Conservation*, *141*(7), 1745–1757. https://doi.org/10.1016/j.biocon.2008.04.024
Brown, S. & Lugo, A. E. (1990). Tropical secondary forests. *Journal of Tropical Ecology*, *6*, 1–32. https://doi.org/10.1017/S0266467400003989
Brüning, L. Z., Krieger, M., Meneses-Pelayo, E., Eisenhauer, N., Ramirez Pinilla, M. P., Reu, B., & Ernst, R. (2018). Land-use heterogeneity by small-scale agriculture promotes amphibian diversity in montane agroforestry systems of northeast Colombia. *Agriculture, Ecosystems and Environment*, *264*, 15–23. https://doi.org/10.1016/j.agee.2018.05.011
Bucciarelli, G. M., Blaustein, A. R., Garcia, T. S., & Kats, L. B. (2014). Invasion complexities: The diverse impacts of nonnative species on amphibians. *Copeia*, *4*, 611–632. https://doi.org/10.1643/OT-14-014
Buck, J. C., Scheessele, E. A., Relyea, R. A., & Blaustein, A. R. (2012). The effects of multiple stressors on wetland communities: pesticides, pathogens and competing amphibians. *Freshwater Biology*, *57,* 61–73. https://doi.org/10.1111/j.1365-2427.2011.02695.x
Burbano-Girón, J., Jantke, K., Molina-Berbeo, M. A., Buriticá-Mejía, N., Urbina-Cardona, J. N., Sánchez-Clavijo, L. M., & Etter, A. (2022). An assessment of spatial conservation priorities for biodiversity attributes: Composition, structure, and function of Neotropical biodiversity. *Biological Conservation, 265*, 109421. https://doi.org/10.1016/j.biocon.2021.109421
Burton, T. M. & Likens, G. E. (1975). Salamander populations and biomass in the Hubbard Brook Experimental Forest, New Hampshire. *Copeia*, *3*, 541–546.
Butchart, S. H. M., Clarke, M., Smith, R. J., Sykes, R. E., Scharlemann, J. P. W., Harfoot, M., … Burgess, N. D. (2015). Shortfalls and solutions for meeting national and global conservation area targets. *Conservation Letters*, *8*(5), 329–337. https://doi.org/10.1111/conl.12158
Button, S. & Borzée, A. (2021). An integrative synthesis to global amphibian conservation priorities. *Global Change Biology*, *27*(19), 4516–4529. https://doi.org/10.1111/gcb.15734
Cabrera-Guzmán, E. & Reynoso, V. H. (2012). Amphibian and reptile communities of rainforest fragments: Minimum patch size to support high richness and abundance. *Biodiversity and Conservation*, *21*(12), 3243–3265. https://doi.org/10.1007/s10531-012-0312-4
Cáceres-Andrade, S. P. & Urbina-Cardona, J. N. (2009). Ensamblajes de anuros de sistemas productivos y bosques en el piedemonte llanero, departamento del Meta, Colombia. *Caldasia*, *31*(1), 175–194.
Camacho-Rozo, C. P. & Urbina-Cardona, N. (2021). Tadpoles inhabiting natural and anthropogenic temporary water bodies: which are the environmental factors that affect the diversity of the assemblages?

Frontiers in Environmental Science, *9*, 1–17. https://doi.org/10.3389/fenvs.2021.667448
Caro, T., Engilis, A., Fitzherbert, E., & Gardner, T. (2004). Preliminary assessment of the flagship species concept at a small scale. *Animal Conservation*, *7*(1), 63–70. https://doi.org/10.1017/S136794300300115X
Carreras Gamarra, M. J., Lassoie, J. P. & Milder, J. (2018). Accounting for no net loss: A critical assessment of biodiversity offsetting metrics and methods. *Journal of Environmental Management*, *220*, 36–43. https://doi.org/10.1016/j.jenvman.2018.05.008
Carvalho, S. B., Brito, J. C., Crespo, E. J., & Possingham, H. P. (2010). From climate change predictions to actions - conserving vulnerable animal groups in hotspots at a regional scale. *Global Change Biology*, *16*(12), 3257–3270. https://doi.org/10.1111/j.1365-2486.2010.02212.x
Catenazzi, A. (2015). State of the world's amphibians. *Annual Review of Environment and Resources*, *40*, 91–119. https://doi.org/10.1146/annurev-environ-102014-021358
Charles, A. (2021). *Communities, conservation and livelihoods*. Gland, Switzerland: IUCN; Halifax, Canada: Community Conservation Research Network, 2021. https://doi.org/10.2305/iucn.ch.2021.01.en
Chazdon, R. L. (2003). Tropical forest recovery: Legacies of human impact and natural disturbances. *Perspectives in Plant Ecology, Evolution and Systematics*, *6*(1–2), 51–71. https://doi.org/10.1078/1433-8319-00042
Chokkalingam, U. & De Jong, W. (2001). Secondary forest: A working definition and typology. *International Forestry Review*, *3*(1), 19–26. https://hdl.handle.net/10568/18318
Christie, A. P., Amano, T., Martin, P. A., Petrovan, S. O., Shackelford, G. E., Simmons, B. I., … Sutherland, W. J. (2020). Poor availability of context-specific evidence hampers decision-making in conservation. *Biological Conservation*, *248*, 108666. https://doi.org/10.1016/j.biocon.2020.108666
Cline, B. B. & Hunter, M. L. (2016). Movement in the matrix: Substrates and distance-to-forest edge affect postmetamorphic movements of a forest amphibian. *Ecosphere*, *7*(2), 1–23. https://doi.org/10.1002/ecs2.1202
Cochrane, M. A. & Barber, C. P. (2009). Climate change, human land use and future fires in the Amazon. *Global Change Biology*, *15*, 601–612. https://doi.org/10.1111/j.1365-2486.2008.01786.x
Cocks, M. (2006). Biocultural diversity: Moving beyond the realm of "indigenous" and "local" people. *Human Ecology*, *34*(2), 185–200. https://doi.org/10.1007/s10745-006-9013-5
Collins, S. J. & Fahrig, L. (2017). Responses of anurans to composition and configuration of agricultural landscapes. *Agriculture, Ecosystems and Environment*, 239, 399–409. https://doi.org/10.1016/j.agee.2016.12.038
Convention on Biological Diversity. (2020). Aichi Biodiversity Targets. Retrieved October 1, 2020, from https://www.cbd.int/sp/targets/
Convention on Biological Diversity. (2022) Monitoring framework for the Kunming-Montreal global biodiversity framework. Montréal: CBD. Available from: https://www.cbd.int/doc/c/179e/aecb/592f67904bf07dca7d0971da/cop-15-l-26-en.pdf
Cook, C. N., Hockings, M. & Carter, R. W. (2010). Conservation in the dark? The information used to support management decisions. *Frontiers in Ecology and the Environment*, *8*(4), 181–188. https://doi.org/10.1890/090020
Cordero-Rivera, A., Velo-Antón, G. & Galán, P. (2007). Ecology of amphibians in small coastal Holocene islands: local adaptations and the effect of exotic tree plantations. *Munibe*, *25*, 94–103.
Cortés-Gómez, A. M., Castro-Herrera, F. & Urbina-Cardona, J. N. (2013). Small changes in vegetation structure create great changes in amphibian ensembles in the Colombian Pacific rainforest. *Tropical Conservation Science*, *6*(6), 749–769. https://doi.org/10.1177/194008291300600604
Cortés-Gómez, A. M., Ramirez, M. P. & Urbina-Cardona, N. (2015). Protocolo para medición de rasgos funcionales en anfibios. In B. Salgado Negret (Ed.), *La ecología funcional como aproximación al estudio, manejo y conservación de la biodiversidad: protocolos y aplicaciones* (pp. 126–179). Bogotá, Colombia: Instituto de Investigación de Recursos Biológicos Alexander von Humboldt.
Cortés-Gomez, A. M., Ruiz-Agudelo, C. A., Valencia-Aguilar, A., & Ladle, R. J. (2015). Ecological functions of Neotropical amphibians and reptiles: A review. *Universitas Scientiarum*, *20*(2), 229–245. https://doi.org/10.11144/Javeriana.SC20-2.efna
Cotten, T. B., Kwiatkowski, M. A., Saenz, D., & Collyer, M. (2012). Effects of an invasive plant, Chinese tallow (*Triadica sebifera*), on development and survival of anuran larvae. *Journal of Herpetology*, *46*(2), 186–193. https://doi.org/10.1670/10-311
Cox, J. L., McKinney, M. L., Fitzpatrick, B. M., Leppanen, C., & Nichols, R. J. (2022). Coniferous conservation supporting a plethora of plethodontids: Implications of conserving eastern hemlock (*Tsuga canadensis*) on southern Appalachian montane salamanders. *Forest Ecology and Management*, *508*, 120010. https://doi.org/10.1016/j.foreco.2022.120010
Crnobrnja-Isailović, J., Jovanović, B., Ilić, M., Ćorović, J., Čubrić, T., Stojadinović, D., & Ćosić, N. (2021). Small hydropower plants' proliferation would negatively affect local herpetofauna. *Frontiers in Ecology and Evolution*, *9*: 610325. https://doi.org/10.3389/fevo.2021.610325
Crowther, T. W., Glick, H. B., Covey, K. R., Bettigole, C., Maynard, D. S., Thomas, S. M., … Bradford, M. A. (2015). Mapping tree density at a global scale. *Nature*, 525(7568), 201–205. https://doi.org/10.1038/nature14967
Crump, M. L. (2015). Anuran reproductive modes: Evolving perspectives. *Journal of Herpetology*, *49*(1), 1–16. https://doi.org/10.1670/14-097
Cushman, S. A. (2006). Effects of habitat loss and fragmentation on amphibians: A review and prospectus. *Biological Conservation*, *128*(2), 231–240. https://doi.org/10.1016/j.biocon.2005.09.031
D'Amico, M., Périquet, S., Román, J., & Revilla, E. (2016). Road avoidance responses determine the impact of heterogeneous road networks at a regional scale. *Journal of Applied Ecology*, *53*(1), 181–190. https://doi.org/10.1111/1365-2664.12572
Dale, V. H., Pearson, S. M., Offerman, H. L., & O'Neill, R. V. (1994). Relating patterns of land-use change to faunal biodiversity in the Central Amazon. *Conservation Biology*, *8*(4), 1027–1036. https://doi.org/10.1046/j.1523-1739.1994.08041027.x
Danielsen, F., Beukema, H., Burgess, N., Parish, F., Brühl, C., Donald, P., … Fitzherbert, E. (2009). Biofuel plantations on forested lands: Double jeopardy for biodiversity and climate. *IOP Conference Series: Earth and Environmental Science*, *6*(24), 242014. https://doi.org/10.1088/1755-1307/6/24/242014
Dare, G. C., Murray, R. G., Courcelles, D. M. M., Malt, J. M., & Palen, W. J. (2020). Run-of-river dams as a barrier to the movement of a stream-dwelling amphibian. *Ecosphere*, *11*(8), e03207. https://doi.org/10.1002/ecs2.3207
Darrah, S. E., Shennan-Farpón, Y., Loh, J., Davidson, N. C., Finlayson, C. M., Gardner, R. C., & Walpole, M. J. (2019). Improvements to the Wetland Extent Trends (WET) index as a tool for monitoring natural and human-made wetlands. *Ecological Indicators*, *99*, 294–298. https://doi.org/10.1016/j.ecolind.2018.12.032
Davies, S. J., Jordaan, M. S., Karsten, M., Terblanche, J. S., Turner, A. A., van Wilgen, N. J., … Measey, J. (2020). Experience and lessons from alien and invasive animal control projects in South Africa. In B. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (pp. 629–663). Switzerland: Springer.
Dayrell, J. S., Magnusson, W. E., Bobrowiec, P. E. D., & Lima, A. P. (2021). Impacts of an Amazonian hydroelectric dam on frog assemblages. *PLoS ONE*, *16*(6), e0244580. https://doi.org/10.1371/journal.pone.0244580
De Lima, M. G. & Gascon, C. (1999). The conservation value of linear forest remnants in central Amazonia. *Biological Conservation*, *91*, 241–247. https://doi.org/10.1016/S0006-3207(99)00084-1
de Melo, L. S. O., Gonçalves-Souza, T., Garey, M. V., & de Cerqueira, D. (2017). Tadpole species richness within lentic and lotic microhabitats: An interactive influence of environmental and spatial factors. *Herpetological Journal*, *27*(4), 339–345.
de Oliveira, S. L., Silva, D. de M., Bastos, R. P., & de Morais, A. R. (2015). Anfíbios anuros nos covais do município de Jataí, Estado de Goiás. *Geoambiente On-line, 24.* https://doi.org/10.5216/revgeoamb.v0i24.34139
De Sy, V., Herold, M., Achard, F., Beuchle, R., Clevers, J. G. P. W., Lindquist, E., & Verchot, L. (2015). Land use patterns and related carbon losses following deforestation in South America. *Environmental Research Letters*, *10*, 124004. https://doi.org/10.1088/1748-9326/10/12/124004
Demaynadier, P. G. & Hunter, M. L. (1998). Effects of silvicultural edges on the distribution and abundance of amphibians in Maine. *Conservation Biology*, *12*(2), 340–352. https://doi.org/10.1046/j.1523-1739.1998.96412.x

