IPBES Global Assessment Chapter 3 Supplementary Online Materials

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S3.1 Quantitative analysis of progress towards the Aichi Targets

S3.1.1 Methods

Datasets for a total of 68 indicators were compiled to assess progress towards the Aichi Targets (Table S3.1). This included all of the those considered by Tittensor et al. (2014), apart from Red List Index for seabirds (replaced by Red List Index showing impacts of fisheries), Protected area coverage of Alliance for Zero Extinction sites and Protected area coverage of Important Bird and Biodiversity Areas (both of which were combined into a new indicator Protected area coverage of Key Biodiversity Areas), Insecticide use (incorporated within a new indicator Pesticde use), rate of mammal and bird extinctions (which has not been updated since the 2000-2010 datapoint), and Protected area coverage of freshwater ecoregions (dropped as it is now judged to be a poor indicator given the large size of these areas and the high proportion of non-freshwater habitat included). Datasets were updated for all bu 22 of these indicators. An additional 16 indicators were included: Red List Index (internationally traded species), Area of tree cover loss (ha), Red List index (forest specialists), Marine trophic index, Nitrogen use balance (kg/km²), Climatic Impact Index for birds, Area of mangrove forest cover (km²), Number of plant genetic resources for food and agriculture secured in conservation facilities, Red List Index (wild relatives of farmed and domesticated species), Percentage change in local species richness, Red List Index (species used for food and medicine), Percentage of global rural population with access to improved water resources, Percentage of countries that have ratified the Nagoya Protocol, Percentage of countries with revised NBSAPs, Species Status Information Index, and Proportion of known species assessed through the IUCN Red List (Table 3.3, S3.1).

Table S3.1 Indicators used in the quantitative analysis of progress towards the Aichi Targets, their characteristics, and projected trends. Numbered target elements correspond to Table 3.3. Indicators marked with \dagger are considered 'IPBES core indicators'. Spatial coverage is scored as poor (1-2 continents, or 3-4 continents and <10 countries), moderate (3-4 continents and \geq 10 countries, or \geq 5 continents and <20 countries), or good (or \geq 5 continents and \geq 20 countries). Asterisks indicate time-series updated since Tittensor et al. (2014) or indicators additional to (or replacing indicators used by) Tittensor et al. (2014).

Strat- egic Goal	Aichi Target	Target Ele- ment	Indicator name	Туре	Spatial coverage	Align- ment	Sampling dates	Projected trend to 2020
А	1	1.1	Biodiversity Barometer (% of respondents that have heard of biodiversity)	Response	Poor	High	2009-2016*	Significant increase
А	1	1.1	Biodiversity Barometer (% of respondents giving correct definition of biodiversity)	Response	Poor	High	2009-2016*	Significant increase
А	1	1.1	Funding towards environmental education (\$)	Response	Good	Low	1995-2010	Non- significant decline
А	1	1.2	Online interest in biodiversity (proportion of google searches)	Response	Good	Medium	2004-2016*	Non- significant decrease

А	2	2.2	Funding towards Environmental Impact Assessment (\$)	Response	Good	Low	1995-2012	Non- significant decrease
А	2	2.4	Number of research studies involving economic valuation	Response	Good	Low	1974-2010	Significant increase
А	3	3.2	World Trade Organisation ' green box' agricultural subsidies (\$)	Response	Good	Medium	1995-2011	Non- significant increase
A	3	3.2	Funding towards institutional capacity building in fisheries (\$)	Response	Good	Low	1995-2012	Non- significant increase
А	4	4.1	Percentage of countries that are Category 1 CITES Parties ⁺	Response	Good	High	1994-2016*	Significant
А	4	4.2	Ecological Footprint (number of earths needed to support human society)†	Pressure	Good	High	1961-2012*	Non- significant increase
А	4	4.2	Red List Index (impacts of utilisation)	Pressure	Good	High	1986-2016*	Significant decrease
А	4	4.2	Red List Index (internationally traded species)	State	Good	Medium	1988-2016*	Significant decrease
А	4	4.2	Human appropriation of net primary productivity (Pg C)	Pressure	Good	Low	1910-2005	Significant increase
А	4	4.2	Human appropriation of fresh water (water footprint) (thousand km3)†	Pressure	Good	High	1995-2009	Significant
В	5	5.1	Wetland Extent Trends Index	State	Good	Medium	1970-2015*	Significant decrease
В	5	5.1	Area of tree cover loss (ha)†	State	Good	High	2001-2016*	Significant increase
В	5	5.1	Percentage natural habitat extent	State	Good	High	1961-2011	Significant decrease
В	5	5.2	Wild Bird Index (habitat specialists)	State	Poor	Low	1968-2014*	Significant decrease
В	5	5.2	Red List index (forest specialists)	State	Good	Low	1988-2016*	Significant decrease
В	6	6.1	Proportion of fish stocks in safe biological limits†	State	Good	High	1974-2013*	Non- significant decrease
В	6	6.1	Marine Stewardship Council certified fisheries (tonnes)†	Response	Good	High	1999-2016*	Significant increase
В	6	6.3	Marine trophic index†	Pressure	Good	High	1960-2014*	Non- significant decrease
В	6	6.3	Red List Index (impacts of fisheries)	Pressure	Good	Medium	1988-2016*	Significant decrease
В	6	6.3	Global effort in bottom-trawling (kW sea-days)	Pressure	Good	Medium	1950-2006	Significant increase
В	7	7.1	Nitrogen use balance (kg/km2)†	Pressure	Good	Low	1961-2011*	Non- significant increase
В	7	7.1	Wild Bird Index (farmland birds)	State	Poor	Medium	1980-2014*	Significant decrease
В	7	7.1	Area of agricultural land under organic production (million ha)	Response	Good	High	1999-2014*	Significant
В	7	7.1	Area of agricultural land under conservation agriculture (thosuand ha)	Response	Good	High	1990-2011	Significant
В	7	7.3	Area of forest under sustainable management: total FSC and PEFC forest management certification (million ha) [†]	Response	Good	High	2000-2016*	Significant increase
В	8	8.1	Red List Index (impacts of pollution)	State	Good	High	1988-2016*	Significant decrease
В	8	8.1	Pesticide use (tonnes)†	Pressure	Good	Medium	2000-2011*	Significant increase
В	8	8.2	Nitrogen surplus (Tg N)	Pressure	Good	Medium	1970-2005	Significant
В	9	9.1	Number of invasive alien species introductions	Pressure	Moderate	Medium	1500-2012	Significant
В	9	9.3	Red List Index (impacts of invasive alien species)	Pressure	Good	Medium	1988-2016*	Significant decrease
В	9	9.4	Percentage of countries with invasive alien species legislation	Response	Good	High	1967-2009	Non- significant increase
В	10	10.1	Percentage live coral cover	State	Good	High	1972-2016*	Non- significant decrease

В	10	10.2	Glacial mass balance (mm water	State	Moderate	Medium	1957-2015*	Significant
В	10	10.2	equivalent) Mean polar sea ice extent (million km2)	State	Good	Medium	1979-2015*	decrease Non-
Б	10	10.2	Mean polar sea ice extent (minion km2)	State	0000	Wedrum	1979-2013	significant decrease
В	10	10.2	Climatic Impact Index for birds	Pressure	Poor	Low	1980-2010*	Non- significant increase
В	10	10.2	Area of mangrove forest cover (km2)	State	Good	Medium	2000-2014*	Significant decrease
С	11	11.1	Percentage of marine and coastal areas covered by protected areas ⁺	Response	Good	High	1990-2016*	Significant
С	11	11.2	Percentage of terrestrial areas covered by protected areas [†]	Response	Good	High	1990-2016*	Significant increase
С	11	11.3	Percentage of Key Biodiversity Areas covered by protected areas	Response	Good	High	1980-2017*	Significant increase
С	11	11.4	Percentage of terrestrial ecoregions covered by protected areas	Response	Good	High	1911-2012	Significant increase
С	11	11.4	Percentage of marine ecoregions covered by protected areas	Response	Good	High	1911-2012	Significant increase
С	11	11.4	Protected area coverage of bird, mammal and amphibian distributions	Response	Good	High	1990-2012	Significant increase
С	11	11.5	Number of protected area management effectiveness assessments [†]	Response	Good	Medium	1990-2013	Significant increase
С	11	11.5	Funding towards nature reserves (\$)	Response	Good	Low	1995-2012	Non- significant increase
С	12	12.2	Living Planet Index	State	Moderate	High	1970-2012*	Significant decrease
С	12	12.2	Red List Index [†]	State	Good	High	1994-2016*	Significant decrease
С	12	12.2	Funding towards species protection (\$)	Response	Good	Low	1995-2012	Non- significant increase
D	13	13.1	Number of plant genetic resources for food and agriculture secured in conservation facilities	Benefit	Good	High	1995-2016*	Significant increase
D	13	13.2	Percentage of terrestrial domesticated animal breeds at risk ⁺	Benefit	Good	High	2000-2013	Significant increase
D	13	13.3	Red List Index (wild relatives of farmed and domesticated species)	Benefit	Good	High	1988-2016*	Significant
D	14	14.1	Percentage change in local species richness	State	Good	Low	1970-2014*	Non- significant
D	14	14.1	Red List Index (species used for food and medicine)	Benefit	Good	Medium	1986-2017*	Significant decrease
D	14	14.1	Red List Index (pollinator species)	Benefit	Good	Low	1988-2016*	Significant decrease
D	14	14.2	Percentage of global rural population with access to improved water resources	Response	Good	Low	1990-2015*	Significant increase
D	16	16.1	Percentage of countries that have ratified the Nagoya Protocol	Response	Good	High	2011-2017*	Significant increase
E	17	17.1	Percentage of countries with revised NBSAPs†	Response	Good	High	2010-2017*	Significant increase
E	19	19.1	Species Status Information Index†	Response	Good	Medium	1980-2014*	Non- significant increase
E	19	19.1	Number of biodiversity papers published	Response	Good	High	1980-2016*	Non- significant increase
E	19	19.1	Proportion of known species assessed through the IUCN Red List †	Response	Good	Medium	2000-2017*	Significant increase
E	19	19.1	Number of species occurrence records in the Global Biodiversity Information Facility	Response	Good	Low	2003-2016*	Significant
Е	19	19.1	Funding committed to environmental research (\$)	Response	Good	Low	1995-2012	Non- significant
E	20	20.1	Funding provided by the Global Environment Facility (\$)	Response	Good	High	1991-2016*	increase Significant increase
Е	20	20.1	Official Development Assistance provided in support of the CBD objectives (\$)	Response	Good	High	2006-2015*	Significant

Е	20	20.1	Global funding committed towards	Response	Good	Medium	1995-2012	Non-
			environmental policy, laws, regulations	-				significant
			and economic instruments (\$)					increase

We assembled a broad suite of indicators to estimate historical trends and project to 2020, building on those used by Tittensor et al (2014) and CBD (2014). Tittensor et al (2014) used the CBD's indicative list (CBD 2012) and scoped more than 160 potential indicators, reviewing them against five criteria: (i) high relevance to a particular Aichi Target and a clear link to the status of biodiversity; (ii) scientific or institutional credibility; (iii) a time series ending after 2010; where unavailable but indicator fills a sizable gap, data ending as near to 2010 as possible; (iv) at least five annual data points in the time series; and (v) broad geographic (preferably global) coverage. Of the 163 potential indicators, 55 met these criteria and were included in Tittensor et al's analysis. We expanded the set to include 68 indicators in total.

Following Tittensor et al (2014), we fitted models to estimate underlying trends using an analysis framework adaptive to the highly variable statistical properties of the indicators. Dynamic linear models (Durbin and Koopman 2001) were fitted to high-noise time series, while parametric multimode averaging (Burnham and Anderson 2002) was used for those with low noise. We projected model estimates and confidence intervals to 2020 to estimate trajectories and rates of change for each indicator (Table S3.2). As most targets lack explicitly quantifiable definitions of "success" for 2020 (and those that have definitions for some components lack them for others), it was not generally possible to measure progress in terms of distance to a defined end point. Therefore, we assigned indicators as relating to states, drivers, responses or nature's contributions to people, and compared projected values in 2020 against modelled 2010 values (underlying trend estimates) for all indicators, while additionally measuring absolute progress where possible. For protected area coverage of the terrestrial environment, marine environment, and Key Biodiversity Areas, we took 17%, 10%, and 100% as thresholds for achievement of the target. For protected area coverage of ecoregions, we assume achievement of the target would require 100% of terrestrial and freshwater ecoregions to have 17% protected area coverage, and 100% of marine ecoregions to have 10% protected area coverage. For protected area management effectiveness assessments, we assume achievement of the target would require all protected areas to have had their management effectiveness assessed.

Selecting indicators for use in the analysis

The first step in the analysis was to identify indicators that could be used to project trends to 2020 by assessing them against the analysis criteria (Tittensor et al 2014):

- (i) substantial relevance to a particular Aichi Target and a clear link between the indicator and the status of biodiversity
- (ii) (scientific or institutional credibility, in terms of the indicator dataset or its underlying methodology being peer-reviewed and generally accepted by the scientific community, it being developed or used by an international public or third sector organization, or being used in previous global assessments of biodiversity trends (e.g. (1))
- (iii) a start point before 2010 and end-point after 2010 where feasible, and where not feasible but the indicator was essential due to a lack of alternatives for the Target, a long series of data points ending as near to 2010 as possible
- (iv) at least 5 annual data points in the time-series

(v) broad geographic (preferably global) coverage.

Statistical modelling framework

Once the indicators used to assess progress towards Aichi Target had been selected, we used an adaptive statistical framework to fit models and project time trends based on the properties of each individual time-series, using the statistical properties of the data to select an appropriate modeling paradigm; noisy time series needed an approach designed to deal with this property, while those with low levels of noise required a separate method (Tittensor et al. 2014).

We first divided the series into two fundamental categories depending on whether or not they exhibited a statistically significant white noise component. We calculated the signal-to-noise ratio by fitting a dynamic linear (state space) model to each time-series based on a random walk plus noise model. Indicators with a low signal-to-noise ratio (i.e. significant noise) were then fitted using linear Gaussian sate-space models (i.e. a Kalman filter and smoother) with a time-varying trend. This approach is specifically developed to filter white noise. Conversely, time-series with a high signal-to-noise ratio (i.e. low noise) were fitted with deterministic models using a multi-model parametric approach assuming an unknown underlying functional form and then model-averaging. Given the desire for a unified approach that could be used to seamlessly compare projections between both these parametric methods, we transformed data where necessary to ensure that the assumptions of Gaussian errors (used in both the multi-model and dynamic linear model approaches). In both cases we visually inspected the residuals for independence of residuals, homogeneity of variance, or contravention of the assumptions of normality; where the latter two occurred, we applied logtransformations and arcsin square root transformations as appropriate. Where autocorrelation was visible in the residuals, we added AR1 and AR2 terms to correct for this.

(i) Dynamic linear models. The dynamic linear model approach fitted a model consisting of global mean with a locally varying trend to each time series. This allowed for a temporally evolving rate parameter within each time-series, thus enabling the model to capture significantly non-linear behavior. Models were fit using maximum likelihood and a Kalman Filter in the R package dlm.

(ii) Multi-model parametric models. The multi-model parametric approach assumes a deterministic trend with an unknown functional form. A total of 18 candidate trend models were fit to each time-series as an ensemble. The models were selected for their ability to fit a wide range of functional forms, such as linear, exponentially increasing or decreasing, asymptotic and others. The ensemble models were ranked according to the Akaike information criterion (AIC) value which is an information theoretic-based goodness of fit statistic and takes into account model fit, complexity, and sample size,

$$AIC = -2\ln[L(\theta_p|y)] + p$$

where *n* is the sample size, $L(\theta_p | y)$ are the likelihood estimates of the model parameters θ_p , given the data *y*, and *p* is the number of free parameters estimated by the model. The overall

model is then represented as a multi-model average of the top-scoring candidate models (as defined below), with weights proportional to their relative AIC score. Small sample size corrected AIC was used to adjust for time series with few data points (5). Normalized multi-model weights for each ensemble model (w_i) were calculated as,

$$w_{i} = \frac{\exp\left(-\frac{1}{2\Delta_{i}}\right)}{\sum_{i=1}^{R} \exp\left(-\frac{1}{2\Delta_{i}}\right)}$$

where R represents the total number of ensemble models, and

$$\Delta_i = AIC_i - AIC_{min},$$

where AIC_i is the AIC score for model i, and AIC_{min} is the minimum (top ranking) AIC score in the ensemble model set. In this manner, the 'best' model of the ensemble is denoted by the largest information-theoretic weight (w_i); this approach selects the model containing the largest amount of information.

Multi-model predicted time trends were derived by calculating a weighted average from the ensemble model predictions as

$$\hat{\theta} = \sum_{i=1}^{R} w_i \hat{\theta}_{i_i}$$

where $\overline{\hat{\theta}}$ is the multi-model averaged prediction, $\hat{\theta}_i$ is the ensemble prediction, and w_i is the standardized weight for model i. The uncertainty of the multi-model predictions were estimated as

$$\widehat{var}(\widehat{\theta}) = \left[\sum_{i=1}^{R} w_i \sqrt{\widehat{var}(\widehat{\theta}_i | g_i) + (\widehat{\theta}_i - \widehat{\theta})^2}\right]^2,$$

where $\hat{var}(\hat{\theta}_i | g_i)$ is the variance of the ensemble prediction.

The parametric models were selected for their ability to fit a wide range of functional forms, such as linear, exponentially increasing or decreasing, asymptotic, and others. Autoregressive terms (of 1st and 2nd order) were tested to ensure that temporal autocorrelation was appropriately accounted for; the autocovariance of each model was plotted to examine residual autocorrelation and autoregressive terms were included if it remained. Models with a delta AIC of less than 2 were included in the model-averaging, with their weight being proportional to their Δ AIC (i.e. models which fit the data less well relative to the 'best' fitting model were down-weighted). By averaging over multiple modes, our approach includes both within-model and between-model uncertainty. If less than 10 data points were available, only two parameter parametric models (not three or four parameter models) were fit. All statistical analyses were carried out in the statistical software R.

It is important to recognize that statistical extrapolations make the assumption of the underlying processes remaining constant into the future, which may or may not be valid. They should therefore be viewed with this assumption clearly in mind. However, we applied relatively conservative and data-driven statistical methods to best represent and forecast the appropriate trends. There is especially high uncertainty for time-series with few data points; we nevertheless felt it was important to include these given the data challenges involved in developing indicators.

We assessed how well aligned each indicator was based on its relevance to a particular Target component. Target components were identified as specific individual textual aims within each Aichi Target. The level of alignment for each indicator with a Target component varied (i.e. some were better proxies than others); we assessed qualitatively whether we considered them to be of 'low', 'medium', or 'high' alignment.

S3.1.2 Indicator factsheets

Details of each indicator that were extrapolated are given in the factsheets below.

Aichi Target 1

Biodiversity barometer

The *Biodiversity barometer* indicator is a tool to gauge global consumer awareness and understanding of biodiversity. The indicator data is derived from national level public surveys implemented by the Union for Ethical BioTrade (UEBT) to measure the level of public awareness of biodiversity. In 2017, the biodiversity barometer survey was conducted with 5,000 consumers in six countries - Brazil, France, Germany, UK and USA. However, only the data from France, Germany, UK and USA form the global indicator, as these have been consistently measured since the inception of the *Biodiversity barometer*. Utilising a temporally shorter data set that includes Brazil does not change the direction or significance of the results.

Model fit

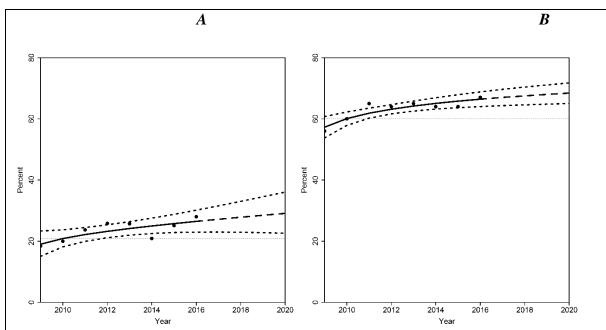


Figure. Modelled trend in the *Biodiversity barometer* from 2009-2016 and statistical extrapolations from 2017 to 2020. A) The percentage of respondents giving a correct definition of biodiversity. B) The percentage of respondents that had heard of the term biodiversity. Both A) and B) show a significant increase between 2010 and 2020. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The *Biodiversity barometer* shows that the level of public awareness of biodiversity in the four headline countries (Germany, France, UK and USA) has risen since 2010 and is projected to continue to rise until 2020, albeit at a slower rate with a levelling off of the trend. Both projections show a significant increase between 2010 and 2020. However, the ability of respondents to provide a correct definition of biodiversity remains low, with fewer than one third of the survey respondents able to define biodiversity correctly in 2015 and 2016; this is projected to increase slightly by 2020 (A). More encouragingly, approximately two thirds of the survey respondents had heard of biodiversity in 2016 and this is projected to increase slightly by 2020 (B).

Strengths

- The indicator is updated annually.
- Results from this indicator are easy to communicate and are directly related to Aichi Target 1.
- The indicator data can be disaggregated to the national and sub-national level, and by gender.

Caveats

- The indicator data at the global level is aggregated from the national level data of just four countries (Germany, France, UK and USA). However steps are being taken to include a more representative set of countries in the future.
- The wording of questions may preclude some survey respondents from showing positive awareness of biodiversity due to lack of understanding of terminology.
- There are <10 data points, making projection uncertain.
- Achieving Aichi Target 1 requires making a distinction between being aware of both the positive values of biodiversity, and how life on Earth may be affected by biodiversity loss. A positive response to the awareness of the term 'biodiversity' does not necessarily translate into awareness of the steps that can be taken in order to conserve biodiversity.

Sampling methodology and data selection

UEBT commissions Ipsos to conduct interviews in the target countries. The survey respondents are chosen from nationally representative samples of people between 16 and 64 years old. The survey results from France, Germany, the UK and the USA form the global indicator. Each year, 1000 consumers are interviewed online in each country and national representative quotas are then used with a weighting to ensure sample representativeness. The survey includes questions regarding: the awareness and understanding of biodiversity; purchasing attitudes regarding the ethical sourcing of biodiversity; the understanding of biodiversity related terms and sources of biodiversity awareness.

References

Union for Ethical BioTrade (UEBT), *Biodiversity Barometer* (Amsterdam, 2017). http://www.biodiversitybarometer.org/

Funding towards environmental education (\$)

Funding towards environmental education (\$) measures international financial flows committed to projects that support environmental education and training. This metric measures the funds committed from a range of multilateral agencies and bilateral donors outside the OECD Development Assistance Committee (DAC), including the World Bank Group, the Global Environment Facility, African Development Bank, Asian Development Bank, Andean Development Corporation, Arab Bank for Economic Development in Africa, Caribbean Development Bank, OPEC Fund for International Development, European Bank for Reconstruction and Development, and various bilateral agencies. **Model fit**

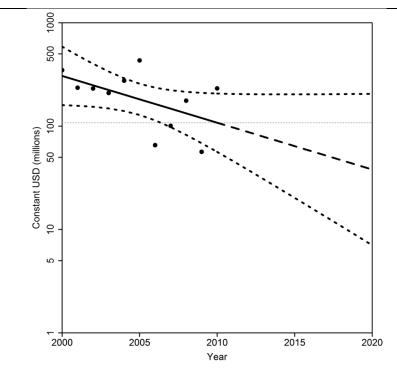


Figure. Modelled trend in *Funding towards environmental education (\$)* 2000-2010 and statistical extrapolation from 2011 to 2020. The trend suggests a declining but non-significant trend between 2010 and 2020. Note that the y-axis is log-scaled. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Funding towards environmental education (\$) has shown a general decline in the last decade and this is extrapolated to continue to 2020, though the difference between 2010 and 2020 is not significant and the confidence in the projection is relatively low.

Strengths

- The metric is based upon a detailed activity categorisation scheme that captures information not previously available. AidData activity codes allow users to identify projects not only according to their dominant purpose, but also by their specific components (i.e. activities). Thus, the granularity of the data allow for more fine-grained analysis of how international development financing is allocated.
- The data included in this analysis covers most large multilateral organizations and represents 45% of all known project-level flows between the years covered.

Caveats

- The project descriptions are sometimes brief and unclear as to the quantity of funds specifically earmarked for indicator activities. As such, this analysis includes the full project commitment amount for a project that had at least one activity relating to the indicator. This almost certainly leads to an over-estimation of the funds that are specifically directed to investment in environmental education.
- Activity codes that identify projects with investment in environmental education are only currently available for certain donors, largely consisting of multilateral agencies and bilateral donors outside of the OECD-DAC.
- This indicator, along with the other AidData financial indicators, do not include internal national spending.

Sampling methodology and data selection

Data were compiled by AidData, an organisation that collects data on international development financing and categorises each project or flow into specific activities and sectors. Data are presented in constant US dollars (set at 2009 levels). Trends were based upon funds committed from 2000-2010 only to account for completeness and reliability concerns with earlier data (Development Co-operation Directorate (DCD) 2008). Additionally, for the purposes of this analysis, we only included donors for whom more than 95% of their projects/activities have received AidData activity codes.

References

Development Co-operation Directorate (DCD), DCD/DAC/STAT (2008) 17/REV1 (2008); www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=DCD/DAC/STAT(200 8)17/REV1&docLanguage=En

Online interest in biodiversity (proportion of google searches)

This indicator shows temporal trends in global awareness of biodiversity through an analysis of searches made on Google. Google Trends compiles data on the frequency of specific search terms inputted into the Google search engine. The data shows the frequency of web searches for the subject of biodiversity, including searches in a variety of languages and topics, normalised against the total number of internet searches over a specific time period.

Model fit

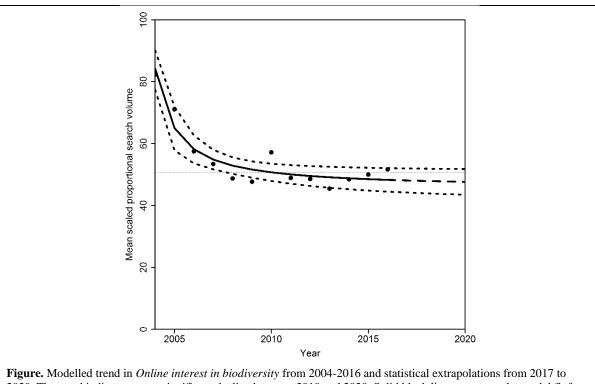


Figure. Modelled trend in *Online interest in biodiversity* from 2004-2016 and statistical extrapolations from 2017 to 2020. The trend indicates a non-significant decline between 2010 and 2020. Solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

This indicator shows that online interest in biodiversity, as measured through the number of searches for biodiversity-related subjects on the Google search engine, has in general decreased since 2004 and is projected to continue to decrease to 2020, albeit at a slower rate with a levelling off of the trend. Although some have challenged these types of Google search data analysis (Ficetola, 2013), they are used in numerous fields and have been shown to be a clear proxy for underlying trends (Mccallum and Bury, 2013).

Strengths

• This data provides a truly global snapshot of interest in biodiversity. Over 3.5 billion searches are undertaken across the world using the Google search engine on a daily basis (Internet Live Stats, 2017).

Caveats

• The search terms and number of languages used in Google Trends is not transparent and it is not possible to analyse a variety of trends and combine due to the proportional nature of the data.

Sampling methodology and data selection

Data were compiled for weekly intervals. Data were then normalised against total internet searches for that week, and then presented as a proportion of the peak in internet searches for the term since data collection began. Extrapolations were calculated using mean values per year.

References

Ficetola, G.F., 2013. Is interest toward the environment really declining? The complexity of analysing trends using internet search data. Biodiversity and conservation, 22(12), pp.2983-2988.

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Internet Live Stats, 2017. Google Search Statistics. [online] Available at: <u>http://www.internetlivestats.com/google-search-statistics/</u>

Aichi Target 2

Funding towards Environmental Impact Assessment (\$)

Funding towards Environmental Impact Assessment (\$) measures international financial flows committed to projects that support Environmental Impact Assessments (EIAs). This metric measures the funds committed from a range of multilateral agencies and bilateral donors outside the OECD Development Assistance Committee (DAC), including the World Bank Group, the Global Environment Facility, African Development Bank, Asian Development Bank, Andean Development Corporation, Arab Bank for Economic Development in Africa, Caribbean Development Bank, OPEC Fund for International Development, European Bank for Reconstruction and Development, and various bilateral agencies.



