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# The conservation of abundance in non-threatened species

**Abstract**

Human modification of the environment is driving declines in population size and distributional extent of much of the world's biota. These declines extend to many of the most abundant and widespread species, where proportionally small declines can result in the loss of vast numbers of individuals, biomass, and interactions. These losses could have major localised effects on ecological and cultural processes and services, without elevating a species' global extinction risk. While most conservation effort is directed at species threatened with extinction in the very near-term, the value of retaining abundance regardless of global extinction risk can readily be justified against many biodiversity or ecosystem-service metrics, including cultural services, at scales from local to global. Here, we characterise the specific challenges facing the identification of conservation priorities for abundant and widespread species, addressing the broader consequences of declines that differ in magnitude and spatial distribution, and where negative effects may be disconnected from the threat process driving declines. Conservation prioritisation for abundant and widespread species shares greater similarity with invasive species risk assessments than extinction risk assessments, because of the importance of local context and per capita effects of abundance on other species. Because priority in conservation planning is usually focussed on preventing extinction of threatened species, the rationale and objectives for incorporating declines in non-threatened species must be clearly articulated, going beyond just extinction risk to encompass the range of harmful effects that are likely to be realised if declines persist or are not reversed. Research should focus on characterising the effects of the local decline in species that are globally non-threatened across a range of ecosystem-services, and quantifying the spatial distribution of these effects through the distribution of abundance. The conservation case for conserving abundance in non-threatened species can be made most powerfully with greater knowledge of the costs of losing this abundance.

## Introduction

Staggering numbers of individuals are being lost as species decline. Between 1980 and 2009, European birds suffered a net loss of ~421 million individuals, equating to a 20% decline in total avian abundance in just three decades (Inger et al. 2015). The vast majority of these individuals (83%) were lost from populations of just a few abundant and widespread species. Similar patterns are evident across the globe and reflect the on-going erosion of abundance affecting a wide range of taxa (Woinarski et al. 2015; Ceballos et al. 2017; Hallmann et al. 2017). The conservation of abundance and biomass represents a major ecological problem that is often not well aligned with extinction-focused conservation targets (Gregory et al. 2005). For example, among the world's bird species with declining population trends, those that are most abundant are overwhelmingly assessed as *Least Concern* with respect to extinction risk (Fig. 1a). Proportionally small declines in these species can, however, result in the loss of vast numbers of individuals and biomass and still not trigger classification on the IUCN Red List (Fig. 1b). For example, the willow warbler *Phylloscopus trochilus* has been estimated to have a global population size of 500 million mature individuals (Fig. 1a), approximating to 5000 tonnes of biomass (~10g adult weight). A 10% decline in population size over 10 years equates to a loss of ~50 million individuals and ~500 tonnes of biomass.

The effects of such losses on extinction risk for the declining species are negligible, yet the broader ecological consequences could be substantial (Van Bael et al. 2008; Säterberg et al. 2013; Maas et al. 2016; Gaston et al. 2018). The arguments for the importance of keeping 'common' species 'common' and the conservation value of abundance *per se* have been well laid out in several notable papers (Gaston & Fuller 2008; Gaston 2010, 2011; Redford et al. 2013). Declines in abundant and widespread species, referred to from here on as 'depletion', are concerning because as populations are depleted, the values, processes and services supported by their abundance and biomass are increasingly likely to be affected (Gaston et al. 2018). All else being equal, the processes and services supported by abundant and widespread species are likely to be of greater magnitude and resonance than those provided by rare and range-restricted species (Winfree et al. 2015; Genung et al. 2017). Furthermore, evidence suggests that restoring large populations (e.g. through habitat restoration) is often unsuccessful due to the scale of management required to achieve positive and sustained population growth (Baker et al. 2012). This creates an imperative to identify declining populations long before they become threatened so that appropriate conservation management and legislation can be targeted to halt further losses.

It is difficult, therefore, to argue against the value of abundance, and the representation of this abundance across broad geographic space, as an important conservation asset independent of global extinction risk. Declines in non-threatened species, however, have received much less attention than species threatened with global extinction in the near future (Redford et al. 2013). Conservation targets are strongly determined by, and evaluated against, global extinction risk (e.g. Hoffmann et al. 2010; Waldron et al. 2017). The exceptions, such as the use of long-term common bird species monitoring to inform conservation policy in the European Union (Gregory et al. 2008), emphasise the dangers of neglecting non-threatened species. The large declines suffered by many common birds (Inger et al. 2015) might have been even more severe without the introduction of appropriate legislation (Sanderson et al. 2015). There are challenges to elevating the importance of conserving the abundance and distributional characteristics of non-threatened species when many other species are far nearer to global extinction. Where conservation resources are limited, trade-offs among different values are necessary and, for many justifiable reasons, priority is often given to those species most threatened with extinction. Against this backdrop, it is important to determine and anticipate when depletion of non-threatened species becomes an urgent conservation priority in local, regional and global contexts.

Here, we characterise the specific challenges facing the conservation of abundance in non-threatened species. Evaluating the significance of the loss of abundance requires understanding the broader consequences of declines of different magnitudes, at different spatial scales, and for species with different trait sets (e.g. body mass, life form, reproductive strategy, generation lengths, foraging guilds, movement behaviours). We focus here specifically on those species that are sufficiently abundant and widespread that they are unlikely to be classified as globally threatened under current extinction-risk frameworks, even if they are suffering marked losses in biomass or numbers of individuals. These species would be assessed as *Least Concern* on the *IUCN Red List* based on geographic range and population size criteria alone, and will have population declining by less than 20% per (the longer of) 10 years or three generations (demarcated in Figure 1b).