Didham, R. K., Kapos, V. & Ewers, R. M. (2012). Rethinking the conceptual foundations of habitat fragmentation research. *Oikos*, *121*(2), 161–170. https://doi.org/10.1111/j.1600-0706.2011.20273.x

Dixo, M. & Metzger, J. P. (2010). The matrix-tolerance hypothesis: an empirical test with frogs in the Atlantic Forest. *Biodiversity and Conservation*, *19*(11), 3059–3071. https://doi.org/10.1007/s10531-010-9878-x

Driscoll, D. A., Armenteras, D., Bennett, A. F., Brotons, L., Clarke, M. F., Doherty, T. S., … Wevill, T. (2021). How fire interacts with habitat loss and fragmentation. *Biological Reviews*, *96*(3), 976–998. https://doi.org 10.1111/brv.12687

Driscoll, D. A., Banks, S. C., Barton, P. S., Lindenmayer, D. B., & Smith, A. L. (2013). Conceptual domain of the matrix in fragmented landscapes. *Trends in Ecology and Evolution*, *28*(10), 605–613. https://doi.org/10.1016/j.tree.2013.06.010

Duarte-Ballesteros, L., Urbina-Cardona, J. N. & Saboyá-Acosta, L. P. (2021). Anuran assemblages and spatial heterogeneity in a paramo ecosystem of Colombia. *Caldasia*, *43*(1), 126–137. https://doi.org/10.15446/caldasia.v43n1.84860

Dudley, N. (2008). *Guidelines for applying protected area management categories*. Gland, Switzerland: IUCN. WITH Stolton, S., Shadie, P., & Dudley, N. (2013). IUCN WCPA Best practice guidance on recognising protected areas and assigning management categories and governance types. Best practice protected area guidelines series No. 21, Gland, Switzerland: IUCN. https://portals.iucn.org/library/node/30018

Dudley, N. & Alexander, S. (2017). Agriculture and biodiversity: a review. *Biodiversity*, *18*(2–3), 45–49. https://doi.org/10.1080/14888386.2017.1351892

Duellman, W. E. (1999). *Patterns of distribution of amphibians: a global perspective*. Baltimore and London: The Johns Hopkins University Press.

Dunn, R. R. (2004). Recovery of faunal communities during tropical forest regeneration. *Conservation Biology*, *18*(2), 302–309. https://doi.org/10.1111/j.1523-1739.2004.00151.x

Echeverría-Londoño, S., Newbold, T., Hudson, L. N., Contu, S., Hill, S. L. L., Lysenko, I., … Purvis, A. (2016). Modelling and projecting the response of local assemblage composition to land use change across Colombia. *Diversity and Distributions*, *22*, 1099–1111. https://doi.org/10.1111/ddi.12478

Ellis, E. C. (2015). Ecology in an anthropogenic biosphere. *Ecological Monographs*, *85*(3), 287–331. https://doi.org/10.1890/14-2274.1

Ernst, R., Linsenmair, K. E. & Rödel, M. O. (2006). Diversity erosion beyond the species level: Dramatic loss of functional diversity after selective logging in two tropical amphibian communities. *Biological Conservation*, *133*(2), 143–155. https://doi.org/10.1016/j.biocon.2006.05.028

Fabian, Y., Bollmann, K., Brang, P., Heiri, C., Olschewski, R., Rigling, A., … Holderegger, R. (2019). How to close the science-practice gap in nature conservation? Information sources used by practitioners. *Biological Conservation*, *235*, 93–101. https://doi.org/10.1016/j.biocon.2019.04.011

Fahrig, L. (2017). Ecological responses to habitat fragmentation per se. *Annual Review of Ecology, Evolution, and Systematics*, *48*, 1–23. https://doi.org/10.1146/annurev-ecolsys-110316-022612

Fahrig, L., Arroyo-Rodríguez, V., Bennett, J. R., Boucher-Lalonde, V., Cazetta, E., Currie, D. J., … Watling, J. I. (2019). Is habitat fragmentation bad for biodiversity? *Biological Conservation*, *230*, 179–186. https://doi.org/10.1016/j.biocon.2018.12.026

Falaschi, M., Melotto, A., Manenti, R., & Ficetola, G. F. (2020). Invasive species and amphibian conservation. *Herpetologica*, *76*(2), 216–227. https://doi.org/10.1655/0018-0831-76.2.216

Farallo, V. R. & Miles, D. B. (2016). The importance of microhabitat: A comparison of two microendemic species of *Plethodon* to the widespread *P. cinereus*. *Copeia*, *104*(1), 67–77. https://doi.org/10.1643/CE-14-219

Faruk, A., Belabut, D., Ahmad, N., Knell, R. J., & Garner, T. W. J. (2013). Effects of oil-palm plantations on diversity of tropical anurans. *Conservation Biology*, *27*(3), 615–624. https://doi.org/10.1111/cobi.12062

Ferrante, L., Baccaro, F. B., Ferreira, E. B., Sampaio, M. F. de O., Santos, T., Justino, R. C., & Angulo, A. (2017). The matrix effect: How agricultural matrices shape forest fragment structure and amphibian composition. *Journal of Biogeography*, *44*(8), 1911–1922. https://doi.org/10.1111/jbi.12951

Ficetola, G. F., Padoa-Schioppa, E. & De Bernardi, F. (2009). Influence of landscape elements in riparian buffers on the conservation of semiaquatic amphibians. *Conservation Biology*, *23*(1), 114–123. https://doi.org/10.1111/j.1523-1739.2008.01081.x

Ficetola, G. F., Siesa, M. E., Manenti, R., Bottoni, L., De Bernardi, F., & Padoa-Schioppa, E. (2011). Early assessment of the impact of alien species: Differential consequences of an invasive crayfish on adult and larval amphibians. *Diversity and Distributions*, *17*(6), 1141–1151. https://doi.org/10.1111/j.1472-4642.2011.00797.x

Fletcher, R. J., Didham, R. K., Banks-Leite, C., Barlow, J., Ewers, R. M., Rosindell, J., … Haddad, N. M. (2018). Is habitat fragmentation good for biodiversity? *Biological Conservation*, *226*, 9–15. https://doi.org/10.1016/j.biocon.2018.07.022

Fog, K. (1997). A survey of the results of pond projects for rare amphibians in Denmark. *Memoranda Societatis pro Fauna et Flora Fennica*, *73*, 91–100.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., … Snyder, P. K. (2005). Global consequences of land use. *Science*, *309*(5734), 570–574. https://doi.org/10.1126/science.1111772

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., … Zaks, D. P. M. (2011). Solutions for a cultivated planet. *Nature*, *478*(7369), 337–342. https://doi.org/10.1038/nature10452

Ford, J., Hunt, D. A. G. A., Haines, G. E., Lewis, M., Lewis, Y., & Green, D. M. (2020). Adrift on a sea of troubles: can amphibians survive in a human-dominated world? *Herpetologica*, *76*(2), 251–256. https://doi.org/10.1655/0018-0831-76.2.251

Fujioka, M. & Lane, S. J. (1997). The impact of changing irrigation practices in rice fields on frog populations of the Kanto Plain, central Japan. *Ecological Research*, *12*(1), 101–108. https://doi.org/10.1007/BF02523615

Galindo-Uribe, D. M., Hoyos-Hoyos, J. M., Isaacs-Cubides, P., Corral-Gómez, N., & Urbina-Cardona, N. (2022). Classification and sensitivity of taxonomic and functional diversity indices of anurans in the Andean coffee cultural landscape. *Ecological Indicators*, *136*, 108650. https://doi.org/10.1016/j.ecolind.2022.108650

Gallice, G. R., Larrea-Gallegos, G. & Vázquez-Rowe, I. (2019). The threat of road expansion in the Peruvian Amazon. *Oryx*, *53*(2), 284–292. https://doi.org/10.1017/S0030605317000412