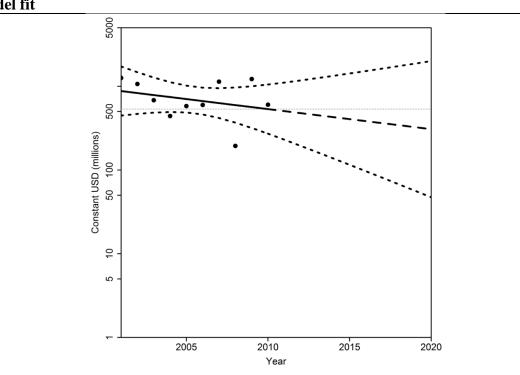


Figure. Modelled trend in *Funding towards Environmental Impact Assessment (\$)* from 1995-2010 and statistical extrapolation from 2011-2020. The trend suggests a non-significant decrease between 2010 and 2020. Note the log scale on the y axis. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Funding towards Environmental Impact Assessment (\$) has shown a general decline in the last decade and this is extrapolated to continue to 2020, though the difference between 2010 and 2020 is not significant and the confidence in the projection is relatively low. **Strengths**

- The metric is based upon a detailed activity categorisation scheme that captures information not previously available. AidData activity codes allow users to identify projects not only according to their dominant purpose, but also by their specific components (i.e. activities). Thus, the granularity of the data allow for more fine-grained analysis of how international development financing is allocated.
- The data included in this analysis covers most large multilateral organizations and represents 45% of all known project-level flows between the years covered.

Caveats

- The project descriptions provided are sometimes brief and unclear as to the quantity of funds specifically earmarked for EIA activities. As such, this analysis includes the full project commitment amount for a project that had at least one activity relating to the indicator. This almost certainly leads to an over-estimation of the funds that are specifically directed to investment in EIAs.
- Activity codes that identify projects with investment in EIAs are only currently available for certain donors, largely consisting of multilateral agencies and bilateral donors outside of the OECD-DAC.
- This indicator, along with the other AidData financial indicators, do not include internal national spending.

Sampling methodology and data selection

Data were compiled by AidData, an organisation that collects data on international development financing and categorises each project or flow into specific activities and sectors. Data are presented in constant US dollars (set at 2009 levels). Trends were based upon funds committed from 2000-2010 only to account for completeness and reliability concerns with earlier data (Development Co-operation Directorate (DCD) 2008). Additionally, for the purposes of this analysis, we only included donors for whom more

than 95% of their projects/activities have received AidData activity codes.

References

Development Co-operation Directorate (DCD), DCD/DAC/STAT (2008) 17/REV1 (2008); www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=DCD/DAC/STAT(2008)17/REV1&docLanguage=En

Number of research studies involving economic valuation

This indicator represents the efforts of the scientific community to measure the economic value of biodiversity. The uptake of such valuations into local and national policy, the focus of Aichi Target 2, is reliant upon this initial assessment and production of assessment strategies by the scientific community. The indicator uses data from the Ecosystem Service Valuation Database (ESVD); a database of monetary values of ecosystem services compiled from primary sources and run by the Ecosystem Services Partnership (Van der Ploeg and de Groot, 2010).

Model fit

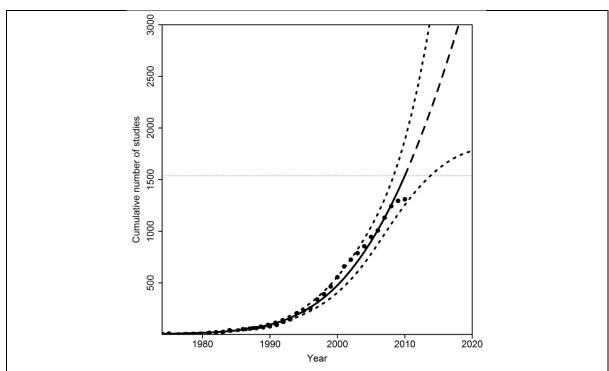


Figure. Modelled trend in *Number of research studies involving economic valuation* from 1974-2010 and statistical extrapolation from 2011-2020. The trend suggests a significant increase in the underlying trend between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The *Number of research studies involving economic valuation* is projected to show a significant increase by 2020, with the overall trajectory accelerating. However, there is considerable uncertainty in the projection, with broad confidence limits for the extrapolation.

Strengths

• The ESVD contains approximately 1300 studies assessing aspects of biodiversity in 71 countries across the globe, and therefore provides one of the most comprehensive databases of its kind (De Groot *et al.* 2012).

Caveats

- The indicator measures interest in the scientific community but does not directly measure uptake of assessments into policy.
- The indicator is based upon a database which was not initially designed to be temporally representative so the trend line may be biased towards more recent studies.

Sampling methodology and data selection

The ESVD is based upon a database compiled for a project undertaken through The Economics of Ecosystems and Biodiversity (TEEB) (Van der Ploeg and de Groot 2010). The primary literature for the TEEB database were gathered from other databases and literature searches, and from recommendations by experts. The indicator looks at the number of studies per year found within the ESVD. The trend is reflected in other datasets such as the EVRI (Environmental Valuation Research Inventory (Christie *et al.* 2012). **References**

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Aichi Target 3

Funding towards institutional capacity building in fisheries (\$)

Funding towards institutional capacity building in fisheries (\$) measures international financial flows committed to projects that support institutional capacity building in fisheries. This metric measures the funds committed from a range of multilateral agencies and bilateral donors outside the OECD Development Assistance Committee (DAC), including the World Bank Group, the Global Environment Facility, African Development Bank, Asian Development Bank, Andean Development Corporation, Arab Bank for Economic Development in Africa, Caribbean Development Bank, OPEC Fund for International Development, European Bank for Reconstruction and Development, and various bilateral agencies.

Model fit

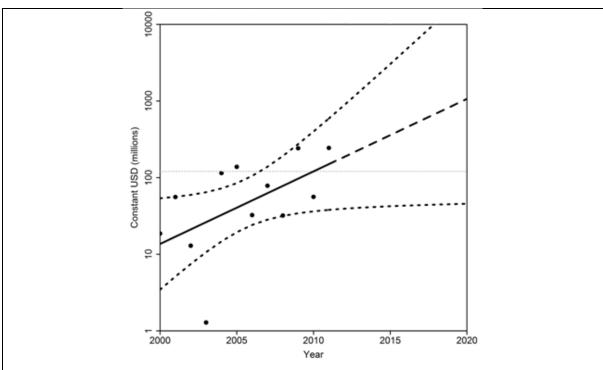


Figure. Modelled trend in *Funding towards institutional capacity building in fisheries (\$)* 2000-2010 and statistical extrapolation from 2011-2020. The trend suggests a non-significant increase between 2010 and 2020. Note the log-scale on the y-axis. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Funding towards institutional capacity building in fisheries (\$) has shown a nonsignificant increase in the last decade and this is extrapolated to continue to 2020, though the difference between 2010 and 2020 is not significant and the confidence in the projections is extremely low.

Strengths

- The metric is based upon a detailed activity categorization scheme that captures information not previously available. AidData activity codes allow users to identify projects not only according to their dominant purpose, but also by their specific components (i.e. activities). Thus, the granularity of the data allow for more fine-grained analysis of how international development financing is allocated.
- The data included in this analysis covers most large multilateral organizations and represents 45% of all known project-level flows between the years covered.

Caveats

- The project descriptions provided are sometimes brief and unclear as to the quantity of funds specifically earmarked for fishery capacity building activities. As such, this analysis includes the full project commitment amount for a project that had at least one activity relating to the indicator. This almost certainly leads to an over-estimation of the funds that are specifically directed to investment in institutional capacity building for fisheries.
- Activity codes that identify projects with investment in capacity building in fisheries are only currently available for certain donors, largely consisting of multilateral agencies and bilateral donors outside of the OECD-DAC.
- This indicator, along with the other AidData financial indicators, do not include internal national spending.

Sampling methodology and data selection

Data were compiled by AidData, an organisation that collects data on international development financing and categorises each project or flow into specific activities and sectors. Data are presented in constant US dollars (set at 2009 levels). Trends were based upon funds committed from 2000-2010 only, to account for completeness and reliability concerns with earlier data (Development Co-operation Directorate (DCD), 2008). Additionally, for the purposes of this analysis, we only included donors for whom more than 95% of their projects/activities have received AidData activity codes.

References

Development Co-operation Directorate (DCD), DCD/DAC/STAT (2008) 17/REV1 2008. www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=DCD/DAC/STAT(200 8)17/REV1&docLanguage=En.

World Trade Organisation 'green box' agricultural subsidies (\$)

Agricultural production is heavily subsidised, in particular in developed countries. In order to reform trade, and to make policies more market-oriented, the World Trade Organisation (WTO) Agreement on Agriculture was established in 1995. The agreement was also intended to improve predictability and security for importing and exporting countries alike. The agreement rests on three pillars: market access, export subsidies and domestic support, and has been classified into different "boxes". Subsidies falling into the amber box (i.e. those that are distorting production and trade) were to be reduced in the period 2000 – 2005 (2010 for all developing countries), while those in the blue box (subsidies designed to limit production but still distort trade) and green box (subsidies not distorting trade and not targeted at specific products, providing direct income to farmers, environmental protection and regional development programmes) could remain. This indicator focuses on the last of these - the green box subsidies - the permitted subsidies which are expected to be the least harmful or beneficial to biodiversity while allowing the financial development of development of development of Meléndez-Ortiz, 2011).

Model fit

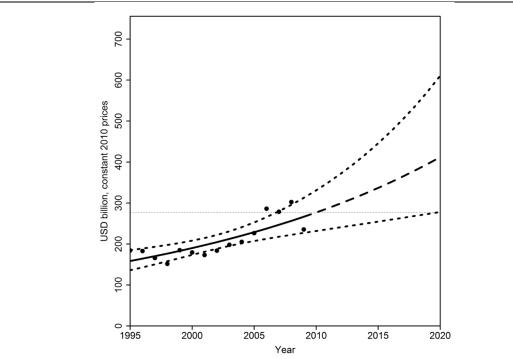


Figure. Modelled trend of *World Trade Organisation 'green box' agricultural subsidies (\$)* 1995-2009 and statistical extrapolation from 2010-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The trend of increased *World Trade Organisation 'green box' agricultural subsidies* (\$) observed over the last decade is projected to continue, with spending in 2020 projected to be approximately double the spending observed in 2000, and about 1.4 times that projected for 2010. The difference between 2010 and 2020 is significant.

Strengths

- The *World Trade Organisation 'green box' agricultural subsidies (\$)* data is gathered from countries across the world and is perhaps the most comprehensive record of spending available.
- "Green box" subsidies encompass environmental protection measures, and, based on a 2013 proposal by the G-33, also land rehabilitation, soil conservation and resource management, as well as drought management and flood control (Meléndez-Ortiz, Bellmann, and Hepburn, eds., 2009).

Caveats

- The consistency of data may be questionable as not all countries report their data in a consistent and regular fashion.
- Green box spending should be the least harmful of subsidies to biodiversity; however, environmental protection and related measures are only one of the support measures included in this category.

Sampling methodology and data selection

Data is compiled and released by the WTO. The total spending for all countries that reported per year was calculated and then converted to constant USD set at 2010 prices. To adjust for variability in the number of countries reporting, a correlation plot against total spending was examined, and outlying years removed until no correlation remained. This process resulted in two years, 2010 and 2011, being removed from the dataset. Note that the amber and blue box data had strong correlations between the numbers of countries reporting and the total spending, such that when these years were removed insufficient data remained to appropriately extrapolate.

References

Goodwin, B.K. and Meléndez-Ortiz, J., 2011. Agricultural Subsidies in the WTO Green Box. World Trade Review, 10(2), p.293.

Meléndez-Ortiz, R., Bellmann, C. and Hepburn, J. eds., 2009. Agricultural subsidies in the WTO green box: ensuring coherence with sustainable development goals. Cambridge University Press.

Aichi Target 4

Percentage of countries that are Category 1 CITES Parties

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) is an international agreement between governments that aims to ensure that international trade in specimens of wild animals and plants does not threaten their survival. Among the conservation agreements with the largest international membership, the 180 Parties to CITES are required to take appropriate measures to enforce the provisions of the Convention and to prohibit trade in specimens in violation of those provisions (Article VIII of the Convention) through the implementation of appropriate policies, legislation and procedures.

The CITES National Legislation Project was established in 1992 to provide legislative analyses and assist Parties to meet the legislative requirements of CITES. Acknowledging that substantial progress has been achieved since its inception, approximately half of the Parties have not yet taken appropriate measures to enforce such provisions of the Convention. In light of this, the indicator has been developed to monitor progress made by the international community towards the development of full legislation for effective implementation of CITES to ensure that international trade in CITES-listed species is sustainable, traceable and legal.

Model fit

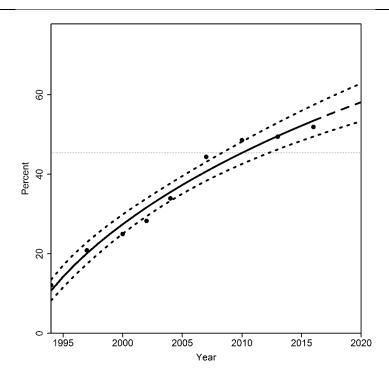


Figure. Modelled trend in the *Percentage of countries that are Category 1 CITES Parties* 1994-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent

data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The projected increase in the *Percentage of countries that are Category 1 CITES Parties* shows an improving commitment from the international community to ensuring that international trade in specimens of wild animals and plants does not threaten their survival. By 2020, it is projected that over 50% of the Parties of CITES will have introduced legislation that will meet the requirements for implementation of CITES - a significant improvement over the 2010 value.

Strengths

• Measures the steps taken by nations towards the prevention of unsustainable consumption of 35,800 CITES-listed species.

Caveats

- The indicator is relevant only for legal international trade in CITES-listed species: not for illegal trade, domestic trade, non- CITES-listed species, or consumption/use of species not resulting in international trade.
- The indicator is very insensitive, measuring only the number of Parties with national legislation consistent with CITES commitments and not the degree of application and enforcement of this legislation, nor the effectiveness of actions taken to reduce unsustainable exploitation.

Sampling methodology and data selection

The Parties are classified under three categories, according to their progress in developing effective legislation for implementing the provisions of the Convention. The indicator is then a measure of the proportion of Category 1 listed Parties relative to those in Categories 2 and 3. The categories are defined as follows:

- Category 1: Legislation that is believed generally to meet the requirements for implementation of CITES.
- Category 2: Legislation that is believed generally to meet one to three of the four requirements for effective implementation of CITES.
- Category 3: Legislation that is believed generally not to meet the requirements for implementation of CITES.

In addition, Parties may be classified as 'under review', during which their legislation is being reviewed as result of new information provided by the member concerned; or as 'pending', normally including new Parties or Parties that have not responded to the Secretariat, for which their legislative analyses are under preparation.

References

https://cites.org/legislation

Ecological Footprint (number of earths needed to support human society)

Direct anthropogenic threats to biodiversity include habitat loss or damage, resource overexploitation, pollution, invasive species and climate change. These direct threats are the result of more distant, indirect drivers of biodiversity loss arising from consumption of resources and the generation of waste. The ultimate drivers of threats to biodiversity are human demands for food, fibre and timber, water, energy and land on which to build infrastructure. As the human population and global economy grow, so do the pressures on biodiversity.

The *Ecological Footprint* measures the demands that our use of ecological assets places on the regenerative capacity of productive ecosystems, measured through a sister indicator called biocapacity (Galli *et al.* 2014). The main aim of the *Ecological Footprint* methodology is thus to promote recognition of ecological limits. This recognition should help safeguard the ecosystems' viability (such as healthy forests, clean air, fertile soils and biodiversity) and life-supporting services.

Model fit

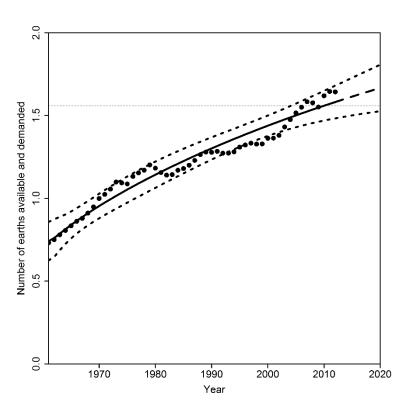


Figure. Modelled trend in the *Ecological Footprint* 1961-2012 and statistical extrapolation from 2013-2020. A value greater than 1.0 represents an utilisation of more resources than the earth can provide; for pressure reduction purposes, reducing the footprint to within the 1.0 threshold (*i.e.* the world's biocapacity) would be ideal. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

An increase in the *Ecological Footprint* represents an increase in humanity's demand on the biosphere's regenerative capacity, which in turn equates to increased pressure on ecosystems and biodiversity and a greater risk of biodiversity loss (Galli *et al.* 2014). If the *Ecological Footprint* exceeds *biocapacity*, then a minimum condition for sustainable consumption is not being met. This means ecosystem stocks are being depleted, and/or emissions are accumulating in the atmosphere and oceans. When this is the case, competition for biological resources and quantitative or qualitative reductions in area for biodiversity will result in biodiversity loss. A reduction in the *Ecological Footprint*, and especially the elimination of overshoot, would indicate reduced pressure on the world's biological resources and a lower risk of biodiversity loss. Unfortunately, the trend line suggests a continued deterioration in the situation such that by 2020 the *Ecological Footprint* will be significantly higher than in 2010.

Strengths

- The indicator captures indirect pressure on biodiversity due to human production, trade and consumption activities. Consumption in one country may have little effect on local ecosystems, but pressure ecosystems from where the product stems (Galli *et al.* 2014).
- The *Ecological Footprint* methodology is continuously being improved and every time a new edition of the results is released (calculated with the most recent methodology), *Ecological Footprint* and *biocapacity* values are back-calculated from the most recent year in order to ensure consistency across the historical time series (Borucke *et al.* 2013).

Caveats

- Data remains limited and assumptions, although documented (Borucke *et al.* 2013), need to be considered.
- For countries with populations fewer than one million, data sets are sometimes incomplete and *Ecological Footprint* results for these nations are therefore not published.

Sampling methodology and data selection

The *Ecological Footprint* tracks human demand on nature in terms of biologically productive areas that a population uses for producing the renewable resources it consumes and absorbing its waste^{*}. This demand is compared to the *biocapacity*, which represents nature's capacity (at global and/or national level) to renew resources and dispose waste (i.e., regenerative capacity). When the *Ecological Footprint* exceeds *biocapacity*, stocks are being depleted, and/or emissions are accumulating in the biosphere (such as CO₂ in the atmosphere and oceans). Thus a minimum condition for sustainable consumption is not being met and the use of natural resources is not within safe ecological limits. *Ecological Footprint* and *biocapacity* calculations are primarily based on data from UN agencies or affiliated organizations such as the Food and Agriculture Organization of the United Nations (FAOSTAT), the UN Statistics Division (UN Commodity Trade Statistics Database), the International Energy Agency (IEA) and other studies in peer reviewed journals as described in (Borucke *et al.* 2013). The Global Footprint Network releases updated National Footprint Accounts each year. Results are published on its websites and in numerous publications including WWF-International's biennial Living Planet Report.

*Due to data limitation, CO2 emissions are the sole waste flow currently tracked by the Ecological Footprint methodology.

References

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Human appropriation of net primary productivity (Pg C)

Human appropriation of net primary productivity (Pg C) is an aggregated indicator that reflects both the amount of area used by humans and the intensity of land use. This may be used to indicate progress against Aichi Target 4 by revealing the measure of impact that human consumption has on natural resources. Net Primary Production (NPP) is the net amount of biomass produced each year by plants and may therefore be used to provide an indication of trophic energy flows in ecosystems. *Human appropriation of net primary productivity* (Pg C) measures to what extent land conversion and biomass harvest alter the availability of NPP (biomass) in ecosystems. It is a prominent measure of the "scale" of human activities compared to natural processes (*i.e.* of the "physical size of the economy relative to the containing ecosystem"). As human harvest of biomass is a major component of this indicator, it is also closely related to socio-economic metabolism as measured by material flow accounts. This indicator relates to land-use change, one of the most important drivers of terrestrial biodiversity loss, although the direct relationship between *Human appropriation of net primary productivity* (Pg C) and biodiversity remains unclear. **Model fit**

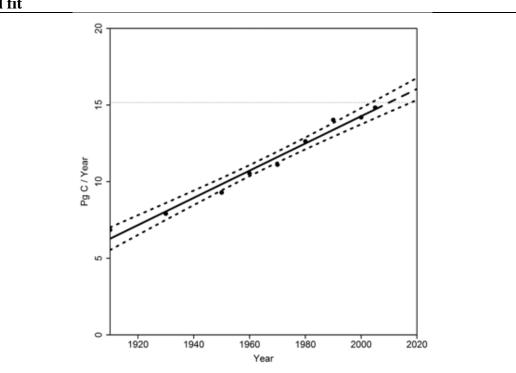


Figure. Modelled trend in the *Human appropriation of net primary productivity (Pg C)* 1910-2005 and statistical extrapolation from 2006-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The increase in the *Human appropriation of net primary productivity (Pg C)* observed since the turn of the century is projected to continue towards 2020, with a significant increase expected in the 2010-2020 period. This will lead to increased pressure on biodiversity and resources. However, note that the data points are temporally infrequent, so there may be a lag in terms of detecting a change due to the low sampling frequency.

Strengths

• *Human appropriation of net primary productivity (Pg C)* is an indicator that can be assessed in a spatially explicit manner, i.e. it is possible to produce maps of *Human appropriation of net primary productivity (Pg C)* that localize the human impact on ecosystems.

Caveats

- Low temporal frequency of data points, and time since last data point (2005)
- A lack of definitive standardization has unfortunately resulted in a range of empirical results (discussed below). This has not only hampered the comparability of results but has also fuelled critiques.
- Although some studies have explored the relationship between biodiversity loss and *Human appropriation of net primary productivity (Pg C)* it still needs to be better understood (Haines-Young, 2009). This indicator does not account for the qualities of the primary productivity appropriated (Smil, 2011). For example, harvesting food crops on land that has been cultivated for centuries is clearly a different appropriation from cutting down a forest stand in a biodiversity hotspot.

Sampling methodology and data selection

Human appropriation of net primary productivity (Pg C) tries to capture the aggregate impact of land use on biomass available in each year in ecosystems. Different definitions of *Human appropriation of net primary productivity (Pg C)* may lead to different empirical results (see Haberl *et al.* 2007).

Human appropriation of net primary productivity (Pg C) is measured as follows:

Human appropriation of net primary productivity (Pg C) = NPP0 – NPPt where NPPt = NPPact - NPPh

NPPO is the potential NPP or the NPP that would be produced by the vegetation in the absence of human interference; NPPt is the NPP that remains in the ecosystems after harvest. In NPPt computation, NPPact is the NPP of the actual vegetation, and NPPh the NPP harvested by humans. Normally, HANPP is expressed as a percentage of potential NPP:

Human appropriation of net primary productivity (Pg C) (%) =

 $\frac{Human appropriation of net primary productivity (Pg C)}{NPP0} \times 100$

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Red List Index (impacts of utilisation)

People depend upon biodiversity and use wildlife in a variety of ways. For example, birds, mammals and amphibians are hunted, trapped and collected for food, sport, pets, medicine, materials (e.g. fur and feathers) and other purposes.

The *Red List Index (impacts of utilisation)* illustrates the changing status of three species groups (birds, mammals and amphibians) owing to the balance between negative trends driven by unsustainable exploitation, and positive trends driven by measures to reduce overexploitation. It excludes changes in status driven by other factors (such as habitat loss or climate change).

Model fit

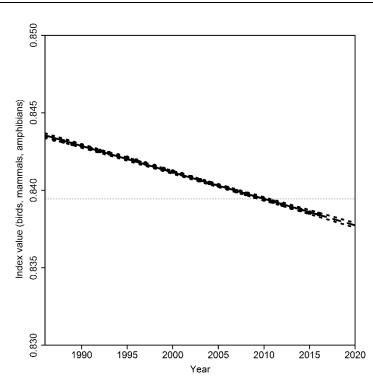


Figure. Modelled trend in the *Red List Index (impacts of utilisation)* 1986-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant decrease between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A Red List Index value of 1.0 equates to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. A Red List Index value of zero indicates that all species have gone extinct. A downwards trend in the graph line (i.e. decreasing Red List Index values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing.

The indicator shows a linear declining trend and is projected to continue to drop significantly over the next three years to 2020. This indicates that levels of utilisation continue to negatively impact on these species and results in a greater risk of extinction for them. However, note that the absolute magnitude of the decline is relatively small, indicating that other pressures are more significant in driving declines in the status of mammals, birds and amphibians.

Strengths

- The Red List Index is based on data on the utilization and extinction risk of a very large proportion of mammals, birds and amphibians worldwide.
- The only global indicator available that is able to disentangle biodiversity trends driven by utilisation from other factors.

Caveats

- The Red List Index is only moderately sensitive, owing to the breadth of Red List categories (Butchart *et al.* 2004, Butchart *et al.* 2005).
- There are very few data points, so there is limited information on which to extrapolate the trend.
- Trends for other taxonomic groups (e.g. utilised plants) are not yet available.
- National versions of this indicator are not yet available: many countries have compiled national red lists (generally for all vertebrate species), but so far few have done this twice or more using consistent methods.

Sampling methodology and data selection

This indicator measures trends in the extinction risk of mammal, bird and amphibian species, and draws on extinction risk assessments and data on utilisation in IUCN and BirdLife International's Species Information Service, which underpins the IUCN Red List.

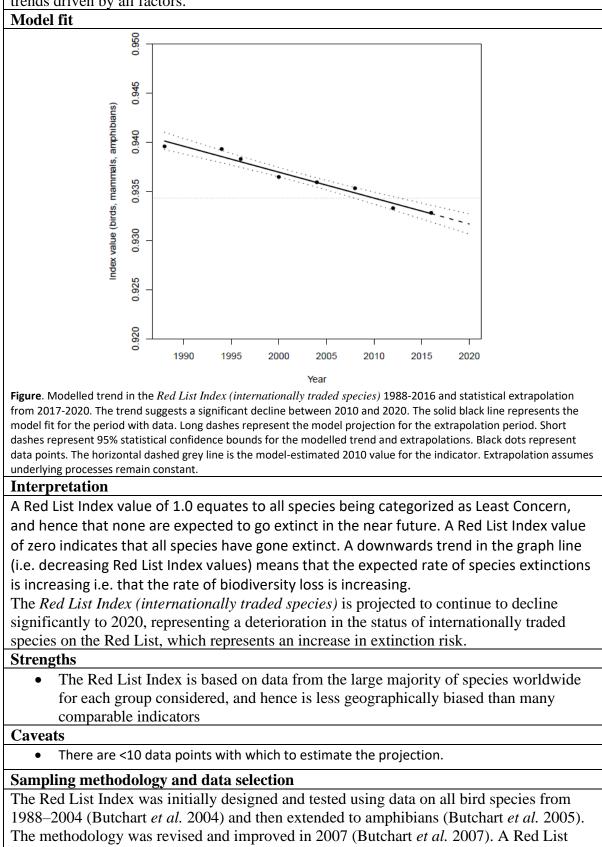
The Red List Index was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A Red List Index for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010). Red List Index trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide.

References

http://www.iucnredlist.org/about/publication/red-list-index

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The Red List Index (internationally traded species) is a disaggregation of RLI data for birds in international trade. It complements two other disaggregated Red List Indices: RLI (trends driven by utilisation) and RLI (species used for food and medicine), but shows trends driven by all factors.



List Index trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide.

References

http://www.iucnredlist.org/about/publication/red-list-index

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Human appropriation of fresh water (water footprint) (thousand km3)

The idea of considering water use along supply chains has gained interest after the introduction of the 'water footprint' concept by Hoekstra in 2002 (Hoekstra and Mekonnen, 2012). The *Water* footprint is an indicator of freshwater use that looks at both direct and indirect use. Reflecting the aim of Aichi Target 4, the concept of the *Water footprint* is rooted in the recognition that human impacts on freshwater systems can ultimately be linked to human consumption, and that issues such as water shortages and pollution can be better understood and addressed by considering production and supply chains as a whole. Many countries have significantly externalised their water footprint, importing water-intensive goods from elsewhere. This puts pressure on the water resources in the exporting regions, where too often mechanisms for water governance and conservation are lacking. Not only governments acknowledge their role in achieving a better management of water resources, but businesses and public-service organisations increasingly recognize their role in the interplay of actors involved in water use and management.

The water footprint of a product is the volume of freshwater used to produce the product, measured over the full supply chain. It is a multidimensional indicator, showing water consumption volumes by source and polluted volumes by type of pollution; all components of a total water footprint are specified geographically and temporally, as a volumetric measure of water consumption and pollution.

Model fit

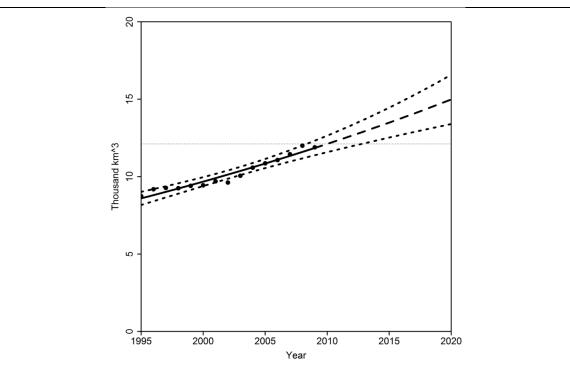


Figure. Modelled trend in the global *Water footprint* 1995-2009 and statistical extrapolation from 2010-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The global consumption and pollution of water is expected to continue to increase significantly to 2020. This will result in increased pressure on human populations, and increased pressure on animal and plant species reliant upon these water sources.