**Depletion and its consequences**

Many of the clearest examples of the effects of depletion on ecosystem services have been observed with localised (or global) extinction of a species, typically following from human exploitation and disturbance (e.g. Soulé et al. 2005, examples therein). These natural experiments reveal the functional roles that species play in an ecosystem through the consequences of their absence. Yet, because of the elevated sensitivity of large-biomass species to extinction, examples are dominated by trophic cascades resulting from the loss of top-predators or large-herbivores (Estes et al. 2011). Large-biomass species, despite typically occurring at low densities, often exert a large *per capita* effect on a broad range of services and the effects of their decline are often dramatic (Estes et al. 2011). For example, changes in the density of sea otters *Enhydra lutris* can rapidly transform a barren ocean floor to a dense kelp forest as the grazing pressure on kelp is reduce through predation by otters on benthic herbivores (Estes et al. 2010). Such effects can be realised with even small changes in the abundance and long before the species is lost from the ecosystem (Estes et al. 2010). Säterberg et al. (2013), for example, showed that more than a quarter of species in real food-webs become functionally extinct before losing 30% of their individuals. Functional extinction, in this case defined as the population size below which another species goes extinction (but see McConkey & O'Farrill (2015) for discussion of complete vs. partial functional extinction), was most likely for large-biomass species.

Equally strong effects, however, can be mediated through sheer numerical abundance, through the total effect of losing many interactions of weak *per capita* strength. For example, local abundance plays an important role in effective seed dispersal (Schupp et al. 2010) and pollination (Garibaldi et al. 2013), and provides a prey-base for higher trophic levels (Delibes-Mateos et al. 2008; Smith et al 2011). The decline in abundance of low trophic level fish species (e.g. anchovy and sardines) in many fisheries has severe impacts on larger predatory species (Smith et al 2011), including fish, birds and mammals (Kaplan et al. 2017). Pulsed abundance can also have major functional effects on ecosystem services (Bauer & Hoye 2014). For example, migrant insectivores can provide economically important top-down control of invertebrate herbivory (López-Hoffman et al. 2017a, 2017b). The life-history traits of many highly abundant species naturally produce large fluctuations in abundance, but evidence suggests that many species are no longer bouncing back after experiencing periods of stringency (Cornulier et al. 2013; Selwood et al. 2015).

Depletion and local extinction are not necessarily synonymous (e.g. Santamaría et al. 2016), but few studies measure the change in ecosystem-service provision as a species' abundance changes. Understanding the form of these function-abundance relationships is critical to predicting when depletion will affect an ecosystem service, with the form of the relationship affecting the sensitivity of an ecosystem function to declines in abundance (Fig. 2a,b). Across a range of ecosystem-services provided by birds, including supporting, regulating and cultural services, Gaston et al. (2018) found the function-abundance relationships to be mostly positive and non-linear. From these examples, it seems likely that many abundance-service relationships will exhibit strong thresholds. It is important to note, however, that these insights are built on relatively few empirical measurements of abundance-service relationships. How these observations generalize more widely is unknown, although the broad pattern of positive and non-linear seems evident across a range of species and services, including seed dispersal (McConkey & Drake 2006) and meso-predator suppression (Newsome et al. 2017).

The drivers of depletion can also act as a selection pressure that drives changes in traits associated with responses to environment change and to effects on ecosystem-services (e.g. Pinsky & Palumb 2014), especially where the loss of individuals is non-random across the population. For example, the preferential removal of large individuals from frugivorous fish populations has resulted in change in size structure of the populations (Costa-Pereira & Galetti 2015). The effects of downsizing the population include the disruption of the mutualistic relationship between tree species that rely on the largest individual fish to disperse their fruit. Whether or not an evolutionary response has been produced in the fish population, the long-term ecological effects for the ecosystem will be large because of the loss of this important function (Costa-Pereira & Galetti 2015). Generally, the presence of many individuals in a population may mask changes to genetic diversity, population structure, or in the distribution of traits. In addition to affecting ecosystem-service provision, these changes might silently increase the vulnerability of the species to environmental changes (Walsh et al. 2006).

**Predicting the effects of depletion**

The loss of individuals from an ecosystem will produce an effect on an ecosystem-service (inclusive of supporting, regulating and provisioning services) that is determined by the average *per capita* effect of the species on this service, the abundance and change in abundance of the species in the ecosystem, and also the form of the relationship between abundance and the service (Fig. 2b; Díaz et al. 2013). Thus, predicting the effects of depletion requires the integration of a suite of species- and context-specific data to scale *per capita* information to understand likely effects of declines of different magnitudes.