Gallmetzer, N. & Schulze, C. H. (2015). Impact of oil palm agriculture on understory amphibians and reptiles: a Mesoamerican perspective. *Global Ecology and Conservation*, *4*, 95–109. https://doi.org/10.1016/j.gecco.2015.05.008

Gardner, T. A., Barlow, J. & Peres, C. A. (2007). Paradox, presumption and pitfalls in conservation biology: the importance of habitat change for amphibians and reptiles. *Biological Conservation*, *138*(1–2), 166–179. https://doi.org/10.1016/j.biocon.2007.04.017

Gardner, T. A., Ribeiro-Júnior, M. A., Barlow, J., Ávila-Pires, T. C. S., Hoogmoed, M. S., & Peres, C. A. (2007). The value of primary, secondary, and plantation forests for a Neotropical herpetofauna. *Conservation Biology*, *21*(3), 775–787. https://doi.org/10.1111/j.1523-1739.2007.00659.x

Gardner, T. A., Von Hase, A., Brownlie, S., Ekstrom, J. M. M., Pilgrim, J. D., Savy, C. E., … Ten Kate, K. (2013). Biodiversity offsets and the challenge of achieving no net loss. *Conservation Biology*, *27*(6), 1254–1264. https://doi.org/10.1111/cobi.12118

Garibaldi, L. A., Oddi, F. J., Miguez, F. E., Bartomeus, I., Orr, M. C., Jobbágy, E. G., … Zhu, C. D. (2021). Working landscapes need at least 20% native habitat. *Conservation Letters*, *14*(2), e12773. https://doi.org/10.1111/conl.12773

Gascon, C., Lovejoy, T. E., Bierregaard, R. O. J., Malcolm, J. R., Stouffer, P. C., Laurance, W. F., … Broges, S. (1999). Matrix habitat and species persistence in tropical forest remnants. *Biological Conservation*, *91*(2–3), 223–229. https://doi.org/10.1016/S0006-3207(99)00080-4

Gelcich, S., Vargas, C., Carreras, M. J., Castilla, J. C., & Donlan, C. J. (2017). Achieving biodiversity benefits with offsets: research gaps, challenges, and needs. *Ambio*, *46*(2), 184–189. https://doi.org/10.1007/s13280-016-0810-9

Geldmann, J., Manica, A., Burgess, N. D., Coad, L., & Balmford, A. (2019). A global-level assessment of the effectiveness of protected areas at resisting anthropogenic pressures. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(46), 23209–23215. https://doi.org/10.1073/pnas.1908221116

Gilbert-Norton, L., Wilson, R., Stevens, J. R., & Beard, K. H. (2010). A meta-analytic review of corridor effectiveness. *Conservation Biology*, *24*(3), 660–668. https://doi.org/10.1111/j.1523-1739.2010.01450.x

Gillespie, G. R., Howard, S., Stroud, J. T., Ul-Hassanah, A., Campling, M., Lardner, B., … Kusrini, M. (2015). Responses of tropical forest herpetofauna to moderate anthropogenic disturbance and effects of natural habitat variation in Sulawesi, Indonesia. *Biological Conservation*, *192*, 161–173. https://doi.org/10.1016/j.biocon.2015.08.034

Gilroy, J. J., Prescott, G. W., Cardenas, J. S., Castañeda, P. G. D. P., Sánchez, A., Rojas-Murcia, L. E., … Edwards, D. P. (2015). Minimizing the biodiversity impact of Neotropical oil palm development. *Global Change Biology*, *21*(4), 1531–1540. https://doi.org/10.1111/gcb.12696

Godfray, H. C. J., Aveyard, P., Garnett, T., Hall, J. W., Key, T. J., Lorimer, J., … Jebb, S. A. (2018). Meat consumption, health, and the environment. *Science*, *361*(6399). https://doi.org/10.1126/science.aam5324

Goldspiel, H. B., Cohen, J. B., McGee, G. G., & Gibbs, J. P. (2019). Forest land-use history affects outcomes of habitat augmentation for amphibian conservation. *Global Ecology and Conservation*, *19*, e00686. https://doi.org/10.1016/j.gecco.2019.e00686

González-del-Pliego, P., Freckleton, R. P., Edwards, D. P., Koo, M. S., Scheffers, B. R., Pyron, R. A., & Jetz, W. (2019). Phylogenetic and trait-based prediction of extinction risk for data-deficient amphibians. *Current Biology*, *29*(9), 1557–1563. https://doi.org/10.1016/j.cub.2019.04.005

González-Fernández, A., González-Salazar, C., Sunny, A., Ruíz-Gutiérrez, F., & Chávez, C. (2022). Determination of priority areas for amphibian conservation in Guerrero (Mexico), through systematic conservation planning tools. *Journal for Nature Conservation*, *68*, 126235. https://doi.org/10.1016/j.jnc.2022.126235

Goosem, M. W., Harding, E. K., Chester, G., Tucker, N., Harriss, C., & Oakley, K. (2010). *Roads in rainforest: Best practice guidelines for planning, design and management*. Cairns, Australia: Reef and Rainforest Research Centre Limited.

Gottsberger, B. & Gruber, E. (2004). Temporal partitioning of reproductive activity in a Neotropical anuran community. *Journal of Tropical Ecology*, *20*(3), 271–280. https://doi.org/10.1017/S0266467403001172

Grant, E. H. C., Miller, D. A. W. & Muths, E. (2020). A synthesis of evidence of drivers of amphibian declines. *Herpetologica*, *76*(2), 101–107. https://doi.org/10.1655/0018-0831-76.2.101

Green, D. M., Lannoo, M. J., Lesbarrères, D., & Muths, E. (2020). Amphibian population declines: 30 years of progress in confronting a complex problem. *Herpetologica*, *76*(2), 97–100. https://doi.org/10.1655/0018-0831-76.2.97

Grill, G., Lehner, B., Thieme, M., Geenen, B., Tickner, D., Antonelli, F., … Zarfl, C. (2019). Mapping the world's free-flowing rivers. *Nature*, *569*(7755), 215–221. https://doi.org/10.1038/s41586-019-1111-9

Guariguata, M. R. & Ostertag, R. (2001). Neotropical secondary forest succession: changes in structural and functional characteristics. *Forest Ecology and Management*, *148*(1–3), 185–206. https://doi.org/10.1016/S0378-1127(00)00535-1

Gurney, G. G., Darling, E. S., Ahmadia, G. N., Agostini, V. N., Ban, N. C., Blythe, J., … I. Jupiter, S. D. (2021). Biodiversity needs every tool in the box: use OECMs. *Nature*, *595*(7869), 646-649. https://doi.org/10.1038/d41586-021-02041-4.

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., … Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. *Science Advances*, *1*(2), 1–10. https://doi.org/10.1126/sciadv.1500052

Hamer, A. J. & McDonnell, M. J. (2008). Amphibian ecology and conservation in the urbanising world: a review. *Biological Conservation*, *141*(10), 2432–2449. https://doi.org/10.1016/j.biocon.2008.07.020

Hansen, N. A., Scheele, B. C., Driscoll, D. A., & Lindenmayer, D. B. (2019). Amphibians in agricultural landscapes: the habitat value of crop areas, linear plantings and remnant woodland patches. *Animal Conservation*, *22*(1), 72–82. https://doi.org/10.1111/acv.12437

Harper, K. A., Macdonald, S. E., Burton, P. J., Chen, J., Brosofske, K. D., Saunders, S. C., … Esseen, P. A. (2005). Edge influence on forest structure and composition in fragmented landscapes. *Conservation Biology*, *19*(3), 768–782. https://doi.org/ 10.1111/j.1523-1739.2005.00045.x

Hartel, T., Scheele, B. C., Rozylowicz, L., Horcea-Milcu, A., & Cogălniceanu, D. (2020). The social context for conservation: Amphibians in human shaped landscapes with high nature values. *Journal for Nature Conservation*, *53*, 125762. https://doi.org/10.1016/j.jnc.2019.125762

Hernández-Ordóñez, O., Santos, B. A., Pyron, R. A., Arroyo-Rodríguez, V., Urbina-Cardona, J. N., Martínez-Ramos, M., … Reynoso, V. H. (2019). Species sorting and mass effect along forest succession: evidence from taxonomic, functional, and phylogenetic diversity of amphibian communities. *Ecology and Evolution*, *9*, 5206–5218. https://doi.org/10.1002/ece3.5110

Hernández-Ordóñez, O., Urbina-Cardona, N. & Martínez-Ramos, M. (2015). Recovery of amphibian and reptile assemblages during old-field succession of tropical rain forests. *Biotropica*, *47*(3), 377–388. https://doi.org 10.1111/btp.12207

Herrera-Montes, A. & Brokaw, N. (2010). Conservation value of tropical secondary forest: A herpetofaunal perspective. *Biological Conservation*, *143*(6), 1414–1422. https://doi.org/10.1016/j.biocon.2010.03.016

Hilje, B. & Aide, T. M. (2012). Recovery of amphibian species richness and composition in a chronosequence of secondary forests, northeastern Costa Rica. *Biological Conservation*, *146*(1), 170–176. https://doi.org/10.1016/j.biocon.2011.12.007

Hillers, A., Veith, M. & Rödel, M. O. (2008). Effects of forest fragmentation and habitat degradation on West African leaf-litter frogs. *Conservation Biology*, *22*(3), 762–772. https://doi.org/10.1111/j.1523-1739.2008.00920.x

Hocking, D. J., Babbitt, K. J. & Hocking, D. J. (2014). Amphibian contributions to ecosystem services. *Herpetological Conservation and Biology*, *9*(1), 1–17.