Strengths

- There are various water footprint studies that have been carried out thus far, from global to national.
- The indicator includes both direct and indirect water use.

Caveats

- Water footprint assessment addresses the issues of freshwater scarcity and pollution. It does not address the issue of flooding. It also does not address the issue of people lacking access to proper clean water supply. Further, the *Water footprint* does not include the use and pollution of seawater.
- The *Water footprint* methodology is still maturing (Chapagain and Tickner, 2012.).

Sampling methodology and data selection

The *Water footprint* has three components: green water footprint; blue water footprint; and grey water footprint, and includes water consumption and pollution throughout the full life cycle: direct, indirect (supply chain) and end-user. Together they provide a comprehensive picture of water use by delineating the source of water consumed – either rainfall/soil moisture or surface/groundwater – and the volume of run-off required for assimilation of pollutants.

The green water footprint is the amount of rainfall or soil moisture consumed and is particularly relevant for agricultural, horticultural and forestry products. The green water footprint of a process is calculated with the following formula:

Green Water Footprint = Green Water Evaporation + G

The blue water footprint is the amount of surface or groundwater which is evaporated, incorporated into a product or otherwise not returned to the same catchment as where abstracted, in the same period as when abstracted. The blue water footprint of a process is calculated as:

Blue Water Footprint = Blue Water Evaporation + Blue Water Incorporation + Lost Return Flow

The grey water footprint is the volume of freshwater that is required to assimilate the load of pollutants discharged based on natural background concentrations and existing ambient water quality standards. It is calculated as:

Grey Water Footprint = Pollutant Load / (Maximum Acceptable Concentration – Natural Concentration)

Water Footprint Assessment (WFA) is a structured process for quantifying and mapping the green, blue and grey water footprint, assessing the sustainability of the water footprint and identifying strategic actions to reduce the water footprint and improve its sustainability.

Water footprints can be assessed at different levels of spatiotemporal detail. At the lowest level of detail, the *Water footprint* is assessed based on multi-year global average water footprint data.

References

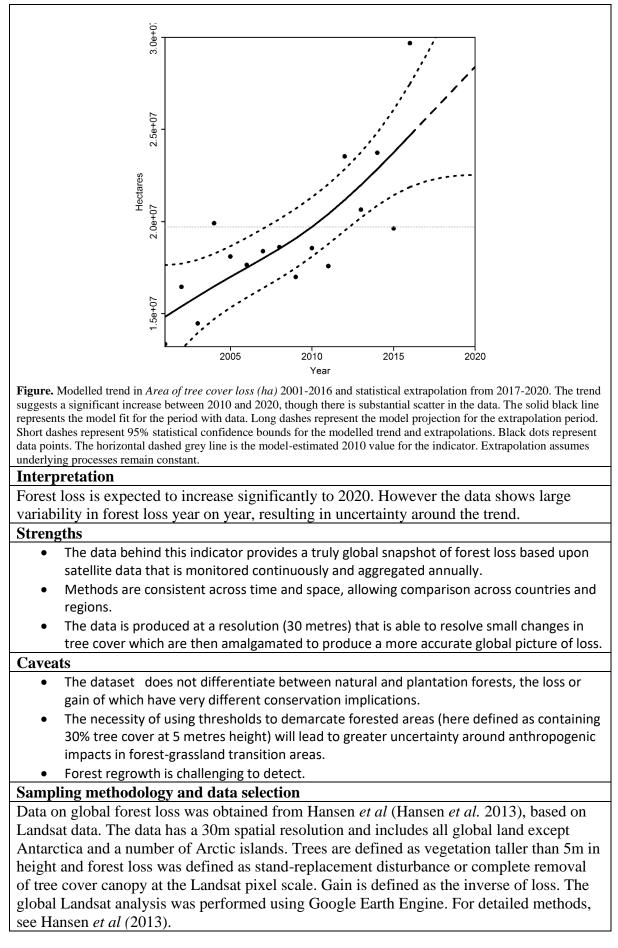
Chapagain, A.K. and Tickner, D., 2012. Water footprint: Help or hindrance? Water Alternatives, 5(3), p.563.

Hoekstra, A.Y. and Mekonnen, M.M., 2012. The water footprint of humanity. Proceedings of the national academy of sciences, 109(9), pp.3232-3237.

Aichi Target 5

Area of tree cover loss (ha)

Forests play a crucial role for maintaining life on earth, through the maintenance of ecological diversity, climate regulation, carbon storage, soil and water protection and provision of resources (fuel, construction materials and medicines) (Heino *et al.* 2015). Despite the importance of forest, deforestation rates remain high, due to agricultural expansion and human population growth (Heino *et al.* 2015). This indicator measures global forest loss, using data obtained from Hansen *et al* (Hansen *et al.* 2013). **Model fit**



References

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R. and Kommareddy, A., 2013. High-resolution global maps of 21st-century forest cover change. Science, 342(6160), pp.850-853. Data available online from <u>http://earthenginepartners.appspot.com/science-2013-global-forest</u>.

Heino, M., Kummu, M., Makkonen, M., Mulligan, M., Verburg, P.H., Jalava, M. and Räsänen, T.A., 2015. Forest loss in protected areas and intact forest landscapes: a global analysis. PloS one, 10(10), p.e0138918.

Percentage natural habitat extent

The conversion of natural habitats to agricultural and urban land is one of the most serious threats to biodiversity and with rising global demand for food through expanding global populations as well as an increase in per capita consumption, the loss of further natural habitat is likely to continue. Conversion of natural habitats to land for human use also puts pressure on intact habitats through fragmentation, eutrophication, alteration of water flows, and the introduction of alien species. This indicator measures the global extent of land which remains natural (i.e. the proportion of the land surface which is non-agricultural; though note that urban area is not accounted for in this indicator).



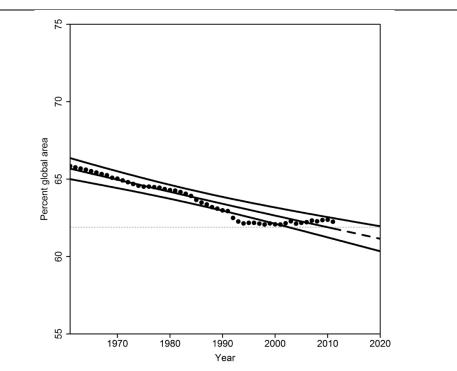


Figure. Modelled trend in *Percentage natural habitat extent* 1961-2011 and statistical extrapolation from 2012-2020. The trend suggests a non-significant decrease between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The extent of global natural habitat is expected to continue to decline, though nonsignificantly, to 2020. It is projected that between 1961 and 2020 there will have been a loss of approximately 6-7% of all natural habitats.

Strengths

• This indicator is compiled using very detailed statistics collected over a long time period.

Caveats

• The data is based upon the amount of natural habitat converted to agriculture only and will therefore underestimate the total loss of habitat due to other causes such as the construction of urban areas. Land which has been abandoned post-agricultural use will also be missed by this indicator.

Sampling methodology and data selection

Data on the global extent of agricultural habitats was collected by the Food and Agricultural Organisation of the United Nations (FAO). Total natural habitat extent was calculated as the proportion of land which has not been converted to agricultural use.

References

Food and Agricultural Organisation of the United Nations (FAO)

Wetland Extent Trends (WET) Index

Wetland ecosystems are of huge value both in terms of their biodiversity and the vital ecosystem services they provide, but studies to assess the status of wetlands suggest that these important habitats are declining in extent around the world. In order to track progress to Aichi Target 5, it is important that work is undertaken to estimate the global baseline rate of decline of wetland extent. The *Wetland Extent Trends (WET) Index* provides a method to estimate broad trends in habitat extent for habitats with incomplete and heterogeneous data. The Index estimates the average rate of change in wetland extent over the recent period of 1970 to 2015 using time-series data from the published scientific literature. The Index enables the rate of loss of wetlands to be estimated, providing an indication of the status of wetlands globally.



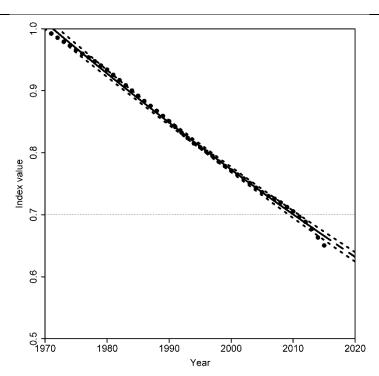


Figure. Modelled trend in the *Wetland Extent Trends (WET) Index* 1970-2015 and statistical extrapolation from 2016-2020. The trend suggests a significant decline between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal

dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

There is a decline in the *Wetland Extent Trends (WET) Index* of 35% between 1970 (which is given a value of 1.0) and 2015. The data also suggests that this rate of loss of wetland is accelerating, and that there will be a significant decline between 2010 and 2020. The Index natural marine/coastal and inland wetlands.

Strengths

- Data can be disaggregated from the global scale to six regions and into three types of wetland.
- Methodology accounts for bias and overrepresentation.

Caveats

- Wetland extent data is unevenly distributed both geographically and thematically *i.e.*, there are more studies of wetlands in Africa than in Oceania and more extensive datasets for mangrove than alpine and tundra wetlands.
- There is variation in the methodology of extent estimation used in the literature.
- There is a general lack of detail in the literature on what wetland has been converted to.
- Some large areas of wetlands are not included e.g. Orinoco and Amazon basins due to lack of data.
- Estimates are based on a sample, and individual time series are not weighted according to size.

Sampling methodology and data selection

- The Wetland Extent Index uses a variation of the Living Planet Index (LPI) methodology (originally developed by WWF for monitoring species abundance (Dixon *et al.* 2016) to aggregate extent trend data from the wetland literature.
- The Index calculates the average change in extent for each year compared to the preceding year, which are then chained together to make an index. The Index starts at an initial value of 1 in 1970 and as with the LPI Index, it can be thought of as a biological analogue of a stock market index.
- The analysis is based on a database containing over 2,000 wetland extent time-series records gathered from a literature search and through personal communication with relevant experts with known data.

- The data is best thought of as a matrix with the possible 'wetland classes' of the data across the *x* axis and the possible 'locality' of the wetland down the *y* axis. The cells of the matrix contain the wetland change time-series data for each unique combination.
- The average trend in wetland extent was calculated for all wetlands in each cell of the matrix for which one or more time-series were available. The average trends for individual locality-wetland class combinations (matrix cells) were then aggregated by region, giving each cell equal weight. The regional aggregations were then themselves averaged to create the global Index.
- The Wetland Extent Trends (WET) Index is weighted according to area estimates of wetland extent at the regional level, based on the Global Lakes and Wetlands Database (GLWD).

References

Dixon, M.J.R., Loh, J., Davidson, N.C., Beltrame, C., Freeman, R., Walpole, M. (2016, with updated data) Tracking global change in ecosystem area: The Wetland Extent Trends index. Biological Conservation 193, 27–35.

Red List index (forest specialists)

This is an indicator of aggregate extinction risk for species dependent on forests (birds, mammals, amphibians and cycads) derived by disaggregation of the Red List Index based on species for which 'Forest' in the Habitats Classification Scheme (http://www.iucnredlist.org/technical-documents/classification-schemes/habitats-classification-scheme-ver3) is classified as of 'major' importance (Butchart *et al.* 2004 PLoS Biology). Although not widely used to date, it can be derived now as an indicator towards Aichi Target 5 and SDG indicator 15.2. It could also be expanded to other habitat-specialist species as useful future.

Model fit

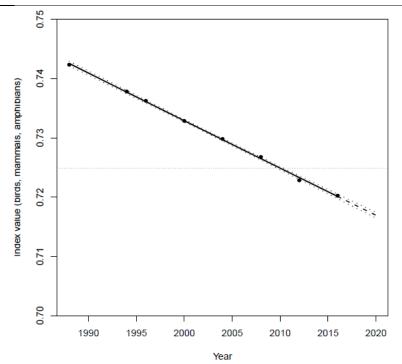


Figure. Modelled trend in the *Red List Index (forest specialists)* 1988-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant decline between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The

horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A Red List Index value of 1.0 equates to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. A Red List Index value of zero indicates that all species have gone extinct. A downwards trend in the graph line (i.e. decreasing Red List Index values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing.

The *Red List Index (forest specialists)* is projected to continue to decline significantly to 2020, representing a deteriorating status of these species on the IUCN Red List. Although the absolute change in index value over time is relatively low.

Strengths

• The Red List Index is based on data from the large majority of species worldwide for each group considered, and hence is less geographically biased than many comparable indicators.

Caveats

- The Red List Index is only moderately sensitive, owing to the breadth of Red List categories (Butchart *et al.* 2004, Butchart *et al.* 2005).
- The attribution of taxa to a specific habitat (e.g. forest) is challenging, which may limit accuracy.

Sampling methodology and data selection

The Red List Index was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A Red List Index for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010). Red List Index trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide.

References

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Wild Bird Index (habitat specialists)

Wild Bird Indices show the average population trends of selected species, based on systematic surveys and monitoring schemes. These data are currently only available for North America and Europe. In these regions, Wild Bird Indices for suites of species that are characteristic of different habitats (forest, grassland, arid land and farmland) have declined. Overall, habitat-specialists have declined by about 25% since 1980. Aichi Target 5 calls for loss of "all natural habitats" to be halved, and degradation and fragmentation to be "significantly reduced". While remote sensing data are useful for quantifying the rate of clearance of forest and some other habitats, they are less useful for quantifying habitat degradation, whereas birds can be useful indicators of environmental health.

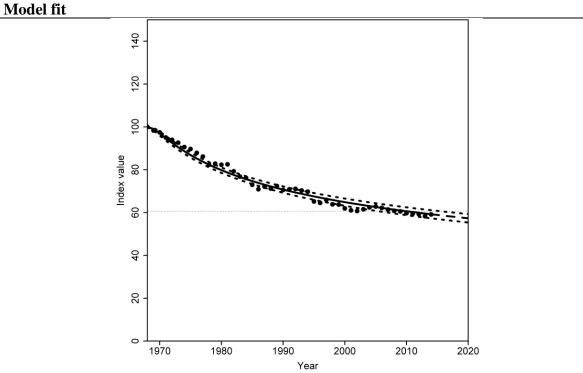


Figure. Modelled trend in the *Wild Bird Index (habitat specialists)* 1968-2014 and statistical extrapolation from 2015-2020. The trend suggests a significant decrease between 2010 and 2020. The Index is set to 100 in 1968. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The declines in habitat specialist species shown by the Wild Bird Indices suggest that habitats in these two regions continue to be degraded, with a significant (though slowing) decline. Trends in many other regions are likely to be similar or worse, and trends for birds are indicative of wider biodiversity declines (Gregory *et al.* 2010).

Strengths

• Based on systematic monitoring and robust sampling.

Caveats

• Trends available only for two temperate developed regions (Europe and North America).

Sampling methodology and data selection

Average population trends of a suite of representative wild birds are measured as an indicator of the general health of the wider environment. Single-species indices are combined to produce a multi-species indicator represented by a single line on a graph, indexed to an arbitrary year for presentational purposes (usually 100 in the start year). Each

species is weighted equally, meaning that the indicator measures changes in species composition (Sheehan *et al.* 2010).

References

Gregory, R.D. and van Strien, A., 2010. Wild bird indicators: using composite population trends of birds as measures of environmental health. Ornithological Science, 9(1), pp.3-22.

Sheehan, D.K., Gregory, R.D., Eaton, M.A., Bubb, P.J. and Chenery, A.M., 2010. The wild bird index – guidance for national and regional use. UNEP-WCMC, Cambridge, UK.

Aichi Target 6

Proportion of fish stocks in safe biological limits

Fisheries are an important source of food, income, jobs, and recreation for people around the world. Global marine fisheries produced just over 80 million tonnes of fish in 2014, providing about 17% of people's animal protein intake, and directly employed about 57 million people world-wide (FAO, 2016), thus making significant contributions to food security and the economy. However, fishing has also impact on fish stocks and their relevant marine ecosystems. With the continued increase of the world population, demand for fish will increase and so will pressure on fish resources. The *Proportion of stocks in safe biological limits* is a measure of the sustainability of fishery resources and is related to maximum sustainable yield (MSY). The *Proportion of fish stocks in safe biological limits* represents those stocks which are not overexploited, depleted, or recovering from overexploitation or depletion.

Model fit

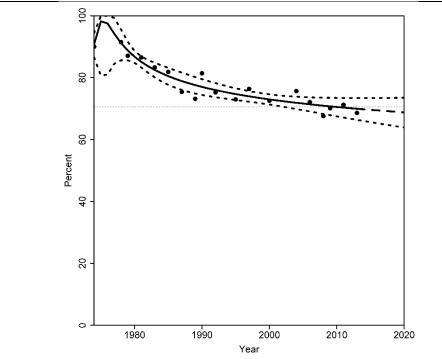


Figure. Modelled trend in the *Proportion of fish stocks in safe biological limits* 1974-2013 and statistical extrapolation from 2014-2020. These represent fish stocks that are not overexploited, depleted, or recovering. The trend suggests a non-significant decline between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Fish stocks outside safe biological limits are those which have been fished down to below the level estimated to produce maximum sustainable yield. Fish stocks within safe biological limits include those which are fully exploited, and so at or close to maximum sustainable production, as well as non-fully exploited stocks. It is predicted that the proportion of fish stocks inside safe biological limits will continue to decline to 2020, though that the decline will not represent a significant change from the 2010 value.

Strengths

- Global data available from 1974 onwards.
- The stocks monitored account for about 80% of global fish landings.

Caveats

- The indicator may not be representative of stocks that were not monitored.
- No national proportions of stocks outside or inside safe biological limits can be calculated from the FAO assessment.

Sampling methodology and data selection

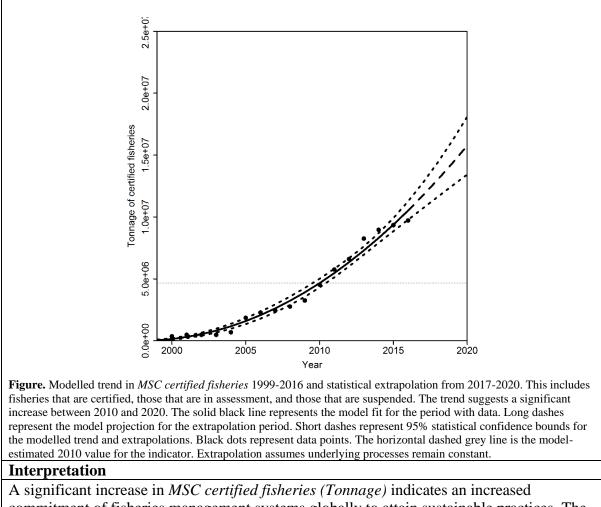
The FAO assessment is based on FAO's statistical areas, i.e. a species within the statistical area is considered an assessment unit, which is different from the classical concept of unit fish stock. The FAO assessment classifies fish stocks into three categories: overexploited; fully exploited; and under-exploited. The percentages were calculated based on the number of stocks for each category at global level. The proportion of fish stocks outside safe biological limits is the percentage of overfished stocks, while the proportion of fish stocks inside safe biological limits is the percentages of fully exploited and under-exploited stocks.

References

FAO (2016). *The State of World Fisheries and Aquaculture 2016* (Food and Agriculture Organization of the United Nations, Rome, 2016).

Marine Stewardship Council certified fisheries (Tonnage)

The increase in the number of *Marine Stewardship Council (MSC) certified fisheries* highlights the continued commitment from fishers, seafood companies, scientists, conservation groups and the public to promote fisheries best practices through certification programs and seafood eco-labelling. The tonnage of fisheries certified through the MSC certification process indicates the level of engagement and commitment of fisheries to strive towards sustainable practices.



commitment of fisheries management systems globally to attain sustainable practices. The indicator shows a positive trend; since 2000, the tonnage of MSC certified fisheries has increased to just under 10,000,000 tons. MSC certified fish represent around 12% of the global marine wild-capture (Marine Stewardship Council 2017).

Strengths

• The global baseline of data available can be disaggregated at the sub-global, regional and national levels.

Caveats

- The MSC data doesn't include aquaculture information as the MSC only certifies wild capture fisheries.
- An increase in tonnage of fisheries does not accurately represent an increase in smallscale fisheries accessing the MSC program.

Sampling methodology and data selection

The *MSC certified fisheries* indicator reveals trends in the tonnage of fisheries certified with the MSC. By the end of 2016, 296 fisheries were certified by the MSC (Marine Stewardship Council 2017). The MSC's standard for sustainable fishing is comprised of three core principles that every fishery in the program must meet (Marine Stewardship Council 2014):

1. Sustainable fish stocks;

2. Minimising environmental impact; and

3. Effective management of the fishery.

In addition, measurable environmental improvements need to be demonstrated for a fishery to keep the MSC certificate as sustainable. Improvements are made by completing action plans

relating to the different MSC performance indicators. MSC certified fisheries are required to complete action plans within the 5 years of certification before full certificate re-assessment. Examples of improvements include reduction in catches to improve stock status, changes in fishing gears to minimize impacts on seabirds and habitats, and more comprehensive research programs to better assess stock and their management.

References

Marine Stewardship Council, 2017. MSC global impacts report 2017. MSC, London.
Marine Stewardship Council, 2014. MSC Fisheries Certification Requirements. Version 2. MSC, London.

Global effort in bottom-trawling (kW sea-days)

Destructive fishing practices directly damage or modify habitat structure and heterogeneity, with resulting impacts on both target and non-target species (Turner *et al.* 1999). The use of bottom trawls has increased globally (Watson *et al.* 2006). Bottom trawls directly impact benthic habitats, and can reduce overall biomass and shift the benthic composition towards small opportunistic species. The use of destructive fishing gears is of particular concern for vulnerable habitats such as coral reefs, which are declining at accelerating rates worldwide (Waycott *et al.* 2008; Burke *et al.* 2011). *Global effort in bottom-trawling (kW sea-days)* therefore serves as indication to the scale of adverse impacts of fisheries on stocks, species and vulnerable ecosystems, which underpin Aichi Target 6.

Model fit

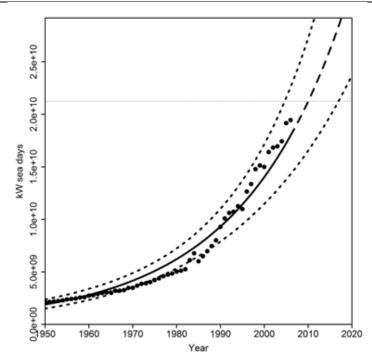


Figure. Modelled trend in the *Global effort in bottom-trawling (kW sea-days)* 1950-2006 and statistical extrapolation from 2007-2020. The trend suggests a significant increase between 2010 and 2020. Solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Projected trends of *Global effort in bottom-trawling (kW sea-days)* show a significant increase to 2020 with an apparent acceleration of total effort. Coupled with the indicator of reduction in the proportion of fish stocks within safe biological limits, this suggests that having all fish stocks that are exploited at or rebuilt to safe biological levels (defined as biomass above biomass at maximum sustainable yield) by 2020 is very unlikely. Overall, although there have been management success stories and positive rebuilding results in some fisheries, the overall global trend suggests increasing exploitation rates due to bottom trawls.

Strengths

• Global data available from 1950 onwards.

Caveats

- This indicator may not reflect the changes in effort of other fishery types (e.g. longliners, purse-seiners).
- The indicator may be sensitive to the assessment of increases in fishing efficiency (see below).

Sampling methodology and data selection

Bottom trawl fishing effort data for the period 1950–2006 were obtained from the FAO, the European Union, and the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR). Data from these diverse and disparate sources were brought together in standardized units based on engine power (watts) and fishing days. From these, all identifiable tuna fisheries effort data were removed to avoid overlap with other sources. Fishing effort reported by agencies and used in this analysis was not initially adjusted for annual efficiency changes. Changes in fishing efficiency can be estimated and fishing effort can be standardized in terms of its effective power (termed effective effort). A conservative annual increase in efficiency of 2.42% has been used based on a prior meta-analysis of published efficiency increases and standardized all effort values to the year 2000.

References

Anticamara, J.A., Watson, R., Gelchu, A. and Pauly, D., 2011. Global fishing effort (1950–2010): trends, gaps, and implications. Fisheries Research, 107(1), pp.131-136.

Burke, L., Reytar, K., Spalding, M. and Perry, A., 2011. Reefs at risk revisited. Turner, S.J., Thrush, S.F., Hewitt, J.E., Cummings, V.J. and Funnell, G., 1999. Fishing impacts and the degradation or loss of habitat structure. Fisheries Management and Ecology, 6(5), pp.401-420.

Watson, R., Revenga, C. and Kura, Y., 2006. Fishing gear associated with global marine catches: II. Trends in trawling and dredging. Fisheries Research, 79(1), pp.103-111.

Waycott, M., Duarte, C.M., Carruthers, T.J., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, A.R. and Kendrick, G.A., 2009.
Accelerating loss of seagrasses across the globe threatens coastal ecosystems.
Proceedings of the National Academy of Sciences, 106(30), pp.12377-12381.

Marine Trophic Index

Fish currently supply the greatest percentage of the world's protein consumed by humans. However, most of the world's fisheries are being fished at levels above their maximum sustainable yield and many regions are severely overfished. The *Marine Trophic Index* (MTI) measures the mean trophic level for all Large Marine Ecosystems and hence indicates the extent of 'fishing down the food webs'. This provides a measure of whether fish stocks, especially of large bodied fish, are being overexploited and whether fisheries are being sustainably managed.

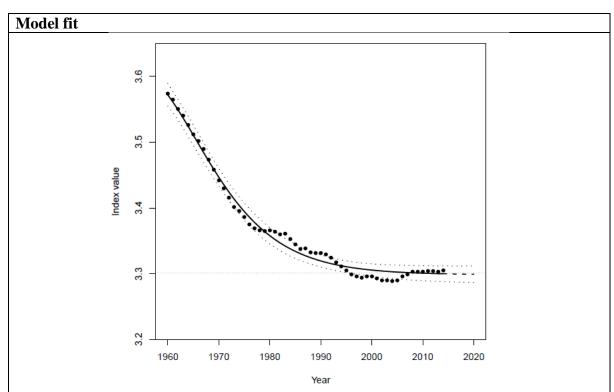


Figure. Modelled trend in *Marine Trophic Index* 1960-2014 and statistical extrapolation from 2015 to 2020. The trend suggests a stabilisation between 2010 and 2020. Note that the y-axis is log-scaled. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The trend MTI has shown a steep decline in value since 1960, but has stabilised in recent years. The decline in index value represents a decline in the abundance and diversity of fish species high in the food chain.

Strengths

• The MTI is a powerful indicator of marine ecosystem integrity and sustainability of fisheries.

• The current data quality is sufficient for global and regional level analyses.

Caveats

- The use of catch composition data as index of relative abundance in the ecosystems
- The quality of the underlying fisheries landings or catch data is poor for some maritime countries (little taxonomic resolution, failure to cover inshore fisheries), and hence the computed index is not as indicative as it could be.

Sampling methodology and data selection

To calculate the MTI, the potential catch that can be obtained given the observed trophic structure of the actual catch is used to assess the fisheries in an initial (usually coastal) region. Actual catch exceeding potential catch indicates exploitation of a new fishing region. The MTI of the new region can then be calculated and subsequent regions are determined in a sequential manner. This method improves upon the use of the Fishing-in-Balance (FiB) index in conjunction with the original MTI calculated over the whole time series because assumptions of fleet and stock stationarity over the entire time series and geographic area are removed. As a default, the Sea Around Us presents the region-based MTI (RMTI) as well as the original MTI/FiB indices in parallel.

References

Pauly D, Watson R (2005) Background and interpretation of the "Marine Trophic Index" as a measure of biodiversity. Philosophical Transactions of the Royal Society-Biological Sciences 360: 415-423.

Red List Index (impacts of fisheries)

Fishing practices can have a number of direct and indirect effects on non-target species for example, as bycatch, mortality in fishing gear, or through disturbance from fishing activities. This disaggregated version of the Red List Index (RLI) shows trends in the status of birds and mammals worldwide driven only by the negative impacts of fisheries or the positive impacts of measures to control or manage fisheries sustainably.

Model fit

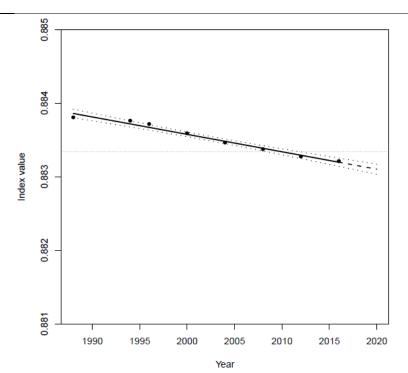


Figure. Modelled trend in the *Red List Index (impacts of fisheries)* 1986-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant decline between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A Red List Index value of 1.0 equates to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. A Red List Index value of zero indicates that all species have gone extinct. A downwards trend in the graph line (i.e. decreasing Red List Index values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing.

The *Red List Index (impacts of fisheries)* shows a decline from 1990, which is projected to continue to 2020, representing a declining status of these species on the IUCN Red List and consequently an increasing extinction risk over time. However, the absolute change in index value over time is relatively low.