System-specific models, for example, mass-balanced food-web models used to predict the flow of energy across trophic levels, have had some success in predicting depletion of both high and low trophic level species (Heithaus et al. 2008). These approaches typically require large amounts of data to parameterise and tend to be associated with economically important components of ecosystems (e.g. fisheries and forestry). The search for more generalizable frameworks for predicting the consequences of environmental change on species and ecosystem function has led to intense focus on species traits (Díaz et al. 2013). Recent efforts to link species traits to specific threats responses ('response-traits') and then to the functional effects resulting from local extinction ('effect traits') have yet to produce highly predictive models (Bartomeus et al. 2018). Realised *per capita* effects have been shown to be highly context specific, and dependent on the ecosystem-service being measured, along with the species-specific traits ('effect traits') that underpin its impact on this service (Wohlgemuth et al. 2017). This is also likely to be true for the form of the abundance-service relationship (Gaston et al. 2018). Thus, the complexity of ecological networks has led to few broadly applicable generalisations, especially when accounting for temporally varying community composition and relative abundance within the species pool, and the range of different perturbations that can affect a system (Santamaría et al. 2016; Bartomeus et al. 2018). As a consequence, at least in the near-term, it may be difficult to make highly specific prediction about the functional consequences resulting from the depletion of specific species.

In general, however, two important insights have been gained from observational and theoretical studies on function-abundance relationships that should inform the motivation for conserving abundance in non-threatened species. These are that (i) functional extinction of a declining population typically occurs before extinction of the species (Galetti et al. 2013; Säterberg et al. 2013), and (ii) functional loss can occur with even proportionally small changes in local abundance (Säterberg et al. 2013; McConkey & O'Farrill 2015). The potential for sudden functional collapse with proportionally small changes in abundance illustrates the risk that comes when little conservation concern is assigned to declines in non-threatened species. This risk is compounded by the difficulties of restoring abundance and the problem that the restoration of abundance does not always restore lost function, e.g. because of hysteresis effects caused by demographic and evolutionary shifts in a system's state (Walsh et al. 2006). Identifying when a species' abundance, and its depletion, is consequential is difficult in many cases, although the complete loss of species with a large body mass, those occupying high trophic levels, or those that dominate in terms of biomass should always raise some concern. More generally, given the emerging understanding of functional importance of abundant and widespread species, a precautionary approach that *a priori* assumes high functional value for species' abundance independent of extinction risk should be adopted as the default position. Conservation prioritisation in this context could perhaps focus on the likelihood of recovering populations to historical abundance baselines, with prioritisation going to species where the recovery of abundance will be most difficult, due equally to political and societal factors, as well as ecological.

**Local depletion as a conservation concern**

The translation of *per capita* ecological effect into population and species-level contributions to ecosystem-services is modulated by spatial and temporal variation in its abundance. Abundance is distributed unevenly across a species' range and can also vary strongly across temporal scales driven, for example, by daily, seasonal, or multi-decadal variation in environmental conditions (Sagarin et al. 2006; Saunders et al. 2017). Higher density and larger distributional extent has the effect of scaling up the number and cumulative strength of interactions associated a species and, potentially, scaling up the impact of losing the species' abundance; although there are many species-specific and distributional factors that will modulate the impact from the depletion of a given species. Depending on the major drivers, the reduction of abundance across a species' range could vary from a highly aggregated to a widely dispersed pattern, and may affect marginal or core areas more strongly (Fig. 2c; Channell & Lomolino 2000). For example, long-term population trends, and therefore conservation priorities, for Australian birds varies considerably between regions (e.g. Fig. 3), likely driven by differences in land use and environmental perturbations across large spatial scales including climate variability, changes to fire regimes and the presence of invasive predators (Evans et al. 2011). At local scales, rapid declines in populations could greatly affect provision of local ecosystem services even if there are negligible effects on global population size. The loss of large frugivores from small forest fragments due to patch area effects can affect seed dispersal and (ultimately) community composition within fragments, even if frugivore abundance seems little changed on average across a larger spatial grain (Galetti et al. 2013). This example highlights the importance of understanding (and quantify) the functional roles of species and how the consequences of depletion scale across the footprint of a species' distribution. This same example illustrates how coarse grain assessments of depletion are likely to overlook local functional extinction, and generally underestimate the importance of depletion.

Declines in local abundance can also have functional consequences that scale beyond the species' occupied range, creating an effect footprint larger than the area of occupancy. This might happen, for example, where depletion of burrowing mammals alters hydrological flows by reducing water infiltration rates, potentially influencing hydrological processes at catchment scales (Wilson & Smith 2015). Many species occur in multiple landscapes through their life cycle, and declines driven by environmental changes in one landscape may have consequences elsewhere (Lundberg & Moberg

2003). The importance of connectivity across large distances (i.e. telecoupling) for maintaining ecological processes and ecosystem services are increasingly being recognised (López-Hoffman et al. 2017a). For example, the pest control (affecting cotton production) and ecotourism services provided by the Mexican free-tailed bat *Tadarida brasiliensis mexicana* in parts of the United States are heavily dependent on maintaining wintering habitat in central and southern Mexico (López-Hoffman et al. 2017a; 2017b). Understanding the spatial processes that maintain species' abundance, and identifying spatial location of declines and movement behaviour of species is vital for understanding and predicting the distribution of potential effects from depletion, and ultimately the wider risk associated with local declines in a species.

Proportional declines measured at coarse scales and reported for a species' global population provide limited and inadequate information to contextualise the significance of declines in abundant and widespread species. From the perspective of species extinction risk, declines that are proportionally small relative to the global population size might be viewed as largely inconsequential across the species' entire occupied range. This might overlook large localised population declines that have substantial, and perhaps widespread, consequences for ecosystem services (López-Hoffman et al. 2017a). This is especially likely when conservation priorities are being developed using spatially coarse information, lacking details on the status of local biodiversity (Dolman et al. 2012) and the linkages between the location of abundance and its effects (López-Hoffman et al. 2017a). Even where evidence for local depletion exists, management intervention aimed at otherwise abundant and widespread species can be perceived as misplaced (Battisti & Fanelli 2015). Such arguments must be reframed in the context of conserving abundance, both because spatial representation is a critical element in maintaining abundance (Gaston & Blackburn 1996) and because the wider abundance of a species has little relevance where local declines precipitate undesired consequences.