Holzer, K. A., Bayers, R. P., Nguyen, T. T., & Lawler, S. P. (2017). Habitat value of cities and rice paddies for amphibians in rapidly urbanizing Vietnam. *Journal of Urban Ecology*, *3*(1), 1–12. https://doi.org/10.1093/jue/juw007

Houlahan, J. E., Findlay, C. S., Schmidt, B. R., Meyer, A. H., & Kuzmin, S. L. (2000). Quantitative evidence for global amphibian population declines. *Nature*, *404*, 752–755. https://doi.org/10.1038/35008052

IFC. (2012). *International Finance Corporation's Guidance Note 6: Biodiversity conservation and sustainable management of living natural resources*. Retrieved December 1, 2022 from https://www.ifc.org/wps/wcm/connect/topics_ext_content/ifc_external_corporate_site/sustainability-at-ifc/policies-standards/performance-standards/ps6

Isaacs Cubides, P. J. & Urbina Cardona, J. N. (2011). Anthropogenic disturbance and edge effects on anuran assemblages inhabiting cloud forest fragments in Colombia. *Natureza & Conservação*, *9*(1), 39–46. https://doi.org/10.4322/natcon.2011.004

IUCN. (2016). *A global standard for the identification of Key Biodiversity Areas, Version 1.0.* Gland, Switzerland: IUCN. https://portals.iucn.org/library/node/46259

IUCN. (2021). The IUCN Red List of Threatened Species Version 2021-2. Retrieved March 30, 2021, from https://www.iucnredlist.org

Jenkins, C. N., Van Houtan, K. S., Pimm, S. L., & Sexton, J. O. (2015). Reply to Brown et al.: Species and places are the priorities for conservation, not economic efficiency. *Proceedings of the National Academy of Sciences of the United States of America*, *112*(32), E4343. https://doi.org/10.1073/pnas.1511375112

Kaneda, T., Greenbaum, C. & Kline, K. (2020). *World population data sheet*. Washington, D.C, USA. Retrieved November 27, 2022 from https://www.prb.org/wp-content/uploads/2020/07/letter-booklet-2020-world-population.pdf

Karlsson, M. & Edvardsson Björnberg, K. (2021). Ethics and biodiversity offsetting. *Conservation Biology*, *35*(2), 578–586. https://doi.org/10.1111/cobi.13603

Kats, L. B. & Ferrer, R. P. (2003). Alien predators and amphibian declines: review of two decades of science and the transition to conservation. *Diversity and Distributions*, *9*(2), 99–110. https://doi.org/10.1046/j.1472-4642.2003.00013.x

Kehoe, L., Kuemmerle, T., Meyer, C., Levers, C., Václavík, T., & Kreft, H. (2015). Global patterns of agricultural land-use intensity and vertebrate diversity. *Diversity and Distributions*, *21*(11), 1308–1318. https://doi.org/10.1111/ddi.12359

Kennedy, C. M., Oakleaf, J. R., Theobald, D. M., Baruch-Mordo, S., & Kiesecker, J. (2019). Managing the middle: a shift in conservation priorities based on the global human modification gradient. *Global Change Biology*, *25*(3), 811–826. https://doi.org/10.1111/gcb.14549

Kiesecker, J. M., Blaustein, A. R. & Belden, L. K. (2001). Complex causes of amphibian population declines. *Nature*, *410*(6829), 681–684. https://doi.org/10.1038/35070552

Knight, A. T., Cowling, R. M., Rouget, M., Balmford, A., Lombard, A. T., & Campbell, B. M. (2008). Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. *Conservation Biology*, *22*(3), 610–617. https://doi.org/10.1111/j.1523-1739.2008.00914.x

Konopik, O., Steffan-Dewenter, I. & Grafe, T. U. (2015). Effects of logging and oil palm expansion on stream frog communities on Borneo, Southeast Asia. *Biotropica*, *47*(5), 636–643. https://doi.org/10.1111/btp.12248

Kremen, C., Williams, N. M. & Thorp, R. W. (2002). Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Sciences of the United States of America*, *99*(26), 16812–16816. https://doi.org/10.1073/pnas.262413599

Kueh, B. H., Maryati, M., Das, I., & Chew, D. (2004). Integrated effort in the prioritization of areas for conservation in Borneo. In B.-H. Kueh, M. Mohamed, I. Das & D. Chew (ed.) *Borneo Biodiversity & Ecosystems Conservation Programme, Proceedings of the BBEC International Conference* (pp. 129-131). Kota Kinabalu, Malaysia: Universiti Malaysia Sarawak.

Kukkala, A. S. & Moilanen, A. (2013). Core concepts of spatial prioritisation in systematic conservation planning. *Biological Reviews*, *88*(2), 443–464. https://doi.org/10.1111/brv.12008

Laurance, W. F., Camargo, J. L. C., Luizão, R. C. C., Laurance, S. G., Pimm, S. L., Bruna, E. M., … Lovejoy, T. E. (2011). The fate of Amazonian forest fragments: A 32-year investigation. *Biological Conservation*, *144*(1), 56–67. https://doi.org/10.1016/j.biocon.2010.09.021

Laurance, W. F., Carolina Useche, D., Rendeiro, J., Kalka, M., Bradshaw, C. J. A., Sloan, S. P., … Zamzani, F. (2012). Averting biodiversity collapse in tropical forest protected areas. *Nature*, *489*(7415), 290–293. https://doi.org/10.1038/nature11318

Laurance, W. F., Lovejoy, T. E., Vasconcelos, H. L., Bruna, E. M., Didham, R. K., Stouffer, P. C., … Sampaio, E. (2002). Ecosystem decay of Amazonian forest fragments: A 22-year investigation. *Conservation Biology*, *16*(3), 605–618. https://doi.org/10.1046/j.1523-1739.2002.01025.x

Laurance, W. F., Sayer, J. & Cassman, K. G. (2014). Agricultural expansion and its impacts on tropical nature. *Trends in Ecology and Evolution*, *29*(2), 107–116. https://doi.org/10.1016/j.tree.2013.12.001

Le Saout, S., Hoffmann, M., Shi, Y., Hughes, A., Bernard, C., Brooks, T. M., … Rodrigues, A. S. L. (2013). Protected areas and effective biodiversity conservation. *Science*, *342*(6160), 803–805. https://doi.org/10.1126/science.1239268

Lehtinen, R. M., Galatowitsch, S. M. & Tester, J. R. (1999). Consequences of habitat loss and fragmentation for wetland amphibian assemblages. *Wetlands*, *19*(1), 1–12. https://doi.org/10.1007/BF03161728

Lehtinen, R. M., Ramanamanjato, J. B. & Raveloarison, J. G. (2003). Edge effects and extinction proneness in a herpetofauna from Madagascar. *Biodiversity and Conservation*, *12*(7), 1357–1370. https://doi.org/10.1023/A:1023673301850

Lemckert, F., Rosauer, D. & Slatyer, C. (2009). A comparison of Australia's anuran records against the reserve system. *Biodiversity and Conservation*, *18*(5), 1233–1246. https://doi.org/10.1007/s10531-008-9542-x

Li, Z., Hou, Y., Cao, J., Ding, Y., & Yuan, X. (2022). What drives green development in China: public pressure or the willingness of local government? *Environmental Science and Pollution Research*, *29*(4), 5454–5468. https://doi.org/10.1007/s11356-021-16059-8

Lindenmayer, D. B. & Westgate, M. J. (2020). Are flagship, umbrella and keystone species useful surrogates to understand the consequences of landscape change? *Current Landscape Ecology Reports*, *5*(3), 76–84. https://doi.org/10.1007/s40823-020-00052-x

López-Bedoya, P. A., Cardona-Galvis, E. A., Urbina-Cardona, J. N., Edwards, F. A., & Edwards, D. P. (2022). Impacts of pastures and forestry plantations on herpetofauna: a global meta analysis. Journal of Applied Ecology, 59, 3038–3048.

https://doi.org/10.1111/1365-2664.14299

Loyola, R. D., Becker, C. G., Kubota, U., Haddad, C. F. B., Fonseca, C. R., & Lewinsohn, T. M. (2008). Hung out to dry: Choice of priority ecoregions for conserving threatened Neotropical anurans depends on life-history traits. *PLoS ONE*, *3*(5), e2120. https://doi.org/10.1371/journal.pone.0002120

Luke, S. H., Slade, E. M., Gray, C. L., Annammala, K. V., Drewer, J., Williamson, J., … Struebig, M. J. (2019). Riparian buffers in tropical agriculture: scientific support, effectiveness and directions for policy. *Journal of Applied Ecology*, *56*(1), 85–92. https://doi.org/10.1111/1365-2664.13280

Mace, G. M., Barrett, M., Burgess, N. D., Cornell, S. E., Freeman, R., Grooten, M., & Purvis, A. (2018). Aiming higher to bend the curve of biodiversity loss. *Nature Sustainability*, *1*(9), 448–451. https://doi.org/10.1038/s41893-018-0130-0

Machovina, B., Feeley, K. J. & Ripple, W. J. (2015). Biodiversity conservation: the key is reducing meat consumption. *Science of the Total Environment*, *536*, 419–431. https://doi.org/10.1016/j.scitotenv.2015.07.022

Maerz, J. C., Brown, C. J., Chapin, C. T., & Blossey, B. (2005). Can secondary compounds of an invasive plant affect larval amphibians? *Functional Ecology*, *19*(6), 970–975. https://doi.org/10.1111/j.1365-2435.2005.01054.x

Maerz, J. C., Blossey, B. & Nuzzo, V. (2005). Green frogs show reduced foraging success in habitats invaded by Japanese knotweed. *Biodiversity and Conservation*, *14*(12), 2901–2911. https://doi.org/10.1007/s10531-004-0223-0

Magnus, R. & Rannap, R. (2019). Pond construction for threatened amphibians is an important conservation tool, even in landscapes with extant natural water bodies. *Wetlands Ecology and Management*, *27*(2–3), 323–341. https://doi.org/10.1007/s11273-019-09662-7

Margules, C. R. & Sarkar, S. (2007). *Systematic conservation planning*. Cambridge, UK: Cambridge University Press.

Marroquín-Páramo, J. A., Suazo-Ortuño, I., Urbina-Cardona, N., & Benítez-Malvido, J. (2021). Cumulative effects of high intensity hurricanes on herpetofaunal assemblages along a tropical dry forest chronosequence. *Forest Ecology and Management*, *479*, 118505. https://doi.org/10.1016/j.foreco.2020.118505

Martín-Torrijos, L., Sandoval-Sierra, J. V., Muñoz, J., Diéguez-Uribeondo, J., Bosch, J., & Guayasamin, J. M. (2016). Rainbow trout (*Oncorhynchus mykiss*) threaten Andean amphibians. *Neotropical Biodiversity*, *2*(1), 26–36. https://doi.org/10.1080/23766808.2016.1151133

Mayhew, S. H., Newman, K., Johnson, D., Clark, E., Hammer, M., Mohan, V., & Ssali, S. (2020). New partnerships, new perspectives: The relevance of sexual and reproductive health and rights for sustainable development. *Health Policy*, *124*(6), 599–604. https://doi.org/10.1016/j.healthpol.2019.03.010

McAlpine, C. A., Etter, A., Fearnside, P. M., Seabrook, L., & Laurance, W. F. (2009). Increasing world consumption of beef as a driver of regional and global change: a call for policy action based on evidence from Queensland (Australia), Colombia and Brazil. *Global Environmental Change*, *19*(1), 21–33. https://doi.org/10.1016/j.gloenvcha.2008.10.008

McDiarmid, R. W. & Altig, R. (1999). *Tadpoles: The biology of anuran larvae*. Chicago, USA: University of Chicago Press.

Measey, G. J. (2011). *Ensuring a future for South Africa's frogs: A strategy for conservation research*. Pretoria, South Africa: South African National Biodiversity Institute.