Strengths

• The Red List Index is based on data from the large majority of species worldwide for each group considered, and hence is less geographically biased than many comparable indicators

Caveats

There are few data points on which to base the projections.

Sampling methodology and data selection

The Red List Index was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A Red List Index for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010). Red List Index trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide.

References

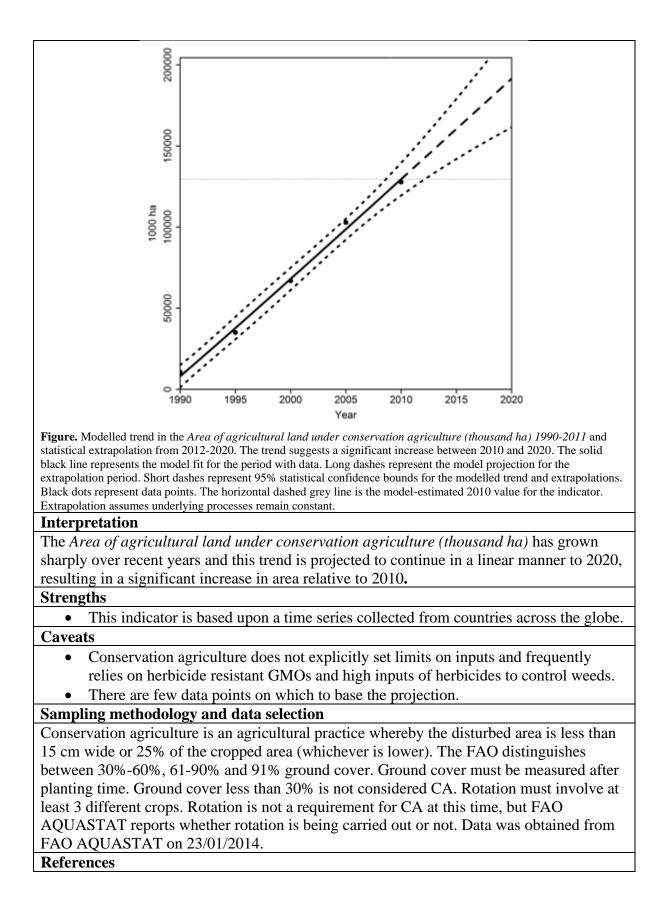
http://www.iucnredlist.org/about/publication/red-list-index

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Aichi Target 7

Area of agricultural land under conservation agriculture (thousand ha)

Conservation Agriculture (CA) is a community of practice that focuses on low tillage, permanent plant cover and crop diversity to reduce environmental impacts and enhance the status of biodiversity in agricultural landscapes. This production system strives to maintain or increase profitability together with high and sustained production levels while concurrently conserving the environment, with a strong focus on soil health. An important aspect of conservation agriculture is the use of a no-tillage system that generally keep soils intact, improves soil diversity, reduces soil erosion, reduces CO₂ emissions from machinery and may improve soil carbon sequestration.



Derpsch, R., Friedrich, T., Kassam, A. and Li, H., 2010. Current status of adoption of notill farming in the world and some of its main benefits. International Journal of Agricultural and Biological Engineering, 3(1), pp.1-25.

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- Soane, B.D., Ball, B.C., Arvidsson, J., Basch, G., Moreno, F. and Roger-Estrade, J., 2012. No-till in northern, western and south-western Europe: A review of problems and opportunities for crop production and the environment. Soil and Tillage Research, 118, pp.66-87.

Area of agricultural land under organic production (million ha)

Organic agricultural practices eliminate many important agricultural pollutants and generally have a positive effect on species diversity in landscapes where they are practiced (Tuck *et al.* 2014). The goals of organic agriculture are generally expressed in terms of broad sustainability, but organic agriculture certification may not include criteria that directly address important issues such as nutrient pollution, soil erosion, crop diversity, land use displacement or economic sustainability and so may not lead to improvements in these criteria (e.g. Leifeld *et al.* 2013).

Model fit

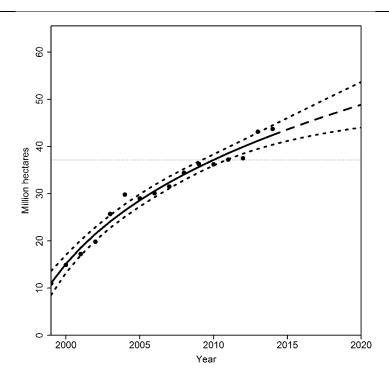


Figure. Modelled trend in the global *Area of agricultural land under organic production (million ha)* 1999-2014 and statistical extrapolation from 2015-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations.

Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The global *Area of agricultural land under organic production (million ha)* is projected to increase significantly by 2020, though the rate may be slowing. This suggests reduced pollutant and fertiliser inputs and hence a potentially beneficial effect on species diversity.

Strengths

• Organic agriculture generally has a positive impact on species diversity in landscapes where it is practiced.

Caveats

- Organic agriculture certification typically does not include criteria that directly address important issues such as nutrient pollution, soil erosion, crop diversity, land use displacement or economic sustainability and may not lead to improvements in these criteria.
- Organic agriculture may give lower crop yields than conventional farming (Leifeld *et al.* 2013), hence requiring more land to grow food.

Sampling methodology and data selection

The Area of agricultural land under organic production (million ha) indicator represents agricultural areas which are certified as organic by the International Federation of Organic Agriculture Movements (IFOAM). Certified organic areas that are already converted, as well as land under conversion, are taken into account, since many data sources do not separate or include the latter (for example Australia, Austria, Germany, Switzerland) and land under conversion is under organic management (Willer and Lernoud 2017). An annual survey is carried out to determine the amount of organic agricultural area by the German Research Institute of Organic Agriculture (FiBL) and IFOAM (FiBL).

References

FiBL, Data collection on organic agriculture world-wide (available at <u>http://www.organic-world.net/statistics/statistics-data-collection.html</u>)

- Leifeld, D. A. Angers, C. Chenu, J. Fuhrer, T. Kätterer, D. S. Powlson, 2013. Organic farming gives no climate change benefit through soil carbon sequestration. Proc. Natl. Acad. Sci. U.S.A. 110, E984 (2013). Medline doi:10.1073/pnas.1220724110
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Nitrogen use balance (kg/km²)

Improvements in nitrogen use efficiency in crop production are critical for addressing the triple challenges of food security, environmental degradation and climate change. Nitrogen input is required to maintain crop production, but inefficient application leads to surplus nitrogen escaping to the environment. This indicator measures agricultural nitrogen-use efficiency.

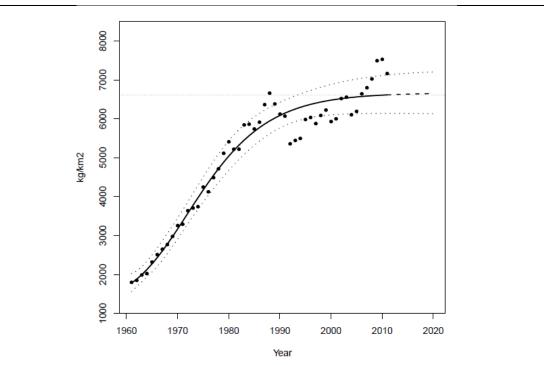


Figure. Modelled trend in the *Nitrogen use balance* (kg/km^2) 1961-2011 and statistical extrapolation from 2012-2020. The trend shows a non-significant increase between 2010 and 2020 which is levelling off. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The extrapolation suggests this indicator will continue to level out from 2011-2020 with only a very slight increase in modelled nitrogen surplus.

Strengths

• The data behind this indicator is comprised of highly accurate national-level data for 113 countries.

Caveats

• This indicator focusses on crop production and therefore does not account for nitrogen surplus produced through livestock management

Sampling methodology and data selection

A nitrogen budget database was established for each country and crop type for 1961 onwards. Nitrogen inputs include fertilizer application, manure application, biological fixation (based on published literature for major legume crops) and atmospheric deposition. Nitrogen outputs are derived from the product of the crop nitrogen content and their yield. The difference between the inputs and the outputs is lost to the environment or remains in the soil. By assuming that over the long term (e.g. over a decade) the average change of the nitrogen in the soil is negligible and is small relative to the annual nitrogen input, then we can assume that the nitrogen surplus is a reasonable index of the nitrogen lost to the environment over the long term.

References

Zhang, X., Davidson, E.A., Mauzerall, D.L., Searchinger, T.D., Dumas, P. and Shen, Y., 2015. Managing nitrogen for sustainable development. *Nature*, *528*(7580), pp.51-59.

Wild Bird Index (farmland birds)

The *Wild Bird Index (farmland birds)* shows the average population trends of species characteristic of farmland, based on systematic surveys and monitoring schemes. These data are currently only available for Europe. As farmland species are reliant upon agricultural habitats, the trends in the *Wild Bird Index (farmland birds)* may be taken as an indication of the sustainability of farming practices.



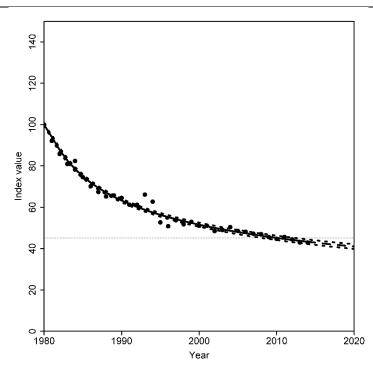


Figure. Modelled trend in the *Wild Bird Index for farmland birds* 1980-2014 and statistical extrapolation from 2015-2020. The trend suggests a significant decrease between 2010 and 2020, though the rate may be slowing. Note: The index is set to 100 in 1980. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Aichi Target 7 calls for agriculture to be managed sustainably. The projected continuing significant decline in the *Wild Bird Index (farmland birds)* indicates that bird species characteristic of farmland habitats in Europe are continuing to be negatively impacted by agricultural practices, and hence there is little evidence of progress towards this target. However, note that while the decrease is significant, it does appear to be slowing over time. Other wildlife groups are likely to be undergoing similar declines, and these trends are likely to be increasingly mirrored in other regions (Gregory *et al.* 2005).

Strengths

• This indicator is based upon systematic monitoring and robust sampling.

Caveats

• Trends are available only for a single temperate developed region.

Sampling methodology and data selection

Average population trends of a suite of representative wild birds are measured as an indicator of the general health of the wider environment. Single-species indices are combined to produce a multi-species indicator represented by a single line on a graph, indexed to an arbitrary year for presentational purposes (usually 100 in the start year). Each species is weighted equally, meaning that the indicator measures changes in species

composition (Sheehan *et al.* 2010). For the purposes of fitting models, data were from 1980 onwards were used.

References

Gregory, R.D., Van Strien, A., Vorisek, P., Meyling, A.W.G., Noble, D.G., Foppen, R.P. and Gibbons, D.W., 2005. Developing indicators for European birds. Philosophical Transactions of the Royal Society of London B: Biological Sciences, 360(1454), pp.269-288.

Sheehan, D.K., Gregory, R.D., Eaton, M.A., Bubb, P.J. and Chenery, A.M., 2010. The wild bird index – guidance for national and regional use. UNEP-WCMC, Cambridge, UK.

Area of forest under sustainable management: total FSC and PEFC forest management certification (million ha)

Commercial forestry and forest conservation are often viewed as being incompatible as they are pursuing different objectives. For example, forestry operations are managing land for continual supply of timber and other forest products, and forest conservationists are managing land for maintaining or restoring biodiversity, ecosystem services and other conservation values. One approach to combining these two objectives in natural resource management is the adoption of multiple use forest management practices, supported by the instrument of forest certification.

The Forest Stewardship Council (FSC) and the Program for the Endorsement of Forest Certification (PEFC) promote sustainable forest management through global systems of certification.

Model fit

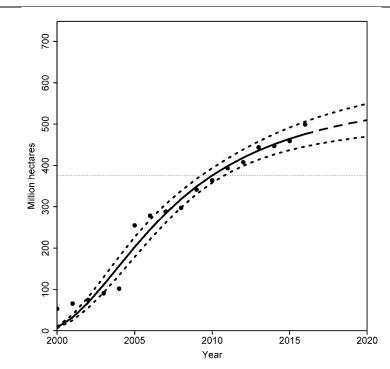


Figure. Modelled trend in the *Area of forest under sustainable management: FSC and PEFC forest management certification* 2000-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant increase in sustainably managed forest between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The Area of forest under sustainable management: FSC and PEFC forest management certification (million ha) indicator measures the area of certified responsibly managed forests, including natural or semi-natural forests that are used to produce timber and non-timber forest products, and forest plantations. The projected trend in the area of certified forest suggests a significant increase by 2020, though the total area certified may be slowing. This represents an increase in the area of commercial forest managed responsibly with respect to biodiversity conservation. This may result in reduced pressures on forest biodiversity within certified areas, which may subsequently reduce biodiversity loss.

Strengths

• The data from the FSC and PEFC provides insight into the use of these global certification programs representing millions of hectares of certified forest around the world.

Caveats

- Many forests that claim to be sustainably managed have not been certified, because the certification process demands time and financial resources that many forest owners, especially those of smaller areas, are unwilling or unable to commit.
- As the data for this indicator is drawn from two independent sources, there is an increased risk of forest area being double counted (i.e. holding both FSC and PEFC certification).
- Certified forests are not only multiple use forest; there are also certified forest plantations, which would be biodiversity poor and bear limited potential for other forest services.

Sampling methodology and data selection

The FSC related data for this indicator originates from the global FSC Certificate Database, which contains up-to-date information as well as public summary reports for all issued certificates, the identification of relevant forest sites and audit results. PEFC data are publicly available from the PEFC online database.

References

https://www.bipindicators.net/indicators/area-of-forest-under-sustainable-managementcertification https://info.fsc.org/

http://www.pefc.co.uk/

Aichi Target 8

Pesticide use (tonnes)

The *Pesticide use* database refers to the use of major pesticide groups (Insecticides, Herbicides, Fungicides, Plant growth regulators and Rodenticides) and relevant chemical families when available. Data refers to quantities of pesticides used in or sold to the agricultural sector for crops and seeds and are expressed in tonnes of active ingredients. However, due to some country reporting practices, the data may be reported by: use or imports in formulated product; sales; distribution or imports for use in the agricultural sector in active ingredients. In these cases it is specified in the country notes. Information on quantities applied to single crops is not available.

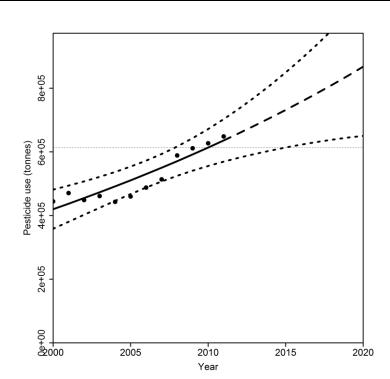


Figure. Modelled *trend in pesticide use* 2000-2011 and statistical extrapolation from 2012-2020. The trend suggests a significant increase in pesticide use between 2010 and 2020. Solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The amount of pesticides used globally has grown over recent years and is projected to continue increasing linearly to 2020, resulting in a significant increase in pesticide usage, relative to 2010.

Strengths

• This indicator provides data collected from countries across the globe.

Caveats

• The reporting by countries is sporadic, making year on year comparisons difficult, and limiting the possibility for inter-country comparisons.

Sampling methodology and data selection

The main source for data collection is the FAO annual questionnaire, sent to the FAO member countries by email in the format of an Excel attachment (when revising a questionnaire a pilot phase is usually carried out for testing and adjusting it according to the feed-backs).

References

http://ref.data.fao.org/dataset?entryId=5e70fee4-fb65-43b6-8da1b6de4626b9bd&tab=about

Food and Agriculture Organization of the United Nations. FAOSTAT. Pesticides (use). (Latest update: Dataset) Accessed (06 Mar 2014). URI: 11 Dec 2017

Red List Index (impacts of pollution)

This indicator shows trends in the status of birds, mammals and amphibians worldwide, but reflects only those trends driven by the negative impacts of pollution or the positive impacts of its control. It is based on assessments of extinction risk for the IUCN Red List, specifically the number of species in each Red List category of extinction risk, and the number moving categories between assessments owing to genuine improvement or deterioration in status driven by pollution or its control. All other changes are excluded, whether from improved knowledge, or genuine impacts of other threats or their control. **Model fit**

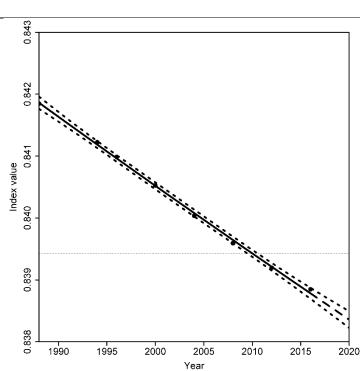


Figure. Modelled trend in the *Red List Index (impacts of pollution)* 1988-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant decrease between 2010 and 2020, though the relative magnitude of the change is very small. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A Red List Index value of 1.0 equates to all species being categorized as Least Concern, and hence none are expected to go extinct in the near future. A Red List Index value of zero indicates that all species have gone extinct. A downwards trend in the graph line (i.e. decreasing Red List Index values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing.

The impacts of pollution on birds, mammals and amphibians worldwide are projected to continue in a linear manner until 2020, with a significant decline in Red List Index values by 2020. However, note that the absolute magnitude of the decline, while significant, is very small, indicating that other drivers of trends are more significant.

Strengths

• The only global indicator available that is able to disentangle trends in species status driven by particular factors.

Caveats

- The Red List Index is only moderately sensitive.
- Trends for other taxonomic groups (e.g. reptiles, plants, invertebrates) are not yet available or have insufficient data for the analysis here.

• There are few data points upon which to base the projection.

Sampling methodology and data selection

The Red List Index was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A Red List Index for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010). Red List Index trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide.

References

http://www.iucnredlist.org/about/publication/red-list-index

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Nitrogen surplus (Tg N)

There are many natural processes that generate nitrogen (N) inputs into ecosystems; in particular, industry, transport and agriculture have greatly increased N inputs (Fowler *et al.* 2013). The soil N budget is often used as an indicator of environmental N pollution (Bouwman *et al.* 2013). In case of positive values of this budget, there is an excess of inputs over outputs, causing an N surplus. This surplus can be lost to the environment via various pathways, including leaching to groundwater, surface runoff or as gaseous losses of ammonia, nitrous oxide or nitric oxide. In agricultural ecosystems, positive soil N budgets or surpluses are caused by inputs of fertilizers, animal manure, and biological N fixation by leguminous crops such as soybeans. In non-agricultural terrestrial ecosystems, N surplus or enrichment primarily arises from wet and dry deposition of N that has been emitted into the air by industry, transport and agriculture (direct effects of N fertilizer addition to agricultural and aquaculture systems are treated in Aichi Target 7). In aquatic ecosystems, N pollution comes primarily from runoff and leaching of fertilizers from agricultural and to a lesser extent from sewage and wet deposition.

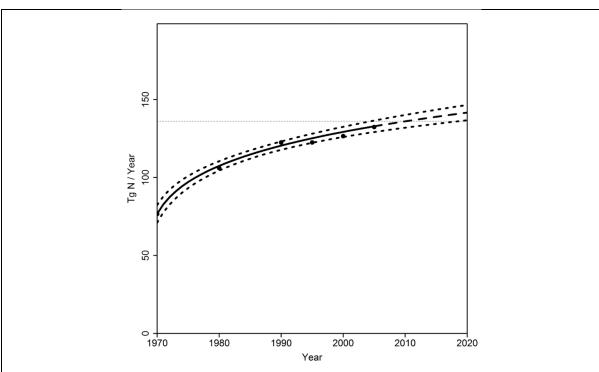


Figure. Modelled trend in the *Nitrogen surplus* 1970-2005 and statistical extrapolation from 2006-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The annual soil nutrient budget includes the N and P inputs and outputs for 0.5 by 0.5 degree grid cells. N inputs include biological N fixation (N_{fix}), atmospheric N deposition (N_{dep}), application of synthetic N fertilizer (N_{fert}) and animal manure (N_{man}). Outputs in the soil N budget include N withdrawal from the field through crop harvesting, hay and grass cutting, and grass consumed by grazing animals (N_{withdr}). The soil N budget (N_{budget}) was calculated as follows:

$$N_{budget} = N_{fix} + N_{dep} + N_{fert} + N_{man} - N_{withdr}$$

It is expected that the N surpluses and thus losses to the environment will increase in the coming decades in a variety of business-as-usual scenarios. Several, primarily industrialized, regions have declining N losses to the environment as a result of increasing efficiency of N utilization and several management strategies aimed at reducing agricultural N losses through leaching, runoff and gaseous losses. At the same time, gaseous N emissions from industry and energy production are projected to be reduced. However, in other, mostly developing, regions, there are large increases in agricultural N use. For example, in Asia, Central and South America and Sub-Saharan Africa the N surpluses in agriculture are projected to increase, while energy and industrial production rapidly increase accompanied by increasing gaseous N emissions and re-deposition.

Strengths

• Although the indicator is not specific about the pathway by which N is injected into the environment, it expresses the pressure of N on the environment, mainly by agricultural production.

Caveats

- This indicator ignores nitrogen accumulation in soil organic matter where there is a surplus, and also ignores nitrogen supply from soil organic matter decomposition in case of nitrogen deficit.
- The data are temporally infrequent, so it may take a longer time to detect changes in trend.

Sampling methodology and data selection

The data are modelled with the IMAGE-GLOBIO models from PBL (2014). How IMAGE models the N surplus can be found on page 274-27 in PBL (2012) and in the most recent IMAGE version 3.0 description (PBL 2014).

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Aichi Target 9

Number of invasive alien species introductions

An "alien species" in this instance refers to a species, subspecies or lower taxon introduced outside its natural past or present distribution; it includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce. "Invasive alien species" is used to mean an alien species whose introduction, establishment and spread threatens biological diversity. This indicator tracks the number of invasive and potentially invasive alien species that have been introduced (and have often become established) in 21 countries over the last 500 years.

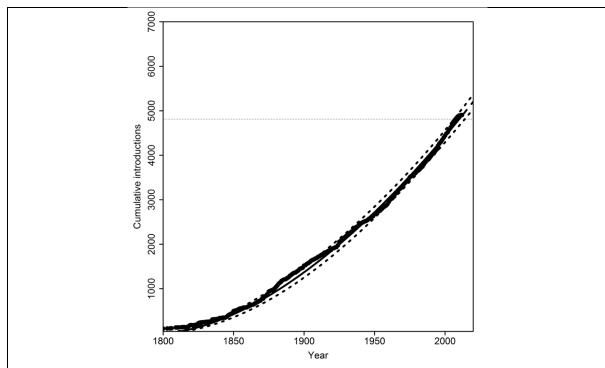


Figure. Modelled trend in the *Number of invasive alien species introductions* 1800-2012 and statistical extrapolation from 2013-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The *Number of invasive alien species introductions* has significantly increased, with no signs of slowing down. The increasing introduction rates of invasive alien species may cause higher establishment rates and are related to increasing international trade and human density. To date, there is an encouraging rise in the adoption of national and international conventions and agreements, regulations and codes of conduct to prevent introduction, establishment, and spread of invasive alien species. Yet, there still exists a gap between international agreements, regulations and measures that are implemented at the national levels, and the implementation of policies themselves.

Strengths

- The dataset is large, encompassing 4,903 introduction records from 3,914 invasive alien species in 21 countries.
- There is a long time-series on which to base extrapolations.

Caveats

- A set of 21 countries distributed around the globe is still not perfectly representative of global patterns. The largest gaps include continental Africa, continental Asia except Israel, and continental Australia.
- For the most recent years, original data may be incomplete due to the time lag between introductions and publication of this information.
- While all taxonomic groups were considered, the majority of the records are plants (>60%), invertebrates, fish, mammals, and birds.

Sampling methodology and data selection

Data were considered from 21 countries that had at least 30 records of species introduction with published year of introduction (at the time of the analyses, i.e. April 2014). These

countries include 9 islands and 12 countries located on continents. Data were fit from the period 1800 onwards.

References

IUCN SSC Invasive Species Specialist Group

Red List Index (impacts of invasive alien species)

This indicator shows trends in the status of birds, mammals and amphibians worldwide, but reflects only those driven by the negative impacts of invasive alien species or the positive impacts of their control. Data are based on assessments of extinction risk for the IUCN Red List, specifically the number of species in each Red List category of extinction risk, and the number moving categories between assessments owing to genuine improvement or deterioration in status driven by impacts of invasive alien species or their control. All other changes are excluded, whether from improved knowledge, or genuine impacts of other threats or their control.

Model fit

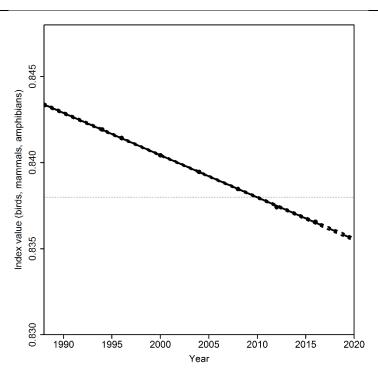


Figure. Modelled trend in the *Red List Index (impacts of invasive alien species)* 1988-2016 and statistical extrapolation from 2013-2020. The trend suggests a significant decrease between 2010 and 2020, though the relative magnitude of the change is very small. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A Red List Index value of 1.0 equates to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. A Red List Index value of zero indicates that all species have gone extinct. A downward trend in the graph line (i.e. decreasing Red List Index values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing.

Aichi Biodiversity Target 9 calls for invasive alien species to be controlled or eradicated. The downward trajectory of the Red List Index for birds, mammals and amphibians showing trends driven by invasive alien species suggests that there will be a significant decrease in their status by 2020. However, note that the absolute magnitude of the decline, while significant, is very small, indicating that other drivers of trends in birds, mammals and amphibians are more significant.

Strengths

• The only global indicator available that is able to disentangle trends in species status driven by particular factors.

Caveats

- The Red List Index is only moderately sensitive.
- Trends for other taxonomic groups (e.g. reptiles, plants, invertebrates) are not yet available or have insufficient data points for the analysis here.
- There are <10 data points on which to base the projections.

Sampling methodology and data selection

The Red List Index was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A Red List Index for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010). Red List Index trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide. Red List Indices have been published showing the negative impacts of invasive species (McGeoch *et al.* 2010).

References

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Percentage of countries with invasive alien species legislation

This indicator measures the *Percentage of countries with invasive alien species legislation*. The global trend in policy response has been positive for the last few decades and the adoption of policies aimed at combatting invasive alien species has significantly increased. As reported in 2010, 55% of the countries signatory to the CBD have enacted invasive alien species relevant national legislation, and 82% of these countries have signed multinational agreements (international conventions, organisation agreements and organisation guidelines) relevant to preventing the spread and promoting the control/eradication of invasive alien species. Among these countries 8% are signatory to all 10 international agreements. For example, the Council of Europe has been developing and adopting codes of conduct addressing some key pathways (e.g. horticulture, botanic gardens, zoos, hunting, or fishing) of invasive alien species. Moreover, once the European regulation on invasive alien species is fully adopted, it will have major implications for neighbouring countries and at a global scale.

Model fit

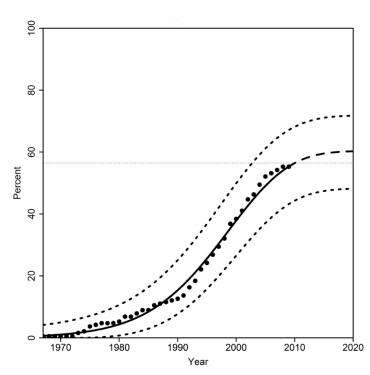


Figure. Modelled trend in the *Percentage of countries with invasive alien species legislation* 1964-2009 and statistical extrapolation from 2010-2020. The trend suggests a non-significant increase between 2010 and 2020, with a projected slowing down in rate. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The projection of the current trend of the *Percentage of countries with invasive alien species legislation* projects a non-significant increase by 2020, with a slowing of the rate of increase in the proportion of countries adopting such legislation. The adoption of national and international policies on invasive alien species may be a first step in combatting the spread of invasive alien species.

Strengths

• This indicator covers 191 countries worldwide.

Caveats

- The adoption of legislation does not necessarily indicate how successfully these policies have been implemented on the ground. There still remains a need for further indicator development to make this link clearer.
- Legislation does not necessarily capture all efforts against invasive alien species that are happening at the national level.

Sampling methodology and data selection

Data for this indicator were produced as follows: all national legislation relevant to controlling invasive alien species was identified for each of the 191 Parties to the CBD. Legislation was considered relevant to the prevention of alien species introductions or to control of invasive alien species if it applied to multiple taxonomic groups and was not exclusively intended to protect agriculture.

References

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Aichi Target 10

Percentage live coral cover

The most widely-gathered metric of coral reef health is the percentage of living coral cover on the reef's surface. This indicator collates datasets from more than 43 countries, representing more than 470 reefs and compassing 1509 records. Aichi Target 10 specifically lists coral reefs as vulnerable ecosystems, and coral reef cover can be used to assess the state of global reefs, though there remains considerable variation between regions, and a strong influence of low and high-frequency stochastic events (e.g. the El Nino Southern Oscillation; ENSO).

Model fit

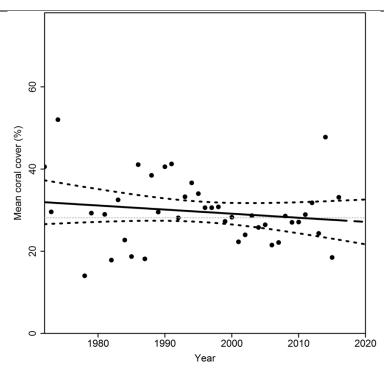


Figure. Modelled trend in the *Percentage live coral cover* 1972-2016 and statistical extrapolation from 2017-2020. The trend suggests a non-significant decrease between 2010 and 2020. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations.

Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The projection of *Percentage live coral cover* suggests a non-significant decline between 2010 and 2020 if underlying processes remain constant. Indeed, the overarching response of coral reefs over the last decade is one of great regional variability in recovery potential. In the Western Indian Ocean, coral reefs were impacted by bleaching events in 1998 and 2016, with 30% of reefs showing evidence of high or severe bleaching, but only 10% showing high or severe mortality (Obura et al., 2017). Therefore although the trend in mortality is not significant globally, the condition of remaining reefs may have severely declined. The Caribbean shows far fewer signs of post-disturbance recovery than reefs of the Indo-Pacific (Roff and Mumby, 2012). Although 75% of the world's reefs are under immediate threat from local impacts and increased sea temperatures (Burke and Bruno, 2010), individual reef trajectories are hugely variable with a notable lack of resilience in the Caribbean. While the most widely-used management tools, Marine Protected Areas (MPAs), are unable to mitigate climate-driven stress, global meta-analyses suggest that coral cover is more stable in MPAs than unprotected areas (Selig and Bruno, 2010). Finally, the drivers of decline are expected to increase in the coming decades, indicating that the underlying processes for this extrapolation may not remain constant.

Strengths

• This indicator is compiled from numerous sources that span the Indian, Pacific, and Caribbean oceans, and numerous reefs of multiple types.

Caveats

- The global average masks considerable underlying variability in the change within ocean basins, regions, and localities.
- Low- and high-frequency stochastic events can have a strong effect on these data (e.g. ENSO).

Sampling methodology and data selection

Coral cover data were collated from published sources, most of which provided mean cover at the scale of individual reefs, although some presented national or even subregional averages. Inconsistent reporting of habitat type and depth prevented a clear assessment of the contribution of local habitat. Data from the Caribbean and Pacific were dominated by forereef habitats (95% and 78% respectively) whereas data from the Indian Ocean were dominated by shallow patch reefs (91%). A dynamic linear model was used to calculate yearly global averages that were then used in the statistical projection framework.

References

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- Burke, L., Reytar, K., Spalding, M. and Perry, A., 2011. Reefs at risk revisited.
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Selig, E.R. and Bruno, J.F., 2010. A global analysis of the effectiveness of marine protected areas in preventing coral loss. PLoS One, 5(2), p.e9278.

Climatic Impact Index for Birds

Birds are useful indicators of the state of the environment as they are sensitive to environmental change, their ecology is well-known and they are relatively easy to survey and count. The *Climatic Impact Index for Birds (CII)* derived from these counts together with other information, quantifies the impacts of recent climate change on the breeding abundance in common birds, accounting for regional variation in both climate impacts and population trends. The CII is relevant to policy makers because it can be used to track biological impacts of climatic warming in near real-time, relating the rate of change in bird populations to observed temperature change and climate drivers.

Model fit

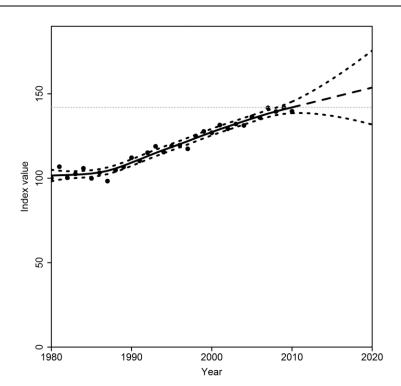


Figure. Modelled trend in the *Climatic impact index for birds* 1980-2010 and statistical extrapolation from 2011-2020. The trend suggests a non-significant increase in impact between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

An upward trend in the graph line for the *Climatic impact index for birds* (i.e. increasing climatic impact index values) means climate change is impacting breeding abundance, however the projected trend between 2010 and 2020 is not significant.

Strengths

- The indicator quantifies impacts of climatic change, one of the most powerful factors shaping future biodiversity.
- The indicator accounts for regional variation in climatic impacts and population trends.

Caveats

- Data is only available for bird species in Europe and North America.
- Data is not available for rare bird species, which may be more vulnerable to climatic impacts.

Sampling methodology and data selection

Developing the indicator involves six steps: (1) selecting species abundance data for analysis; (2) fitting species' distribution models to species' occurrence data and concurrent long-term mean climate values for a single fixed time period, and applying those models to annual climate data to determine how climate suitability has changed for each species in each country or state in which it occurs; (3) checking that these climate suitability trends are informative predictors of abundance trends; (4) deriving composite multispecies abundance indices for each state or country, separately for species with positive climate suitability trends (hereafter, the CST+ group) and for those with negative climate suitability trends (the CST– group); (5) amalgamating country- or state-level information to produce subcontinental CST+ and CST– indices; and (6) contrasting the CST+ and CST– indices to produce a climate impact indicator (CII), which reflects the divergent fates of species favoured and disadvantaged by climate change.

References

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Glacial mass balance (mm water equivalent)

The understanding of fluctuations in the extent of the cryosphere is important not only to the large number of highly-specialised species which depend upon its existence, but also as a striking indicator of global climate change and as a predictor of likely impacts on sea level changes (Jacob *et al.* 2012) and effects on ocean and atmospheric currents. Variations in the extent of glaciers and amount of melt-water run-off from them can have dramatic impacts on local communities through changes to local hydrological processes (WGMS 2012). This indicator examines changes to the mass balance of 'reference' glaciers worldwide (WGMS 2016).

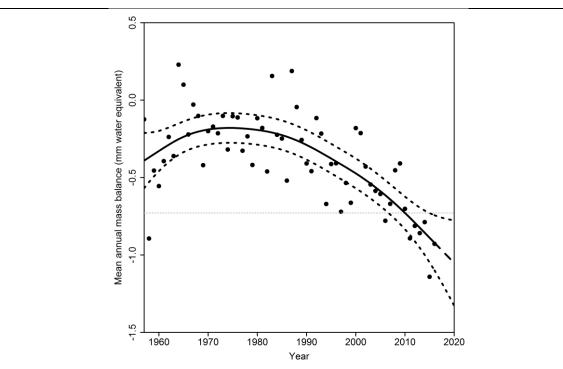


Figure. Modelled trend in *Glacial mass balance (mm water equivalent)* 1957-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant decline between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Glacial mass balance (mm water equivalent) is expected to decrease significantly until 2020, though there is substantial scatter around the data.

Strengths

• A readily understandable and measurable indicator with a long time series.

Caveats

- The volume of glaciers only addresses a small part of the total habitat loss due to the warming of the cryosphere.
- Impacts to the cryosphere are spatially variable, and difficult to describe concisely with available metrics.

Sampling methodology and data selection

The annual change in the *Glacial mass balance (mm water equivalent)* is calculated as a mean of the changes in selected reference glaciers for which there exist long-term repeated measures (Braithwaite, 2002). A negative value indicates that, on average, glaciers around the world have decreased in volume over the year. With constant climatic conditions, the balances would tend towards zero (WGMS 2012). Up to 37 glaciers are used to calculate the glacier mass balance, but as the number of glaciers monitored has varied in the past, records were sub-setted to only include sampling since 1970 where the number of glaciers monitored has remained relatively constant (34±3). The mass balance of each glacier is calculated through a combination of surface area measures (through aerial photographs, ground mapping, and satellite imagery), together with volume and density measures calculated on the ground. Data was obtained from http://geodata.grid.unep.ch/extras/graph_glaciers.php on 31/01/2018.

References

http://geodata.grid.unep.ch/extras/graph_glaciers.php

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WGMS, in ICSU (WDS)/IUGG (IACS)/UNEP/UNESCO/WMO, M. Zemp et al., Eds. (World Glacier Monitoring Service, Zurich, Switzerland, 2012).

Area of mangrove forest cover (km2)

Area of mangrove forest cover (km2) provides a standardized spatial dataset that monitors mangrove cover globally at high spatiotemporal resolutions. These data can be used to improve monitoring of mangrove carbon stocks and establish baseline local mangrove forest inventories required for payment for ecosystem service initiatives such as REDD+ or the voluntary carbon market.

Model fit

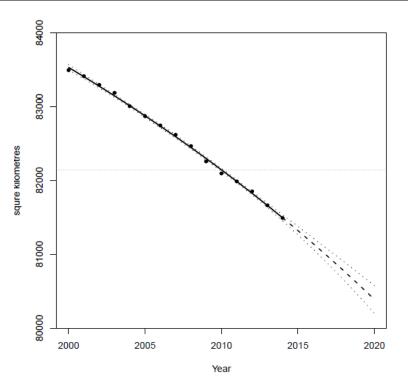


Figure. Modelled trend in *Area of mangrove forest cover (km2)* from 2000-2014 and statistical extrapolations from 2014 to 2020. The trend indicates a significant decline between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The indicator shows that global mangrove forest cover has decreased significantly since 2000 and is projected to continue to decline significantly to 2020, with the rate of decline increasing over time.

Strengths
• These data can be used to drive the mangrove research agenda, particularly as it pertains to monitoring of mangrove carbon stocks and the establishment of baseline local mangrove forest inventories required for payment for ecosystem service initiatives (Hamilton and Casey 2016).
• Data is derived from global remotely sensed products with high spatio-temporal granularity, meaning data can be accurately compared across countries and regions.
Caveats
• There is potential for errors of commission, where non-mangrove trees are incorrectly recorded as mangrove, and errors of omission, where mangrove trees exist but are not recorded.
 Mangrove trees less than 5m tall are not included in this data.
• This data has not undergone under a validation process so may lack accuracy.
Sampling methodology and data selection
The Global Forest Change database, the Terrestrial Ecosystems of the World database and the Mangrove Forests of the World database were synthesized to extract mangrove forest cover at high spatial and temporal resolutions. The new database was then used to monitor mangrove cover at the global, national and protected area scales.
References
Hamilton, S.E. & Casey, D., 2016. Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21).

Global Ecology and Biogeography doi: 10.1111/geb.12449.

Mean polar sea ice extent (million km2)

The *Mean polar sea ice extent* indicator tracks changes of ice cover in the Arctic and Antarctic oceans. The National Snow and Ice Data Center has been monitoring daily changes in sea ice extent for more than 40 years. The extent of sea ice undergoes yearly fluctuations, with minimum levels experienced at each pole in their autumn and maximum extents experienced at each pole in their spring. The enriched waters caused by the summer and autumn melt of the sea ice are important to a range of invertebrates which support large breeding populations of a variety of birds, mammals and fish (Loeb *et al.* 1997). The maximum extent of sea ice in the colder months is vital to a number of highly-specialised vertebrate populations. Variations in the yearly cycles of sea ice extent are known to decrease the breeding success of these species reliant on the cryosphere habitat (Kovacs *et al.* 2011). **Model fit**

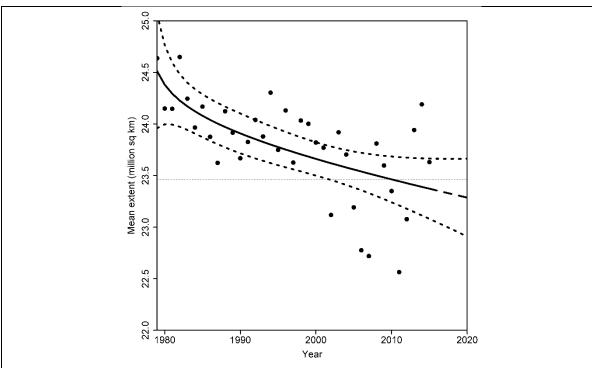


Figure. Modelled trend in *Mean polar sea ice extent* 1979-2015 and statistical extrapolation from 2016-2020. The trend suggests a continued though non-significant decline between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

It is predicted that *Mean polar sea ice extent* will continue to decrease through to 2020, with considerable implications for those species that depend on the yearly cycles of sea ice for specific life history events or processes.

Strengths

• The National Snow and Ice Data Center provides high resolution data collected at regular intervals using a comparable methodology for over 40 years. As such the *Mean polar sea ice extent* indicator provides an accurate and comprehensive dataset for analysis of changes in polar habitat.

Caveats

• Trends in Arctic and Antarctic sea ice extent are not well correlated, and their aggregation may lessen the robustness of the analysis.

Sampling methodology and data selection

The Sea Ice Index compiles weekly and monthly data on Antarctic and Arctic sea ice extent. Sea Ice Index products are derived from two data sets: the Near-Real-Time DMSP SSM/I-SSMIS Daily Polar Gridded Sea Ice Concentrations and the Sea Ice Concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave Data. On monthly extent images, ice ends and water begins where the concentration estimates of grid cells in the gridded average, or mean, concentration field for that month drop below 15 percent. This spatial data is then converted to numerical data using an algorithm developed by NASA (for further details see (Hanna *et al.* 2013, Fetter *et al.* 2002)). Monthly data was used for this analysis as it is dataset is less prone to the impacts of short-term weather (Fetter *et al.* 2002). Yearly data were calculated for each pole by extracting the mean extent per year from the 12 monthly data points. Global data was then produced for each year by summing the northern and southern data points.

References

Fetterer, F., Knowles, K., Meier, W. and Savoie, M., 2002. Sea ice index. Natl Snow and Ice Data Center, Boulder, CO) Available at http://nsidc. org/data/g02135

Hanna, E., Navarro, F.J., Pattyn, F., Domingues, C.M., Fettweis, X., Ivins, E.R., Nicholls, R.J., Ritz, C., Smith, B., Tulaczyk, S. and Whitehouse, P.L., 2013. Ice-sheet mass balance and climate change. Nature, 498(7452), pp.51-59.

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Aichi Target 11

Percentage of terrestrial, marine and coastal areas covered by protected areas

The *Percentage of terrestrial, marine and coastal areas covered by protected areas* helps to track progress in the establishment of a global comprehensive protected area network. Protected areas can provide multiple benefits for biodiversity conservation and sustainable development. They are widely recognized as a major tool for the conservation of species and ecosystems. The biodiversity they protect provides a range of goods and services essential to human well-being. They also help to safeguard natural resources and areas of cultural importance that local communities and indigenous peoples depend on.

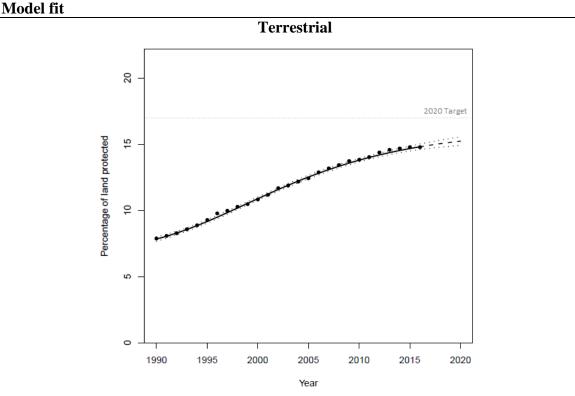


Figure. Modelled trend in *Percentage of terrestrial areas covered by protected areas* 1990-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey ling represents the target of 17% by 2020 under Aichi Target 11. Extrapolation assumes underlying processes remain constant.

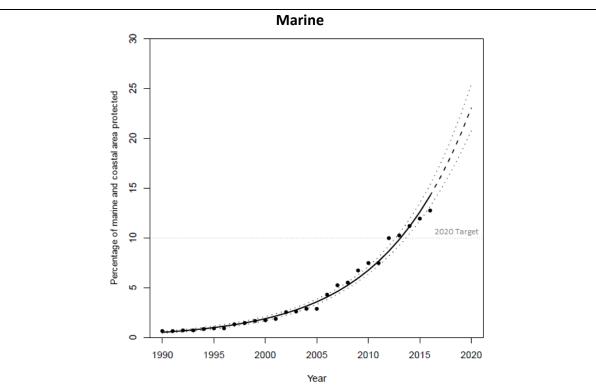


Figure. Modelled trend in *Percentage of marine and coastal areas covered by protected areas* 1990-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey ling represents the target of 10% by 2020 under Aichi Target 11. Extrapolation assumes underlying processes remain constant.

Interpretation

The *Percentage of terrestrial, marine and coastal areas covered by protected areas* indicator measures a policy response to biodiversity loss. An increase in protected area coverage indicates increased efforts by governments and civil society to protect land and sea areas with a view to achieving the long-term conservation of biodiversity with associated ecosystem services and cultural values. The area of the earth's surface that is protected is projected to increase significantly on both land and sea by 2020, with the rate accelerating in the oceans and decelerating slightly on land. By 2020 marine areas covered by protected areas are projected to reach over 20%, while terrestrial areas will only reach around 15%.

Strengths

- The data are available as a time series at all scales (global, regional and national) from 1872 onwards; and can be separately expressed for marine and terrestrial areas at each of these levels.
- Governments are encouraged to continue submitting their protected areas data to the World Database on Protected Areas (WDPA) to ensure accurate representation.

Caveats

- Gaps and/or time lags in reporting protected area data to the WDPA need to be addressed in order to reduce differences in globally and regionally/nationally derived indicator values.
- The indicator uses the year of establishment of protected areas to track change over time. However, some protected areas a missing years of establishment, or may change if there is a change to the site's status (e.g. the designation changes, the boundaries are reviewed or the name changes).

- Future commitments by governments to protect areas are not included in the database, yet may affect the projected trajectory.
- Different methodological approaches to analysing, and revisions of data underlying, the WDPA may produce varying results.
- Data can provide inaccurate values for sites which are partly terrestrial and marine, due to the absence of boundaries between the two.

Sampling methodology and data selection

Protected area coverage statistics were calculated using the December 2016 version of the WDPA. The analysis included all protected areas designated at a national level, those under regional agreements (e.g. Natura 2000 network), and those under international conventions or agreements (e.g. Natural World Heritage sites). UNESCO Man and the Biosphere Reserves, protected areas with a status of "proposed" or "not reported", and sites reported as points without an associated area were removed from the analysis. UNESCO Man and the Biosphere Reserves (MAB reserves) were removed on the basis that their buffer areas and transition zones may not comply with the IUCN protected area definition. Moreover, most core areas of MAB reserves overlap with existing protected areas. A GIS analysis is used to calculate terrestrial and marine protection. A global protected area layer is created by buffering the points recorded in the WDPA based on their reported areas and combining them with the polygons recorded in the WDPA. This layer is overlaid with country boundaries, coastlines and/or buffered coastlines to obtain the absolute and relative coverage of protected areas at national, regional and global scales. Time series are created by grouping the global protected area layer by the known year of establishment of protected areas recorded in the WDPA.

References

IUCN and UNEP-WCMC (2016), The World Database on Protected Areas (WDPA) [Online], Cambridge, UK: UNEP-WCMC. Available at: www.protectedplanet.net.

Percentage Key Biodiversity Areas covered by protected areas

Protected areas are delineated locations that are recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values. Key Biodiversity Areas (KBAs) are sites that contribute to the global persistence of biodiversity, of which ove 15,000 have been identified on land and at sea. This indicator shows trends over time in the degree to which KBAs are covered by protected areas.

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Figure. Modelled trend in the Percentage Key Biodiversity Areas covered by protected areas 1980-2016 and statistical								
extrapolation from 2017-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation								
period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots								
represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.								
Interpretation	sunden	ying proces		istant.				
The significant increase of Percentage Key Biodiversity Areas covered by protected								
areas shows that KBAs are increasingly covered by protected areas over time.								
However, the data show that less than half of each KBA is covered on average, and one-third of sites have zero coverage. Recent slowing growth in coverage may								
						th in covera	age may	
	m tim	e lags in	capturing	data in the W	DPA.			
Strengths The Wor	ld Dat	abase on	Protected	Areas (WDF	PA) is a com	nrehensive	database with	
• The World Database on Protected Areas (WDPA) is a comprehensive database with inputs from all nations on earth.								
 The Wold Database of Key Biodiversity Areas is a global database of important 								
sites for biodiversity in terrestrial, freshwater and marine realms, from nearly all								
countries worldwide.								
Caveats								
• The WDPA is biased towards state managed protected areas.								
• Only KBAs identified using data for birds (Important Bird and Biodiversity Areas)								
and for highly threatened taxa (birds, mammals, amphibians, corals, conifers and selected reptiles) restricted to single sites (Alliance for Zero Extinction sites) are								
geographically comprehensive; KBAs for other taxa have mainly been identified in								
	biodiversity hotspots.							
Sampling methodology and data selection								
The indicator is calculated from data derived from a spatial overlap between digital								
polygons for protected areas from the World Database on Protected Areas (WDPA, IUCN								
and UNEP-WCMC, 2016) and digital polygons for Key Biodiversity Areas (from the World Database of Key Biodiversity Areas, including Important Bird and Biodiversity								
Areas, Alliance for Zero Extinction Sites, and other Key Biodiversity Areas; available at								

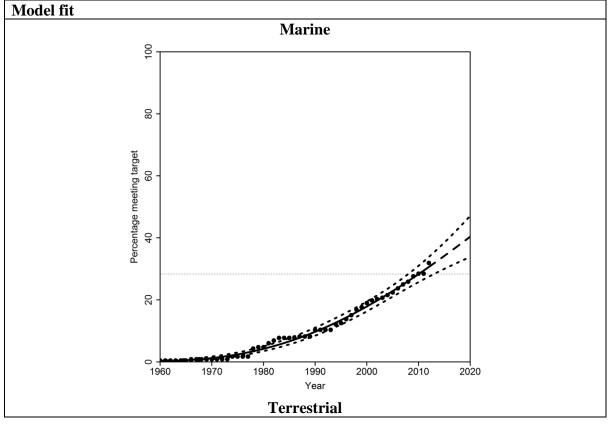
www.keybiodiversityareas.org and through the Integrated Biodiversity Assessment Tool at https://www.ibat-alliance.org/ibat-conservation/login). The indicator shows temporal trends in the mean % of each Key Biodiversity Area that is covered by protected areas. This is calculated using data on the year of protected area establishment recorded in the WDPA. As this is unknown for c.14% of terrestrial protected areas, a year was randomly assigned from another protected area within the same country, or for countries with less than five protected areas with known year of establishment, from all terrestrial PAs, and then this procedure was repeated 1,000 times, and the median was plotted.

References

IUCN and UNEP-WCMC, 2016. The World Database on Protected Areas (WDPA) [Online], Cambridge, UK: UNEP-WCMC. Available at: www.protectedplanet.net.

Protected area coverage of marine and terrestrial ecoregions

Ecoregions are ecologically and geographically defined areas supporting characteristic, geographically distinct assemblages of natural communities and species. This indicator shows the percentage of marine and terrestrial ecoregions that meet a threshold level of protection (17% for terrestrial; 10% for marine).



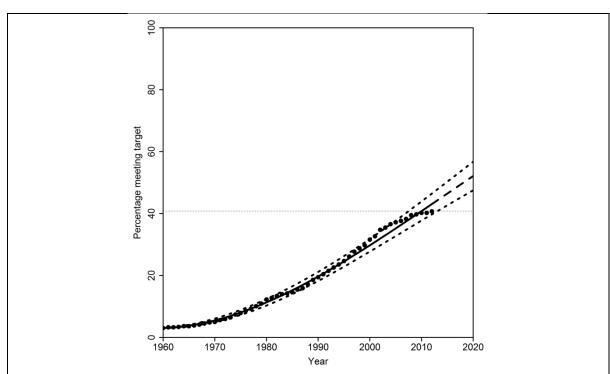


Figure. Modelled trend in the *Protected area coverage of marine and terrestrial ecoregions* 1935-2012 and statistical extrapolation from 2012-2020. The trend suggests a significant increase between 2010 and 2020 in all three ecosystems. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Both ecosystems are projected to show a significant increase in coverage by 2020, though with the rates of addition varying from exponential in the marine realm, to decelerating in the terrestrial. Protected areas cover a growing number of the world's ecoregions and a growing proportion of each of them: currently 55% of terrestrial ecoregions and 37% of marine ecoregions have at least 10% coverage and 7% of terrestrial and 7% of marine ecoregions have at least 75% coverage (Butchart *et al.* unpublished data). On the other hand, 7% of terrestrial and 28% of marine ecoregions have less than 1% coverage of protected areas (Butchart *et al.* unpublished data). Protected area coverage varies widely across ecoregions.

Strengths

• The first assessment of protected area coverage of marine ecoregions.

Caveats

- Gaps and limitations of the WDPA generate uncertainty and probably lead to an underestimate of coverage.
- Inevitably, these are limited indicators of the actual protection afforded to these systems.

Sampling methodology and data selection

Trends in coverage over time were based on spatial overlays of ecoregion polygons with protected area polygons, and the dates of establishment of protected areas documented in the WDPA. Protected areas lacking such data were assigned a date at random from other protected areas in the same country, and this was iterated 1,000 times with the median line taken as the indicator.

References

Analysis produced by BirdLife International, Cambridge Conservation Initiative, UNEP-WCMC

Protected area coverage of bird, mammal and amphibian distributions

The *Protected area coverage of bird, mammal and amphibian distributions* helps to track progress in the degree to which the protected area network adequately covers the distributions of species, using these three groups as surrogates for biodiversity more generally. It shows trends over time in the percentage of species for which protected areas achieve at least a target level of coverage, with the targets set individually for each species depending on their range size.

Model fit

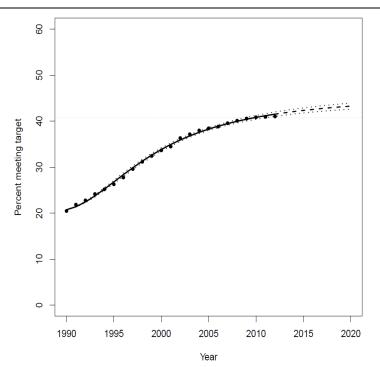


Figure. Modelled trend in the *Protected area coverage of bird, mammal and amphibian distributions* 1990-2012 and statistical extrapolation from 2013-2020. Target levels of coverage were based on the approach of (*103*). The values represent the mean of the three taxa. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

For individual species groups, protected area coverage of species distributions has grown in recent decades, but remains inadequate. The proportion of species meeting target levels of coverage ranges from 55% for birds to 23% for amphibians, and current growth in protected area coverage is inadequate for this element of Aichi Target 11 to be met by 2020. However, the indicator does suggest that the percentage protected area coverage is likely to increase significantly by 2020.

Strengths

• Measures the degree to which protected areas cover species, one of the fundamental units of biodiversity

Caveats

- Assessment is based upon broad maps of distribution extent, and not refined metrics related to occupancy
- Gaps and limitations of the WDPA generate uncertainty and probably lead to an underestimate of coverage.
- Target levels of coverage for each species are determined solely by their distribution size and are not related to species-specific needs taking into account biology and threats
- This approach does not include any information on marine taxa, or invertebrates.

Sampling methodology and data selection

Target levels of protected area coverage for each species were set following (Rodrigues *et al.* 2004), ranging from 100% of the distribution for species with ranges <1,000 km2 to 10% for species with distributions >250,000 km2 (capped at 1 million km2). Trends in coverage over time were based on spatial overlays of distribution maps with protected area polygons, and the dates of establishment of protected areas documented in the WDPA. Protected areas lacking such data were assigned a date at random from other protected areas in the same country, and this was iterated 1,000 times with the median line taken as the indicator. Data were fit from 1990 onwards.

References

Rodrigues, A.S., Akcakaya, H.R., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Chanson, J.S., Fishpool, L.D., Da Fonseca, G.A., Gaston, K.J. and Hoffmann, M., 2004.
Global gap analysis: priority regions for expanding the global protected-area network. *AIBS Bulletin*, 54(12), pp.1092-1100

Funding towards nature reserves (\$)

Funding towards nature reserves (\$) indicator looks at financial commitments globally toward achieving these goals. The funds committed towards protected areas will provide an insight into future extent and management effectiveness trends as funds committed now will be used for these tasks in the future. The funds have been committed from a wide range of funding sources including: the World Bank; the Organisation for Economic Cooperation and Development (OECD); the World Health Organisation (WHO); nation states; multilateral donors such as the African Development Bank; and non-governmental organisations (NGOs).

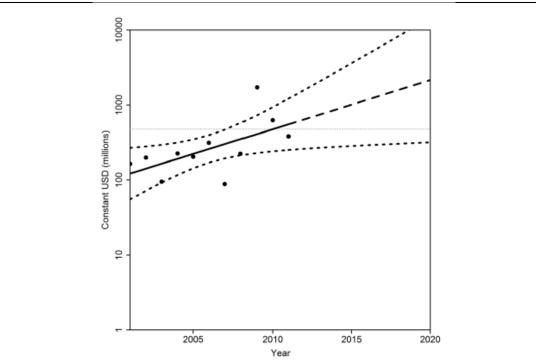


Figure. Modelled trend in the *Funding towards nature reserves* (\$) 1995-2011 and statistical extrapolation from 2012-2020. The trend suggests no significant change between 2010 and 2020. Note the log scale on the y-axis. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Effective management of protected areas relies, at least in part, on adequate funding. *Funds towards nature reserves (\$)* are predicted to show a non-significant increase to 2020, though the confidence in the projection is low.