## Integrating non-threatened species into conservation priorities

Concern for abundant and widespread species is not controversial. We argue, however, that the relative urgency and importance of declines in the abundance of non-threatened species is difficult to place in context against other compelling conservation priorities because the motivation and knowledge base necessary to provide this context is underdeveloped compared to those other priorities (e.g. extinction risk, invasive species). Below, we discuss how the conservation of abundance in non-threatened species might be integrated into global conservation priorities.

### *Prioritising non-threatened species*

The objectives of conserving threatened species are generally well-defined. This, combined with a well-established empirical and theoretical understanding of the drivers of extinction, has resulted in the success of threatened species lists for identifying and prioritising threatened species conservation (Rodrigues et al. 2006). While a formal 'threat' listing for 'common' species in decline has been suggested (Gaston & Fuller 2008), the basis for such a risk-framework is not as obvious as the formulation of the IUCN Red List for Threatened Species. The IUCN Red List classifies extinction risk using species-specific population (e.g. trend, distribution, population size) and trait (e.g. generation length) information (Mace et al. 2008). *Risk* is the product of *Impact* and the *Likelihood* of this impact occurring. The Red List is similarly devised to quantify extinction risk, but because the impact is considered constant across species (i.e. the impact of extinction for that species is the same across all species), risk is simply equal to the likelihood of extinction. For abundant and widespread species, the consequences of a decline for other species – including humans – are of over-riding importance and these factors are not easily formulated into a risk assessment with global relevance. These effects are spatially explicit, depending on the distribution of abundance and the distribution of the functional effects of this abundance. Greatest concern would be raised by declines, or projected declines, in a species that exerts a strong functional effect on a valued ecosystem-service that is likely to be lost due to declines (Fig. 2bi-iii), regardless of a species' broader extinction risk.

As such, conservation prioritisation for abundant and widespread species shares greater similarity with invasive species risk assessments than extinction risk assessments, because the focus must be on location and *per capita* effect of a species' abundance on other species or services rather than the global extinction risk of the species itself (e.g. Parker et al. 1999). Invasive species risk assessments, which are typically produced for discrete geographic areas (e.g. protected areas to countries) rather than globally, might provide a template for developing approaches for prioritise risk from depletion, especially in the estimation of impact across space related to per capita or per area effects (Barney et al. 2013). For example, figure 2 shows how local impacts from depletion (e.g. at the scale of, for example, protected areas or ecoregions), could be predicted for a species using demographic forecasts (Fig. 2a) and information on the form and strength of the abundance-services relationship (Fig. 2b). The integration of this information, perhaps across multiple ecosystem-services (Barney et al. 2013), could be used to assess the relative depletion risk for the most locally common species. As for invasive species risk assessments, data limitations provide a major obstacle to effective implementation. Below we discuss these data gaps.

*Knowledge gaps*

To implement a prioritisation approach for depletion, or even simply to increase the awareness of the problem, much more information is required on the per capita effects and abundance-service relationships of the most widespread and abundant species. Soule *et al.* (2005) advocated the preservation of functional abundance for strongly interacting species, identified as those that cause impacts on other species when they decline in local abundance. The suggestion here is that species can be identified as strongly interacting based on effects resulting from their decline in abundance, and then minimum abundance thresholds can be established form management. We currently lack much of the fundamental information to identifying these species and thresholds. Without such information, there is no way of identifying priorities across species and in different environmental contexts. Empirical and theoretical studies must be employed to greater effect in understanding the consequences of depletion so that choices and trade-offs in conservation investment can be informed by an evaluation of risk from alternative actions (Säterberg et al. 2013; Harvey et al. 2017; Gaston et al. 2018). For many ecosystem-services, the abundance-service relationships and per capita effect might be predicted based on existing ecological theory, e.g. using well-studied species' functional-responses to resource availability (e.g. Dawes & Souza 2013). The null position, in the absence of such information for most species and contexts, must be to assume all abundant and widespread species are functionally important (Gaston & Fuller 2008). In this case, information on the likelihood of recovering abundance once lost would be particularly valuable for identifying conservation priorities.

The availability of spatially and temporally explicit information on abundance must also be a priority to enable predictions of where and when services are likely to be impacted by depletion. Population trends of widespread species can vary greatly across their range (Devenish et al. 2017) and impacts can be distributed away from the drivers of depletion due to species' movements or other environmental flows. Information collected to assess extinction risk typically does not provide the spatial information necessary to place declines in abundant and widespread species in the appropriate context vis-à-vis the size and distribution of effects on ecosystem services. Extent of occurrence and global population trends do not provide the spatial resolution necessary to detect such variation, and area of occupancy at resolutions suitable for conservation planning and threat assessment (e.g. typically <10 km; Keith et al. 2018) is often poorly estimated or unknown across all or part of species' ranges. The collation of information on the distribution and abundance of threatened species is made systematic through the framework of the IUCN Red List, but this does not necessarily provide the same motivation for the collection or collation of information on species that are abundant and widespread (e.g. Fig. 4). The need for basic distribution and demographic data is partly being addressed by citizen science initiatives and through the integration with remotely sensed environmental data (Jetz et al. 2012). Taxonomic and geographical biases mean that for the majority of species and in many parts of the world distribution and abundance data is not current available (Meyer et al. 2016).

Resolving spatial variation in occupancy or abundance, and the spatial variation in rates of declines, is vital for identifying the threats. While factors driving declines in threatened species may also be causing depletion in similar abundant and widespread species, such concordance cannot be assumed. Conservation strategies that perform well for threatened species will not necessarily align well with the measures needed to halt declines in abundant and widespread species. Much will depend on the drivers of declines. The establishment of protected areas is the dominant strategy for the conservation of many threatened species but, given that conservation reserve systems typically occupy only 10-20% of regions or ecosystems and that management of conservation reserves often targets threatened species within them, this is unlikely to prevent the erosion of abundance in more widespread species. The depletion of species outside of protected areas can even undermine the performance of the protected area. For example, the closure of a small South African fishery was not predicted to produce positive population growth for the endangered African penguin *Spheniscus demersus* because the population was dependent on prey availability across a scale larger than this single fishery (Sherley et al. 2015). The practical challenges of maintaining space for processes that promote abundance are higher for wide ranging species, which are often poorly served by existing protected areas and often require conservation across political boundaries (Runge et al. 2014). Some successes have resulted from reducing threats in the wider landscape (e.g. Gamero et al. 2017).

However, the continued decline of Europe's most common bird species, despite considerable targeted conservation management, indicates that research and policy initiatives in this area are a priority.

Perhaps the principal influence of the *IUCN Red List* on species conservation has been to focus attention on the issue of the emerging extinction crisis and to stimulate the collation of information to inform species-level threat assessment (Rodrigues et al. 2006). Similar attention might be focused on the emerging abundance crisis were the estimation and reporting of absolute changes in abundance and biomass routine (Gaston & Fuller 2008). The headline figures reported by Inger et al. (2015) are easy to interpret and leave no doubt about dire state of Europe's avifauna. Population size estimates are currently lacking for most species, except where economic value is linked to abundance (e.g. fisheries, forestry) or where density estimates are critical for management (i.e. livestock densities in protected areas). However, population sizes could be estimated for many species from even coarse-scale AOO and habitat/season specific estimates of population density (e.g. Hui et al. 2009). Abundance/biomass trends would convey a different, but equally powerful, message to that of existing population trend indices (e.g. World Wildlife Fund, 2016), which typically report the average rate of population change across a suite of species, without describing changes in total abundance and biomass.

**Conclusions**

The value of abundance can be justified from multiple perspectives. At present, it may be challenging to quantify and rank the ecological significance of depletion in abundant species and to define thresholds of depletion for conservation prioritisation. Abundant species often occur across vast geographic space with declines often driven by varied combinations of multiple interacting threats operating across different parts of their distributions. There are, however, clear steps that could be taken to better understand the status of abundant and widespread species, and of the relative priority for conservation action to address depletion in such species.

1) The quantification of the contribution of abundant and widespread species to ecological processes and services is necessary to understand and communicate the consequences of losing abundance in non-threatened species. This includes increasing the effort to quantify links between threatening processes and undesired effects of depletion where movement behaviour or other environmental flows create a spatial disconnect. A clearer understanding of the consequences of losing abundance in non-threatened species, even in a broad sense, will help to define the objectives of elevating the priority of these species relative to other conservation challenges.

2) The conservation of abundant and widespread species requires information on the spatial distribution and trends in abundance that can feed into conservation planning at scales from local to global. These requirements are not necessarily met by data collection and collation aimed at assessing species extinction risk, but should be identified specifically against the objectives aligned with the need for conserving abundance in non-threatened species (i.e. impacts on other species). Increasing citizen science and remote sensing data, along with advances in modelling species distributions and trends, will benefit these actions greatly. Emphasis should be placed on collecting information that can be used to estimate abundance (not just presence at a coarse scale), increasing data collection in under-sampled regions, and in establishing historical baselines for species' abundance to avoid shifting baseline problems when setting conservation goals.

3) More systematic assessments of the threats that are causing depletion in abundant and widespread species are required. Such assessment should complement assessments of threats that are causing imperilment of threatened species, and then be used to prioritise management responses that can best

address the needs for threat reduction for non-threatened and threatened species. Such assessments would guide when fundamentally different threat management responses are required for reversing declines in abundant and widespread species cf. rare species.

Finally, the conservation narrative, dominated by rarity and extinction risk, must be augmented with a message that highlights the ecological and societal value of abundance regardless of global extinction threat.

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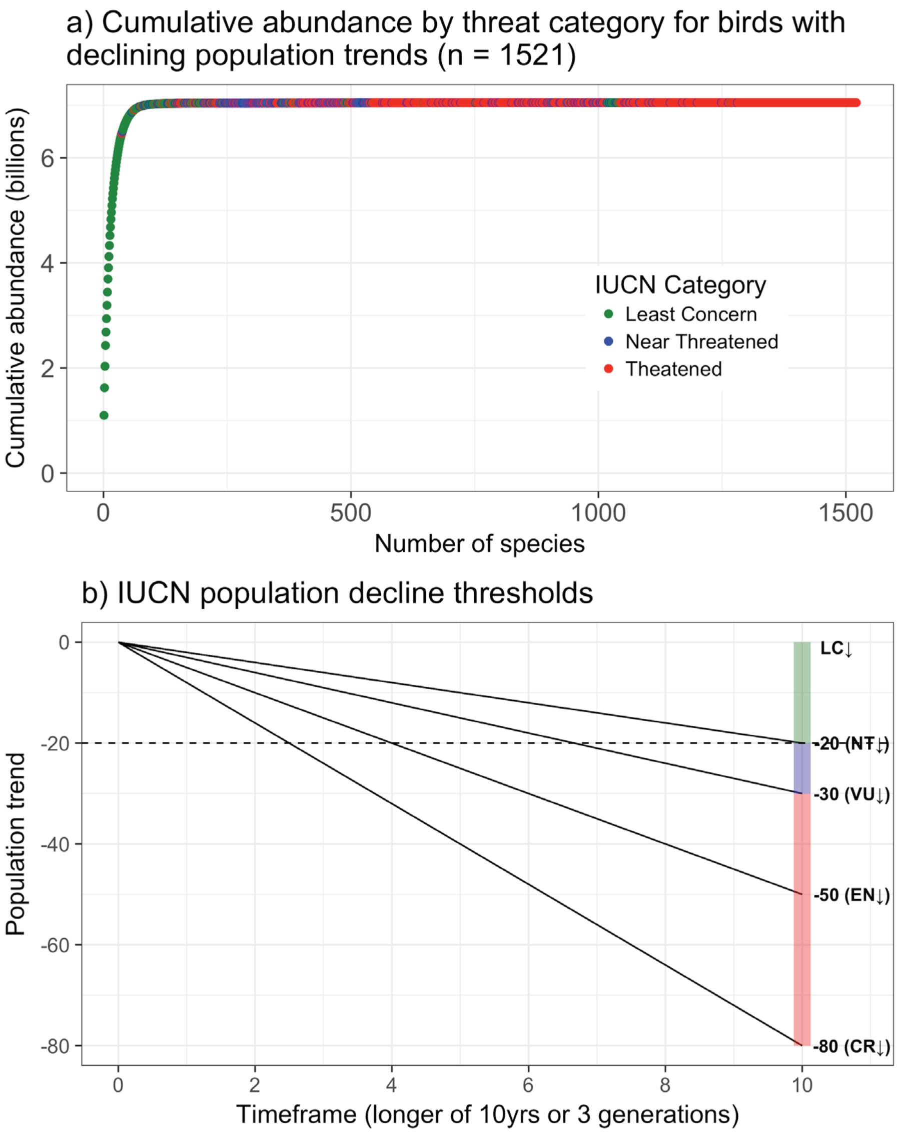
Figure 1. The conservation context of the decline in global abundance: (a) ca. 90% of the total abundance for the world's declining bird species (those with population size estimates, *n* = 2141) derives from only 108 species, almost exclusively assessed as Least Concern (LC = 100; NT = 4; TH [CR, EN, VU] = 4) on the *IUCN's Red List of Threatened species* (data supplied by BirdLife International April 2016). (b) In order for a species that is abundant and widespread to become listed as threatened under the IUCN's criteria it must have undergone a decline of at least 30% over the longer of 10 years or 3 generations. Thus, vast numbers of individual can be lost (and continue to be lost) from LC species without changing their extinction risk status.  CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern.

Figure 2. Functional effects of local depletion can occur prior to species loss. As a local population density declines through time (a) the emergence of consequences (b) measured against a functional value / ecosystem-service metric (e.g. species richness, carbon sequestration, economic value) will depend on the relationship between the species density (e.g. per capita effect) and the functional value metric (b). Non-linear relationships (i) and (iii) will be frequently observed because of complex density-dependent interactions among species, but linear 'mass effects' might also be observed where consequences are a function of biomass (ii). The size of the species' contribution to the overall ecosystem-service will vary, i.e. high (i) vs. low (iv), and this will affect the impact realised as the species declines. The sensitivity of the system or process to the depletion of abundance (i.e. 50% loss of local abundance, demarcated on (a) and (b) by ----) will depend on these relationships with, for example, functional loss occurring at lower levels of depletions across the forms (i) to (iii). For example, 50% loss of local abundance results in no function loss (i), partial function loss (ii), or complete function loss (iii). Together, the rate of population decline, the form of the abundance-function relationship, and the size of the total effect of abundance could be formulated into a risk framework to assess relative local risk among species (c).

Figure 3. Species' long-term population trends can differ in sign across their range, as illustrated by the degree of consensus in the direction of bird population trends (1999-2013) between major biogeographic regions of south-eastern Australia. Values given in the region intersections show the percentage of species that show a consistent trend direction (positive, negative, stable) between these regions. For example, of the 45 species found in all three regions, the regional trends of 33% of these species show consistency in direction across the three regions. n = total number of shared species between regions given. *N* = number of species in each regions with an estimated trend. Data from State of Australian Birds 2015 regional reports (http://birdlife.org.au/education-publications/publications/state-of-australias-birds).

Figure 4. Species threat status can motivate the collection and collation of population data, as illustrated by the percentage of species in each extinction risk category, across five major taxonomic classes, for which there is an estimate of (a) Area of Occupancy (AOO), (b) Extent of Occurrence (EOO) and (c) Population size. The total number of species assessed in each taxonomic class and threat category is given in (a) above the relevant column. Data come from two major databases (IUCN as of March 2016; Aves data provide by BirdLife International April 2016).

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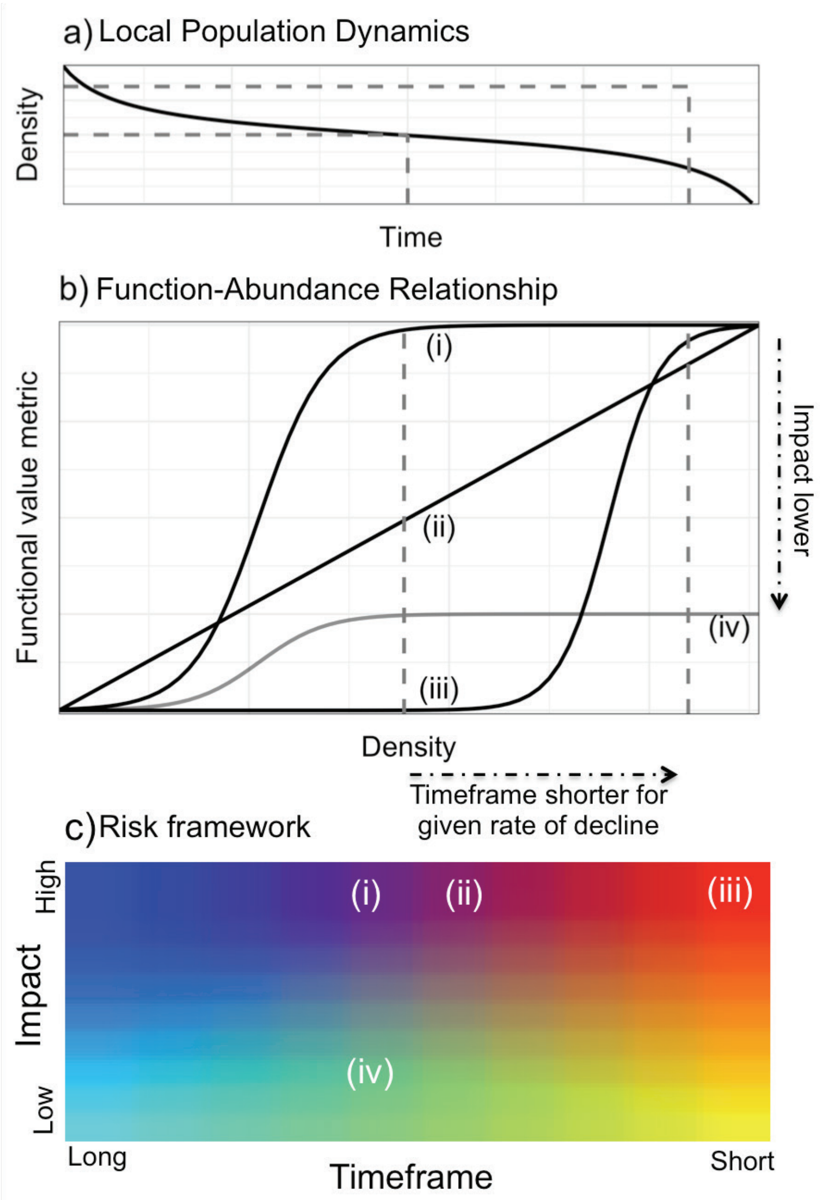


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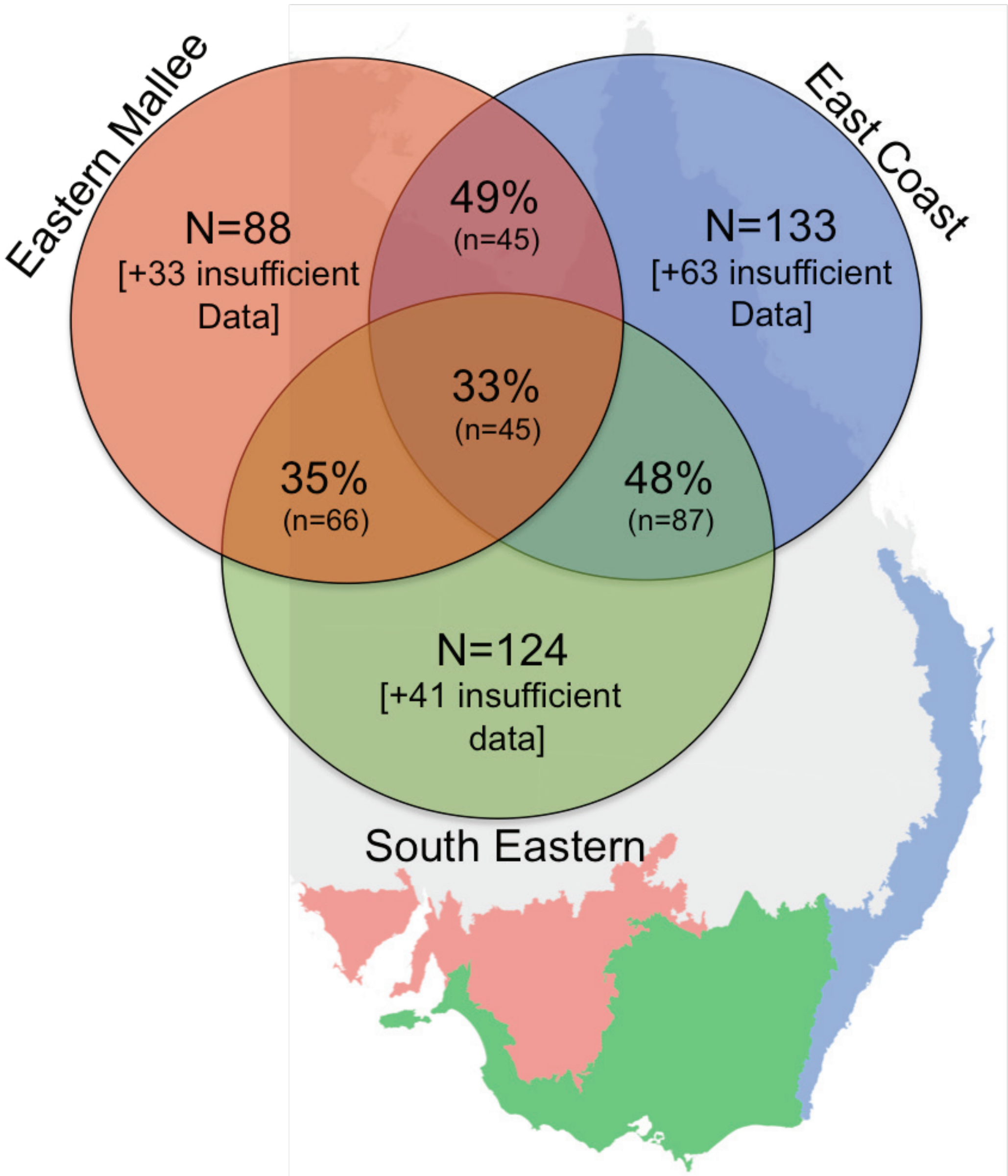


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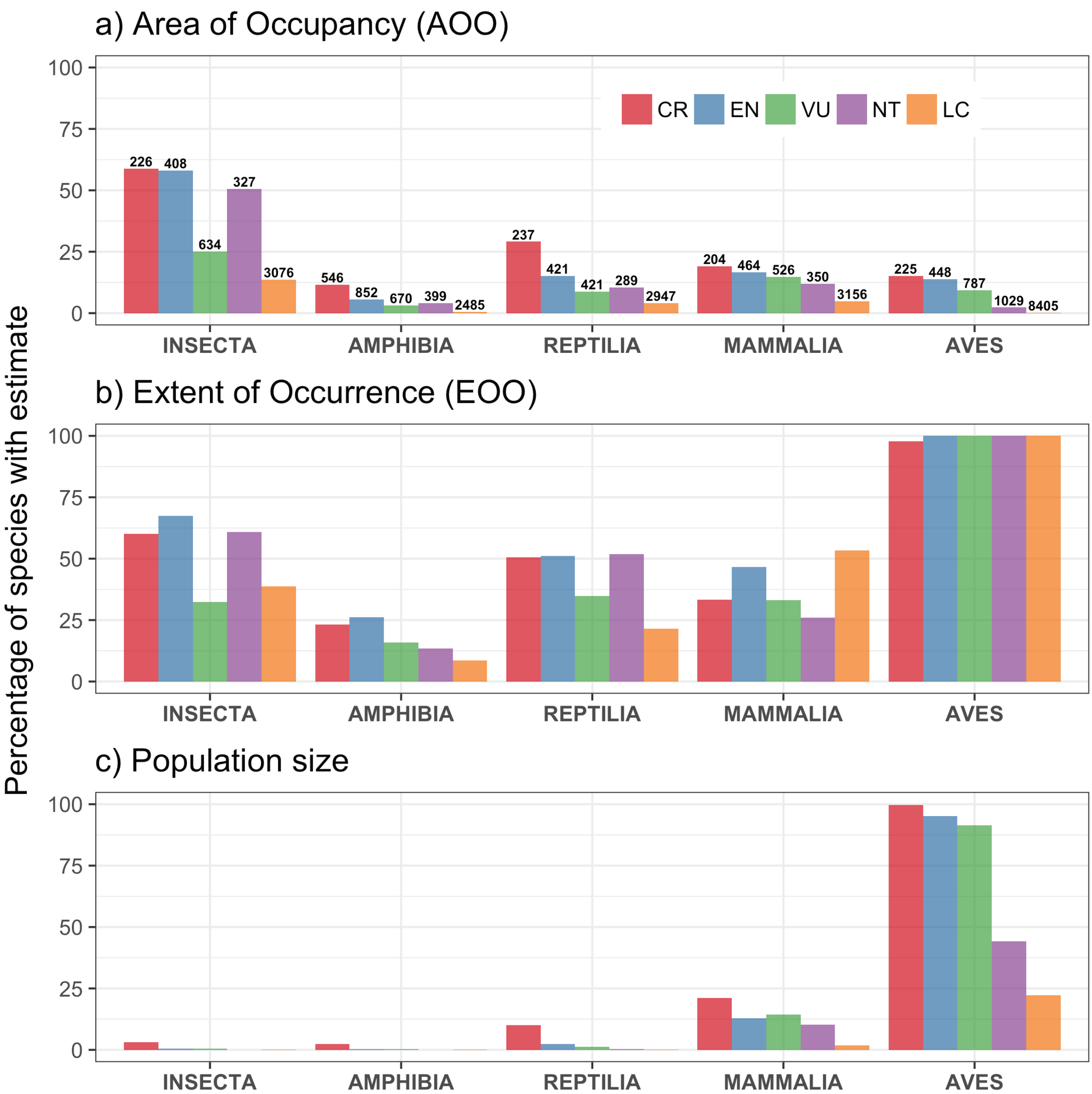


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