Meegaskumbura, M., Bossuyt, F., Pethiyagoda, R., Manamendra-Arachchi, K., Bahir, M., Milinkovitch, M. C., & Schneider, C. J. (2002). Sri Lanka: an amphibian hot spot. *Science*, *298*(5592), 379–379. https://doi.org/10.1126/science.298.5592.3

Meine, C., Soulé, M. & Noss, R. F. (2006). "A mission-driven discipline": the growth of conservation biology. *Conservation Biology*, *20*(3), 631–651. https://doi.org/10.1111/j.1523-1739.2006.00449.x

Melo, F. P. L., Arroyo-Rodríguez, V., Fahrig, L., Martínez-Ramos, M., & Tabarelli, M. (2013). On the hope for biodiversity-friendly tropical landscapes. *Trends in Ecology and Evolution*, *28*(8), 462–468. https://doi.org/10.1016/j.tree.2013.01.001

Mendenhall, C. D., Frishkoff, L. O., Santos-Barrera, G., Pacheco, J., Mesfun, E., Mendoza Quijano, F., … Pringle, R. M. (2014). Countryside biogeography of Neotropical reptiles and amphibians. *Ecology*, *95*(4), 856–870. https://doi.org/10.1890/12-2017.1

Menéndez-Guerrero, P. A., Davies, T. J. & Green, D. M. (2020). Extinctions of threatened frogs may impact ecosystems in a global hotspot of anuran diversity. *Herpetologica*, *76*(2), 121–131. https://doi.org/10.1655/0018-0831-76.2.121

Mitchell, J. C., Breisch, A. R. & Buhlmann, K. A. (2006). *Habitat management* *guidelines for amphibians and reptiles of the Northeastern United States*. Technical Publication HMG-3. Partners in Amphibian and Reptile Conservation (PARC). Retrieved November 25, 2022 from http://northeastparc.org/habitat-management-guidelines/

Moor, H., Bergamini, A., Vorburger, C., Holderegger, R., Bühler, C., Egger, S., & Schmidt, B. R. (2022). Bending the curve: simple but massive conservation action leads to landscape-scale recovery of amphibians. *Proceedings of the National Academy of Sciences of the United States of America, 119*(42): e2123070119. https://doi.org/10.1073/pnas.2123070119

Moore, R. D. (2011). *Protecting the smaller majority: Amphibian conservation case studies*. Conservation International and the IUCN/SSC Amphibian Specialist Group.

Mora, F., Martínez-Ramos, M., Ibarra-Manríquez, G., Pérez-Jiménez, A., Trilleras, J., & Balvanera, P. (2015). Testing chronosequences through dynamic approaches: time and site effects on tropical dry forest succession. *Biotropica*, *47*(1), 38–48. https://doi.org/10.1111/btp.12187

Murcia, C. (1995). Edge effects in fragmented forests: implications for conservation. *Trends in Ecology & Evolution*, *10*(2), 58–62. https://doi.org/10.1016/S0169-5347(00)88977-6

Murrieta-Galindo, R., González-Romero, A., López-Barrera, F., & Parra-Olea, G. (2013). Coffee agrosystems: An important refuge for amphibians in central Veracruz, Mexico. *Agroforestry Systems*, *87*(4), 767–779. https://doi.org/10.1007/s10457-013-9595-z

Murrieta-Galindo, R., López-Barrera, F., González-Romero, A., & Parra-Olea, G. (2013). Matrix and habitat quality in a montane cloud-forest landscape: Amphibians in coffee plantations in central Veracruz, Mexico. *Wildlife Research*, *40*(1), 25–35. https://doi.org/10.1071/WR12076

Muths, E., Chambert, T., Schmidt, B. R., Miller, D. A. W., Hossack, B. R., Joly, P., … Grant, E. H. C. (2017). Heterogeneous responses of temperate-zone amphibian populations to climate change complicates conservation planning. *Scientific Reports*, *7*(1), 1–10. https://doi.org/10.1038/s41598-017-17105-7

Naito, R., Sakai, M., Natuhara, Y., Morimoto, Y., & Shibata, S. (2013). Microhabitat use by *Hyla japonica* and *Pelophylax porosa brevipoda* at Levees in rice paddy areas of Japan. *Zoological Science*, *30*(5), 386–391. https://doi.org/10.2108/zsj.30.386

Navas, C. A. & Otani, L. (2007). Physiology, environmental change, and anuran conservation. *Phyllomedusa*, *6*(2), 83–103. https://doi.org/10.11606/issn.2316-9079.v6i2p83-103

Neckel-Oliveira, S. & Gascon, C. (2006). Abundance, body size and movement patterns of a tropical treefrog in continuous and fragmented forests in the Brazilian Amazon. *Biological Conservation*, *128*(3), 308–315. https://doi.org/10.1016/j.biocon.2005.09.037

Newbold, T., Hudson, L. N., Arnell, A. P., Contu, S., Palma, A. De, Ferrier, S., … Zhang, H. (2016). Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. *Science*, *353*(6296), 288–291. https://doi.org/10.1126/science.aaf2201

Nori, J., Lemes, P., Urbina-Cardona, N., Baldo, D., Lescano, J., & Loyola, R. (2015). Amphibian conservation, land-use changes and protected areas: a global overview. *Biological Conservation*, *191*, 367–374. https://doi.org/10.1016/j.biocon.2015.07.028

Nunes, A. L., Fill, J. M., Davies, S. J., Louw, M., Rebelo, A. D., Thorp, C. J., … Measey, J. (2019). A global meta-analysis of the ecological impacts of alien species on native amphibians. *Proceedings of the Royal Society B*, *286*, 20182528. https://doi.org/10.1098/rspb.2018.2528

Ochoa-Ochoa, L. M., Bezaury-Creel, J. E., Vázquez, L.-B., & Flores-Villela, O. (2011). Choosing the survivors? A GIS-based triage support tool for micro-endemics: Application to data for Mexican amphibians. *Biological Conservation*, *144*(11), 2710–2718. https://doi.org/10.1016/j.biocon.2011.07.032

Ochoa-Ochoa, L. M., Urbina-Cardona, J. N., Vázquez, L. B., Flores-Villela, O., & Bezaury-Creel, J. (2009). The effects of governmental protected areas and social initiatives for land protection on the conservation of Mexican amphibians. *PLoS ONE*, *4*(9), e6878. https://doi.org/10.1371/journal.pone.0006878

Ochoa-Ochoa, L. M., Urbina-Cardona, N. & Flores-Villela, O. (2011). Planning amphibian conservation in Mexico. *FrogLog*, *97*, 51–52.

Oddone Aquino, A. G. H. E. & Nkomo, S. L. (2021). Spatio-temporal patterns and consequences of road kills: a review. *Animals*, *11*(3), 1–23. https://doi.org/10.3390/ani11030799

Oscarson, D. B. & Calhoun, A. J. K. (2007). Developing vernal pool conservation plans at the local level using citizen-scientists. *Wetlands*, *27*, 80–95. https://doi.org/10.1672/0277-5212(2007)27[80:DVPCPA]2.0.CO;2

Oteros-Rozas, E., Ruiz-Almeida, A., Aguado, M., González, J. A., & Rivera-Ferre, M. G. (2019). A social-ecological analysis of the global agrifood system. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(52), 26465–26473. https://doi.org/10.1073/pnas.1912710116

Palomo, I., Montes, C., Martín-López, B., González, J. A., García-Llorente, M., Alcorlo, P., & Mora, M. R. G. (2014). Incorporating the social-ecological approach in protected areas in the Anthropocene. *BioScience*, 64(3), 181–191. https://doi.org/10.1093/biosci/bit033

Pawar, S. S., Rawat, G. S. & Choudhury, B. C. (2004). Recovery of frog and lizard communities following primary habitat alteration in Mizoram, Northeast India. *BMC Ecology*, 4(10), 1–18. https://doi.org/10.1186/1472-6785-4-10

Pawaskar, U. S., Raut, R. D. & Gardas, B. B. (2018). Assessment of consumer behavior towards environmental responsibility: a structural equations modeling approach. *Business Strategy and the Environment*, 27(4), 560–571. https://doi.org/10.1002/bse.2020

Payn, T., Carnus, J. M., Freer- Smith, P., Kimberley, M., Kollert, W., Liu, S., Orazio, C., Rodriguez, L., Silva, L. N., & Wingfield, M. J. (2015). Changes in planted forests and future global implications. *Forest Ecology and Management*, *352*, 57–67. https://doi.org/10.1016/j.foreco.2015.06.021

Pearce, F. & Madgwick, F. J. (2020). *Water lands: A vision for the world's wetlands and their people*. The Netherlands: Harper Collins.

Pearman, P. B. (1997). Correlates of amphibian diversity in an altered landscape of Amazonian Ecuador. *Conservation Biology*, *11*(5), 1211–1225. https://doi.org/10.1046/j.1523-1739.1997.96202.x

Pence, G. K. Q. (2017). *The Western Cape Biodiversity spatial plan: Technical report*. Cape Town, South Africa: Capenature.

Perfecto, I. & Vandermeer, J. (2008). Biodiversity conservation in tropical agroecosystems: A new conservation paradigm. *Annals of the New York Academy of Sciences*, *1134*, 173–200. https://doi.org/10.1196/annals.1439.011

Perfecto, I. & Vandermeer, J. (2010). The agroecological matrix as alternative to the land-sparing/agriculture intensification model. *Proceedings of the National Academy of Sciences of the United States of America*, *107*(13), 5786–5791. https://doi.org/10.1073/pnas.0905455107

Petranka, J. W. & Holbrook, C. T. (2006). Wetland restoration for amphibians: Should local sites be designed to support metapopulations or patchy populations? *Restoration Ecology*, *14*(3), 404–411. https://doi.org/10.1111/j.1526-100X.2006.00148.x

Pfeifer, M., Lefebvre, V., Peres, C. A., Banks-Leite, C., Wearn, O. R., Marsh, C. J., … Ewers, R. M. (2017). Creation of forest edges has a global impact on forest vertebrates. *Nature*, *551*(7679), 187–191. https://doi.org/10.1038/nature24457

Phillips, H. R. P., Halley, J. M., Urbina-Cardona, J. N., & Purvis, A. (2018). The effect of fragment area on site-level biodiversity. *Ecography*, *41*(7), 1220–1231. https://doi.org/10.1111/ecog.02956

Pineda, E. & Halffter, G. (2004). Species diversity and habitat fragmentation: frogs in a tropical montane landscape in Mexico. *Biological Conservation*, *117*(5), 499–508. https://doi.org/10.1016/j.biocon.2003.08.009

Pineda, E., Moreno, C., Escobar, F., & Halffter, G. (2005). Frog, bat, and dung beetle diversity in the cloud forest and coffee agroecosystems of Veracruz, Mexico. *Conservation Biology*, *19*(2), 400–410. https://doi.org/10.1111/j.1523-1739.2005.00531.x

Pinero-Rodríguez, M. J., Fernández-Zamudio, R., Arribas, R., Gomez-Mestre, I., & Díaz-Paniagua, C. (2021). The invasive aquatic fern *Azolla filiculoides* negatively impacts water quality, aquatic vegetation and amphibian larvae in Mediterranean environments. *Biological Invasions*, *23*(3), 755–769. https://doi.org/10.1007/s10530-020-02402-6

Poore, J. & Nemecek, T. (2018). Reducing food's environmental impacts through producers and consumers. *Science*, *360*, 987–992. https://doi.org/10.1126/science.aaq0216

Pope, S. E., Fahrig, L. & Merriam, H. G. (2000). Landscape complementation and metapopulation effects on leopard frog populations. *Ecology*, *81*(9), 2498–2508. https://doi.org/10.2307/177470

Pouzols, F. M., Toivonen, T., Minin, E. Di, Kukkala, A. S., Kullberg, P., Kuustera, J., … Moilanen, A. (2014). Global protected area expansion is compromised by projected land-use and parochialism. *Nature*, *516*(7531), 383–386. https://doi.org/10.1038/nature14032

Pratihar, S., Clark, H. O., Dutta, S., Khan, M. S., Patra, B. C., Ukuwela, K. D. B., … Das, S. (2014). Diversity and conservation of amphibians in South and Southeast Asia. *Sauria*, *36*, 9–59.

Queiroz, C. de S., da Silva, F. R., & Rossa-Feres, D. de C. (2015). The relationship between pond habitat depth and functional tadpole diversity in an agricultural landscape. *Royal Society Open Science*, *2*(7), 150165. https://doi.org/10.1098/rsos.150165

Rabinowitz, D., Cairns, S. & Dillon, T. (1986). Seven forms of rarity and their frequency in the flora of the British Isles. In M. E. Soulé (Ed.), *Conservation biology: the science of scarcity and diversity*. Sunderland, USA: Sinauer Associates, Inc.

Ramsar Convention on Wetlands. (2018). *Global wetland outlook: State of the world's wetlands and their services to people 2018*. Gland, Switzerland. Retrieved November 15, 2022 from https://www.global-wetland-outlook.ramsar.org/outlook

Rannap, R., Lõhmus, A. & Briggs, L. (2009). Restoring ponds for amphibians: A success story. *Hydrobiologia*, *634*, 87–95. https://doi.org/10.1007/s10750-009-9884-8

Resasco, J. (2019). Meta-analysis on a decade of testing corridor efficacy: what new have we learned? *Current Landscape Ecology Reports*, *4*, 61–69. https://doi.org/10.1007/s40823-019-00041-9

Ribeiro, J., Colli, G. R. & Soares, A. (2019). Landscape correlates of anuran functional connectivity in rice crops: a graph-theoretic approach. *Journal of Tropical Ecology*, *35*(3), 118–131. https://doi.org/10.1017/S026646741900004X

Rice, R. A. & Greenberg, R. (2000). Cacao cultivation and the conservation of biological diversity. *Ambio*, *29*(3), 167–173. https://doi.org/10.1579/0044-7447-29.3.167

Ricketts, T. H., Dinerstein, E., Boucher, T., Brooks, T. M., Butchart, S. H. M., Hoffmann, M., … Wikramanayake, E. (2005). Pinpointing and preventing imminent extinctions. *Proceedings of the National Academy of Sciences*, *102*(51), 18497–18501. https://doi.org/10.1073/pnas.0509060102

Riemann, J. C., Ndriantsoa, S. H., Raminosoa, N. R., Rödel, M. O., & Glos, J. (2015). The value of forest fragments for maintaining amphibian diversity in Madagascar. *Biological Conservation*, *191*, 707–715. https://doi.org/0.1016/j.biocon.2015.08.020

Riemann, J. C., Ndriantsoa, S. H., Rödel, M. O., & Glos, J. (2017). Functional diversity in a fragmented landscape — Habitat alterations affect functional trait composition of frog assemblages in Madagascar. *Global Ecology and Conservation*, *10*, 173–183. https://doi.org/10.1016/j.gecco.2017.03.005

Ries, L., Fletcher Jr, R. J., Battin, J., & Sisk, T. D. (2004). Ecological responses to habitat edges: mechanisms, models, and variability explained. *Annual Review of Ecology, Evolution, and Systematics*, *35*, 491-522. https://doi.org/10.1146/annurev.ecolsys.35.112202.130148

Rios-López, N. & Aide, T. M. (2007). Herpetofaunal dynamics during secondary succession. *Herpetologica*, *63*(1), 35–50. https://doi.org/10.1655/0018-0831(2007)63[35:HDDSS]2.0.CO;2

Ritchie, H. & Roser, M. (2019). Land Use. Retrieved October 23, 2022 from OurWorldInData.org website: https://ourworldindata.org/land-use

Roach, N. S., Urbina-Cardona, N. & Lacher, T. E. (2020). Land cover drives amphibian diversity across steep elevational gradients in an isolated Neotropical mountain range: Implications for community conservation. *Global Ecology and Conservation*, *22*, e00968. https://doi.org/10.1016/j.gecco.2020.e00968

Rodrigues, A. S. L., Akçakaya, H. R., Andelman, S. J., Bakarr, M. I., Boitani, L., Brooks, T. M., … Yan, X. (2004). Global gap analysis: Priority regions for expanding the global protected-area network. *BioScience*, *54*(12), 1092–1100. https://doi.org/10.1641/0006-3568(2004)054[1092:GGAPRF]2.0.CO;2

Rodrigues, A. S. L., Andelman, S. J., Bakarr, M. I., Boitani, L., Brooks, T. M., Cowling, R. M., … Yan, X. (2004). Effectiveness of the global protected area network in representing species diversity. *Nature*, *428*, 640–643. https://doi.org/10.1038/nature02459

Roelants, K., Gower, D. J., Wilkinson, M., Loader, S. P., Biju, S. D., Guillaume, K., … Bossuyt, F. (2007). Global patterns of diversification in the history of modern amphibians. *Proceedings of the National Academy of Sciences of the United States of America*, *104*(3), 887–892. https://doi.org/10.1073/pnas.0608378104

Roni, P. (2003). Responses of benthic fishes and giant salamanders to placement of large woody debris in small Pacific Northwest streams. *North American Journal of Fisheries Management*, *23*(4), 1087–1097. https://doi.org/10.1577/m02-048

Rowley, J., Brown, R., Bain, R., Kusrini, M., Inger, R., Stuart, B., … Phimmachak, S. (2010). Impending conservation crisis for Southeast Asian amphibians. *Biology Letters*, *6*(3), 336–338. https://doi.org/10.1098/rsbl.2009.0793

Ruhí, A., San Sebastian, O., Feo, C., Franch, M., Gascón, S., Richter-Boix, À., … Llorente, G. (2012). Man-made Mediterranean temporary ponds as a tool for amphibian conservation. *International Journal of Limnology*, *48*(1), 81–93. https://doi.org/10.1051/limn/2011059

Russildi, G., Arroyo-Rodríguez, V., Hernández-Ordóñez, O., Pineda, E., & Reynoso, V. H. (2016). Species- and community-level responses to habitat spatial changes in fragmented rainforests: Assessing compensatory dynamics in amphibians and reptiles. *Biodiversity and Conservation*, *25*(2), 375–392. https://doi.org/10.1007/s10531-016-1056-3

Saboyá Acosta, L. P. & Urbina-Cardona, J. N. (2023). Current state of knowledge of Páramo amphibians in Colombia: Spatio temporal trends and information gaps to be strengthened for effective conservation. *Tropical Conservation Science*, *16*, 19400829231169984. https://doi.org/10.1177/19400829231169984

Sangermano, F., Bol, L., Galvis, P., Gullison, R. E., Hardner, J., & Ross, G. S. (2015). Habitat suitability and protection status of four species of amphibians in the Dominican Republic. *Applied Geography*, *63*, 55–65. https://doi.org/10.1016/j.apgeog.2015.06.002

Santos-Barrera, G. & Urbina-Cardona, J. N. (2011). The role of the matrix-edge dynamics of amphibian conservation in tropical montane fragmented landscapes. *Revista Mexicana de Biodiversidad*, *82*, 679–687. https://doi.org/10.22201/ib.20078706e.2011.2.463

Saunders, D. A., Hobbs, R. J. & Margules, C. R. (1991). Biological consequences of ecosystem fragmentation: A review. *Conservation Biology*, *5*(1), 18–32. https://doi.org/10.1016/0006-3207(92)90725-3

Scheffers, B. R., Brunner, R. M., Ramirez, S. D., Shoo, L. P., Diesmos, A., & Williams, S. E. (2013). Thermal buffering of microhabitats is a critical factor mediating warming vulnerability of frogs in the Philippine Biodiversity Hotspot. *Biotropica*, *45*(5), 628–635. https://doi.org/10.1111/btp.12042

Scheffers, B. R., Edwards, D. P., Diesmos, A., Williams, S. E., & Evans, T. A. (2014). Microhabitats reduce animal's exposure to climate extremes. *Global Change Biology*, *20*(2), 495–503. https://doi.org/10.1111/gcb.12439

Schleicher, J., Peres, C. A., Amano, T., Llactayo, W., & Leader-Williams, N. (2017). Conservation performance of different conservation governance regimes in the Peruvian Amazon. *Scientific Reports*, *7*, 11318. https://doi.org/10.1038/s41598-017-10736-w

Schloegel, L. M., Picco, A. M., Kilpatrick, A. M., Davies, A. J., Hyatt, A. D., & Daszak, P. (2009). Magnitude of the US trade in amphibians and presence of *Batrachochytrium dendrobatidis* and ranavirus infection in imported North American bullfrogs (*Rana catesbeiana*). *Biological Conservation*, *142*, 1420–1426. https://doi.org/10.1016/j.biocon.2009.02.007

Schmidt, B. R., Brenneisen, S. & Zumbach, S. (2020). Evidence-based amphibian conservation: A case study on toad tunnels. *Herpetologica*, *76*(2), 228–239. https://doi.org/10.1655/0018-0831-76.2.228

Schmidt, B. R. & Zumbach, S. (2008). Amphibian road mortality and how to prevent it: a review. In J. C. Mitchell, R. E. Jung Brown, & B. Bartholomew (Eds.), *Urban Herpetology* (pp. 157–167). St. Louis, Missouri: Society for the Study of Amphibians and Reptiles. https://doi.org/10.5167/uzh-10142

Schneider-Maunoury, L., Lefebvre, V., Ewers, R. M., Medina-Rangel, G. F., Peres, C. A., Somarriba, E., … Pfeifer, M. (2016). Abundance signals of amphibians and reptiles indicate strong edge effects in Neotropical fragmented forest landscapes. *Biological Conservation*, *200*, 207–215. https://doi.org/10.1016/j.biocon.2016.06.011

Scott, D. E., Metts, B. S., & Whitfield Gibbons, J. (2008). Enhancing amphibian biodiversity on golf courses with seasonal wetlands. In J. C. Mitchell, R. E. Jung Brown, & B. Bartholomew (Eds.), *Urban herpetology* (pp. 285–292). Society for thc Study of Arnphibians and Reptiles.

Scriven, S. A., Gillespie, G. R., Laimun, S., & Goossens, B. (2018). Edge effects of oil palm plantations on tropical anuran communities in Borneo. *Biological Conservation*, *220*, 37–49. https://doi.org/10.1016/j.biocon.2018.02.006

Semlitsch, R. D. & Bodie, J. R. (2003). Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles. *Conservation Biology*, *17*(5), 1219–1228. https://doi.org/10.1046/j.1523-1739.2003.02177.x

Simon, J. A., Snodgrass, J. W., Casey, R. E., & Sparling, D. W. (2009). Spatial correlates of amphibian use of constructed wetlands in an urban landscape. *Landscape Ecology*, *3*, 361–373. https://doi.org/10.1007/s10980-008-9311-y

Skowno, A. L., Poole, C. J., Raimondo, D. C., Sink, K. J., Van Deventer, H., Van Niekerk, L., … Driver, A. (2019). National biodiversity assessment 2018: The status of South Africa's ecosystems and biodiversity. Synthesis report. South African National Biodiversity Institute. In *South African National Biodiversity Institute*. Pretoria, South Africa: South African National Biodiversity Institute, an entity of the Department of Environment, Forestry and Fisheries. Retrieved October 14, 2022 from http://bgis.sanbi.org/NBA/NBA2011_metadata_formalprotectedareas.pdf%5Cnpapers2://publication/uuid/786A77C5-B11A-4F8D-B139-F3F626EBC802

Smith, R. J., Bennun, L., Brooks, T. M., Butchart, S. H. M., Cuttelod, A., Di Marco, M., … Scaramuzza, C. A. d. M. (2019). Synergies between the key biodiversity area and systematic conservation planning approaches. *Conservation Letters*, *12*, e12625. https://doi.org/10.1111/conl.12625

Smith, R. K., Meredith, H. M. R. & Sutherland, W. (2021). Amphibian conservation. In W. J. Sutherland, L. V. Dicks, S. O. Petrovan, & R. K. Smith (Eds.), *What works in conservation 2021*. Cambridge, UK: Open Book Publishers. https://doi.org/10.11647/OBP.0267

Smith, R. K., Meredith, H. M. R. & Sutherland, W. J. (2019). Amphibian conservation. In W. J. Sutherland, L. V. Dicks, N. Ockendon, S. O. Petrovan, & R. K. Smith (Eds.), *What works in conservation 2019* (pp. 9–65). Cambridge, UK: Open Book Publishers. https://doi.org/10.11647/obp.0179

Smith, R. K. & Sutherland, W. J. (2014). *Amphibian conservation: Global evidence for the effects of interventions*. Exeter, UK: Pelagic Publishing.

Soto-Sandoval, Y., Suazo-Ortuño, I., Urbina-Cardona, N., Marroquín-Páramo, J., & Alvarado-Díaz, J. (2017). Efecto de los estadios sucesionales del bosque tropical seco sobre el microhábitat usado por *Agalychnis dacnicolor* (Anura: Phyllomedusidae) y *Smilisca fodiens* (Anura: Hylidae). *Revista de Biologia Tropical*, *65*(2), 777–798. https://doi.org/10.15517/rbt.v65i2.24706

South African National Biodiversity Institute (SANBI). (2015). *The business case for biodiversity stewardship. A report produced for the Department of Environmental Affairs*. SANBI, Pretoria, South Africa.

South African National Biodiversity Institute (SANBI) and Wildlands Conservation Trust. (2015). *Case study: Biodiversity partnership area. uMzimvubu Catchment Partnership Programme*. SANBI, Pretoria, South Africa.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., & de Haan, C. (2006). *Livestock's long shadow: environmental issues and options*. Rome, Italy: Food and Agriculture Organisation (FAO).

Stuart, S. N., Chanson, J. S., Cox, N. A., Young, B. E., Rodrigues, A. S. L. L., … Waller, R. W. (2004). Status and trends of amphibian declines and extinctions worldwide. *Science*, *306*(5702), 1783–1786. https://doi.org/10.1126/science.1103538

Stumpel, A. H. P. & van der Voet, H. (1998). Characterizing the suitability of new ponds for amphibians. *Amphibia-Reptilia*, *19*, 125–142. https://doi.org/10.1163/156853898x00421

Suazo-Ortuño, I., Alvarado-Díaz, J. & Martínez-Ramos, M. (2008). Effects of conversion of dry tropical forest to agricultural mosaic on herpetofaunal assemblages. *Conservation Biology*, *22*(2), 362–374. https://doi.org/10.1111/j.1523-1739.2008.00883.x

Suazo-Ortuño, I., Alvarado-Díaz, J., Mendoza, E., López-Toledo, L., Lara-Uribe, N., Márquez-Camargo, C., … Rangel-Orozco, J. D. (2015). High resilience of herpetofaunal communities in a human-modified tropical dry forest landscape in western Mexico. *Tropical Conservation Science*, *8*(2), 396–423. https://doi.org/10.1177/194008291500800208

Suazo-Ortuño, I., Urbina-Cardona, N., Lara-Uribe, N., Marroquín-Páramo, J., Soto-Sandoval, Y., Rangel-Orozco, J., … Alvarado-Díaz, J. (2018). Impact of a hurricane on the herpetofaunal assemblages of a successional chronosequence in a tropical dry forest. *Biotropica*, *50*(4), 649–663. https://doi.org/10.1111/btp.12544

Sutherland, W. J., Dicks, L. V., Petrovan, S. O., & Smith, R. K. (2021). *What works in conservation 2021*. Cambridge, UK: Open Book Publishers. https://doi.org/10.11647/OBP.0267

Sutherland, W. J., Pullin, A. S., Dolman, P. M., & Knight, T. M. (2004). The need for evidence-based conservation. *Trends in Ecology and Evolution*, *19*(6), 305–308. https://doi.org/10.1016/j.tree.2004.03.018

Tarrant, J. & Armstrong, A. J. (2013). Using predictive modelling to guide the conservation of a critically endangered coastal wetland amphibian. *Journal for Nature Conservation*, *21*, 369–381. https://doi.org/10.1016/j.jnc.2013.03.006

Tarrant, J., Kruger, D. & du Preez, L. H. (2016). Do public attitudes affect conservation effort? Using a questionnaire-based survey to assess perceptions, beliefs and superstitions associated with frogs in South Africa. *African Zoology*, *51*(1), 13–20. https://doi.org/10.1080/15627020.2015.1122554

Thompson, M. E. & Donnelly, M. A. (2018). Effects of secondary forest succession on amphibians and reptiles: A review and meta-analysis. *Copeia*, *106*(1), 10–19. https://doi.org/10.1643/CH-17-654

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., … Swackhamer, D. (2001). Forecasting agriculturally driven global environmental change. *Science*, *292*(5515), 281–284. https://doi.org/10.1126/science.1057544

Tocher, M. D., Gascon, C. & Zimmerman, B. L. (1997). Fragmentation effects on a central Amazonian frog community: A ten-year study. In W. F. Laurance & R. O. Bierregaard (Eds.), *Tropical forest remnants: Ecology, management and conservation of fragmented communities* (pp. 124–137). Chicago, USA: University of Chicago Press.

Toledo, L. F., Becker, C. G., Haddad, C. F. B., & Zamudio, K. R. (2014). Rarity as an indicator of endangerment in Neotropical frogs. *Biological Conservation*, *179*, 54–62. https://doi.org/10.1016/j.biocon.2014.08.012

Trimble, M. J. & van Aarde, R. J. (2014). Amphibian and reptile communities and functional groups over a land-use gradient in a coastal tropical forest landscape of high richness and endemicity. *Animal Conservation*, *17*(5), 441–453. https://doi.org/10.1111/acv.12111

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., & Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecology Letters*, *8*(8), 857–874. https://doi.org/10.1111/j.1461-0248.2005.00782.x

Tscharntke, T., Tylianakis, J. M., Rand, T. A., Didham, R. K., Fahrig, L., Batáry, P., … Westphal, C. (2012). Landscape moderation of biodiversity patterns and processes - eight hypotheses. *Biological Reviews*, *87*(3), 661–685. https://doi.org/10.1111/j.1469-185X.2011.00216.x

Tsianou, M. A. & Kallimanis, A. S. (2016). Different species traits produce diverse spatial functional diversity patterns of amphibians. *Biodiversity and Conservation*, *25*(1), 117–132. https://doi.org/10.1007/s10531-015-1038-x

UNESCO. (2018). Mount Nimba Strict Nature Reserve. Retrieved December 1, 2021 from https://whc.unesco.org

Urbina-Cardona, N. (2008). Conservation of Neotropical herpetofauna: research trends and challenges. *Tropical Conservation Science*, *1*(4), 359–375. https://doi.org/10.1177/194008290800100405

Urbina-Cardona, N., Bernal, E. A., Giraldo-Echeverry, N., & Echeverry-Alcendra, A. (2015). El monitoreo de herpetofauna en los procesos de restauración ecológica: indicadores y métodos. In M. Aguilar-Garavito & W. Ramirez (Eds.), *Monitoreo a procesos de restauración ecológica aplicado a ecosistemas terrestres* (pp. 134–147). DC, Colombia: Instituto de Investigación de Recursos Biológicos Alexander von Humboldt. https://doi.org/10.13140/RG.2.1.4400.7129

Urbina-Cardona, N. & Flores-Villela, O. (2010). Ecological-niche modeling and prioritization of conservation-area networks for Mexican herpetofauna. *Conservation Biology*, *24*(4), 1031–1041. https://doi.org/10.1111/j.1523-1739.2009.01432.x

Urbina-Cardona, N. & Londoño-M, M. C. (2003). Distribución de la comunidad de herpetofauna asociada a cuatro áreas con diferente grado de perturbación en la Isla Gorgona, Pacífico colombiano. *Revista de La Academia Colombiana de Ciencias*, *27*(102), 105–113.

Urbina-Cardona, N. & Loyola, R. D. (2008). Applying niche-based models to predict Endangered-Hylid potential distributions: are Neotropical protected areas effective enough? *Tropical Conservation Science*, *1*(4), 417–445. https://doi.org/10.1177/194008290800100408

Urbina-Cardona, N., Olivares-Pérez, M. & Reynoso, V. H. (2006). Herpetofauna diversity and microenvironment correlates across a pasture-edge-interior ecotone in tropical rainforest fragments in the Los Tuxtlas Biosphere Reserve of Veracruz, Mexico. *Biological Conservation*, *132*, 61–75. https://doi.org/10.1016/j.biocon.2006.03.014

Valdez, J. W., Klop-Toker, K., Stockwell, M. P., Fardell, L., Clulow, S., Clulow, J., & Mahony, M. J. (2019). Informing compensatory habitat creation with experimental trials: A 3-year study of a threatened amphibian. *Oryx*, *53*(2), 310–320. https://doi.org/10.1017/S0030605317000394

Valencia-Aguilar, A., Cortés-Gómez, A. M. & Ruiz-Agudelo, C. A. (2013). Ecosystem services provided by amphibians and reptiles in Neotropical ecosystems. *International Journal of Biodiversity Science, Ecosystem Services and Management*, *9*(3), 257–272. https://doi.org/10.1080/21513732.2013.821168

Van Buskirk, J. (2012). Permeability of the landscape matrix between amphibian breeding sites. *Ecology and Evolution*, *2*(12), 3160–3167. https://doi.org/10.1002/ece3.424

van der Ree, R., Jaeger, J. A. G., van der Grift, E. A., & Clevenger, A. P. (2011). Effects of roads and traffic on wildlife populations and landscape function: road ecology is moving toward larger scales. *Ecology and Society*, 16(1), 48. https://doi.org/10.5751/ES-03982-160148

Van Dyke, F., Berthel, A., Harju, S. M., Lamb, R. L., Thompson, D., Ryan, J., … Dreyer, G. (2017). Amphibians in forest pools: Does habitat clustering affect community diversity and dynamics? *Ecosphere*, 8(2), e01671. https://doi.org/10.1002/ecs2.1671

van Wilgen, B. W. (2009). The evolution of fire and invasive alien plant management practices in fynbos. *South African Journal of Science*, 105, 335–342. https://doi.org/10.4102/sajs.v105i9/10.106

Vannini, A., Bruni, G., Ricciardi, G., Platania, L., Mori, E., & Tricarico, E. (2018). *Gambusia holbrooki,* the 'tadpolefish': The impact of its predatory behaviour on four protected species of European amphibians. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *28*(2), 476–484. https://doi.org/10.1002/aqc.2880

Varumo, L., Yaneva, R., Koppel, T., Koskela, I. M., Garcia, M. C., Sozzo, S., … Dictor, M. C. (2020). Perspectives on citizen engagement for the EU post-2020 biodiversity strategy: an empirical study. *Sustainability*, *12*(4), 1532. https://doi.org/10.3390/su12041532

Vasconcelos, T. S., Santos, T. G., Rossa-Feres, D. C., & Haddad, C. F. B. (2009). Influence of the environmental heterogeneity of breeding ponds on anuran assemblages from southeastern Brazil. *Canadian Journal of Zoology*, *87*(8), 699–707. https://doi.org/10.1139/Z09-058

Venter, O., Fuller, R. A., Segan, D. B., Carwardine, J., Brooks, T., Butchart, S. H. M., … Watson, J. E. M. (2014). Targeting global protected area expansion for imperiled biodiversity. *PLoS Biology*, *12*(6), e1001891. https://doi.org/10.1371/journal.pbio.1001891

Vergara-Ríos, D., Montes-Correa, A. C., Urbina-Cardona, J. N., De Luque-Villa, M., Cattan, P. E., & Granda-Rodríguez, H. D. (2021). Local community knowledge and perceptions in the Colombian Caribbean towards amphibians in urban and rural settings: Tools for biological conservation. *Ethnobiology and Conservation*, *10*, 24. https://doi.org/10.15451/ec2021-05-10.24-1-22

Walker, L. R., Wardle, D. A., Bardgett, R. D., & Clarkson, B. D. (2010). The use of chronosequences in studies of ecological succession and soil development. *Journal of Ecology*, *98*(4), 725–736. https://doi.org/10.1111/j.1365-2745.2010.01664.x

Watling, J. I., Arroyo-Rodríguez, V., Pfeifer, M., Baeten, L., Banks-Leite, C., Cisneros, L. M., … Fahrig, L. (2020). Support for the habitat amount hypothesis from a global synthesis of species density studies. *Ecology Letters*, *23*(4), 674–681. https://doi.org/10.1111/ele.13471

Watling, J. I. & Braga, L. (2015). Desiccation resistance explains amphibian distributions in a fragmented tropical forest landscape. *Landscape Ecology*, *30*(8), 1449–1459. https://doi.org/10.1007/s10980-015-0198-0

Watling, J. I., Nowakowski, A. J., Donnelly, M. A., & Orrock, J. L. (2011). Meta-analysis reveals the importance of matrix composition for animals in fragmented habitat. *Global Ecology and Biogeography*, *20*(2), 209–217. https://doi.org/10.1111/j.1466-8238.2010.00586.x

Wells, K. D. (2007). *The ecology and behavior of amphibians*. Chicago, USA: University of Chicago Press.

Whitfield, S. M. & Pierce, M. S. F. (2005). Tree buttress microhabitat use by a Neotropical leaf-litter herpetofauna. *Journal of Herpetology*, *39*(2), 192–198. https://doi.org/10.1670/219-04A

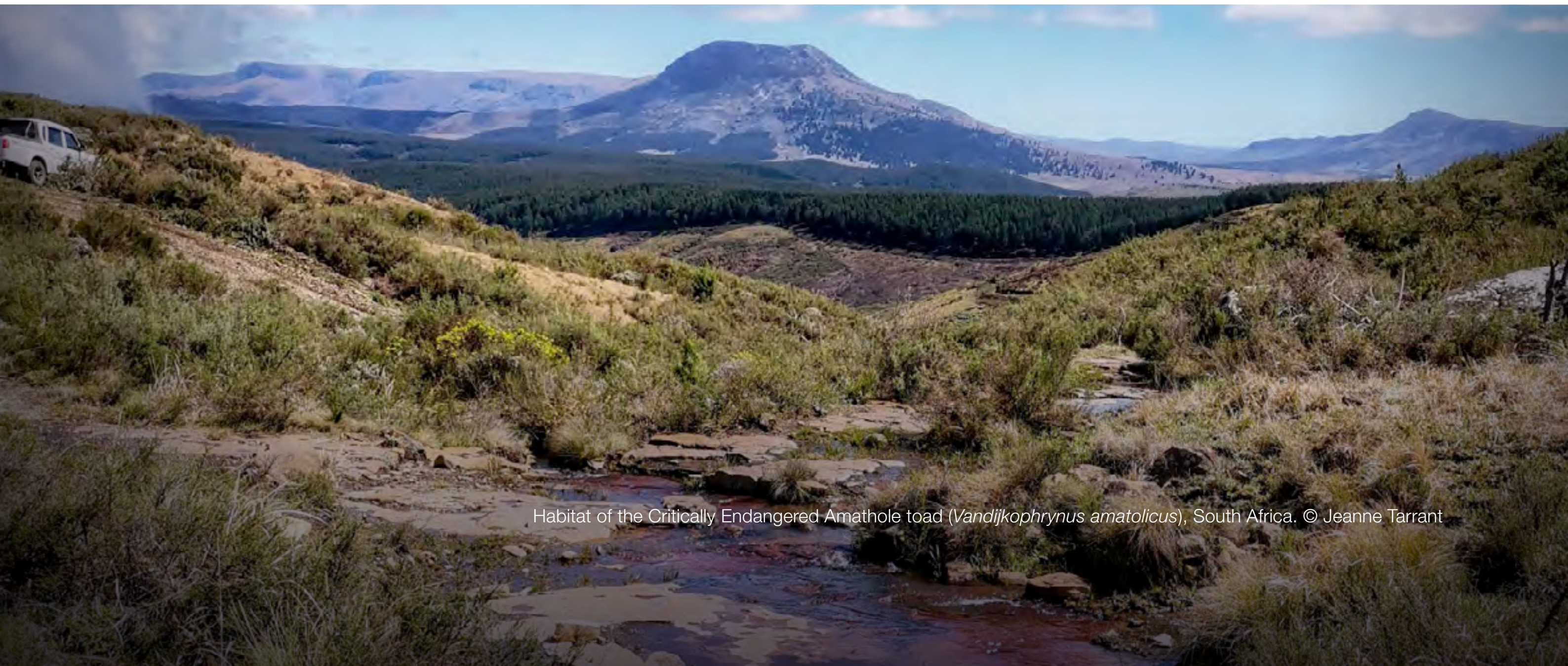
World Bank. (2019). *Forest-smart mining: Offset case studies*. 76. Retrieved October 11, 2022 from https://www.profor.info/sites/profor.info/files/FOREST-SMART MINING_Offset Case Studies.pdf

World Conservation Society. (2020). *Proyecto Estrategia de Conservación de Anfibios de Colombia*. Bogotá, Colombia: World Conservation Society.

WWF. (2020). *Living Planet Report 2020: Bending the curve of biodiversity loss*. Gland, Switzerland: World Wildlife Fund.

Zabala-Forero, F. & Urbina-Cardona, N. (2021). Respuestas de la diversidad taxonómica y funcional a la transformación del paisaje: relación de los ensamblajes de anfibios con cambios en el uso y cobertura del suelo. *Revista Mexicana de Biodiversidad*, *92*, e923443. https://doi.org/10.22201/ib.20078706e.2021.92.3443

zu Ermgassen, S. O. S. E., Baker, J., Griffiths, R. A., Strange, N., Struebig, M. J., & Bull, J. W. (2019). The ecological outcomes of biodiversity offsets under "no net loss" policies: A global review. *Conservation Letters*, 12(6), e12664. https://doi.org/10.1111/conl.1266



Habitat of the Critically Endangered Amathole toad (*Vandijkophrynus amatolicus*), South Africa. © Jeanne Tarrant

Eleutherodactylus nortoni is a Critically Endangered species that occurs in the Tiburon Peninsula of Haiti and Sierra de Bahoruco in the Dominican Republic and is threatened by the loss of the few remnant cloud forests where it lives. © Ariadne Angulo