Strengths

- The metric is based upon a detailed activity categorization scheme that captures information not previously available. AidData activity codes allow users to identify projects not only according to their dominant purpose, but also by their specific components (*i.e.* activities). Thus, the granularity of the data allow for more fine-grained analysis of how funds towards nature reserves are allocated.
- The data included in this analysis covers most large multilateral organizations and represents 45% of all known project-level flows between the years covered.

Caveats

- The project descriptions provided are sometimes brief and unclear as to the quantity of funds specifically earmarked for nature reserves. As such, this analysis includes the full project commitment amount for a project that had at least one activity relating to the indicator. This almost certainly leads to an over-estimation of the funds that are specifically directed to funding nature reserves.
- Activity codes that identify projects with investment in nature reserves are only currently available for certain donors, largely consisting of multilateral agencies and bilateral donors outside of the OECD-DAC.
- This indicator, along with the other AidData financial indicators, do not include internal national spending.

Sampling methodology and data selection

Data were compiled by AidData, an organisation that collects data on international development financing and categorises each project or flow into specific activities and

sectors. Data are presented in constant US dollars (set at 2009 levels). Trends were based upon funds committed from 2000-

2010 only to account for completeness and reliability concerns with earlier data (Development Co-operation Directorate, 2008). Additionally, for the purposes of this analysis, we only included donors for whom more than 95% of their projects/activities have received AidData activity codes.

References

Development Co-operation Directorate (DCD), 2008. DCD/DAC/STAT (2008)17/REV1. Available

at: <u>www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=DCD/DAC/STAT(2</u> 008)17/REV1&docLanguage=En

Number of protected area management effectiveness assessments

Protected areas will only be able to significantly contribute to biodiversity conservation if they are managed effectively. Standardised repeat assessments of management effectiveness have become a powerful tool to support adaptive and effective management of protected areas over time. They help to ensure that protected areas meet their conservation objectives and deliver the desired conservation outcomes. This is of critical importance in meeting Aichi Target 11 as the declaration of a protected area does not always result in adequate protection.

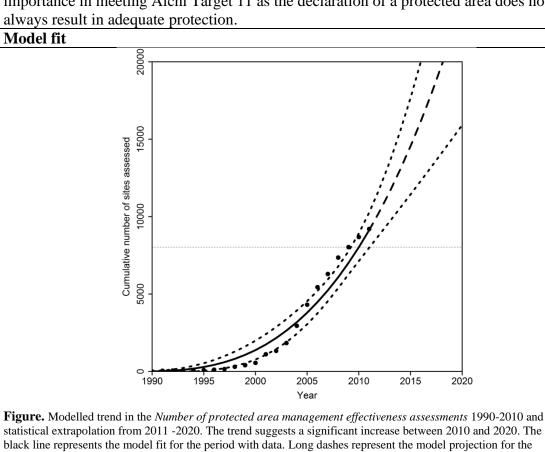


Figure. Modelled trend in the *Number of protected area management effectiveness assessments* 1990-2010 and statistical extrapolation from 2011 -2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Assessments of management inputs and actions, as measured using various management effectiveness tools, have increased dramatically over the past decade, with over 8,000 sites now assessed and hundreds being added each year, particularly in regions where the GEF is actively supporting protected area projects and where countries have adopted regular assessments as part of their business cycle. This trend is projected to continue, resulting in a significantly increased number of sites assessed by 2020. Results from different protected areas show a very wide range of scores, and a recent assessment of 4,100 protected areas designated 13% as having 'clearly inadequate' management, 62% as having 'basic management' and 24% as having 'sound management'. However, repeat assessments suggest that management effectiveness scores are generally increasing over time, especially where the results of assessments are used to address shortcomings and improve management.

Strengths

- This global database can be used to report at national, regional and global levels.
- Improvements in polygon coverage in the WDPA and linking of protected area assessments to the WDPA enable assessment of the total area where assessments have been undertaken as well as the number of sites assessed.

Caveats

- The undertaking of a management effectiveness assessment, although an important first step, does not equate to the area being adequately protected.
- As these assessments usually focus on management inputs and actions and do not measure positive outcomes for biodiversity, further analyses to assess measures of biodiversity outcomes for protected areas are needed to fully report on this element of Aichi target 11.

Sampling methodology and data selection

The global study on management effectiveness was conducted through the auspices of the IUCN World Commission on Protected Areas (WCPA) working with the support of partners and co-workers across the world in government and non-government organisations. A database was compiled of where and when individual assessments of management effectiveness has been undertaken, with associated metadata on methodology and indicators used. Over 11,000 assessments from 8,000 sites in 228 countries, derived from more than 50 methodologies were recorded. 2012 and 2013 were excluded from the fitting process because at the time of modelling the records were incomplete for those years. Data were fit from 1990 onwards.

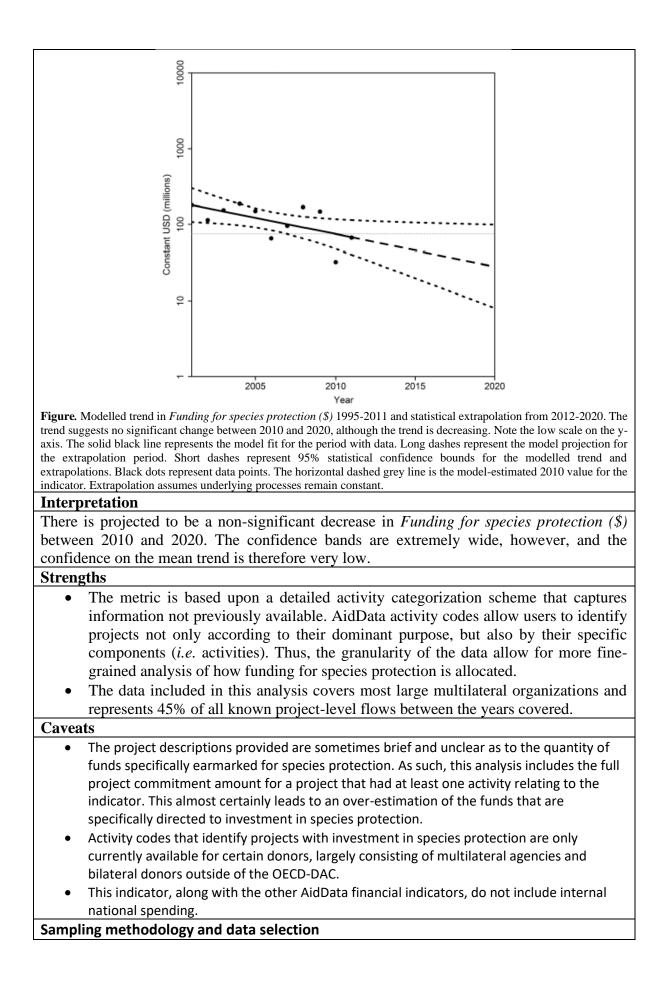
References

Leverington, F., Costa, K.L., Pavese, H., Lisle, A. and Hockings, M., 2010. A global analysis of protected area management effectiveness. Environmental management, 46(5), pp.685-698.

Aichi Target 12

Funding for species protection (\$)

This indicator measures the funds committed towards the conservation of threatened species or habitats. The funds have been committed from a wide range of funding sources including: the World Bank; the OECD; WHO; nation states; multilateral donors such as the African Development Bank; and NGOs.



Data were compiled by AidData, an organisation that collects data on international development financing and categorises each project or flow into specific activities and sectors. Data are presented in constant US dollars (set at 2009 levels). Trends were based upon funds committed from 2000-2010 only to account for completeness and reliability concerns with earlier data (Development Co-operation Directorate, 2008). Additionally, for the purposes of this analysis, we only included donors for whom more than 95% of their projects/activities have received AidData activity codes.

References

Development Co-operation Directorate (DCD), 2008. DCD/DAC/STAT (2008)17/REV1. Available at: www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=DCD/DAC/STAT(2008)17/ REV1&docLanguage=En

Living Planet Index

Wild species are under pressure across all biomes and regions of the world. These declines ultimately result from humanity's demands on the biosphere which result in habitat loss, over-exploitation, pollution, spread of invasive species and climate change. Decline in species populations not only threatens biodiversity, but also ecosystem services which the human race depends on for a multitude of purposes including provision of food, medicine and basic materials. The *Living Planet Index* (LPI) measures trends in vertebrate populations of threatened and non-threatened species and is used as a proxy for monitoring biodiversity change in different habitats.

Model fit

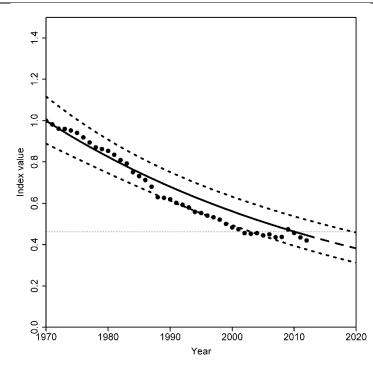


Figure. Modelled trend in the *Living Planet Index* 1970-2012 and statistical extrapolation from 2013-2020. The trend suggests a significant decrease between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A decrease in the LPI represents an overall reduction of species populations, meaning species have declined more, on average, than increased in abundance. This implies that diversity will have reduced, even if none of those species populations has declined to zero (extinction). The projected trend suggests that there will be a significant decrease in the LPI by 2020, although there is some suggestion that the rate may be slowing.

Strengths

- The LPI is not only a global index but can also be calculated for selected regions, nations, biomes or taxonomic groups, provided that there are sufficient data available.
- Data are available as a global time series 1970 onwards (plus Arctic time series from 1970).

Caveats

- At present data submitted by nations and regions must be sent directly to the responsible organizations for the LPI (WWF and ZSL). However, work is currently underway to make the database available online, in the hope that this will encourage nations and regions to submit their data to produce both their own indicators and strengthen the global indicator.
- No invertebrate species are included.

Sampling methodology and data selection

The LPI was calculated using time-series data on more than 9000 populations of over 2,600 species of mammal, bird, reptile, amphibian and fish from all around the globe. The changes in the population of each species were aggregated and shown as an index relative to 1970, which was given a value of 1. The LPI can be thought of as a biological analogue of a stock market index that tracks the value of a set of stocks and shares traded on an exchange.

The Global LPI is the aggregate of two equally-weighted indices of vertebrate populations - the temperate and the tropical LPIs – calculated as the geometric mean of the two. The tropical LPI consists of the terrestrial and freshwater species populations found in the Afrotropical, Indo-Pacific and Neotropical ecosystems and marine species populations from the zone between the Tropics of Cancer and Capricorn. The temperate LPI includes all terrestrial and freshwater species populations from the Palaearctic and Nearctic ecosystems, and marine species north and south of the tropics. In the tropical and temperate LPIs the overall trends in terrestrial, freshwater and marine species are given equal weight. The results of the LPI are published biennially in the Living Planet Report.

References

Collen, B.E.N., Loh, J., Whitmee, S., McRae, Amin, R. and Baillie, J.E., 2009. Monitoring change in vertebrate abundance: the Living Planet Index. Conservation Biology, 23(2), pp.317-327.

Loh, J., Green, R.E., Ricketts, T., Lamoreux, J., Jenkins, M., Kapos, V. and Randers, J., 2005. The Living Planet Index: using species population time series to track trends in biodiversity. Philosophical Transactions of the Royal Society of London B: Biological Sciences, 360(1454), pp.289-295.

Red List Index (birds, mammals, amphibians, corals, cycads

Species are the most intuitive unit of biodiversity, one which resonates with the public and about which we have a relatively good understanding. The IUCN Red List is a well-established and respected system for classifying species by their relative risk of extinction and has been widely recognised as an important component of the suite of indicators needed to track progress towards the 2020 Aichi Targets. The *Red List Index (RLI)* shows changes in the overall extinction risk of sets of species over time, and relates to the rate at which species move through IUCN Red List categories towards or away from extinction. It is calculated from the number of species in each category (Least Concern, Near Threatened, Vulnerable, Endangered, Critically Endangered, Extinct), and the number changing categories between assessments as a result of genuine improvement or deterioration in status (category changes owing to improved knowledge or revised taxonomy are excluded). Tracking the net movement of species through the Red List categories provides a useful metric of changing biodiversity status.



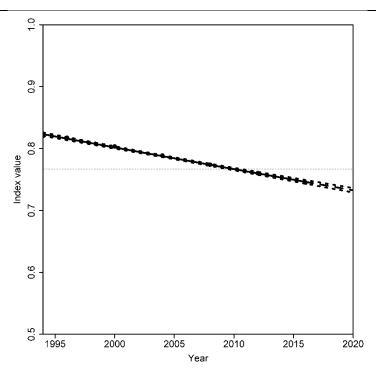


Figure. Modelled trend in the *Red List Index (birds, mammals, amphibians and corals)* various 1994-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant decrease between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A *Red List Index* value of 1.0 equates to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. A *Red List Index* value of zero indicates that all species have gone extinct. A downwards trend in the graph line (i.e. decreasing *Red List Index* values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing. The projection suggests that this will be the case with a significant linear decline until 2020.

Strengths

- The *Red List Index* is fundamental to measure progress towards achieving Aichi Target 12 as it shows trends in survival probability (the inverse of extinction risk) of five major taxonomic groups.
- The indicator is currently used at all levels Global, Regional, Sub-global, National, Sub-national/local (e.g. Szabo *et al.* 2012) – the latter from the increasing suite of countries whiuch have produced comparable red lists for the same species groups at least twice, or for all countries/regions using a procedure that disaggregates the global index, weighting each species in each national/regional index by the proportion of its range in that country/region
- The *Red List Index* can also be disaggregated to show trends for species in different biogeographic ecosystems, political units, ecosystems, habitats, taxonomic groups and for species relevant to different international agreements and treaties.
- The *Red List Index* is based on data from the large majority of species worldwide for each group considered, and hence is less geographically biased than many comparable indicators.

Caveats

• The taxonomic coverage of the *Red List Index* is fairly limited, albeit growing rapidly. A sampled approach to Red Listing has been developed (Baillie *et al.* 2008) to assess the relative extinction risk of additional vertebrate, invertebrate and plant groups, with Red List Indices to be developed in due course.

Sampling methodology and data selection

The *Red List Index* was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A *Red List Index* for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010) and for cycads in 2016. *Red List Index* trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide.

A method for calculating an aggregated *Red List Index* based on the data for multiple taxonomic groups was developed and published in (Butchart *et al.* 2010). More specifically, Red List Indices have been published showing the negative impacts of invasive species (McGeoch *et al.* 2010), and the positive impacts of conservation action (Hoffmann *et al.* 2010) and protected areas (Butchart *et al.* 2012). A *Red List Index* to show the impact of a single conservation institution was published by Young *et al.* (2014). The spatial distribution of the *Red List Index* was mapped by Rodrigues *et al.* (2014). A *Red List Index* for pollinators was published by Regan *et al.* (2015).

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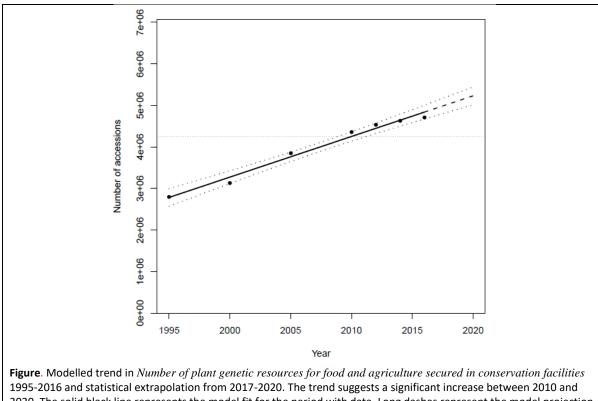
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Aichi Target 13

Number of plant genetic resources for food and agriculture secured in conservation facilities

The measure of trends in ex situ conserved materials provides an overall assessment of the extent to which we are managing to maintain and/or increase the total genetic diversity available for future use and thus protect it from any permanent loss of genetic diversity which may occur on-farm and in the natural habitat. This information is key to support the livelihood of the world's population with sufficient, diverse and nutritious diets long into the future.



1995-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Number of plant genetic resources for food and agriculture secured in conservation facilities has increased since 1995 and is projected to continue to increase significantly to 2020. This positive trend approximates to an increase in secured agro-biodiversity.

Strengths

- The indicator contains data from 71 countries for plants and 128 countries for animals
- For both plant and animal components geographic disaggregation is possible at global, regional and national levels.

Caveats

- The indicator does not account for the addition of duplicates of samples already conserved, or deletion of redundant duplicates, which may lead to overestimating losses or gains in accessions.
- Loss of viability of conserved material that is not detected will contribute to an overestimate in number of accessions reported.
- There are <10 data points with which to estimate the projection.

Sampling methodology and data selection

Computation Method:

Plant genetic resources

The plant component of the indicator is calculated as the total number of unique accessions of plant genetic resources secured in medium to long term conservation facilities. This should include all the accessions in base collections, and unique accessions stored in medium term conservation facilities, as active collections, only when these accessions should be considered to become part of the national base collections.

Animal genetic resources

For the animal component the indicator is calculated as the number of local breeds stored within a gene-bank collection with an amount of genetic material stored which is required to reconstitute the breed (based on the Guidelines on Cryo-conservation of animal genetic resources, FAO, 2012, http://www.fao.org/docrep/016/i3017e/i3017e00.htm).

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Percentage of terrestrial domesticated animal breeds at risk

Genetic diversity in livestock species is important to agriculture and food production because it enables livestock to be raised in a wide range of production environments and to provide a wide range of products and services (food, fibres, manure, draught power etc.). Livestock genetic diversity is threatened by various factors including the trend towards greater homogeneity in the world's livestock production systems and a lack of appropriate management strategies and policies. Planning measures to promote the sustainable use, development and conservation of animal genetic resources requires information on the diversity of these resources nationally and internationally.

The *Percentage of terrestrial domesticated animal breeds at risk* indicator is intended to show whether or not the objective of maintaining the genetic diversity of farmed and domesticated animals has been met, using three indicators of genetic diversity: number of locally adapted breeds, breed extinction risk status, and the proportion of exotic breeds present.

Model fit

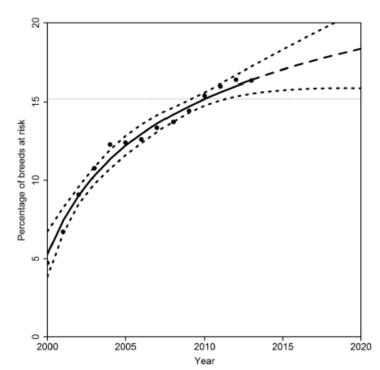


Figure. Modelled trend in *Percentage of terrestrial domesticated animal breeds at risk* 2000-2013 and statistical extrapolation from 2014-2020. The trend suggests a significant increase in the percentage of breeds at risk between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The full indicator consists of three elements: number of locally adapted breeds; proportion of breeds classified as at risk, not at risk and unknown; and proportion of the total population accounted for by locally adapted and exotic breeds. Elements one and three have not yet been fully calculated, because countries have not yet classified their breeds as locally adapted or exotic. For element two, an increase in the percentage of breeds reported to the FAO categorized as at risk or extinct indicates a decline in livestock diversity. However, the large share of breeds with unknown population status contributes to the uncertainty of the indicator.

There is predicted to be a significant increase in the percentage of breeds at risk between 2010 and 2020. However, the slope of the curve is flattening, which implies that countries are taking action. Indicators describing the status of implementing the Global Plan of Action for Animal Genetic Resources presented in 2012 showed gaps in policies, institutions and capacity building related to genetic diversity of terrestrial domesticated animals and severe problems in funding the implementation of relevant actions in most of the reporting countries. Therefore, a lack of policies considering animal genetic diversity, a lack of institutions, capacity building and funding can be seen as the main factors hindering the achievement of Aichi Target 13.

Strengths

Available as a time series since 1980s (despite many gaps), the data can be disaggregated at the regional and national levels, as well as by livestock species.

Caveats

- Despite 182 countries having contributed, the population dataset remains incomplete.
- Data updates are insufficiently regular at present to allow for an accurate assessment of trends. The timescales of present trends, therefore, depend on the regularity of countries entering population data into the Domestic Animal Diversity Information System (DAD-IS) and the completeness of historical data.
- Based on the aggregation of breed risk-status data, it is important to bear in mind that breed diversity does not fully reflect genetic diversity, because it does not account for within breed diversity or for how closely breeds are related to each other. Measuring the effects of genetic dilution through uncontrolled cross-breeding, a substantial threat to diversity, is a particular problem that is not yet captured in DAD-IS.

Sampling methodology and data selection

In the absence of direct measures at a genetic level, the main method used to estimate trends in the diversity of terrestrial domesticated animals is to monitor aggregate changes in breed risk status, i.e. changes in the proportion of breeds categorized as being at risk of extinction. This is achieved using data from the Domestic Animal Diversity Information System (DAD-IS), maintained by FAO. DAD-IS covers more than 30 species used for food and agriculture and includes data on the size and structure of breed populations. Data collection began in 1987 in Europe. From 1996 onwards, DAD-IS has been continuously open to all countries for online data entry; for which input of historical data is encouraged.

About 16% of the approximately 8,200 breeds that have been reported to FAO as of January 2014 are classified as being at risk of extinction based on the most recently available population figures -8% are already extinct. For another 54%, no population data are available and therefore risk status is unknown. For reasons of data completeness, we fit the model to data from the year 2000 onwards.

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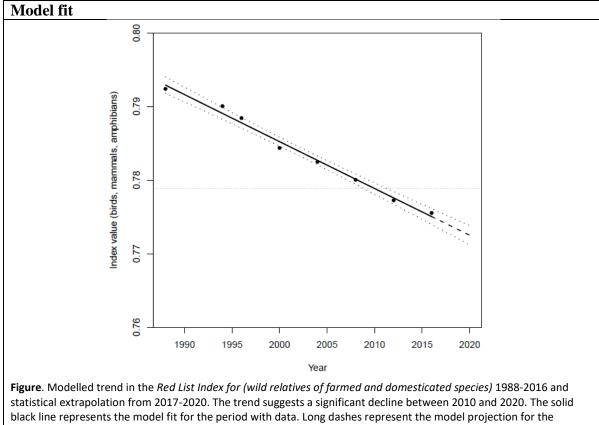
www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=DCD/DAC/STAT(200 8)17/REV1&docLanguage=En

Red List Index (wild relatives of farmed and domesticated species)

The *Red List Index (wild relatives of farmed and domesticated species)* is a disaggregation of RLI data for birds and mammals that are wild relatives of domesticated species.

The indicator is directly relevant to Aichi Biodiversity Target 13 "by 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socio-economically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity." The Red List Index tracks trends in survival probability (the inverse of extinction risk) which is broadly correlated with genetic diversity. Given no data are available on genetic diversity per se, this indicator is the best available proxy for this part of Target 13.

For the same reasons, it is also directly relevant to SDG Target 2.5 "by 2020, maintain the genetic diversity of seeds, cultivated plants and farmed and domesticated animals and their related wild species".



extrapolation period. Short dashes represent 95% statistical confidence bounds for the model projection for the extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A Red List Index value of 1.0 equates to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. A Red List Index value of zero indicates that all species have gone extinct. A downwards trend in the graph line (i.e. decreasing Red List Index values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing.

The *Red List Index (wild relatives of farmed and domesticated species)* shows a significant decline from 1988, this trend is projected to continue in a linear manner to 2020. This represents a deteriorating status of these species on the IUCN Red List.

Strengths

• The Red List Index is based on data from the large majority of species worldwide for each group considered, and hence is less geographically biased than many comparable indicators.

Caveats

• There are <10 data points with which to estimate the projection.

Sampling methodology and data selection

The Red List Index was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A Red List Index for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010). Red List Index trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide.

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Aichi Target 14

Percentage change in local species richness

The extent to which biodiversity change in local assemblages contributes to global biodiversity loss is poorly understood. This indicator includes 100 time series from over 6.1 million species occurrence records from biomes across Earth to assess how species richness within assemblages is changing through time. There are 35,613 species represented, encompassing mammals, birds, fish invertebrates, and plants. The indicator identifies quantified patterns of temporal diversity, measured as change in local diversity. The geographical distribution of study locations is global, and includes marine, freshwater, and terrestrial biomes, extending from the polar regions to the tropics in both hemispheres. **Model fit**

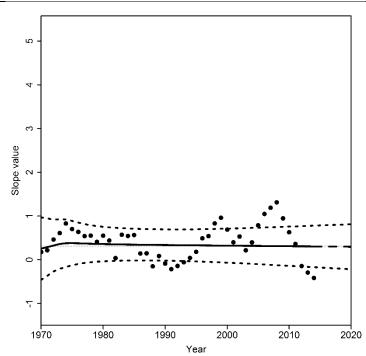


Figure. Modelled trend in the Percentage change in local species richness 1970-2014 and statistical extrapolation from 2015-2020. The trend suggests no significant change between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

This indicator is projected to show no significant trend to 2020, representing no change in local species richness over time. This suggests that, whilst global biodiversity may be declining, local richness is showing no substantial trend, potentially due to species invasions and biotic homogenization. In relation to Aichi Target 14, local species richness can be interpreted as a surrogate for the degree to which ecosystems providing essential services are being effectively safeguarded (albeit with caveats about species invasions and biotic homogenization).

Strengths

• All metrics are calculated from the original local time series data from around the globe, rather than relying on published summary statistics, and thus are able to standardize sampling effort within each time series.

Caveats

- Not all data sets have constant species richness.
- Local richness does not reflect global richness.

- Data are not necessarily from heavily perturbed or impacted sites, which may show different trends.
- The indicator does not account for sampling bias.

Sampling methodology and data selection

This indicator quantifies change in biodiversity through time using temporal trends in a diversity. Temporal a diversity is a measure of diversity within a sample. It can be measured as species richness or with related diversity metrics that take species abundances into account. To measure temporal change in a diversity, the slope of the long-term relationship between diversity and time is calculated for each time series. Multiple time-series were aggregated by fitting a model estimating overall slope within a moving time window, resulting in a single annual data point to which the extrapolation approach was fitted.

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Red List Index (pollinator species)

Biodiversity provides many different ecosystem services at local to global scales. Most services are difficult to link to individual species but pollination is an exception, with multiple studies showing that exclusion of particular groups of pollinator species leads to reduction in crop productivity and value.

The Red List Index can be disaggregated to show trends in survival probability for subsets of species that are known to be pollinators. It is based on data from the IUCN Red List – the number of species in each Red List category of extinction risk, and the number moving categories between assessments owing to genuine improvement or deterioration in status.

Model fit

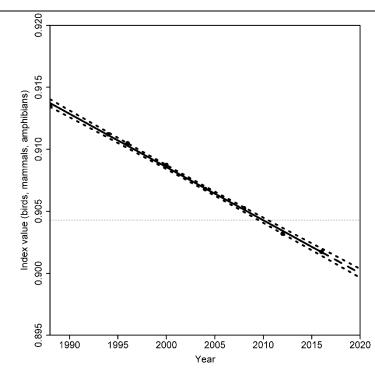


Figure. Modelled trend in the *Red List Index (pollinator species)* 1988-2016 and statistical extrapolation from 2017-2020. The trend implies a significant decrease between 2010 and 2020, although the absolute magnitude of the change is small. The solid black line represents the model fit for the period with data. Long dashes represent the model projection

for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A Red List Index value of 1.0 equates to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. A Red List Index value of zero indicates that all species have gone extinct. A downwards trend in the graph line (i.e. decreasing Red List Index values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing.

The *Red List Index (pollinator species)* among birds (e.g. sunbirds and New World warblers) and mammals (e.g. some bats and rodents) shows declining trends, indicating these species are moving faster towards extinction. However, overall they are less threatened than non-pollinator species (for which the Red List Index has lower values), perhaps reflecting the fact that average body size is larger among non-pollinators, and that large-bodied species tend to be more threatened.

Mammals and birds form only a minority of all pollinators, but data for the many pollinator species among insect groups are currently not available (though an assessment for bumblebees is currently in preparation). It is likely, however, that they too are in decline. Aichi Target 14 calls for "ecosystems that provide essential services" to be "restored and safeguarded". The decline in the *Red List Index (pollinator species)* implies that ecosystems supporting them are not currently being adequately safeguarded.

The *Red List Index (pollinator species)* is projected to continue to decrease significantly towards 2020. However, the absolute magnitude of the change is small but nevertheless implies substantial increases in extinction risk for the species considered.

Strengths

• The only indicator available reflecting trends in the status of nearly all pollinator species worldwide in the two taxonomic groups.

Caveats

- The Red List Index is only moderately sensitive
- Trends for other taxonomic groups, particularly invertebrates are not yet available. Sampling methodology and data selection

The Red List Index was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A Red List Index for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010). Red List Index trends can be calculated for any set of species that has been assessed at least twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide. An RLI for pollinators was published by Regan *et al.* (2015).

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Red List Index (species used for food and medicine)

Biodiversity provides many different ecosystem services to people, at local to global scales. This version of the Red List Index is based only on data for birds, mammals and amphibians that are known to be used by people for food or medicine. It shows changes in the aggregate extinction risk of these species over time. The decline in the index indicates that these species are moving ever faster towards extinction owing to a combination of unsustainable use and other pressures, such as habitat loss driven by unsustainable agriculture, logging and commercial and residential development.

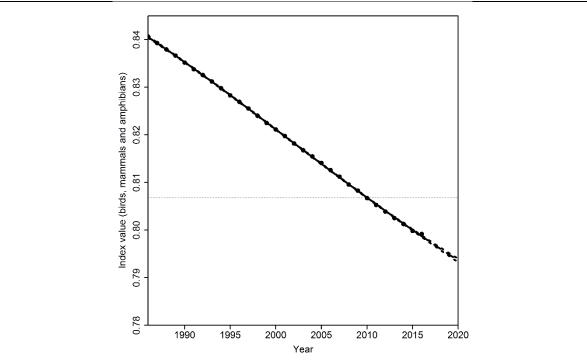


Figure. Modelled trend in the *Red List Index (species used for food and medicine)* 1986-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant decrease between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

A Red List Index value of 1.0 equates to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. A Red List Index value of zero indicates that all species have gone extinct. A downwards trend in the graph line (i.e. decreasing Red List Index values) means that the expected rate of species extinctions is increasing i.e. that the rate of biodiversity loss is increasing.

The *Red List Index (species used for food and medicine)* is in decline and is projected to continue to decline to 2020, representing a deteriorating IUCN Red List status and an increasing extinction risk for these species.

Strengths

- Data can be disaggregated to regional and national levels.
- The Red List Index is based on data from the large majority of species worldwide for each group considered, and hence is less geographically biased than many comparable indicators.

Caveats

- The Red List Index is only moderately sensitive, owing to the breadth of Red List categories (Butchart *et al.* 2004, Butchart *et al.* 2005).
- Trends for other taxonomic groups (e.g. reptiles, plants, invertebrates) are not yet available or have insufficient data for the analysis here.

Sampling methodology and data selection

The Red List Index was initially designed and tested using data on all bird species from 1988–2004 (Butchart *et al.* 2004) and then extended to amphibians (Butchart *et al.* 2005). The methodology was revised and improved in 2007 (Butchart *et al.* 2007). A Red List Index for mammals was added in 2008 and for corals in 2010 (Butchart *et al.* 2010). Red List Index trends can be calculated for any set of species that has been assessed at least

twice for the IUCN Red List. For the set of species considered, trends are based on information from all non-Data Deficient species worldwide.

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Percentage of global rural population with access to improved water resources

Access to an improved water source for the rural population refers to the percentage of the population in these areas using an improved drinking water source that reduces risks of disease transmission of lack of available water close to home. Improved drinking water sources include: piped water on premises (piped household water connection located inside the user's dwelling, plot or yard), and other improved drinking water sources (public taps or standpipes, tube wells or boreholes, protected dug wells, protected springs, and rainwater collection).

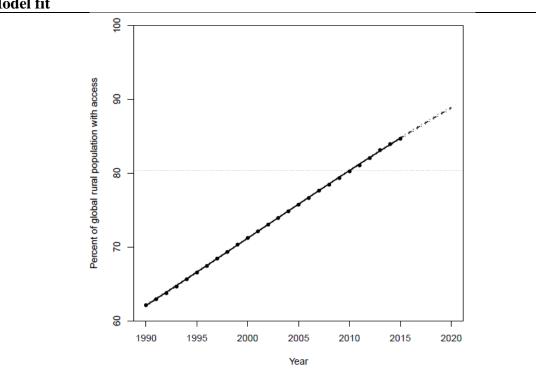


Figure. Modelled trend in *Percentage of global rural population with access to improved water resources* from 1990-2015 and statistical extrapolations from 2016 to 2020. The trend indicates a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The Percentage of global rural population with access to improved water resources shows a positive trend from 1990 to 2015, and is projected to increase in a linear manner to around 90% by 2020.

Strengths

- The indictor reports on global data, which can be disaggregated at the national level to show country-specific trends.
- Data is based on information from users, which provides more accurate information than data from service providers.

Caveats

The data on access to an improved water source measure the percentage of the population with ready access to water for domestic purposes. Access to drinking water from an improved source does not ensure that the water is safe or adequate, as these characteristics are not tested at the time of survey. But improved drinking water technologies are more likely than those characterised as unimproved to provide safe drinking water and to prevent contact with human excreta. While information on access to an improved water source is widely used, it is extremely subjective, and such terms as safe, improved, adequate, and reasonable may have different meaning in different countries despite official WHO definitions. Even in high-income countries treated water may not always be safe to drink. Access to an improved water source is equated with connection to a supply system; it does not take into account variations in the quality and cost (broadly defined) of the service.

Sampling methodology and data selection

The data are derived by the Joint Monitoring Programme of the World Health Organization (WHO) and United Nations Children's Fund (UNICEF) based on national censuses and nationally representative household surveys. The coverage rates for water and sanitation are based on information from service users on the facilities their households actually use, rather than on information from service providers, which may include non-functioning systems. While the estimates are based on use, the Joint Monitoring Programme reports use as access, because access is the term used in the Millennium Development Goal target for drinking water and sanitation. WHO/UNICEF define an improved drinking-water source as one that, by nature of its construction or through active intervention, is protected from outside contamination, in particular from contamination with faecal matter. Improved water sources include piped water into dwelling, plot or yard; piped water into neighbour's plot; public tap/standpipe; tube well/borehole; protected dug well; protected spring; and rainwater.

References

https://data.worldbank.org/indicator/SH.H2O.SAFE.RU.ZS https://www.indexmundi.com/facts/indicators/SH.H2O.SAFE.RU.ZS

Aichi Target 15 No indicator extrapolations available.

Aichi Target 16

Percentage of parties that have ratified the Nagoya Protocol

Aichi Target 16 requires that the Nagoya Protocol enters into force by 2015. The Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization to the Convention on Biological Diversity is an international agreement which aims at sharing the benefits arising from the utilization of genetic resources in a fair and equitable way. It entered into force on 12 October 2014, 90 days after the date of deposit of the fiftieth instrument of ratification. For Target 16 to be met, 50 countries must ratify the Protocol by October 2015 at the latest.

Model fit

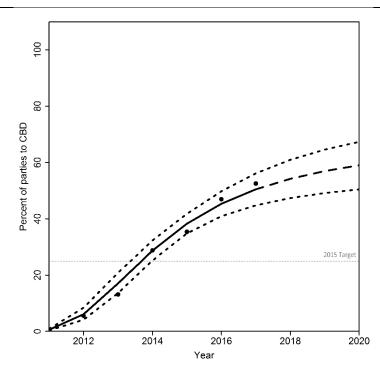


Figure. Modelled trend in the *percentage of parties to the CBD who have ratified the Nagoya Protocol* 2011-2017 and statistical extrapolation from 2018-2020. The trends suggests a significant increase between 2011 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey ling represents the target of 25% by 2015 under Aichi Target 16. Extrapolation assumes underlying processes remain constant.

Interpretation

Data from 2011 to 2017 under the indicator *Percentage of parties that have ratified the Nagoya Protocol* shows that the 2015 target of 50 countries ratifying the protocol (25% of parties to the CBD) has been met. Extrapolation of the trend to 2020 indicates that around 53% of parties will have ratified the protocol by 2020, a significant increase from 2015-2020.

Strengths

• This indicator is directly relevant to Target 16 and provides a straightforward, global insight into the ratification of the Nagoya Protocol

Caveats

• As the Nagoya Protocol was agreed in 2010, there are only data points available under this indicator from 2011 onwards.

Sampling methodology and data selection

As of 2017 data reported to the CBD, 104 Parties to the CBD have ratified the Nagoya Protocol. This indicator includes all countries that have either ratified, acceded to, approved or accepted the Protocol are therefore Parties to it.

References

https://www.cbd.int/abs/nagoya-protocol/signatories/default.shtml

Aichi Target 17

Percentage of countries with developed or revised NBSAPs

Each Party to the CBD is obliged to develop a National Biodiversity Strategy and Action Plan (NBSAP), which is the mechanism for implementing the Convention at the national level. Since the adoption of the Strategic Plan for Biodiversity 2011-2020, Parties have been revising and updating their NBSAPs to bring them in to line with the Strategic Plan and its twenty Aichi Biodiversity Targets. This indicator directly monitors progress towards Aichi Target 17, by measuring how many CBD Parties have developed and revised their NBSAPs in line with the Strategic Plan by 2015.

Model fit

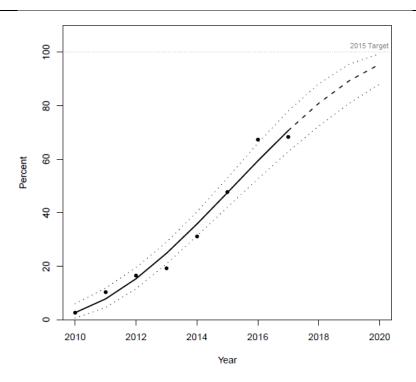


Figure. Modelled trend in the *percentage of countries with developed or revised NBSAPs* 2010-2017 and statistical extrapolation from 2018-2020. The trends suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey ling represents the target of 100% by 2020 under Aichi Target 17. Extrapolation assumes underlying processes remain constant.

Interpretation

The *percentage of countries with developed or revised NBSAPs* has risen significantly since 2010 and is expected to continue to rise leading up to 2020. As of 18 Jan 2018, 189 out 196 Parties (96%) have developed NBSAPs, of these, 139 take the Strategic Plan for Biodiversity (2011-2020) into account. However, by 2015 only 77 countries (39%) had developed NBSAPs indicating that the target of 100% by 2015 was not met. **Strengths**

• This indicator is directly relevant to Target 17 and provides a straightforward, global insight into the adoption of the Convention into national policy

Caveats

The indicator does not give any indication of the effectiveness of the NBSAPs
There are <10 data points on which the trend is calculated

Sampling methodology and data selection

This is indicator is based on a count of the number of NBSAPs that have been submitted to the CBD and whether those submitted post-2010 incorporate the Strategic Plan for Biodiversity 2011-2020.

References

https://www.cbd.int/nbsap/

Aichi Target 18

No indicator extrapolations available.

Aichi Target 19

Number of biodiversity papers published

The *Number of biodiversity papers published* indicator reveals trends in scientific research and transfer of scientific knowledge through an analysis of scientific publications on the topic of biodiversity. Data on the number of published papers with biodiversity in the title was amalgamated using the Web of Science scientific citation indexing service.

Model fit

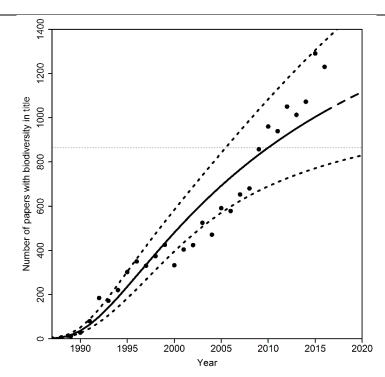


Figure. Modelled trend in the *Number of biodiversity papers published* 1987-2016 and statistical extrapolation from 2017-2020. The trend suggests a non-significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The number of biodiversity papers published annually is predicted to increase, though not significantly, by 2020.

Strengths

• The Web of Science index is a comprehensive archive of scientific biological publications in many different languages.

Caveats

 Searches were conducted for journals with 'biodiversity' in the title, but this technique is likely to miss manuscripts which reported research focussing on biodiversity, but excluded the word from their title. The effectiveness of this as a proxy for all biodiversity-focussed papers is unknown.

Sampling methodology and data selection

Searches for the word 'biodiversity' in the title of the publication were undertaken through the Web of Science search engine and the number of manuscripts published per year were recorded. Searches were undertaken from 1970 to 2016, but only searches from 1987 produced any records.

References

http://wok.mimas.ac.uk/

Funding committed to environmental research (\$)

This indicator provides insight into the funding available for the development and transfer of environmental knowledge. This indicator has two components. The first component measures global funds towards environmental research based in institutions of higher learning, governmental agencies, and other research institutions – such projects will further the science base relating to biodiversity. The second component focuses on the funds committed to environmental education for local communities, schools, and other non-experts – such projects will further the transference and application of biodiversity knowledge.

The funds have been committed from a wide range of funding sources including: the World Bank; the OECD; WHO; nation states; multilateral donors such as the African Development Bank; and NGOs.

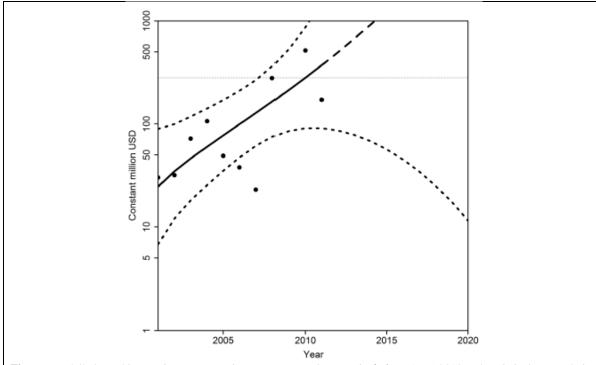


Figure. Modelled trend in *Funding committed to environmental research* (\$) from 1995-2010 and statistical extrapolation from 2011-2020. The trend suggests a non-significant increase between 2010 and 2020. Note the log scale on the y axis. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Funding committed to environmental research (\$) is projected to show a non-significant increase between 2010 and 2020, though the uncertainty around the projection is extremely high.

Strengths

- The metric is based upon a detailed activity categorization scheme that captures information not previously available. AidData activity codes allow users to identify projects not only according to their dominant purpose, but also by their specific components (i.e. activities). Thus, the granularity of the data allow for more fine-grained analysis of how funds for environmental research are allocated.
- The data included in this analysis covers most large multilateral organizations and represents 45% of all known project-level flows between the years covered.

Caveats

- The project descriptions provided are sometimes brief and unclear as to the quantity of funds specifically earmarked for environmental research activities. As such, this analysis includes the full project commitment amount for a project that had at least one activity relating to the indicator. This almost certainly leads to an over-estimation of the funds that are specifically directed to investment in environmental research.
- Activity codes that identify projects with investment in environmental research are only currently available for certain donors, largely consisting of multilateral agencies and bilateral donors outside of the OECD-DAC.
- This indicator, along with the other AidData financial indicators, do not include internal national spending.

Sampling methodology and data selection

Data were compiled by AidData, an organisation that collects data on international development financing and categorises each project or flow into specific activities and

sectors. Data are presented in constant US dollars (set at 2009 levels). Trends were based upon funds committed from 2000-2010 only to account for completeness and reliability concerns with earlier data (Development Co-operation Directorate, 2008). Additionally, for the purposes of this analysis, we only included donors for whom more than 95% of their projects/activities have received AidData activity codes.

References

Development Co-operation Directorate (DCD), 2008. DCD/DAC/STAT(2008)17/REV1. Available at:

www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=DCD/DAC/STAT(200 8)17/REV1&docLanguage=En

Number of species occurrence records in the Global Biodiversity Information Facility

The Global Biodiversity Information Facility (GBIF) is an international open data infrastructure, funded by governments. Founded in 2001, its mission is to provide free and open access to biodiversity data via the Internet, to benefit research and policy. Data from a wide variety of sources can be discovered and accessed via a global portal (www.gbif.org) and web services, as well as through national and thematic web portals and online applications. The data published through GBIF includes species occurrence data from digitized natural history specimen collections, observations from citizen science networks, surveys and research projects, historic literature and a range of other sources. GBIF also deals with names and taxonomic checklists, as well as structured metadata describing biodiversity datasets. As of 2017, more than 1,000 institutions shared data through GBIF, and more than 250 peer-reviewed research papers per year cite GBIF as a source of data. There are 42 voting participants and 48 associate participants including countries, economies and international organizations.

Through collaboration between its Participant nodes and the secretariat, GBIF also fulfils a capacity enhancement role by sharing skills, open-source software, tools and best practices on the mobilization and use of biodiversity data. Therefore, the number of GBIF records indicator reflects the status and trends of shared biodiversity knowledge, science base and technologies to which Aichi Target 19 refers.

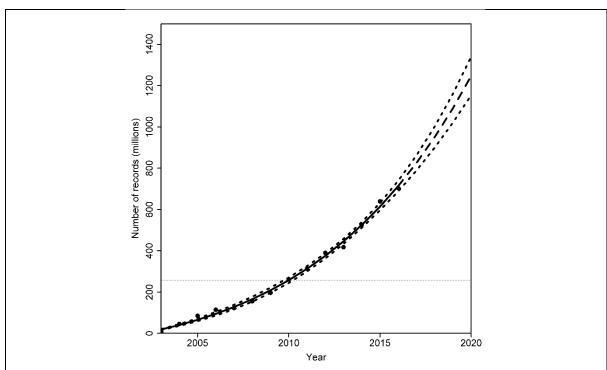


Figure. Modelled trend in the *Number of species occurrence records in the Global Biodiversity Information Facility* 2001-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Number of species occurrence records in the Global Biodiversity Information Facility is projected to grow at an accelerating rate to 2020, suggesting a significant increase in shared biodiversity knowledge.

Strengths

• The metric is based on a continuous time-series of reliable data.

Caveats

- The metric represents only one aspect of data sharing through GBIF, namely the global number of published occurrence records, and does not reflect other parameters such as taxonomic coverage (e.g. number of species), record completeness or geographic biases.
- The metric relates only to datasets currently registered through GBIF, mainly from institutions based in countries currently participating in GBIF. It excludes many digitized records mobilized through other online networks that are not currently linked to GBIF (e.g. Species Link <u>http://splink.cria.org.br</u>), or that have not recently updated the data served through the GBIF network (e.g. the Ocean Biogeographic Information System <u>http://iobis.org</u>).

Sampling methodology and data selection

The number of records available through GBIF over time was calculated by taking a snapshot of the species occurrence records in the GBIF data index at annual intervals since 2003.

References

https://www.gbif.org/the-gbif-network

Proportion of known species assessed through the IUCN Red List

The IUCN Red List has a >50 year history, with its underlying methodology robustly published in the scientific literature. The indicator of *Proportion of known species assessed through the IUCN Red List* has been tracked for many years through the summary statistics updated several times annually on the IUCN Red List website

(http://www.iucnredlist.org/about/summary-statistics), although has not been published into the scientific literature in its own right.

Model fit

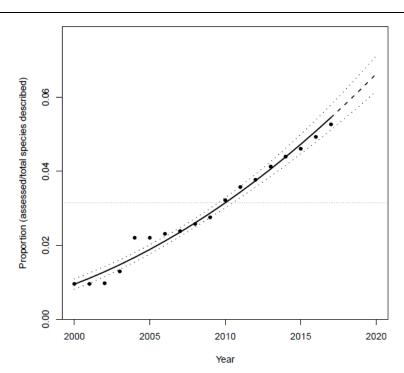


Figure. Modelled trend in the *Proportion of known species assessed through the IUCN Red List* 2000-2017 and statistical extrapolation from 2018-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

This indicator is projected to continue to increase significantly to 2020, representing a large increase in Red List assessed species. This will improve knowledge on the extinction risk of species, aiding conservation initiatives by highlighting those at greater risk.

Strengths

- Automatically updated with all IUCN Red List updates, usually three times per year.
- Complements other Red List Index indicators
- Summary statistics also available by taxonomic group and country

Caveats

• Not all taxonomic groups have been completely assessed.

Sampling methodology and data selection

This indicator is automatically updated with all IUCN Red List updates, usually three times per year.

References

http://www.iucnredlist.org/about/summary-statistics

Species Status Information Index

Primary species occurrence records are essential for monitoring the status and trends of biodiversity, but remain limited and biased in their availability. The *Species Status Information Index (SSII)* measures coverage of mobilized biodiversity data, i.e. its ability to represent the taxonomic, spatial, and temporal variation in biodiversity. The *SSII* thus quantifies the growth in the shared evidence base available and used for advancing knowledge about the distribution of species, and their associated functions, in space and time. The indicator is calculated annually at near global scale for an array of species groups.

Model fit

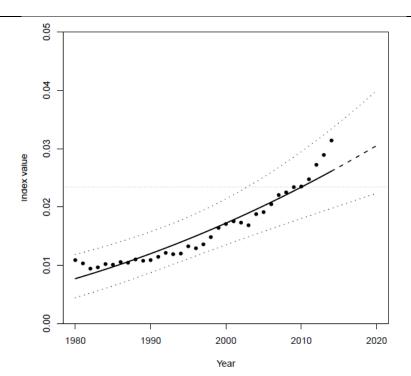


Figure. Modelled trend in the *Species Status Information Index* 1980-2014 and statistical extrapolation from 2015-2020. The trend suggests a non-significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

The *Species Status Information Index* shows an increasing upward trend, which reflects the significant mobilization of data from GBIF.

Strengths

- Data can be disaggregated to spatial levels ranging from small regions, countries, biomes.
- The indicators is made up of millions of observation records for birds, mammals and amphibians

Caveats

- Data only exists for terrestrial birds, mammals and amphibians.
- Several data holders do not make their data accessible to GBIF, meaning this indictor is not an entirely comprehensive source of information.
- The indicator only covers species distribution data and does not include information on other aspects of critical relevance for conservation, such as species abundances, ranging behaviour or conservation status.

Sampling methodology and data selection

The *SSII* characterizes coverage of data mobilized for a given species group, country and year by setting it in relation to expert expectation across a standardized grid. Several metrics are available, with 'Assemblage-level Coverage' the most encompassing. It measures how well, on average, available presence data characterize the makeup of grid cell assemblages and thus the evidence available to quantify status and changes in the makeup of communities, and their associated aggregate functions.

Specifically, the metric is defined as the proportion of species expected to occur in a cell that have been recorded in a given year, averaged across all cells in a country. In a given year, a value of 100% would suggest that at least one record is available for all species expected in each of a country's grid cells, i.e. complete assemblage structure coverage for this spatial resolution. This metric can be shown as a country average, or for individual cells, to identify within-country data gaps and biases that may inform future sampling and mobilization.

Calculations are performed over a standardized global grid of ca. 150-km resolution at the equator for which expert expectations are deemed broadly reliable. Expert expectation information is provided by Map of Life, where the sources used are assumed to be broadly characteristic for the past 35 years. Synonym lists were carefully developed to match species names in presence data to names used for the expert information.

References

Meyer, C., H. Kreft, R. Guralnick, and W. Jetz. 2015. Global priorities for an effective information basis of biodiversity distributions. Nature communications 6: 8221

Aichi Target 20

Funding provided by the Global Environment Facility (\$)

The Global Environment Facility (GEF) brings together funds from 183 countries and international institutions, non-governmental organisations and the private sector to support projects related to biodiversity, climate change, international waters, land degradation, the ozone layer and persistent organic pollutants. The GEF also serves as a financial mechanism for a number of UN environmental conventions including: the Convention on Biological Diversity (CBD); United Nations Framework Convention on Climate Change (UNFCCC); the Stockholm Convention on Persistent Organic Pollutants (POPs); UN Convention to Combat Desertification (UNCCD); and the Montreal Protocol. The *Funding provided by the Global Environment Facility* (\$) indicator measures the funds that GEF has invested in biodiversity work and sustainable development initiatives.

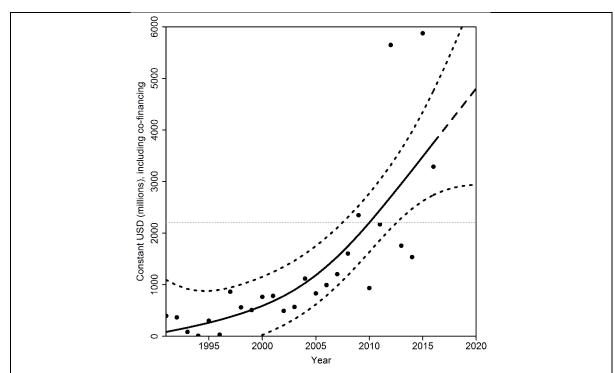


Figure. Modelled trend in *Funding provided by the Global Environment Facility* (\$) 1991-2016 and statistical extrapolation from 2017-2020. The trend suggests a significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Funding provided by the Global Environment Facility (\$) is projected to increase significantly between 2010 and 2020. This implies an increase in investment in biodiversity work and sustainable development initiatives by countries, international institutions, non-governmental organisations and the private sector.

Strengths

Data has been amalgamated from 183 countries across the world.

Caveats

• Inconsistency of data - earlier years report 'Project Closure' rather than 'Under implementation' or 'IA approved' as recorded more recently.

Sampling methodology and data selection

The Global Environment Facility's dataset was downloaded from www.thegef.org on 10/04/2017. GEF projects with a focal area of 'biodiversity' were selected. The indicator uses data of the total amount of GEF funding (recorded per country and in regional or global projects), and the amount of co-financing (comprising the total cash and in-kind resources committed by governments, other multilateral or bilateral sources, the private sector, NGOs, the project beneficiaries and the concerned GEF agency). Total funds in US dollars (made constant at 2016 inflation rates) was used as the *GEF Funding* metric; to provide a measure of outreach in the development of biodiversity values and reflect the national focus of this target.

References

www.thegef.org

Global funding committed towards environmental policy, laws, regulations and economic instruments (\$)

Global funding committed to environmental policy, laws, regulations and economic instruments (\$) measures international financial flows committed to projects that support environmental policy and laws. This metric measures the funds committed from a range of multilateral agencies and bilateral donors outside the OECD Development Assistance Committee (DAC), including the World Bank Group, the Global Environment Facility, African Development Bank, Asian Development Bank, Andean Development Corporation, Arab Bank for Economic Development in Africa, Caribbean Development Bank, OPEC Fund for International Development, European Bank for Reconstruction and Development, and various bilateral agencies.

Model fit

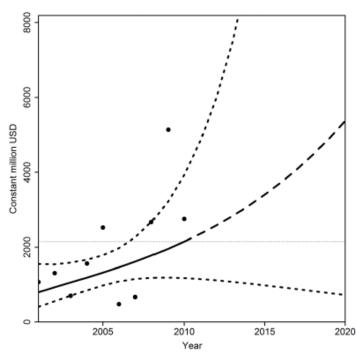


Figure. Modelled trend in *Global funding committed to environmental policy, laws, regulations and economic instruments* (\$) from 1995-2010 and statistical extrapolation from 2011-2020. The trend suggests a non-significant increase between 2010 and 2020. The solid black line represents the model fit for the period with data. Long dashes represent the model projection for the extrapolation period. Short dashes represent 95% statistical confidence bounds for the modelled trend and extrapolations. Black dots represent data points. The horizontal dashed grey line is the model-estimated 2010 value for the indicator. Extrapolation assumes underlying processes remain constant.

Interpretation

Global funding committed to environmental policy, laws, regulations and economic instruments (\$) is projected to show a non-significant increase between 2010 and 2020, though confidence in the projection is relatively low.

Strengths

- The metric is based upon a detailed activity categorisation scheme that captures information not previously available. AidData activity codes allow users to identify projects not only according to their dominant purpose, but also by their specific components (i.e. activities). Thus, the granularity of the data allow for more fine-grained analysis of how funds committed to environmental policy, laws, regulations and economic instruments are allocated.
- The data included in this analysis covers most large multilateral organizations and represents 45% of all known project-level flows between the years covered.

Caveats

- The project descriptions provided are sometimes brief and unclear as to the quantity of funds specifically earmarked for these activities. As such, this analysis includes the full project commitment amount for a project that had at least one activity relating to the indicator. This almost certainly leads to an over-estimation of the funds that are specifically directed to environmental policy or laws.
- Activity codes that identify projects with investment in environmental policy or laws are only currently available for certain donors, largely consisting of multilateral agencies and bilateral donors outside of the OECD-DAC.
- This indicator, along with the other AidData financial indicators, do not include internal national spending.

Sampling methodology and data selection

Data were compiled by AidData, an organisation that collects data on international development financing and categorises each project or flow into specific activities and sectors. Data are presented in constant US dollars (set at 2009 levels). Trends were based upon funds committed from 2000-2010 only to account for completeness and reliability concerns with earlier data. Additionally, for the purposes of this analysis, we only included donors for whom more than 95% of their projects/activities have received AidData activity codes.

References

Development Co-operation Directorate (DCD), 2008. DCD/DAC/STAT (2008)17/REV1. [Online]

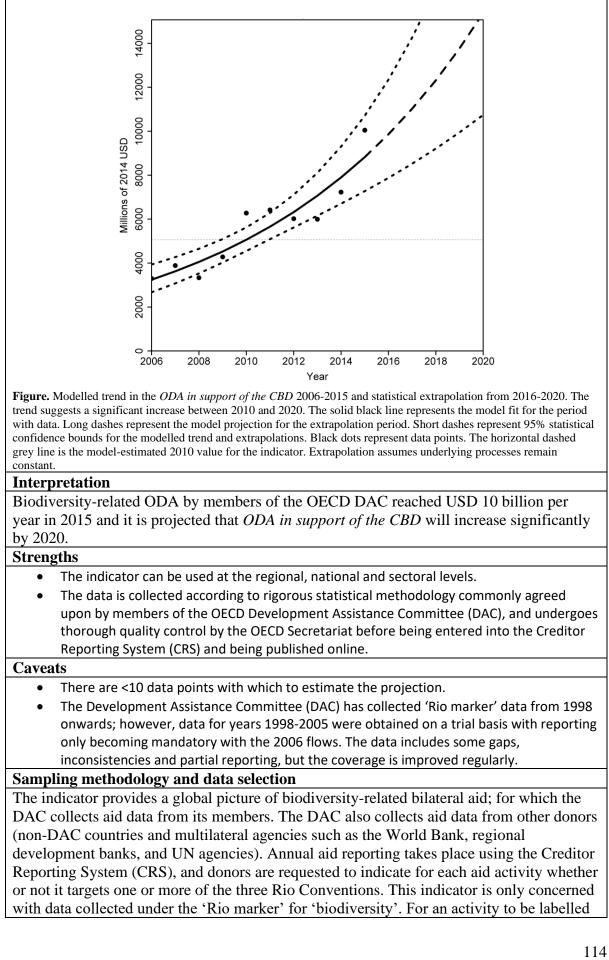
Available at:

www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=DCD/DAC/STAT(200 8)17/REV1&docLanguage=En

Official Development Assistance provided in support of the CBD objectives (\$)

Adequate access to resources is essential for effective implementation of the Convention on Biological Diversity (CBD). The *Official Development Assistance (ODA)* indicator tracks the transfer of bilateral aid to developing countries for the effective implementation of their commitments under the CBD, thus monitoring one component of resource mobilization for the Strategic Plan for Biodiversity 2011-2020.

Model fit



with this 'Rio marker' it must promote one of the three objectives of the CBD: the conservation of biodiversity, sustainable use of its components, or fair and equitable sharing of the benefits of the utilisation of genetic resources. When assigning the 'Rio markers' donors use the scoring system: 0 = Not targeted, 1 = Significant objective, 2 = Principal objective. Donors are also asked to report on the breakdown of their aid activities by sector, recipient country and region, income group, and aid instrument used (grants, loans, and equity investment). This means that the indicator data can be disaggregated to look at the breakdown of aid activities according to these different criteria, e.g. between sectors (i.e. forestry, agriculture, etc.). Note, however, that marker data do not allow exact quantification of aid allocation or spending on biodiversity. They give an upper-bound estimate of bilateral biodiversity aid commitments, and describe the extent to which OECD DAC members address the objectives of the CBD in their aid programmes. Historical data (1998-2015) were taken from the OECD DAC Creditor Reporting System (Feb 2017). Due to the issues with completeness of records 1998-2005, only data from 2006 onwards were used.

References

OECD, 2015. Biodiversity related official development assistance 2015. Available at: <u>https://www.oecd.org/dac/environment-development/Biodiversity-related-ODA.pdf</u> OECD, 2009. Measuring aid targeting the objectives of the Rio Conventions. Available at: <u>https://www.oecd.org/dac/stats/documentupload/49778699.pdf</u> Detailed activity level data are available online: http://oe.cd/RioMarkers

S3.1.2.1 Results of extrapolations of regional indicators

For a small selection of six indicators, trends to 2020 were extrapolated for each of the IPBES regions (Table S3.2).

Strat	Aichi	Indicator	Projected trend to 2020				
-egic Goal	Target Comp- onent		Global	Africa	Americas	Asia- Pacific	Europe & Central Asia
В	5.1	Area of tree cover loss (ha)	Significant increase	Significant increase	Non- significant trend	Non- significant inrease	Non- significant decrease
В	6.1	Marine Stewardship Council certified fisheries (tonnes)	Significant increase	Significant decrease	Significant increase	Significant increase	Significant increase
В	6.3	Marine trophic index	Non- significant decrease	Significant decrease	Non- significant decrease	Non- significant decrease	Significant increase
В	8.1	Pesticide use (tonnes)	Significant increase	Non- significant increase	Significant increase	Significant increase	Non- significant increase
С	11.3	Percentage of Key Biodiversity Areas covered by protected areas	Significant increase				
E	19.1	Species Status Information Index	Non- significant increase	Non- significant increase	Non- significant increase	Non- significant increase	Significant increase

Table S3.2 Selected indicator trends for different reegions

S3.2 Methods for literature search for assessment of progress towards Aichi Targets

ISIS Web of Knowledge was searched using key words (as 'topic') that were tailored for each Aichi Target. No data limit was set. Results were ordered by date (most recent first). The first 250 hits were evaluated (or all hits if the search returned fewer than 250. Relevant papers were selected; these excluded case studies (except in some cases of long longitudinal experiments), and papers addressing the global, regional, biome, ecosystem or, exceptionally, country level were prioritized. Several combinations of keywords were tested until a combination was found that resulted in sufficient papers.

Table S3.3 Search terms used for literature search for assessment of progress towards Aichi Targets.

Aichi Target	Search terms	No. hits	No. selected
1	'Aichi target 1' OR 'biodiversity' OR 'awareness' AND 'values' OR 'systematic review' OR 'synthesis' OR 'indicator' AND 'conservation' AND 'environmental education'	29	13
2	'Aichi target 2' OR 'biodiversity' OR 'conservation' OR 'review' OR 'systematic review' OR 'progress' OR 'indicator' AND 'national planning' OR 'development'		7
3	'Aichi target 3' OR 'biodiversity' OR 'conservation' OR 'review' OR 'systematic review' OR 'funding' AND 'sustainability' OR 'fisheries' AND 'agriculture' AND 'capacity building' OR 'harmful subsidies'	70	11
4	'Aichi target 4' OR 'stakeholders' OR 'red list' OR 'review' OR 'systematic review' OR 'CITES' AND 'ecological footprint' OR 'HANPP'	27	9
5	'Aichi target 5' OR 'meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' OR 'indicator' AND 'protected area' OR 'ecoregion' OR 'key biodiversity areas'	675	22
	'Aichi target 5' OR 'meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'forest loss' OR 'forest extent' OR 'deforestation' OR 'forest transition'	838	39
	'Aichi target 5' OR 'meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'mangrove loss' OR 'mangrove degradation' OR 'mangrove fragmentation	13	6
	'Aichi target 5' OR 'meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'savannah'	191	5

	'Aichi target 5' OR 'meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'semiarid ecosystem*'	24	4
6	'Aichi Target 6' OR 'systematic review' OR 'biodiversity habitat index' OR 'key biodiversity areas' OR 'species habitat index' AND 'marine invertebrates' OR 'aquatic plants'	4	0
	'Aichi Target 6' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'marine invertebrates' OR 'aquatic plants'	795	3
	'Aichi Target 6' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'fisheries' OR '*fishing' OR 'fishery'	3725	32
7	 'Aichi target 7' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'agriculture' OR 'aquaculture' OR 'forestry' AND 'biodiversity' OR 'land use' 	1441	31
8	'Aichi target 8' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'nutrient pollution' OR 'phosphorus' OR 'Nitrogen' AND 'biodiversity'	394	30
9	'Aichi target 9' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'invasive species' OR 'allien species' AND 'biodiversity'	244	38
10	'Aichi target 10' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'coral reefs' OR 'ocean acidification' AND 'climate change' AND 'biodiversity'	63	36
11	'Aichi target 11' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'protected area*' OR 'protected area system*' OR 'protected area coverage' OR 'area-based conservation' OR 'protected landscape*' OR 'protected seascape*' AND 'biodiversity' OR 'ecosystem services'	579	49
12	'Aichi Target 12' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'IUCN red list' OR 'CITES Appendices' OR 'endangered species' OR 'critically endangered species' OR 'vulnerable species'	747	12
	'Aichi Target 12' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'Red list index' OR 'Wild Bird index' OR 'Living planet index'	7	4
13	'Aichi target 13' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'cultivated crops' OR 'domesticated plants' OR 'cultivated plants' OR 'domesticated animals' OR 'farmed animals' OR 'landrace*' AND' genetic diversity' OR 'genetic erosion'	52	15
14	 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'aichi target 14' OR 'ecosystem services' OR 'biodiversity AND 'local communities' OR 'local people' OR 'indigenous communities' OR 'indigenous people' 	153	12

15	'Aichi target 15' OR 'meta-analy*is' OR 'metaanaly*is' OR 'review' OR 'systematic review' OR 'synthesis' AND 'ecosystem resilience' OR 'ecosystem conservation' OR 'ecosystem restoration' OR 'reforestation'	391	29
16	'Aichi target 16' OR 'Nagoya protocol'	110	12
17	'Aichi target 17' OR 'policy' OR 'national biodiversity strategy ' AND 'action plan'	15	15
18	'Aichi target 18' OR 'biodiversity' OR 'progress' AND 'indigenous and local communities' OR 'traditional knowledge	232	20
19	'Aichi target 19' OR 'biodiversity' OR 'progress' AND 'natural resources' OR 'science' AND 'trends'	157	14
20	'Aichi target 20' OR 'strategic plan' OR 'financial resources'	431	10

S3.3 Extended review of the Aichi Biodiversity Targets and Indigenous Peoples and Local Communities

Aichi Target 1: By 2020, at latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably.

Methods: The text below is based on a literature review using the following search terms as topics: ("Indigenous Community" OR "Indigenous Peoples" OR "Local Community" or "Aboriginal") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("environmental awareness" OR "values of biodiversity" OR "conservation awareness" OR "conservation outreach" OR "public environmental awareness" OR "environmental education" OR ("Aichi Target 1"). The search was run in Web of Science yielding 108 papers of which 55 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions).

IPLCs have played a crucial role in raising awareness of the diverse values of biodiversity from local to global scales (Sakakibara 2009; Bali & Kofinas 2014; Rathwell & Armitage 2016; Athayde 2017; Singh et al. 2017). They have substantially contributed to initiate, maintain and strengthen initiatives for communicating, educating and raising awareness about biodiversity at multiple levels (FPP & CBD 2016; Janif et al. 2016; Horton 2017; Veríssimo et al. 2018). These initiatives vary in format, extent and scope, including organization of cultural events and festivals (Cruikshank 1997; Langton & Rhea 2005; Singh Negi 2010; Fernández-Llamazares & Cabeza 2017), production of written and audiovisual materials (Iseke & Moore 2011; Bali & Kofinas 2014), and promotion of intercultural dialogue around the values of biodiversity (De Groot & Zwaal 2007; Rozzi et al. 2015; Herman 2016). Many of these actions have been channeled and orchestrated through IPLC organizations and networks, such as the International Indigenous Forum on Biodiversity (IIFB) and the Traditional Knowledge Information Portal (TKIP) of the Convention on Biological Diversity, both of which inform general audiences about IPLC views in relation to the global biodiversity agenda (FPP & CBD 2016). Increasing

presence of IPLCs on social media (e.g., Sandoval-Almazan & Gil-Garcia 2014; Carlson et al. 2017) is also contributing to give visibility to conflicts around biodiversity all over the world (Benyei et al. 2017; Örestig & Lindgren 2017).

IPLC-led awareness-raising campaigns often reveal conceptualizations of nature that differ substantially from Western epistemologies (Lewis & Sheppard 2005; Beckford et al. 2010; Brown 2013; Fernández-Llamazares & Cabeza 2017). While monetary valuation has become an important tool to raise awareness of biodiversity in Western contexts (e.g., de Groot et al. 2012; Lienhoop et al. 2015), there is well-established evidence that IPLCs have largely contributed to promote recognition towards the intrinsic values of nature, acknowledging it has an spiritual dimension for them (Jeeva et al. 2006; Clark & Slocombe 2009; Parotta & Trosper 2012; Chen & Gilmore 2015; Aniah & Yelfaanibe 2016). Research from all over the world indicates that many IPLC cultures are underpinned by eco-centric values and holistic nonmaterialistic worldviews (Kelbessa 2005; Mercer et al. 2007; Snodgrass et al. 2007; White 2010; Voeller 2011; Gratani et al. 2016). IPLC narratives on the environment often build on philosophical concepts such as the mutual reciprocity between humans and nature (Nadasdy 2007; Kohn 2013; Wall Kimmerer 2011), webs of relationality and kin (Salmon 2000; Viveiros de Castro 2007; Aiyadurai 2016), lack of a nature-culture divide (De La Cadena 2010; Caillon et al. 2017), promotion of relational approaches to nature (Kopenawa & Albert 2013; Comberti et al. 2015), as well as a powerful stewardship ethics (Dove 2011; Gammage 2011). IPLC communication strategies often emphasize the sentient nature of the land through which humans and non-humans derive agency (Brown 2013; Kohn 2013; Allison 2017). Several authors have argued that promoting recognition towards IPLC holistic and eco-centric values can facilitate novel ways of conceptualizing and achieving global sustainability (Jackson et al. 2008; Hawke 2012; Gratani et al. 2016; Herman 2016; Powys Whyte 2016).

The arts have been particularly conducive at celebrating IPLC values of biodiversity, relaying them to global audiences and inspiring social reflectivity about, and action towards, sustainability (Sakakibara 2009; Bali & Kofinas 2014; Rathwell & Armitage 2016; Athayde 2017; Horton 2017). The cultural manifestations of IPLCs, including songs, arts and place-based oral history, often transmit the idea that the relation between humans and nature should be one of respect, gratitude and reciprocity (Snodgrass et al. 2007; Clark & Slocombe 2009; White 2010; Herman-Mercer et al. 2016; Fernández-Llamazares et al. 2017). For instance, IPLC oral traditions have been recognized as central to revitalize biocultural diversity (Packer et al. 2007; Ryan 2015; Brown 2013; Herrmann et al. 2013). By integrating knowledge with emotions, IPLC cultural forms offer a platform to establish emotional connections with the landscape, helping to cultivate a sense of place (Cruikshank 2001, 2012, 2013; Sakakibara 2008; Fernández-Llamazares & Cabeza 2017; Singh et al. 2017).

Lack of awareness of biodiversity and its multiple values has been implicated amongst the main drivers of the current conservation crisis (Lunney 1998; Balmford et al. 2002; Buijs et al. 2008; Lindemann-Maties & Bose 2008; Snaddon et al. 2008). There is well-established evidence that many IPLCs currently face cultural and economic pressures that threaten their deep and intimate connections with the environment (Collings et al. 1998; Kishigami 2004; Godoy et al. 2005; Ford et al. 2006, 2007, 2010; Reyes-García et al. 2007, 2013, 2014; Luz et al. 2015, 2017). Erosion of knowledge about the natural world has been linked to growing isolation from it (Iseke & Moore 2011; Shen et al. 2012; Herman-Mercer et al. 2016; Tang & Gavin 2016). Rapid socioeconomic changes amongst IPLCs generally result in disconnection with biodiversity and decreased knowledge about its multiple values (Papworth et al. 2009; Kandari et al. 2014; Fernández-Llamazares et al. 2015, 2016). Educational programs and outreach

activities promoting ILK revitalization have been found to help avoiding shifts in IPLC values of biodiversity, building biocultural self-esteem and promoting intergenerational exchange of knowledge about biodiversity (Aikenhead 2001; McCarter et al. 2014; McCarter & Gavin 2014; Gavin et al. 2015; Tang & Gavin 2016; Wilder et al. 2016; López-Maldonado & Berkes 2017).

Awareness-raising initiatives narrowly framing biodiversity in monetary terms have often downplayed the importance of intrinsic, cultural and other non-economic values of biodiversity of crucial importance for IPLCs (Gómez-Baggethun & Ruiz-Pérez 2011; Christie et al. 2012; Kallis et al. 2013; Jax et al. 2013; Boeraeve et al. 2015; Pascual et al. 2017; Díaz et al. 2018). Monetary valuation of biodiversity and NCPs is increasingly emphasized in NGO awarenessraising texts and policy reports (Tangerini & Soguel 2004; Pinfold 2011; Brander & van Beukering 2013; WWF-Dalberg 2013), whereas the intangible benefits of biodiversity continue to be largely overlooked in communication strategies about biodiversity (Boeraeve et al. 2015; Hausmann et al. 2016). Similarly, advertisement campaigns by pro-environmental nature conservation NGOs have often used threatening messages to raise awareness of biodiversity (Vasi & Macy 2003; Weberling et al. 2011; Weinstein et al. 2015), often failing to capitalize upon IPLC cultural values and their own intrinsic motivation to conserve nature (Jim & Xu 2002; Spiteri & Nepal 2007; van der Ploeg et al. 2011; García-Amado et al. 2013; Hazzah et al. 2014). Similar patterns have been found in the context of climate change communication programs amongst IPLCs (Marin & Berkes 2013; Rudiak-Gould 2014; Fernández-Llamazares et al. 2015). To address some of these challenges, innovative art-based participatory methods are starting to emerge to better engage IPLCs in conservation and stimulate interest in biodiversity (Osnes 2013; Heras & Tàbara 2014, 2016; Bali & Kofinas 2016). Similarly, education programs integrating ILK in school programs are also playing a significant role in promoting awareness of the multiple values of biodiversity amongst IPLCs (Kimmerer 2002; Simpson 2002; Eder 2007; Castagno & Brayboy 2008; Cebrián & Noguera 2010; Glasson et al. 2010; Reyes-García et al. 2010; Ruiz-Mallén et al. 2010; McCarter & Gavin 2011, 2014; Hamlin 2013; Calderon 2014; Thomas et al. 2014; Abah et al. 2015; Mokuku 2017). Additionally, IPLCs are also engaging in numerous ecotourism initiatives worldwide, which largely contribute to raise awareness about biodiversity (Bookbinder et al. 1998; Stem et al. 2003; Lai & Nepal 2006; Rozzi et al. 2006; Stronza & Gordillo 2008; Espeso-Molinero et al. 2016; Bluwstein 2017; Mendoza-Ramos & Prideaux 2017), particularly among urban citizens, often inviting them to re-assess their lifestyles in relation to the environment (Fredrikson 2001; Cheng et al. 2014; Cheung & Folk 2014).

Aichi Target 2: By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems. Methods: The text below is based on a literature review using the following search terms as topics: "indigenous people\$" OR "traditional ecological knowledge" AND "policy influence". The string resulted in 409 search results in Topic search on Science Direct and SCOPUS with a subscription at McGill University, of which 61 were from 2017 and relevant to the topic. Additional papers were also selected from the authors' own literature database.

Past and current levels of species richness and density directly depend on ILK, however colonial and post-colonial regimens that have economically benefited from these improvements have systematically erased, neglected and minimized the importance of these local practices for environmental management (Toledo, 2013). Through the Aichi targets and

SDGs, UNEP aims to make amends for these historical damages and to bring human societies to heightened levels of appreciation, respect and inclusion of IPLCs environmental governance institutions. However, despite numerous efforts from IPLCs in communicating messages about the terms of their inclusion and environmental governance based upon reciprocity (Belfer et al., 2017; Raatikainen and Barron, 2017), little or no advancement has been achieved in that direction. For instance, although Standing Rock Sioux Tribe members have repeatedly tried to communicate the importance of their territory in maintaining water flows and local biodiversity levels, the US Government has prioritized the construction of a 1,200 mile long oil pipe that crosses not only sacred lands but also the Missouri River (Raffensperger, 2014). Another example of how IPLCs contribute to enhanced biodiversity levels is found in the African savannahs, where pastoralist communities have historically coexisted with wildlife and, through co-evolutionary practices, have promoted the expansion of pastures that not only benefit their herds but attract wild herbivores and carnivores leading to the great migration in the Trans-Mara protected land network (Hesse & McGregor, 2006). Despite the co-creation of this abundance, the Kenyan and Tanzanian pastoralist communities have been marginalized and forced to change their ancestral livelihoods in favor of industrial livestock production, justified under the umbrella of economic efficiency and development (Fratkin and Roth, 2005).

Many IPLCs around the world share situations of exclusion and inequality (Ban Ki-Moon, 2014, message on the International Day of the World's Indigenous Peoples; Suma and Grossman, 2017). In spite of this bleak reality, some IPLCs values have been mainstreamed into national and local policy, recognizing non-human actors as legitimate stakeholders with equal legal standing as human beings (Haraway, 2016). Thus, some countries have put together institutions that are in line with IPLCs world views and that recognize the rights of ecosystems to exist, reproduce, and thrive. Examples include the Ecuadorian and Bolivian Constitutions and the New Zealand's recognition of Te Urewa legal personhood. However, despite the merits of this "rights approach to nature protection", implementing it has proven difficult as ecosystems do not have a voice in courtrooms when their existence is at risk (McNeill, 2017; Temper and Martinez-Alier, 2016). While in some countries IPLCs have taken the lead and used these governmental efforts as an opportunity to mainstream their values, this has also proven contentious as power asymmetries within IPLCs and between IPLCs and government institutions has led to the imposition and reification of value systems (Jacobs et al., 2016; Bidder et al., 2016; Griewald et al. 2017). For example, Sumak Kawsay is a Quechua term that means "living well". In recent years the term has been appropriated as "buen vivir" to advance a political agenda not necessarily aligned with IPLCs core values (Perreault 2017).

Although IPLCs have contributed to our current levels of biodiversity and their contributions have been recognized in several occasions, our society does not value IPLCs enough for their past and continuing efforts (Belfer et al. 2017) and IPLCs rights are being trampled again and again in favor of so call development (Escobar, 2011). For example, IPLC environmental governance regimens are based in shared institutions which manage and maintain their collective territory (Trosper, 2009). The constant efforts of the Nation state to privatize common lands (Dell'Angelo et al. 2017) and the imposition of a western narrative of the commons as a failure and always imminent tragedy (Hardin, 1970) has led to the vanishing of common property regimens throughout our world (Raatikainen and Barron, 2017). Aichi target 2 does not address the most important and pressing issue relating IPLCs and biodiversity conservation: the protection and promotion of common property regimens throughout the world as means of re-claiming a lost relation between humans and our natural environment (Shiva, 1997; Torkar and McGregor, 2012). A shift from top-down

environmental policy to bottom-up inclusive socio-ecological policy requires: (i) the recognition of the importance of socially and historically contextualized scientific knowledge in the development and implementation of innovative environmental policy (Pascual et al 2016, Kolinjivadi et al, 2016); (ii) the expansion of our value system to include relational values along with utilitarian and non-utilitarian values in nature (Chan et al 2006; Kosoy and Corbera, 2010); and (iii) the inclusion of non-human stakeholders as legitimate actors in the socio-ecosystem (Saito, 2017; Culinam, 2011).

Traditional Forest Management: T'Souke Hills and Victoria City. For over 100 years, T'Souke Hills have provided water to the Greater Victoria Area, which has been owned and administered by the Capital Regional District (CRD). Despite past and current land management efforts to ensure constant water supply to the Greater Vitoria Area, recent studies show that water volumes will not be sufficient to cover demand in the near future (Smiley et al, 2016). Furthermore, studies on water quality changes associated with distribution of forest ages show that nitrogen exports increases with forest maturity leading to trade-offs in management between water quality and forest cover (Zhu and Mazumder, 2008).

T'Sou-ke is a First Nation in South Vancouver Island who have struggled to survive while their lands and resources have been taken away, first by the Crown and now by the Government in British Columbia. Despite these limitations, the T'Sou-ke Nation has developed a road map to achieve the sovereignty and security they require by focusing on three pillars: food sovereignty, energy sovereignty, and cultural sovereignty (T'Sou-ke Constitution). Access to their traditional territories and maintenance of their historical practices in these territories is one of the main pillars for the flourishing of this Nation. In particular, access to the T'Sou-ke Hills – or as they are locally known the Blue Camas Hills is a crucial step towards food and cultural sovereignty. Fire has been used to create patches in forested areas (Derr, 2014), enabling First Nations women access to the Blue Camas (*Camassia spp.*), which was one of the most important goods for trade along with salmon on the West Coast of North-America before the settlers took and controlled land for agriculture (Gritzner, 1994; Storm and Shebitz, 2006).

It is therefore imperative to tackle the conflict between securing long term access to drinking water to an ever-growing population in the Greater Victoria area while granting access and use rights to the T'Souke Nation over their territories. Hence, a novel agreement between the T'Sou-Ke Nation and the CRD is required, one that grants the T'Sou-Ke Nation rights to their ancestral lands while guaranteeing a shared responsibility for the provision of drinking water to not only the human population in greater Victoria but also to all other species, plant and animal, that share this common territory. Only the recognition of the Blue Camas Hills as a commons under T'Sou-Ke stewardship will lead to the long term maintenance and flourishing of this ecosystem and their derived ecosystem services including biodiversity.

Aichi Target 3: By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the Convention and other relevant international obligations, taking into account national socio economic conditions.

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous community" OR "indigenous people" OR "local community" OR "aboriginal" OR "farmer") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("subsidy"

OR "incentives") AND ("harmful" OR "negative impact") AND ("biodiversity" OR "ecosystem" OR "biodiversity loss") OR ("Environmental harmful incentives" OR "Environmental harmful subsidies" OR "Environmental harmful practices") OR ("Aichi Biodiversity Target 3"). The search was run in Web of Science yielding 59 papers of which 37 are relevant to the topic.

There is some evidence that by maintaining and integrating ILK into incentives for biodiversity conservation, IPLC might contribute to sustainable resource management. For example, in Northwest Yunnan, China, regions inhabited by IPLCs show lower biodiversity pressures and higher abundance of endangered musk deer (Moschus spp.) than other regions, with musk deer conservation in protected areas benefiting from traditional management practices (Li, Bleisch, and Jiang 2016). It is also growingly agreed that, by examining the different histories of fine-scale management practices in an area conservation planners could advance traditional management practices as a conservation strategy (Kay et al. 2017). For instance, past and current grazing practices have different effects over biodiversity, for which incentive programs related to grazing practices that consider a diversity of approaches could have higher conservation impact (Kay et al. 2017). Because local management practices might affect biodiversity and ecosystem services, there have been calls for incentives to induce people to reduce harmful effects on ecosystem services (Baskaran, Cullen, and Colombo 2009, Baskaran, Cullen, and Takatsuka 2009, Chaves and Riley 2001, Palm-Forster, Swinton, and Shupp 2017). It should be noticed, however that researchers have suggested that traditional management systems might need to be improved using scientific and institutional inputs to meet the challenges arising from local development and environmental conservation (Chandrasekhar et al. 2007). Finally, because the negative effects on biodiversity and ecosystem services by management practices are associated with locally non-adapted practices, such as those showing a lack of attachment with culture or beliefs related to biodiversity (Dominguez, Zorondo-Rodríguez, and Reyes-García 2010), the design of effective policies that will induce people to adopt more environmentally friendly practices needs to be grounded on the relative values attached by people to detrimental environmental impacts (Baskaran, Cullen, and Colombo 2009).

Non-socially and -ecologically tailored biodiversity management practices affect biodiversity and IPLC that depend on it for their livelihoods (Abdollahzadeh, Sharifzadeh, and Damalas 2016)(Ribeiro et al. 2014, Diaz et al. 2015, Roder et al. 2008, Acharya et al. 2015). For instance, in the Iberian cereal steppes economic incentives (e.g., decoupling payments from production) promote shifts from the traditional cereal-fallow-sheep system towards specialized livestock grazing systems, with consequent declines in land-use heterogeneity and associated biodiversity (Ribeiro et al. 2014). Additionally, traditional biodiversity management practices are often lost due to the introduction of western practices, as illustrated by the replacement of chinampas (raised beds) by subsidized plastic greenhouses in Mexico (Merlin-Uribe et al. 2013). In the same vein, environmentally harmful subsidies reduce the attractiveness and effectiveness of instruments that achieve 'no net loss' of biodiversity and reconciling nature conservation with economic development goals (Santos et al. 2015). For instance, harmful subsidies can affect habitat banking and tradable development rights and reduce the effectiveness of those instruments and their ability to ensure equitable allocation of the benefits and costs (Santos et al. 2015). Policies focusing on small-scale production, elimination of harmful subsidies, and redirection of resources allocated to operation costs have been proposed to address harmful effects of subsidies on IPLC (Sabau and Boksh 2017). The successful integration of IPLC and their management practices into conservation planning needs policy-makers willingness to accept changes and invest is such changes (von Haaren and Reich 2006, Richards 1996, Ribeiro et al. 2014). For instance, although participatory approaches have often been proposed for protected areas management, few governments commit the resources needed to effectively implement this approach, with the consequent loss of opportunity to provide positive incentives for conservation (e.g., ecotourism and community-based management conservation) (Richards 1996). The integration of IKP under a participatory approach has also been complicated by policies and land legislation which sent out negative or ambiguous signals to IPLC (Richards 1996). Incentive measures for biodiversity conservation cannot be evaluated and compared outside the context of institutional performance and relationships (Wells 1998, Diaz et al. 2015). The institutional framework includes a variety of organizations operating on different spatial scales, where IPLC are key stakeholders. Conservation and management strategies should promote IKP with likely positive conservation and social impacts. Better understanding of traditional management may open up new opportunities for biodiversity conservation in much wider tracts of unprotected and human-dominated lands (Li, Bleisch, and Jiang 2016). Nevertheless, increasing knowledge on the topic it is not enough to achieve conservation and social goals, if it is not understood and accepted by policy makers (Diaz et al. 2015, Richards 1996, Wells 1998). