Target 12: Preventing Extinctions and Improving Species Conservation Status

By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.

Preface

The report on modelling and scenarios for GBO3 summarized predicted biodiversity loss under future scenarios based on four key measures: (i) species extinctions, (ii) changes in species abundance, (iii) habitat loss, and (iv) changes in the distribution of species, functional species groups or biomes (Leadley et al. 2010, Pereira et al. 2010). In this report, habitat loss is considered under the targets that comprise Strategic Goal B (‘Reduce the direct pressures on biodiversity’). Here, we consider extinction risk and conservation status. For extinction risk, we use measures based on the threat status (measured using the IUCN Red List) of species, such as the Red List Index (Butchart et al., 2004). We also consider changes in species population sizes and distributions, and changes in the species diversity and species composition of ecological communities. Change in the diversity and composition of communities are considered despite the focus of this target on the status of known threatened species, because changes in local diversity accumulate to cause global loss of species, including of known threatened species. We do not consider genetic diversity here because of the focus of the target on conserving species. Genetic diversity is considered under Chapter 13, although only for cultivated plants, domesticated animals and their wild relatives. The long-term conservation of species and of biodiversity more broadly will depend on conserving genetic diversity (e.g. Forest et al., 2007), and thus genetic diversity should be considered in future reports and target-setting. Freshwater environments are lacking much of the information required to present an equivalent assessment to those for the terrestrial and marine environments, necessitating a more qualitative treatment.

1. Are we on track to achieve the 2020 target?

1.a Status and trends to date

Observed extinctions have generally increased over the last 200 years (Barnosky et al. 2011). For birds and mammals, there has been a slowing in the apparent rate of extinction in the last 50 years (Figure 12.1A), although uncertainty about whether and when a species became extinct is large (see ‘Uncertainties’ below), and there is generally a long lag time between a species becoming extinct and being recorded as such. For freshwater fish species on the other hand, observed numbers of extinctions have increased unabated over the last 100 years (Figure 12.1B). Consistent data on extinctions for other taxonomic groups are not available, but the
number of extinctions since 1500 is very high. For example, at least 34 amphibians became
extinct during this period, 2 extinct in the wild and 165 possibly extinct (Stuart et al., 2008). At
least 310 of the approximately 6800 species of molluscs assessed by IUCN have become extinct
since 1500, nearly 5% of the total assessed (www.iucnredlist.org). In order to prevent
extinctions of known threatened species in the near future (the main aim of Target 12) will
require the protection of the sites containing the last populations of critically endangered
species (Alliance for Zero Extinction sites): Coverage of these by protected areas has increased
steadily over the past 25 years (Figure 12.2 D; Butchart et al., 2012). Protection of Important
Bird Areas has increased rapidly in the same time period, although from a much lower starting
point (Figure 12.2 E). All measures to protect known threatened species will require significant
investments; there have been no clear trends in funding for the protection of species in the last
15 years (Figure 12.2 F).
Figure 12.1. Median and 95% confidence intervals of the number of extinctions in 25-year intervals from 1800 to the present of (A) mammal and bird (data source: BirdLife International, 2014) species and (B) freshwater fish (in total, 47 mammal species, 90 bird species and 32 fish species were declared extinct, and 27 mammals and 13 birds presumed extinct since 1800). When an exact extinction date was not given, an interval during which a declared extinction is thought to have occurred was used; otherwise, the last sighting of the species was considered as the earliest date and the latest date was assumed to be 30 years after last sighting. For the 43 species presumed extinct, we considered the last sighting as the earliest date of extinction and added the average time between last sighting and the declaration of extinction of other species to derive the latest date; C) Red List Index (RLI) for the world’s birds (red, 9869 species), mammals (purple, 4556), amphibians (green, 4355) and corals (blue, 704); and an aggregate estimate (black; Butchart et al., 2010); declines correspond to increases in extinction risk (Butchart, S. H. M. et al., unpublished data). The RLI ranges from 0 (all species are extinct) to 1 (all species are considered “Least Concerned”). Confidence intervals around the aggregate RLI trend represent 95% confidence intervals, and were based on multiple sources of underlying uncertainty – see Butchart et al., 2010; D) reconstructed trend in extinction risk for carnivores (284 species) and ungulates (262 species) in the last 40 years. The Red List status of all species was retrospectively assessed applying the current IUCN criteria to information on past population and geographic range size, structure, and trend, as well as habitat loss and other threats, available from historical IUCN Red Data Books and Action Plans (Di Marco et al. in press; doi: 10.1111/cobi.12249).

The average extinction risk of assessed species – measured as the Red List Index – increased steadily over the past 40 years with no signs of slowing (Figure 12.1C, D), although increased attention and investment towards threatened species has prevented some critically endangered species from going extinct (Butchart et al., 2006; Hoffmann et al. 2010). Among terrestrial species, amphibians have a high level of threat and are declining in conservation status strongly (Figure 12.1C), with 32% of species threatened and 40% declining according to the IUCN (www.iucnredlist.org; Stuart et al., 2008). For plants, comprehensive assessments of extinction risk are only available for gymnosperms, for which 41% of species are considered threatened (www.iucnredlist.org); there are no reported trends in extinction risk of plants at present. For flowering plants, only 6% of the approximately 270,000 known species have been assessed (including all cacti); however, of these 56% (31% of cacti) are considered threatened. The conservation status of terrestrial invertebrate species and fungi is poorly known at a global scale, but assessments are available for certain regions suggesting that significant proportions of species are threatened with extinction: 9% of European butterflies (van Swaay et al., 2010), 15% of European dragonflies (Kalkman et al., 2010), 11% of European saproxylic beetles (Nieto & Alexander, 2009) and 16% of British fungi (Evans et al., 2006). Freshwater species are also showing strong declines, with 32% of freshwater vertebrates and 32% of decapods at risk of extinction (Collen et al. 2014). In the marine realm, over 550 species of marine fishes and invertebrates are listed on the IUCN Red List as Critically Endangered, Endangered, and Vulnerable (www.iucnredlist.org). This is an underestimate, owing to insufficient data with which to assess the extinction risk of many marine organisms. As with trends in the extinction risk of species, population trends – as measured by the Living Planet Index (Figure 12.2C), the
Wild Bird Index of habitat specialist bird species (cross-reference to Target 5) and the Wildlife Picture Index (O’Brien et al. 2010) – continue to decline.

For all terrestrial vertebrate groups, habitat loss because of agriculture, aquaculture and logging is a stress responsible for the decline of the greatest number of species (Hoffmann et al., 2010). Reptiles and amphibians are particularly sensitive to habitat degradation because of their comparatively low dispersal ability, relatively small home ranges and thermoregulatory constraints (Kearney et al., 2009). For mammals and birds, hunting is the next greatest threat (Hoffman et al., 2010). For amphibians, invasive species and disease (in particular the chytrid fungal pathogen Batrachochytrium dendrobatidis; Cheng et al. 2011) are the main drivers of decline after habitat loss, although the interaction of these threats with climate change is likely to exacerbate amphibian decline in the near future (Hof et al. 2011; Pounds et al. 2006).

Invasive alien species are also a major threat to birds, particularly those on oceanic islands (BirdLife International 2014). Disease is also an important threat for certain other taxonomic groups (for example, white nose syndrome in bats). The relative threat posed to other animal taxonomic groups by different human activities is less well known. For invertebrates – bees in particular – pesticides appear to be a serious threat (e.g. Gill et al., 2012).

Plant species are mainly affected by loss, degradation and increased fragmentation of habitats, and alien invasive species (Bilz et al., 2011). In the future, the threat from climate change is predicted to grow (Bilz et al., 2011; Giam et al., 2010). Genetic erosion and extinction have been identified as important threats to the crop wild relatives and to plant populations that occur on islands (Bilz et al., 2011). Aquatic plants are affected by ecosystem modification and loss caused by the transformation of wetland habitats, and the intensification of agricultural activities accompanied by eutrophication and pollution (Bilz et al., 2011). In some countries the collection of wild plant species (as medicinal plants, for food, or for their beauty or value for collectors) is causing a loss of species and a reduction of their reproductive success (Bilz et al., 2011).

While the global assessment of marine species is still ongoing, data are available on the threats driving species loss for several well studied groups. The chondrichthynes (sharks and rays) are overexploited through targeted fisheries as well as incidental by-catch (Dulvy et al. 2014). In addition, half of the 69 high-value sharks and rays in the global fin trade are threatened (53.6%, n = 37), while low-value fins often enter trade as well, even if meat demand is the main fishery driver (Dulvy et al. 2014). Similarly, for parrotfishes and surgeonfishes, species that play critical roles in coral reef ecosystems, 40% (73 species) of known species are impacted by small and large scale fisheries and 6% are recorded to be affected by habitat modification, 3% by pollution and 1% by by-catch (Comeros-Raynal et al. 2012). For groupers and wrasses, 12% of the 163 species assessed are considered to be at risk of extinction if current trends continue, with 13% considered near-threatened, under the IUCN Red List criteria. However, 30% of species could not be assessed owing to insufficient data. The major driver of extinction risk in this group is overfishing, with poor or no fishery management (Sadovy de Mitcheson et al. 2012). For
seabirds, particularly albatrosses and large petrels, longline and gillnet fisheries present a severe threat (Croxall et al., 2012).

At the global scale, limited data prevent comparable reporting for freshwater species, especially freshwater fishes and invertebrates. However, based on current data, the main threat to freshwater vertebrates and decapods is habitat loss and degradation, affecting 80% of species. This is followed by pollution (50% of threatened species) and exploitation (40% of threatened species) (Collen et al. 2014). The combined effects of overexploitation and habitat degradation are acute for freshwater-dependent chondrichthys, with over one-third (36.0%) of the 90 obligate and euryhaline freshwater chondrichthyans considered threatened (Dulvy et al. 2014).

The degradation of coastal, estuarine and riverine habitats threatens 14% of sharks and rays: through residential and commercial development (22 species, including river sharks Glyphis spp.); mangrove destruction for shrimp farming in Southeast Asia (4 species, including Bleeker’s variegated stingray Himantura undulata); dam construction and water control (8 species, including the Mekong freshwater stingray Dasyatis laosensis), and pollution (20 species) are noted as driving species declines (Dulvy et al. 2014).

Importantly, within broad groups of species, the effect of threats will not fall evenly on different species. In terrestrial environments, large-bodied, slow-breeding species with strict habitat and dietary requirements have been shown to be more adversely impacted by habitat loss (e.g. Vetter et al., 2011; Newbold et al., 2013), and to be at greater risk of extinction (Cardillo et al., 2005; Davidson et al., 2009) than other species. Similarly, among marine fishes, turtles and mammals, large-bodied, late maturing species have been shown to be more sensitive to fishing and pollution than other species (Reynolds et al. 2005; Cheung et al. 2005; 2007; Davidson et al., 2012; Norse et al. 2012; Maxwell et al. 2013), while marine mammals are also sensitive to coastal stressors, such as habitat loss (Davidson et al., 2012; Maxwell et al. 2013).
1.b Projecting forward to 2020

A. Mammal and bird extinctions

B. Red List Index for birds, mammals and amphibians (aggregate)

C. Living Planet Index

D. Protected area coverage of A2E sites

E. Protected area coverage of IBA sites

F. Funds towards species protection
Statistical extrapolation to 2020 of 6 key measures of the extinction, extinction risk and conservation status of species: observed extinction rates of birds and mammals (A), the aggregate Red
List Index of birds, mammals and amphibians (B), the Living Planet Index (C), the representation by protected areas of sites whose protection could avert the extinction of known threatened species: Alliance for Zero Extinction sites (AZEs; D) and Important Bird Areas (IBAs; E) and funds for the protection of species (F). Long dashes represent extrapolation period. Short dashes represent 95% confidence bounds. Horizontal dashed grey line represents model-estimated 2010 value for indicator. Extrapolation assumes underlying processes remain constant. Visconti et al. (unpublished data) (A); S. H. M. Butchart et al. (unpublished data); B. Collen et al. (unpublished data) and described in Collen et al. (2009) (C); S. H. M. Butchart et al. (unpublished data) and described in Butchart et al. (2012) (D–E) and AidData (http://aiddata.org/) (F). Extrapolations are based on the assumption that underlying mechanisms continue to follow trends. Methods for model fitting are described in the introductory chapter.

Concerted conservation action has been shown to be effective in reducing the extinction risk of vertebrate species (Butchart et al., 2006; Hoffmann et al., 2010), and further action might prevent some extinctions that would otherwise occur by 2020. However, extrapolations suggest that it is very unlikely that all extinctions of known threatened (bird and mammal) species will be prevented by 2020 (Figure 12.2 A). Indeed, many species are at high risk of imminent extinction (e.g. Wake, 2012) and the level of resourcing required to prevent extinctions of known threatened species is an order of magnitude greater than current investment (McCarthy et al., 2012). Furthermore, many undescribed species have already, or will by 2020, become extinct without our knowledge (Mora et al. 2011). The global rate of extinctions might be slowing (Figure 12.2 A); however, at least for birds the rate of extinctions in continental areas may be accelerating (Szabo et al., 2012) and lags in reporting might lead to an underestimate of recent extinctions. On the other hand, the rate of extinction of freshwater fish species is likely to continue increasing (Figure 12.1 B); however, as noted elsewhere, data for freshwater species are very limited. Furthermore, extinction risk – as measured by the Red List Index for birds, mammals and amphibians – is predicted to continue to increase (Figure 12.2 B), as are population trends, as measured by the Living Planet Index (Figure 12.2 C; Collen et al. 2009) and Wild Bird Index (cross-reference to Target 5). On the other hand, coverage of Alliance for Zero Extinction sites by protected areas is predicted to increase, although based on the current rate of increase it is unlikely that 25% of sites will be protected before 2020 (Figure 12.2 D; Butchart et al. 2012). Coverage of Important Bird Areas is predicted to increase rapidly, exceeding 25% coverage of sites by 2020 (Figure 12.2 E). The incomplete coverage of assessments of marine species and the short time series of existing data preclude a numerical extrapolation of marine species’ trends to 2020. Future trends in funding for species protection are difficult to predict (Figure 12.2 F).

In terrestrial, marine and freshwater environments, habitat destruction, fragmentation and degradation (hereafter “habitat loss”) are likely to remain major stresses on biodiversity until 2020 and beyond (Alkemade et al., 2009; Green et al., 2005; Jetz et al., 2007; Martinuzzi et al. 2013a,b). In addition, for both marine and freshwater species, over-exploitation is and will remain a major threat (Pitcher and Cheung 2013, see Target 6). Many studies have predicted the
impact that habitat loss will have in the future on the ranges (e.g., Jetz et al., 2007; Cheung et al., 2009), population trends (e.g., WWF, 2012) and extinction risk (Bird et al., 2011) of species, and on the diversity of ecological communities (Gibson et al., 2011; Allan 2004; Cheung et al., 2009; Newbold et al., in prep.). Moreover, emerging threats such as deep-sea mining may further increase the extinction risk associated with habitat changes (Boschen et al., 2013).

Short-term future projections of the extinction risk of species as a result of projected habitat loss generally predict a worsening situation. However, improvements can be seen under some scenarios. Under business-as-usual scenarios, species within ecological communities are projected to continue declining in abundance on average (Fig. 12.3A; Alkemade et al., 2009), and the number of species within communities is also projected to decrease (Fig. 12.3B). The projections suggest that global populations of carnivore and ungulate species will continue decreasing steeply (Fig. 12.3C), and that these species will lose substantial proportions of their ranges (Visconti et al., 2011). This leads to a predicted increase in species’ extinction risk (Fig. 12.3D; Di Marco et al. in press; Visconti et al., submitted; see also Millennium Ecosystem Assessment, 2005). Regional model predictions under business-as-usual scenarios mirror these results: for example, in the Brazilian Amazon approximately 2%, on average, of vertebrate species are predicted to become locally extinct as a result of habitat loss, with a further 12% committed to extinction (Wearn et al., 2012), while the percentage of threatened bird species in the same region is predicted to increase from 3% to between 8 and 11% (Bird et al., 2011).

Under the Rio+20 scenarios (Netherlands Environmental Assessment Agency, 2010), designed to mitigate biodiversity losses, losses of within-community diversity (abundances and numbers of species) are slowed to some extent, but not prevented (Fig. 12.3A, B). Population size and extinction risk trends are reversed under these mitigation scenarios in the short term, caused by a net gain in natural habitat in Africa and South-East Asia, which are hotspots of carnivore and ungulate richness (Fig. 12.3C, D). The scenarios assume that the natural habitat gained is biotically equivalent to primary natural habitat; relaxing this assumption would lessen the modelled effectiveness of mitigation. In the Brazilian Amazon, the local extinction of vertebrate species is predicted to decrease by one- to two-thirds under scenarios that assume a reduced rate of deforestation (Wearn et al., 2012).
Figure 12.3. Predicted changes in the global average of local abundance and species richness of ecological communities, and of average population trends of carnivores and ungulates (calculated as a geometric mean of the ratio between population size at a given time and population size in 1970, following the Living Planet Index methods; Collen et al., 2009) and extinction risk (Red List Index – RLI) from 2000 to 2050, under three of the Rio+20 scenarios (Netherlands Environmental Assessment Agency, 2010; see also chapter 0). Values are scaled to equal 100% in 2000. The population trends and
extinction risk projections under the Rio+20 scenarios are the ‘maximum physiological dispersal’
projections of terrestrial carnivore and ungulate species from Visconti et al. (in review). The modelled
indicators measure slightly different aspects of conservation status, hence the differences in projected
trends. GLOBIO projects mean species abundance (MSA) across all taxonomic groups relative to pristine
conditions as a response to multiple pressures including habitat loss, climate change and human
disturbance. MSA does not allow for increases in species abundance and measures “naturalness”. Local
total abundance (PREDICTS model), global population trends of species and extinction risk (RLI) do not
have this constraint. PREDICTS and GLOBIO measure local losses, which are aggregated spatially across
grid cells, while RLI and LPI trends are measures of global species decline, which are aggregated across
species. The extrapolated trend in extinction risk (RLI) for carnivores and ungulates was done by fitting a
natural spline interpolation to the data in Fig 12.1B. Vertical bars show uncertainty in 2050 (shown only
for 2050 for clarity); uncertainty estimates were not available for the GLOBIO or Visconti LPI projections.

While positive changes to land-use policies (e.g. investing in forest restoration or modifying
investments in crop production) can affect trends, there are limits to what can be achieved. In
the United States, for example, the basic economic and demographic factors shaping land-use
change are powerful and even fairly dramatic policy changes have been shown to lead to only
moderate deviations from a business-as-usual scenario (Radeloff et al. 2012). However, some
policy tools will be easier to enact, highlighting that opportunities exist for exploring different
policy options that can have more substantial impacts on the reduction of habitat loss.

2. What needs to be done to reach the Aichi Target?

2.a. Actions

Human actions have greatly increased current extinction rates. Reducing the threat of human-
induced extinction requires action to address the direct and indirect drivers of change. As such,
actions taken to attain the targets under Strategic Goals A and B also have the potential to
contribute to the attainment of this target. Despite individual success stories, there is no sign of
an overall reduced risk of extinction across groups of species; however there are very large
regional differences. Against this background, possible key actions to accelerate progress
towards this target include:

(a) Identifying and prioritizing species for conservation activities based on
assessments of species conservation status (Target 19);

(b) Filling gaps in existing national, regional and global species conservation status
assessments (Target 19);

(c) Developing and implementing species action plans that include specific
conservation actions aimed directly at particular threatened species, for example through
restrictions on hunting and trade, captive breeding and reintroductions;
(d) Developing more representative and better-managed protected area systems, prioritizing sites of special importance to biodiversity such as Alliance for Zero Extinction Sites and Key Biodiversity Areas (Target 11);

(e) Reducing the loss, degradation and fragmentation of habitats (Target 5), and actively restoring degraded habitats (Target 15);

(f) Promoting fishing practices that take account of the impact of fisheries on marine ecosystems and non-targeted species (Target 6);

(g) Controlling or eradicating invasive alien species and pathogens (Target 9);

(h) Reducing pressures on land use through sustainable land-use practices (Target 7); and

(i) Reducing pressures from trade, by increasing awareness among potential consumers of products from threatened species (Target 1), and through actions agreed under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (Target 4).

Most of the Convention’s programmes of work as well as the Global Strategy for Plant Conservation and the Global Taxonomy Initiative provide guidance that is relevant to this target.

To prevent further extinctions of known threatened species, substantial conservation investment is needed across terrestrial, freshwater and marine ecosystems: it is estimated that investment needs to increase by an order of magnitude in order to reduce the extinction risk of known threatened species (McCarthy et al., 2012). It is essential to continue monitoring the status of species that have already been assessed, and to obtain information on the distribution and extinction risk of less well-studied species to make assessments and future projections of the status of these species. Assessments for species in all environments, but especially for freshwater and marine species, are a work in progress (see Carrizzo et al. 2013 and http://sci.odu.edu/gmsa/index.html), and many critical regions have not been evaluated. Investments in completing these assessments are crucial for conservation decision-makers to be able to adequately represent, evaluate and conserve lesser-known taxa.

Actions aimed at the conservation of threatened species can be broadly categorized as species-level, which are aimed directly at threatened populations (e.g. legislation on hunting and trade, vaccinations, captive breeding and reintroductions) and site, ecosystem, or landscape-level, which are directed at species' habitat (e.g. protected areas, invasive species control, forest management) (Boyd et al., 2008). Species-level actions are generally targeted at the protection of a single species, although protection of habitat thus achieved might also protect other
species (Skaala et al., 2014), while broader site, ecosystem, and landscape-level actions are likely to benefit multiple species (Boyd et al. 2008). In the last 3 decades, species-level conservation actions reduced the extinction risk of nearly 30 species of vertebrates (Hoffmann et al., 2010). Among habitat-level actions, management and protection of sites had a positive effect on 36 species; invasive species control on 16, legislation and education on 14, and hunting and trade management on 8 species of terrestrial vertebrates (Hoffmann et al. 2010). Further such actions will help to avoid extinction and improve the conservation status of species in the future.

Among threatened terrestrial vertebrates, 20% can be protected in single sites, i.e. the entire global population is restricted to, and thus assumed able to be conserved at, sites managed for conservation as a single unit (Boyd et al. 2008). Approximately 60% can be protected through a network of sites, which suggests that effectively managed protected areas aimed at protecting the remaining populations of these species, coupled with site-level management to address the causes of their decline is the most important action to achieve target 12 (Boyd et al. 2008). It is also essential to increase knowledge about the status of other taxonomic groups (Stuart et al., 2010). In order to effectively protect freshwater biodiversity, it will be necessary to target protected areas towards freshwater habitats (see Target 11). Similarly, effectively managed Marine Protected Areas have been shown to allow recovery of fish biomass, particularly of predatory species (Edgar et al. 2014), although the contribution of marine protected areas to reduction in extinction risk has not been quantified. Protection of Alliance for Zero Extinction sites (AZEs) and Important Bird and Biodiversity Areas has increased over time, although progress has slowed recently (Figure 11.2F; Butchart et al 2012). Increasing efforts to protect these sites will help to avert extinctions in the near future.

Species-level conservation needs to be complemented by landscape- or ecosystem-scale policy measures aimed at reducing habitat loss, over-exploitation, pollution, and impact of invasive species and pathogens. In terrestrial and inland water environments, habitat loss is by far the greatest threat to animal and plant species (Hoffman et al. 2010). Therefore, actions aimed at stopping habitat loss and mitigating fragmentation (see chapter 5), and actively restoring degraded habitat (see chapter 15) will be critical for the persistence of many terrestrial and inland water species. The highest density of endangered terrestrial species is in Southeast Asia, owing to deforestation and consequent conversion to cropland and wood plantations, and to direct exploitation of plant and animal species (Orme et al., 2005; Sodhi et al., 2010). For terrestrial species, this region, together with other regions of high endemcity, such as global plant biodiversity hotspots (Myers et al. 2000) and tropical islands, require immediate attention.

Effectively managed Marine Protected Areas are shown to result in significant recovery of fish biomass, particularly for predatory species (Edgar et al. 2014), although the contribution of marine protected areas to reduction in extinction risk has not been quantified. Increased biomass in marine protected areas is also shown to increase ecosystem resilience to climate change effect (Bates et al. 2013). In addition, ecosystem-based fisheries management that
consider the fisheries management in the context of global and local environmental changes and other human impacts should be implemented.

Invasive species are also a major threat: for example, for declining terrestrial invertebrates, invasive species are listed as a major threat for 15% of species (Hoffmann et al., 2010). Therefore, targeted efforts at eradicating invasive species, especially cats and rats on islands, are urgently required to prevent imminent extinctions (see also Target 9). Although the contribution of marine invasive species to extinction has not been quantified, marine invasion should be prevented through a number of measures. These measures include effective control of ballast water discharge, improved public education, monitoring and removal actions to eliminate or suppress invasive species.

For more than 300 amphibian species affected by the chytrid fungus *Batrachochytrium dendrobatidis* (Vredenburg et al. 2010), and several other critically endangered species with very small and declining populations, captive breeding will be required until the causes of decline are removed or mitigated sufficiently to permit reintroduction (Boyd et al. 2008; Stuart et al., 2008).

Given the stress placed on native freshwater and marine fishes and invertebrates by unsustainable harvesting, there is an urgent need to reduce such harvesting and to develop more sustainable methods and approaches that would improve the status of many threatened species and improve livelihoods for humans (see also targets 6, 7 and 14). Use of destructive fishing methods, particularly on vulnerable habitats, should be prevented. By-catch of vulnerable and endangered species such as some sharks, seabirds, sea turtles needs to be substantially reduced. To improve current population trends it will be critical to complement actions towards sustainable harvests with effective placement and management of protected areas, some of which would be beneficially established through locally managed areas (see also Target 11), to protect critical habitats and to restore abundance and increase resilience of populations.

Habitats and species are rarely affected by single pressures, and therefore multiple coordinated actions are required. Species-level management is therefore best coordinated through action plans. These have been produced and updated for several taxonomic groups by the International Union for Conservation of Nature (https://www.iucn.org/about/work-programmes/species/publications/species_actions_plans/), other NGOs and regional and local government authorities worldwide. A notable example is the Global Strategy for Plant Conservation (GSPC), established by the Convention on Biological Diversity and updated for the period 2011-2020 http://www.cbd.int/gspc/. The incorporation of GSPC targets and species actions plans into NBSAPs (National Biodiversity Strategies And Action Plans; cross-reference to Target 17) and their timely implementation will be critical to prevent the extinction of many known threatened species.
2.b. Costs and Cost-benefit analysis

Global estimates of the costs of meeting Target 12 suggest that US$3.4 to 4.8 billion will be required per year (McCarthy et al., 2012; High Level Panel, 2014). This estimate was based on extrapolation of the estimated cost of actions needed to improve the conservation status (Red List status) of a sample of 211 threatened bird species combined with data on the relative costs of conservation actions for birds and a wide range of other taxa (McCarthy et al., 2012). Assuming that conservation actions undertaken for each species are entirely independent of one another, it is estimated that improving the status of all bird species will cost $1.23 billion per year (McCarthy et al., 2012). Recognizing that some conservation actions will benefit species other than the target species, total costs are estimated at $0.88 billion per year (McCarthy et al., 2012). Extrapolating these costs from the 1,115 globally threatened bird species to the 13,452 other known threatened species, it is estimated that improving the status of all known threatened species will cost between $3.41 billion and $4.76 billion per year (McCarthy et al., 2012). Current funding is only 12% of that required (McCarthy et al., 2012).

Quantifying the total value of the benefits provided by biodiversity to human society, and thus the economic benefits of preventing extinctions and meeting Target 12, is impossible. However, almost all analyses that have been carried out have suggested that the benefits of conservation actions outweigh the costs. For example, pollination services provided by insect species have been estimated to be worth $19 to 21 billion per year in the European Union alone (High Level Panel, 2014). Furthermore, it has been estimated that 2.5% to 16% of all jobs in the European Union depend on the environment to some degree and that 5.8% of jobs in sub-Saharan Africa depend on tourism, much of which is nature-based (High Level Panel, 2014). It has been estimated that a network of protected areas that adequately conserved biodiversity would achieve a benefit-to-cost ration of 100:1 (Balmford et al., 2002).

3. What are the implications for biodiversity in 2020?

Extinction of species, both local and global, will have profound effects on ecological communities more broadly and on the functioning of ecosystems. The non-random loss of species from ecological communities leads to those communities becoming homogenized and dominated by certain functional types of species (Newbold et al., in press). The loss of key species from communities can lead to altered interactions among species and ultimately to trophic cascades (Estes et al., 2011). Finally, the local extinction of species from ecological communities will impair the functioning of ecosystems: recent meta-analyses have shown that more diverse communities function more resiliently over space and time, in the face of environmental changes (Isbell et al., 2011).
4. What do scenarios suggest for 2050?

Land-use change and over-exploitation will remain the major drivers of terrestrial species loss until 2050, but with climate change increasing in importance over time (Alkemade et al., 2009; Collen et al., 2014). Most of the Rio+20 scenarios (Netherlands Environmental Assessment Agency, 2010) predict further declines in population trends of species and in the average local diversity of ecological communities, and further increases in species’ extinction risk, although these changes are slowed or in some cases reversed under scenarios that assume some sort of mitigation efforts (consumption or technology changes) (Figure 12.3).

Climate change may affect species directly, through their physiological tolerance, or indirectly through changes in vegetation (Powell & Lenton, 2013). Marine species are also threatened by ocean acidification and hypoxia (Vaquer-Sunyer & Duarte, 2008; Godbold & Calosi, 2012). The combined effects of these stressors may further exacerbate the effects of climate change on marine biodiversity (Mora et al. 2013). The frequency and intensity of extreme climate events is also likely to have a major impact on future fisheries production in both inland and marine systems. Shifts in the migration phenology of many species important for commercial and recreational fisheries have been attributed to climate change, including: Pacific salmon (Quinn and Adams 1996), Atlantic salmon (Juanes et al. 2004) and smelt (Ahas and Aasa 2006). There are strong interactions between the effects of fishing and the effects of climate because fishing reduces the age, size, and geographic diversity of populations and the biodiversity of marine ecosystems (Brander 2007), which makes species more vulnerable to the potential effects of climate change. Inland (freshwater) fisheries are additionally threatened by changes in precipitation and water management (Palmer et al. 2008; Strayer and Dudgeon 2010).

In all environments, synergistic effects of multiple drivers could further increase biodiversity loss. For example, the impact of habitat loss on species has been shown to be worsened by climate change (Mantyka-Pringle et al. 2012), the invasion and spread of exotic plants has been shown to be more likely given higher rates of land-use change (Chytrý et al. 2012). These results suggest that future biodiversity assessments should consider the interacting effects of multiple threats to biodiversity loss, rather than treating the effects of drivers as being additive. Distribution shifts driven by climate change will alter biodiversity patterns (Lawler et al., 2009) and may affect trophic interactions, although the implications of the latter for extinction risk are not yet clear.

Biodiversity in some marine habitats, such as coral reefs (cross-reference to Target 10), is particularly sensitive to projected climate change and ocean acidification. At a global scale, the potential impact of climate change on freshwater biodiversity remains poorly understood, but is projected to present a growing challenge to the integrity and function of freshwater systems (Dudgeon et al. 2006).
In the technical report on modelling and scenarios for the Global Biodiversity Outlook 3 (Leadley et al., 2010), models predicting future changes (to 2050) in extinction rates, average abundance of species within ecological communities, and species distributions were reviewed. In summary:

projected extinction rates ranged from values similar to current ones (for models of projected species-specific habitat loss) to two orders of magnitude larger (for models based on the species-area relationship); models of projected species abundance (all based on the GLOBIO model, Alkemade et al. 2009) predicted a mean decline of 9-17% in abundance by 2050; for both species loss and decrease in abundance, the socio-economic scenarios reviewed only made small differences in the predicted outcomes; all studies of changes in species distributions (mostly based on niche models or global vegetation models) predict distributional shifts that result in changes of biotic communities and potentially the creation of new communities, yet there is great uncertainty on the rate and extent of change.

Since the publication of the Global Biodiversity Outlook 3, several studies have advanced our understanding of biodiversity scenarios for 2050.

There is a consensus that there will be widespread local extinctions of species in both marine and terrestrial environments, driven by climate change (Fig. 12.4; 12.5; Cheung et al. 2009; Bellard et al. 2012), which are also likely to trigger cascade effects through co-extinctions of dependent species (Brook et al. 2008), which could result in loss of ecosystem services (Hooper et al. 2012; Tilman et al. 2012).

**Figure 12.4.** Projections biodiversity loss owing to climate change (and other drivers). The width of the box indicates the generality of the predictions with respect to spatial scale and taxonomic breadth. The box is delimited by the upper and lower boundaries of the intermediate scenario, while the whiskers indicate the highest and lowest biodiversity losses across all scenarios. The highest estimates of local losses are obtained when considering direct effects of climate on species by projecting their bioclimatic envelope (e.g. Thomas et al. 2004; Thuiller et al. 2005) and at the lowest end when considering only
indirect effects through changes in land cover (Jetz et al., 2007). Reproduced from Bellard et al. (2013).

Note, this figure needs to be replaced with a high-resolution version during copyediting.

Figure 12.5. Proportion of exploited marine fishes and invertebrates (1066 spp.) predicted to become locally extinct given predicted climate change under the SRES A2 emissions scenario by 2050, relative to 2000. Adapted from Cheung et al. (2009). Note: updated versions of these models are still being run but will be ready in time for the final version.

Species can survive climate change by shifting their ranges or by adapting, either through evolutionary change (i.e. changes in behavioural, physiological or ecological traits) or through phenotypic plasticity (i.e., the species already possess the required traits to survive under new climatic conditions and these traits are selected for within the existing pool).

For terrestrial species, the velocity of climate change (Loarie et al. 2009) is expected to outpace the dispersal ability of most species, across several studied taxa (Bertrand et al. 2011; Devictor et al. 2012; Schloss et al. 2012). Species with narrow altitudinal ranges and low thermal tolerance, especially those inhabiting high mountains, are predicted to incur local extinctions in several regions of the world (Laurance et al., 2011; Dullinger et al., 2012). Furthermore, for terrestrial species to adapt evolutionarily to climate change would require rates of niche evolution that are more than 10,000 times faster than those typically observed (Quintero & Wiens 2013). However, a recent study revised upwards many previous estimates of species ability to shift their range (Chen et al. 2011), and several marine and freshwater groups appear able to keep pace with climate change (Kappes & Haase 2012; Kinlan & Gaines 2003).

Projected changes vary substantially in different parts of the world owing to variation in the different drivers of biodiversity change. Under business-as-usual scenarios, particularly strong declines are predicted in Africa, because of expanding agriculture, livestock production and forestry (Jetz et al. 2007; Visconti et al. 2011; Fig. 12.6). Large declines of terrestrial species are also predicted in the Amazon, a region with very low spatial climatic gradients that is predicted to experience no-analog future climates (Williams et al. 2007; Fig. 12.6), and which is rich in vertebrate species with high intrinsic vulnerability to climate change (Foden et al. 2013). Finally, large declines and turnover rates are predicted in areas rich in elevational specialists (Laurance
et al. 2011) such as the Andes for mammals (Lawler et al. 2009; Schloss et al. 2012) and the
Himalaya for birds (Jetz et al. 2007). For marine fish and invertebrate species the areas with
highest expected local extinctions by 2050 are sub-polar regions, the tropics and semi-enclosed
seas (e.g. the Mediterranean and the Red Sea; Fig. 12.5); while the areas with highest number
of expected invasions are the Arctic and Southern Ocean (Cheung et al. 2009). Inland waters
remain one of the most highly threatened ecosystems (Vörösmarty et al. 2010), and regardless
of the land use change scenario considered the biodiverse catchments of the southeast United
States are expected to see dramatic urban expansion (Martinuzzi et al. 2013b). Under some
scenarios the southeast United States is also expected to see crop expansion that would further
fragment and pollute critical freshwater habitats (Martinuzzi et al. 2013b).

Figure 12.6. Modelled change in several measures of biodiversity (average population trends of carnivore
and ungulate species, calculated following the methods used by the Living Planet Index – LPI; Visconti et
al., in review; average local species richness; Newbold et al., in prep.; and average local abundance of
species; Alkemade et al., 2009) between 2000 and 2050 under a baseline scenario (left panels) and the
‘technology change’ Rio+20 scenario (right panels) (scenarios presented in Netherlands Environmental
Assessment Agency, 2010). Local abundance and LPI measures included both direct effects of climate change and indirect effects (through land cover change). Local species richness change accounted for indirect effects. For these maps, LPI was calculated for each grid cell by aggregating population trends in projected population size within the cell for all carnivore and ungulate species. This contrasts with the LPI calculations in figure 12.3 where the trend in population size for each species were calculated globally and aggregated across all species. In LPI calculations, the local extinctions where replaced with 1% of the maximum population size to avoid calculating a geometric mean with a zero. Because changes in LPI are sensitive to species richness, grid cells with <10 species of carnivore and ungulate species were removed from the LPI analyses.

A number of studies have focused on particular regions. Range shifts and contractions are predicted by 2050 for two-thirds of European breeding birds (Barbet-Massin et al. 2012), for tree species in France (Cheaib et al. 2012) and for Alpine plants (with an almost 50% reduction in range size by 2100; Dullinger et al. 2012). Contractions and shifts in the distributions of European plants, birds and mammals are expected to be similar across taxonomic groups, because sensitivity to climate change is not strongly correlated with phylogeny (Thuiller et al. 2011). In Australia, 67% of Australian savanna bird species ranges are projected to decrease. However, migratory and tropical-endemic birds are predicted to benefit from climate change with increasing distributional area. Richness hotspots of tropical savanna birds are also expected to move, increasing in southern savannas and southward along the east coast of Australia, but decreasing in the arid zone (Reside et al. 2012).

Long-term projections of land use change under the Rio+20 scenarios indicate that habitat loss will continue to pose a threat to biodiversity under the business-as-usual scenario (Fig. 12.3 A-D), particularly in Africa and Central Asia (Figure 12.6; Visconti et al., submitted). Regional models that predict the impacts of land-use change on species extinction similarly predict a worsening situation under business-as-usual scenarios: in the Brazilian Amazon 10% of species, on average, are predicted to become locally extinct as a result of forest loss, with a further 27% committed to extinction (Wearn et al., 2012). In the mitigation scenarios, short-term trends in the diversity of ecological communities (see previous section) are continued (Fig. 12.3 A, D; Newbold et al., in prep.), driven by continued land-use change, which is captured better than climate change in the GLOBIO and PREDICTS model frameworks. For population trends and extinction risk of carnivores and ungulates, the short-term gains begin to be reversed by 2050 as climate change overcomes the beneficial effects of reduced land-use change (Figure 12.3 C, D; Visconti et al., 2011). Similarly, scenarios for the Brazilian Amazon that predict reduced rates of deforestation lead to reduced local extinctions of species (by 37-57% for actual extinctions and 61-82% for extinction debt, depending on the scenario adopted; Wearn et al., 2012).
5. Uncertainties

There are, unavoidably, many uncertainties in the various methods used to make predictions about the future of biodiversity. However, all of the extrapolations and models reviewed here confer a high degree of confidence in the assertion that current trends in extinction risk of species, and in the conservation status of species and ecological communities, are strongly negative. Furthermore, all of the models employing business-as-usual scenarios predict further strong declines throughout this century.

Status and trends

Estimates of past extinction rates are uncertain for several reasons. First, the number of field biologists is small relative to the number of species, and therefore extinction rates can be estimated only for a few well-studied and possibly atypical groups (mainly vertebrates), while extinctions can go undetected in species-rich but poorly studied groups (Balmford et al. 2003). Second, being confident that a species is actually extinct requires levels of survey effort that very often exceed available resources even for very well-studied groups (Butchart et al., 2006; Scott et al., 2008). Finally, species do not immediately respond to human pressures, and extinction can be delayed for centuries (Tilman et al., 1994). Even for species that are almost certainly extinct, knowing exactly when extinction occurred is difficult, and most known extinctions are accompanied by a range of likely dates.

All of the measures used to assess status and trends were biased toward vertebrate species in terrestrial environments, and therefore our knowledge of recent changes in the status of invertebrate species, and all species in freshwater and marine environments, is much more limited.

Projections

There are fundamental differences between extrapolating the past trend of an indicator into the future, and modelling future trends based on scenarios of the underlying pressures, which leads to large differences in projected outcomes. The statistical extrapolations used to project trends to 2020 assume that the extent to which targets will be achieved is only a function of the inertia in the underlying mechanisms that underpin the trends, while the models based on scenarios of how the pressures will change explicitly project the underlying mechanisms underpinning the trends which therefore can take different trajectories respect to the past. For example in the case of the Red List Index, the extrapolations assumed a constant trend in the indicator, and therefore a further increase in extinction risk, while the more process-driven models assumed a constant trend in the pressures, a slower short-term decline in population size and therefore a reduction in extinction risk.
For scenario-based modelling, there is great variation in projected future extinction rates both within and between studies, with three factors explaining much of this variation. First, the degree of land use and climate change predicted by different scenarios explains much of the variation in projected extinctions within studies. For example, Thomas et al. (2004) projected vertebrate extinctions are 11 to 34% for 0.8° to 1.7°C global warming versus 33 to 58% for >2.0°C warming (the magnitude of these predicted losses has been disputed since, and they are based on species distribution models, which are subject to numerous sources of uncertainty – see e.g. Thuiller et al., 2004; Araújo & Guisan, 2006). Second, an important contribution to the broad range of projections within studies is different assumptions about species life-history traits, especially displacement rates (e.g., the highest projected extinction rates are 38% with unlimited migrations rates versus 58% with no migration in Thomas et al. 2004) and habitat specificity (e.g. in Malcolm et al. 2006 the highest extinction rates are 7% for broad habitat specificity versus 43% for narrow habitat specificity), emphasizing the need for research on these fundamental aspects of species ecology and their incorporation into global models (Foden et al. 2013; Thuiller et al. 2008). Third, there is a substantial degree of uncertainty in the climate and land-use change models themselves, which contributes to uncertainty in projected rates of extinction or biodiversity loss.

Species response models

A large fraction of the variation in predicted outcomes for biodiversity among studies arises from differences between modelling approaches. For example, Sekercioglu et al. (2008), using a logistic model of extinction risk as a function of range size, predicted ten times more birds extinction than Jetz et al. (2007), using a linear, mechanistic model of extinction as a function of habitat suitability. The assumed linear relationships between habitat and population decline, which underlie many predictions of global extinctions (e.g. Jetz et al., 2007; Visconti et al., in review), may lead to underestimates of species global extinction risk (Di Fonzo et al. 2013).

Studies using bioclimatic envelope models tend to project larger range contractions and increase in extinction risk under future climate change than other approaches. This is likely to be in part due to the largely untested assumption that species will not survive climatic conditions never experienced before, whereas species might adapt to climate change through phenotypic plasticity or micro-evolution (Charmantier et al. 2008; Boutin & Lane 2014). However, there are a number of other known limitations of species distribution models, which could contribute to their predicting relatively large changes (Araújo & Guisan, 2006).

Species-area relationship models tend to give larger measures of extinction risk (Pimm & Raven 2000; Thomas et al. 2004) because they are based on the accumulation of species with expanding sampling area, which may not accurately reflect the scaling of species extinction with reduced area (Lewis, 2006; He & Hubbell 2011; but see e.g. Pereira et al., 2012; Pimm & Brooks, 2013), and have in at least one case (Thomas et al., 2004) been criticized for misapplying the Red List criteria (Akçakaya et al., 2006). Species-area relationships also measure species
committed to extinction. However, the lag time between being committed to extinction and actually going extinct may range from decades to centuries (Stork, 2010; Wearn et al., 2012).

Spatially-explicit metapopulation models probably make more conservative and robust estimates of extinction risk by avoiding several of the assumptions described above (Pearson et al. 2014), but are computationally intensive and require data available for only a fraction of species.

Global change models

Additional uncertainty in model projections arises from their coverage of threats affecting species. The LPI and RLI projections for large mammals (Figure 12.3; 12.6) only accounted for land use and climate change despite direct persecution being an important threat for these species. The PREDICTS model projections (Figure 12.3; 12.6) were based on only land-use change and indirect impacts of climate change through biome shifts. None of the terrestrial models reviewed here accounted for direct harvesting of species, and the Rio+20 scenarios did not include future projections of human population density, which could act as a proxy for pressure from direct harvesting in terrestrial environments. The qualitative differences between the Rio+20 scenarios used here are unlikely to be affected by the inclusion of direct harvest, because several factors (low food security, poor access to food markets and a high proportion of people living in rural areas) mean that direct harvest of species is likely to be greatest in the business-as-usual scenario.

For marine species, the projections (Figure 12.5) focused on climate change as a driver. Addition of other threats, particularly fishing and habitat loss might modify the rate of local extinction.

Finally, there is uncertainty in the outputs from climate or Earth System models used to drive the projections, particularly at the finer spatial scale often used in biodiversity modelling (Stock et al., 2011). Models often do not consider trophic interactions in detail, or the scope for evolutionary adaptation. Overall, based on fundamental biological and ecology theory, observations and model projections, the large-scale direction of local extinction indicated by model projections is more robust, while the projected magnitude and finer-scale patterns more uncertain.
6. Dashboard – Progress towards Target

<table>
<thead>
<tr>
<th>Target Elements</th>
<th>Status</th>
<th>Comment</th>
<th>Confidence</th>
</tr>
</thead>
<tbody>
<tr>
<td>Extinction of known threatened species has been prevented</td>
<td></td>
<td>Further extinctions likely by 2020, e.g. for amphibians and fish. For bird and mammal species some evidence measures have prevented extinctions</td>
<td>Low</td>
</tr>
<tr>
<td>The conservation status of those species most in decline has been improved and sustained</td>
<td></td>
<td>Red List Index still declining, no sign overall of reduced risk of extinction across groups of species. Very large regional differences</td>
<td>High</td>
</tr>
</tbody>
</table>

Complied by: Tim Newbold, Piero Visconti, Carlo Rondinini, William Cheung and Stephanie Januchowski-Hartley, with contributions from Andy Purvis and Stuart Butchart.

Extrapolations: Derek Tittensor

NBSAPs and National Reports: Kieran Mooney / CBD Secretariat

Dashboard: Tim Hirsch

6. References


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1 This provides a current assessment of progress towards the Aichi Biodiversity Target based on the material presented in this chapter and the expert judgement of the authors of the GBO-4 Technical Report. It is subject to change as additional material becomes available, including information from national reports, NBSAPs and the BIP partnership.


