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.

75   Here, we characterise the specific challenges facing the conservation of abundance in non-threatened

76   species. Evaluating the significance of the loss of abundance requires understanding the broader

77   consequences of declines of different magnitudes, at different spatial scales, and for species with

78   different trait sets (e.g. body mass, life form, reproductive strategy, generation lengths, foraging

79   guilds, movement behaviours). We focus here specifically on those species that are sufficiently

80   abundant and widespread that they are unlikely to be classified as globally threatened under current

81   extinction-risk frameworks, even if they are suffering marked losses in biomass or numbers of

82   individuals. These species would be assessed as *Least Concern* on the *IUCN Red List* based on

83   geographic range and population size criteria alone, and will have population declining by less than

84   20% per (the longer of) 10 years or three generations (demarcated in Figure 1b).

85

86   **Depletion and its consequences**

87   Many of the clearest examples of the effects of depletion on ecosystem services have been observed

88   with localised (or global) extinction of a species, typically following from human exploitation and

89   disturbance (e.g. Soulé et al. 2005, examples therein). These natural experiments reveal the

90   functional roles that species play in an ecosystem through the consequences of their absence. Yet,

91   because of the elevated sensitivity of large-biomass species to extinction, examples are dominated by

92   trophic cascades resulting from the loss of top-predators or large-herbivores (Estes et al. 2011).

93   Large-biomass species, despite typically occurring at low densities, often exert a large *per capita*

94   effect on a broad range of services and the effects of their decline are often dramatic (Estes et al.

95   2011). For example, changes in the density of sea otters *Enhydra lutris* can rapidly transform a

96   barren ocean floor to a dense kelp forest as the grazing pressure on kelp is reduce through predation

97   by otters on benthic herbivores (Estes et al. 2010). Such effects can be realised with even small

98   changes in the abundance and long before the species is lost from the ecosystem (Estes et al. 2010).

99   Säterberg et al. (2013), for example, showed that more than a quarter of species in real food-webs

100 become functionally extinct before losing 30% of their individuals. Functional extinction, in this case

101 defined as the population size below which another species goes extinction (but see McConkey &

102 O'Farrill (2015) for discussion of complete vs. partial functional extinction), was most likely for

103 large-biomass species.

104

105 Equally strong effects, however, can be mediated through sheer numerical abundance, through the

106 total effect of losing many interactions of weak *per capita* strength. For example, local abundance

107 plays an important role in effective seed dispersal (Schupp et al. 2010) and pollination (Garibaldi et

108 al. 2013), and provides a prey-base for higher trophic levels (Delibes-Mateos et al. 2008; Smith et al

109 2011). The decline in abundance of low trophic level fish species (e.g. anchovy and sardines) in

110 many fisheries has severe impacts on larger predatory species (Smith et al 2011), including fish,

111 birds and mammals (Kaplan et al. 2017). Pulsed abundance can also have major functional effects on

112 ecosystem services (Bauer & Hoye 2014). For example, migrant insectivores can provide

113 economically important top-down control of invertebrate herbivory (López-Hoffman et al. 2017a,

114 2017b). The life-history traits of many highly abundant species naturally produce large fluctuations

115 in abundance, but evidence suggests that many species are no longer bouncing back after

116 experiencing periods of stringency (Cornulier et al. 2013; Selwood et al. 2015).

117

118 Depletion and local extinction are not necessarily synonymous (e.g. Santamaría et al. 2016), but few

119 studies measure the change in ecosystem-service provision as a species' abundance changes.

120 Understanding the form of these function-abundance relationships is critical to predicting when

121 depletion will affect an ecosystem service, with the form of the relationship affecting the sensitivity

122 of an ecosystem function to declines in abundance (Fig. 2a,b). Across a range of ecosystem-services

123 provided by birds, including supporting, regulating and cultural services, Gaston et al. (2018) found

124 the function-abundance relationships to be mostly positive and non-linear. From these examples, it

125 seems likely that many abundance-service relationships will exhibit strong thresholds. It is important

126 to note, however, that these insights are built on relatively few empirical measurements of

127 abundance-service relationships. How these observations generalize more widely is unknown,

128 although the broad pattern of positive and non-linear seems evident across a range of species and

129 services, including seed dispersal (McConkey & Drake 2006) and meso-predator suppression

130 (Newsome et al. 2017).

131

132 The drivers of depletion can also act as a selection pressure that drives changes in traits associated

133 with responses to environment change and to effects on ecosystem-services (e.g. Pinsky & Palumb

134 2014), especially where the loss of individuals is non-random across the population. For example,

135 the preferential removal of large individuals from frugivorous fish populations has resulted in change

136 in size structure of the populations (Costa-Pereira & Galetti 2015). The effects of downsizing the

137 population include the disruption of the mutualistic relationship between tree species that rely on the

138 largest individual fish to disperse their fruit. Whether or not an evolutionary response has been

139 produced in the fish population, the long-term ecological effects for the ecosystem will be large

140 because of the loss of this important function (Costa-Pereira & Galetti 2015). Generally, the presence

141 of many individuals in a population may mask changes to genetic diversity, population structure, or

142 in the distribution of traits. In addition to affecting ecosystem-service provision, these changes might

143 silently increase the vulnerability of the species to environmental changes (Walsh et al. 2006).

144

145 **Predicting the effects of depletion**

146 The loss of individuals from an ecosystem will produce an effect on an ecosystem-service (inclusive

147 of supporting, regulating and provisioning services) that is determined by the average *per capita*

148 effect of the species on this service, the abundance and change in abundance of the species in the

149 ecosystem, and also the form of the relationship between abundance and the service (Fig. 2b; Díaz et

150 al. 2013). Thus, predicting the effects of depletion requires the integration of a suite of species- and

151 context-specific data to scale *per capita* information to understand likely effects of declines of

152 different magnitudes.

153

154 System-specific models, for example, mass-balanced food-web models used to predict the flow of

155 energy across trophic levels, have had some success in predicting depletion of both high and low

156 trophic level species (Heithaus et al. 2008). These approaches typically require large amounts of data

157 to parameterise and tend to be associated with economically important components of ecosystems

158 (e.g. fisheries and forestry). The search for more generalizable frameworks for predicting the

159 consequences of environmental change on species and ecosystem function has led to intense focus on

160 species traits (Díaz et al. 2013). Recent efforts to link species traits to specific threats responses

161 ('response-traits') and then to the functional effects resulting from local extinction ('effect traits')

162 have yet to produce highly predictive models (Bartomeus et al. 2018). Realised *per capita* effects

163 have been shown to be highly context specific, and dependent on the ecosystem-service being

164 measured, along with the species-specific traits ('effect traits') that underpin its impact on this

165 service (Wohlgemuth et al. 2017). This is also likely to be true for the form of the abundance-service

166 relationship (Gaston et al. 2018). Thus, the complexity of ecological networks has led to few broadly

167 applicable generalisations, especially when accounting for temporally varying community

168 composition and relative abundance within the species pool, and the range of different perturbations

169 that can affect a system (Santamaría et al. 2016; Bartomeus et al. 2018). As a consequence, at least in

170 the near-term, it may be difficult to make highly specific prediction about the functional

171 consequences resulting from the depletion of specific species.

172

173 In general, however, two important insights have been gained from observational and theoretical

174 studies on function-abundance relationships that should inform the motivation for conserving

175 abundance in non-threatened species. These are that (i) functional extinction of a declining

176 population typically occurs before extinction of the species (Galetti et al. 2013; Säterberg et al.

177 2013), and (ii) functional loss can occur with even proportionally small changes in local abundance

178 (Säterberg et al. 2013; McConkey & O'Farrill 2015). The potential for sudden functional collapse

179 with proportionally small changes in abundance illustrates the risk that comes when little

180 conservation concern is assigned to declines in non-threatened species. This risk is compounded by

181 the difficulties of restoring abundance and the problem that the restoration of abundance does not

182 always restore lost function, e.g. because of hysteresis effects caused by demographic and

183 evolutionary shifts in a system's state (Walsh et al. 2006). Identifying when a species' abundance,

184 and its depletion, is consequential is difficult in many cases, although the complete loss of species

185 with a large body mass, those occupying high trophic levels, or those that dominate in terms of

186 biomass should always raise some concern. More generally, given the emerging understanding of

187 functional importance of abundant and widespread species, a precautionary approach that *a priori*

188 assumes high functional value for species' abundance independent of extinction risk should be

189 adopted as the default position. Conservation prioritisation in this context could perhaps focus on the

190 likelihood of recovering populations to historical abundance baselines, with prioritisation going to

191 species where the recovery of abundance will be most difficult, due equally to political and societal

192 factors, as well as ecological.

193

194 **Local depletion as a conservation concern**

195 The translation of *per capita* ecological effect into population and species-level contributions to

196 ecosystem-services is modulated by spatial and temporal variation in its abundance. Abundance is

197 distributed unevenly across a species' range and can also vary strongly across temporal scales driven,

198 for example, by daily, seasonal, or multi-decadal variation in environmental conditions (Sagarin et

199 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

225 2003). The importance of connectivity across large distances (i.e. telecoupling) for maintaining

226 ecological processes and ecosystem services are increasingly being recognised (López-Hoffman et

227 al. 2017a). For example, the pest control (affecting cotton production) and ecotourism services

228 provided by the Mexican free-tailed bat *Tadarida brasiliensis mexicana* in parts of the United States

229 are heavily dependent on maintaining wintering habitat in central and southern Mexico (López-

230 Hoffman et al. 2017a; 2017b). Understanding the spatial processes that maintain species' abundance,

231 and identifying spatial location of declines and movement behaviour of species is vital for

232 understanding and predicting the distribution of potential effects from depletion, and ultimately the

233 wider risk associated with local declines in a species.

234

235 Proportional declines measured at coarse scales and reported for a species' global population provide

236 limited and inadequate information to contextualise the significance of declines in abundant and

237 widespread species. From the perspective of species extinction risk, declines that are proportionally

238 small relative to the global population size might be viewed as largely inconsequential across the

239 species' entire occupied range. This might overlook large localised population declines that have

240 substantial, and perhaps widespread, consequences for ecosystem services (López-Hoffman et al.

241 2017a). This is especially likely when conservation priorities are being developed using spatially

242 coarse information, lacking details on the status of local biodiversity (Dolman et al. 2012) and the

243 linkages between the location of abundance and its effects (López-Hoffman et al. 2017a). Even

244 where evidence for local depletion exists, management intervention aimed at otherwise abundant and

245 widespread species can be perceived as misplaced (Battisti & Fanelli 2015). Such arguments must be

246 reframed in the context of conserving abundance, both because spatial representation is a critical

247 element in maintaining abundance (Gaston & Blackburn 1996) and because the wider abundance of a

248 species has little relevance where local declines precipitate undesired consequences.

249

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

274   projected declines, in a species that exerts a strong functional effect on a valued ecosystem-service

275   that is likely to be lost due to declines (Fig. 2bi-iii), regardless of a species' broader extinction risk.

276

277   As such, conservation prioritisation for abundant and widespread species shares greater similarity

278   with invasive species risk assessments than extinction risk assessments, because the focus must be on

279   location and *per capita* effect of a species' abundance on other species or services rather than the

280   global extinction risk of the species itself (e.g. Parker et al. 1999). Invasive species risk assessments,

281   which are typically produced for discrete geographic areas (e.g. protected areas to countries) rather

282   than globally, might provide a template for developing approaches for prioritise risk from depletion,

283   especially in the estimation of impact across space related to per capita or per area effects (Barney et

284   al. 2013). For example, figure 2 shows how local impacts from depletion (e.g. at the scale of, for

285   example, protected areas or ecoregions), could be predicted for a species using demographic

286   forecasts (Fig. 2a) and information on the form and strength of the abundance-services relationship

287   (Fig. 2b). The integration of this information, perhaps across multiple ecosystem-services (Barney et

288   al. 2013), could be used to assess the relative depletion risk for the most locally common species. As

289   for invasive species risk assessments, data limitations provide a major obstacle to effective

290   implementation. Below we discuss these data gaps.

291

292   *Knowledge gaps*

293   To implement a prioritisation approach for depletion, or even simply to increase the awareness of the

294   problem, much more information is required on the per capita effects and abundance-service

295   relationships of the most widespread and abundant species. Soule *et al.* (2005) advocated the

296   preservation of functional abundance for strongly interacting species, identified as those that cause

297   impacts on other species when they decline in local abundance. The suggestion here is that species

298   can be identified as strongly interacting based on effects resulting from their decline in abundance,

299 and then minimum abundance thresholds can be established form management. We currently lack

300 much of the fundamental information to identifying these species and thresholds. Without such

301 information, there is no way of identifying priorities across species and in different environmental

302 contexts. Empirical and theoretical studies must be employed to greater effect in understanding the

303 consequences of depletion so that choices and trade-offs in conservation investment can be informed

304 by an evaluation of risk from alternative actions (Säterberg et al. 2013; Harvey et al. 2017; Gaston et

305 al. 2018). For many ecosystem-services, the abundance-service relationships and per capita effect

306 might be predicted based on existing ecological theory, e.g. using well-studied species' functional-

307 responses to resource availability (e.g. Dawes & Souza 2013). The null position, in the absence of

308 such information for most species and contexts, must be to assume all abundant and widespread

309 species are functionally important (Gaston & Fuller 2008). In this case, information on the

310 likelihood of recovering abundance once lost would be particularly valuable for identifying

311 conservation priorities.

312

313 The availability of spatially and temporally explicit information on abundance must also be a priority

314 to enable predictions of where and when services are likely to be impacted by depletion. Population

315 trends of widespread species can vary greatly across their range (Devenish et al. 2017) and impacts

316 can be distributed away from the drivers of depletion due to species' movements or other

317 environmental flows. Information collected to assess extinction risk typically does not provide the

318 spatial information necessary to place declines in abundant and widespread species in the appropriate

319 context vis-à-vis the size and distribution of effects on ecosystem services. Extent of occurrence and

320 global population trends do not provide the spatial resolution necessary to detect such variation, and

321 area of occupancy at resolutions suitable for conservation planning and threat assessment (e.g.

322 typically <10 km; Keith et al. 2018) is often poorly estimated or unknown across all or part of

323 species' ranges. The collation of information on the distribution and abundance of threatened species

324   is made systematic through the framework of the IUCN Red List, but this does not necessarily

325   provide the same motivation for the collection or collation of information on species that are

326   abundant and widespread (e.g. Fig. 4). The need for basic distribution and demographic data is partly

327   being addressed by citizen science initiatives and through the integration with remotely sensed

328   environmental data (Jetz et al. 2012). Taxonomic and geographical biases mean that for the majority

329   of species and in many parts of the world distribution and abundance data is not current available

330   (Meyer et al. 2016).

331

332   Resolving spatial variation in occupancy or abundance, and the spatial variation in rates of declines,

333   is vital for identifying the threats. While factors driving declines in threatened species may also be

334   causing depletion in similar abundant and widespread species, such concordance cannot be assumed.

335   Conservation strategies that perform well for threatened species will not necessarily align well with

336   the measures needed to halt declines in abundant and widespread species. Much will depend on the

337   drivers of declines. The establishment of protected areas is the dominant strategy for the

338   conservation of many threatened species but, given that conservation reserve systems typically

339   occupy only 10-20% of regions or ecosystems and that management of conservation reserves often

340   targets threatened species within them, this is unlikely to prevent the erosion of abundance in more

341   widespread species. The depletion of species outside of protected areas can even undermine the

342   performance of the protected area. For example, the closure of a small South African fishery was not

343   predicted to produce positive population growth for the endangered African penguin *Spheniscus*

344   *demersus* because the population was dependent on prey availability across a scale larger than this

345   single fishery (Sherley et al. 2015). The practical challenges of maintaining space for processes that

346   promote abundance are higher for wide ranging species, which are often poorly served by existing

347   protected areas and often require conservation across political boundaries (Runge et al. 2014). Some

348   successes have resulted from reducing threats in the wider landscape (e.g. Gamero et al. 2017).

349 However, the continued decline of Europe's most common bird species, despite considerable

350 targeted conservation management, indicates that research and policy initiatives in this area are a

351 priority.

352

353 Perhaps the principal influence of the *IUCN Red List* on species conservation has been to focus

354 attention on the issue of the emerging extinction crisis and to stimulate the collation of information to

355 inform species-level threat assessment (Rodrigues et al. 2006). Similar attention might be focused on

356 the emerging abundance crisis were the estimation and reporting of absolute changes in abundance

357 and biomass routine (Gaston & Fuller 2008). The headline figures reported by Inger et al. (2015) are

358 easy to interpret and leave no doubt about dire state of Europe's avifauna. Population size estimates

359 are currently lacking for most species, except where economic value is linked to abundance (e.g.

360 fisheries, forestry) or where density estimates are critical for management (i.e. livestock densities in

361 protected areas). However, population sizes could be estimated for many species from even coarse-

362 scale AOO and habitat/season specific estimates of population density (e.g. Hui et al. 2009).

363 Abundance/biomass trends would convey a different, but equally powerful, message to that of

364 existing population trend indices (e.g. World Wildlife Fund, 2016), which typically report the

365 average rate of population change across a suite of species, without describing changes in total

366 abundance and biomass.

367

368 **Conclusions**

369 The value of abundance can be justified from multiple perspectives. At present, it may be

370 challenging to quantify and rank the ecological significance of depletion in abundant species and to

371 define thresholds of depletion for conservation prioritisation. Abundant species often occur across

372 vast geographic space with declines often driven by varied combinations of multiple interacting

373 threats operating across different parts of their distributions. There are, however, clear steps that

15

374 could be taken to better understand the status of abundant and widespread species, and of the relative

375 priority for conservation action to address depletion in such species.

376

377 1) The quantification of the contribution of abundant and widespread species to ecological processes

378 and services is necessary to understand and communicate the consequences of losing abundance in

379 non-threatened species. This includes increasing the effort to quantify links between threatening

380 processes and undesired effects of depletion where movement behaviour or other environmental

381 flows create a spatial disconnect. A clearer understanding of the consequences of losing abundance

382 in non-threatened species, even in a broad sense, will help to define the objectives of elevating the

383 priority of these species relative to other conservation challenges.

384

385 2) The conservation of abundant and widespread species requires information on the spatial

386 distribution and trends in abundance that can feed into conservation planning at scales from local to

387 global. These requirements are not necessarily met by data collection and collation aimed at

388 assessing species extinction risk, but should be identified specifically against the objectives aligned

389 with the need for conserving abundance in non-threatened species (i.e. impacts on other species).

390 Increasing citizen science and remote sensing data, along with advances in modelling species

391 distributions and trends, will benefit these actions greatly. Emphasis should be placed on collecting

392 information that can be used to estimate abundance (not just presence at a coarse scale), increasing

393 data collection in under-sampled regions, and in establishing historical baselines for species'

394 abundance to avoid shifting baseline problems when setting conservation goals.

395

396 3) More systematic assessments of the threats that are causing depletion in abundant and widespread

397 species are required. Such assessment should complement assessments of threats that are causing

398 imperilment of threatened species, and then be used to prioritise management responses that can best

399   address the needs for threat reduction for non-threatened and threatened species. Such assessments

400   would guide when fundamentally different threat management responses are required for reversing

401   declines in abundant and widespread species cf. rare species.

402

403   Finally, the conservation narrative, dominated by rarity and extinction risk, must be augmented with

404   a message that highlights the ecological and societal value of abundance regardless of global

405   extinction threat.

406

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570   Figure 1. The conservation context of the decline in global abundance: (a) ca. 90% of the

571   total abundance for the world's declining bird species (those with population size estimates, n

572   = 2141) derives from only 108 species, almost exclusively assessed as Least Concern (LC =

573   100; NT = 4; TH [CR, EN, VU] = 4) on the *IUCN's Red List of Threatened species* (data

574   supplied by BirdLife International April 2016). (b) In order for a species that is abundant and

575   widespread to become listed as threatened under the IUCN's criteria it must have undergone

576   a decline of at least 30% over the longer of 10 years or 3 generations. Thus, vast numbers of

577   individual can be lost (and continue to be lost) from LC species without changing their

578   extinction risk status. CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT

579   = Near Threatened; LC = Least Concern.
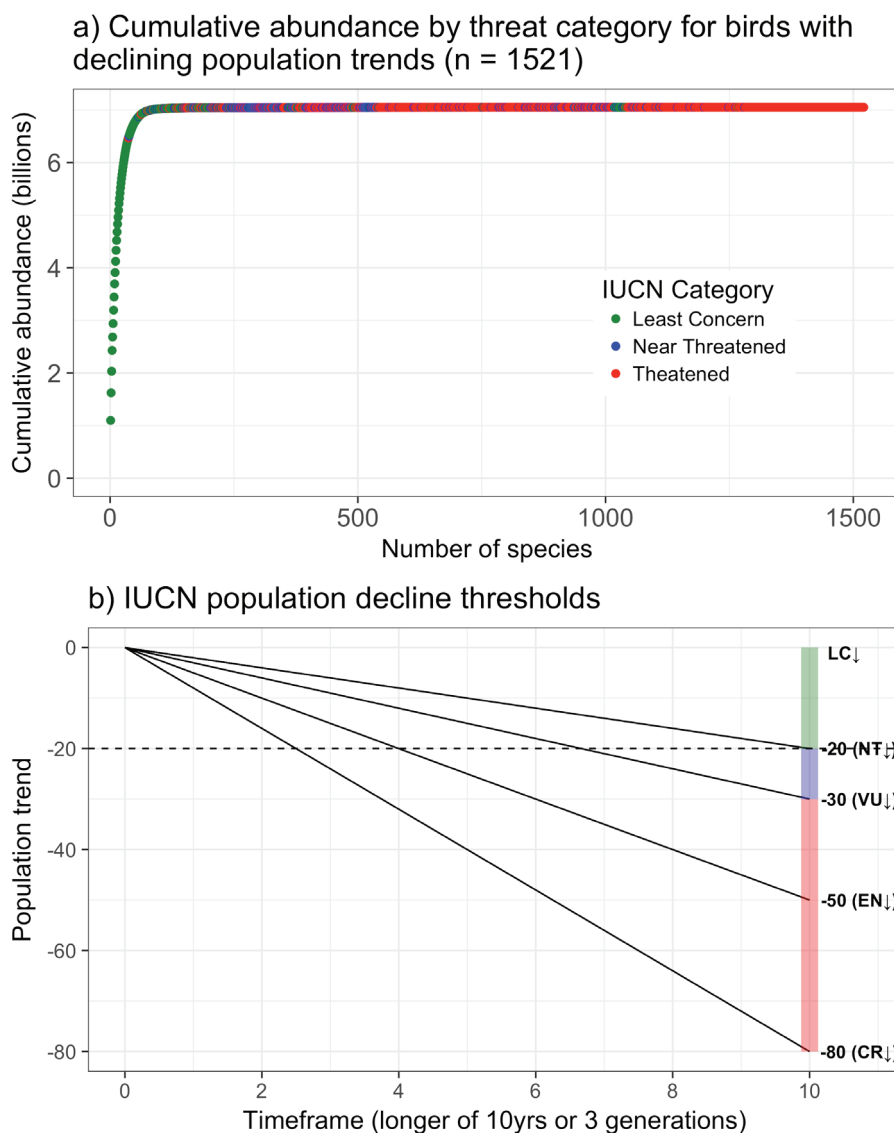
580

581   Figure 2. Functional effects of local depletion can occur prior to species loss. As a local

582   population density declines through time (a) the emergence of consequences (b) measured

583   against a functional value / ecosystem-service metric (e.g. species richness, carbon

584   sequestration, economic value) will depend on the relationship between the species density

585   (e.g. per capita effect) and the functional value metric (b). Non-linear relationships (i) and

586   (iii) will be frequently observed because of complex density-dependent interactions among

587   species, but linear 'mass effects' might also be observed where consequences are a function

588   of biomass (ii). The size of the species' contribution to the overall ecosystem-service will

589   vary, i.e. high (i) vs. low (iv), and this will affect the impact realised as the species declines.

590   The sensitivity of the system or process to the depletion of abundance (i.e. 50% loss of local

591   abundance, demarcated on (a) and (b) by ----) will depend on these relationships with, for

592   example, functional loss occurring at lower levels of depletions across the forms (i) to (iii).

593   For example, 50% loss of local abundance results in no function loss (i), partial function loss

594   (ii), or complete function loss (iii). Together, the rate of population decline, the form of the

595   abundance-function relationship, and the size of the total effect of abundance could be

596   formulated into a risk framework to assess relative local risk among species (c).

597

598   Figure 3. Species' long-term population trends can differ in sign across their range, as

599   illustrated by the degree of consensus in the direction of bird population trends (1999-2013)

600   between major biogeographic regions of south-eastern Australia. Values given in the region

601   intersections show the percentage of species that show a consistent trend direction (positive,

602   negative, stable) between these regions. For example, of the 45 species found in all three

603   regions, the regional trends of 33% of these species show consistency in direction across the

604   three regions. n = total number of shared species between regions given. N = number of

605   species in each regions with an estimated trend. Data from State of Australian Birds 2015

606   regional reports (http://birdlife.org.au/education-publications/publications/state-of-australias-

607   birds).

608

609   Figure 4. Species threat status can motivate the collection and collation of population data, as

610   illustrated by the percentage of species in each extinction risk category, across five major

611   taxonomic classes, for which there is an estimate of (a) Area of Occupancy (AOO), (b)

612   Extent of Occurrence (EOO) and (c) Population size. The total number of species assessed in

613   each taxonomic class and threat category is given in (a) above the relevant column. Data

614   come from two major databases (IUCN as of March 2016; Aves data provide by BirdLife

615   International April 2016).

a) Cumulative abundance by threat category for birds with declining population trends (n = 1521)

b) IUCN population decline thresholds

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.
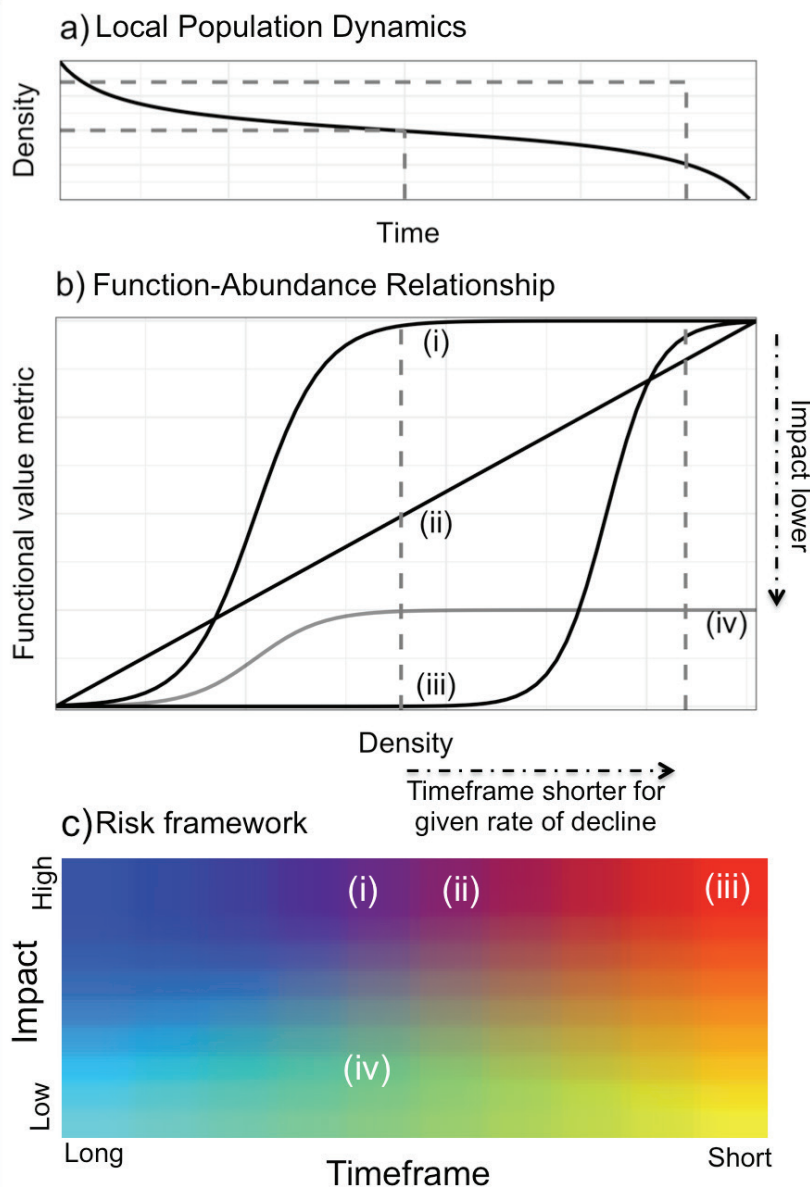
a) Local Population Dynamics

b) Function-Abundance Relationship

c) Risk framework

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

635   realised as the species declines. The sensitivity of the system or process to the depletion of

636   abundance (i.e. 50% loss of local abundance, demarcated on (a) and (b) by ----) will depend on these

637   relationships with, for example, functional loss occurring at lower levels of depletions across the

638   forms (i) to (iii). For example, 50% loss of local abundance results in no function loss (i), partial

639   function loss (ii), or complete function loss (iii). Together, the rate of population decline, the form of

640   the abundance-function relationship, and the size of the total effect of abundance could be

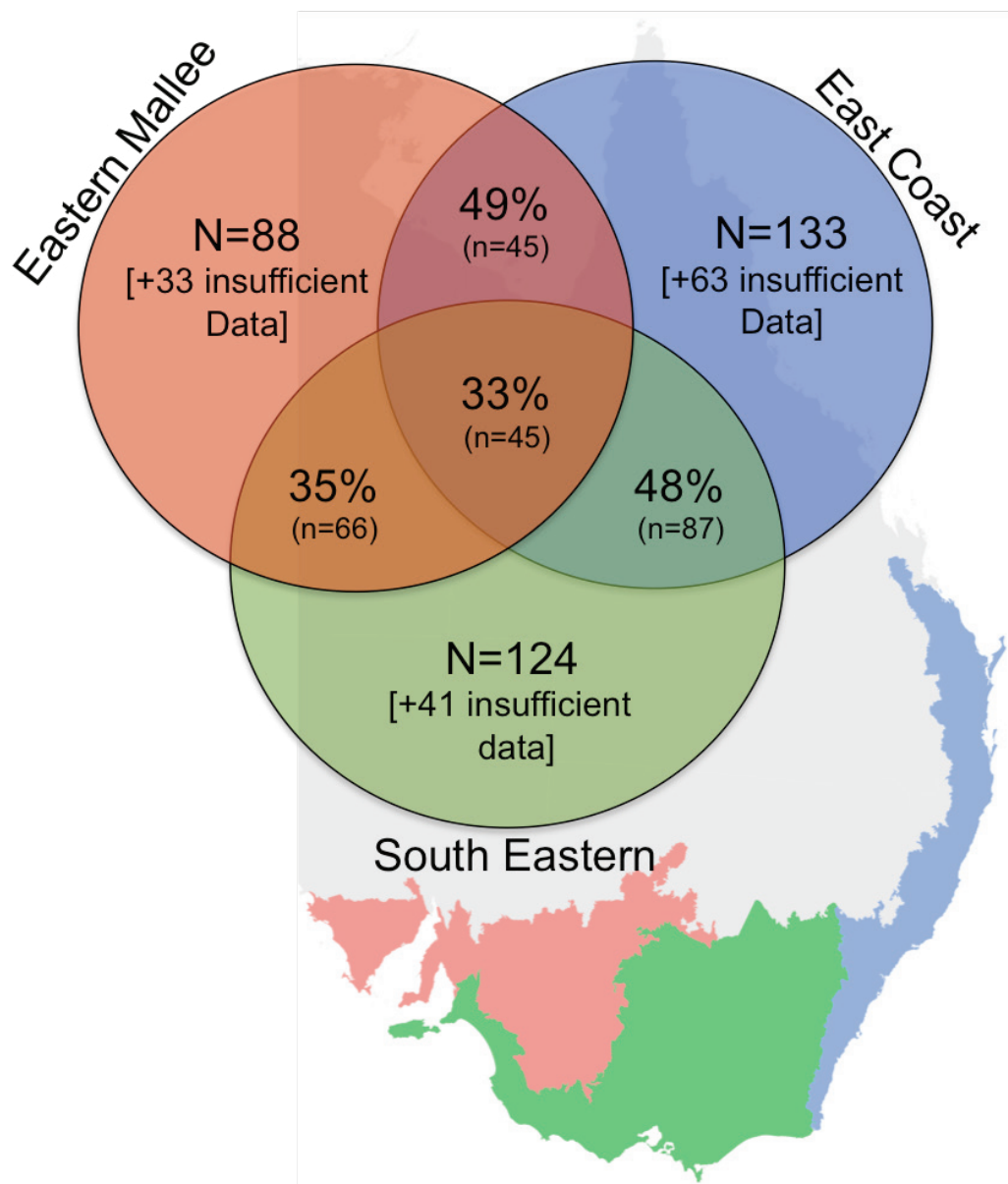641   formulated into a risk framework to assess relative local risk among species (c).

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).

a) Area of Occupancy (AOO)

b) Extent of Occurrence (EOO)
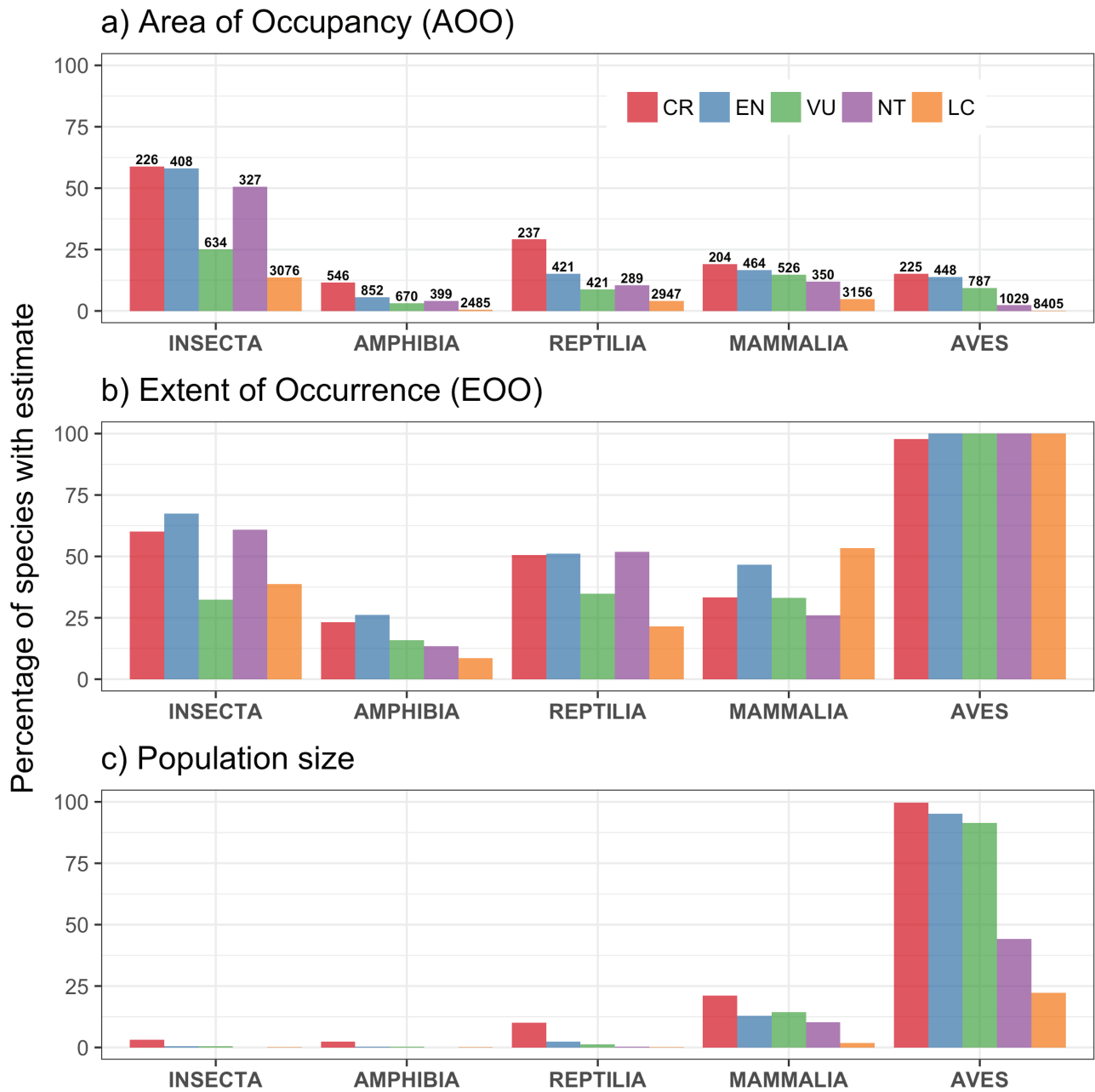
c) Population size

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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).

660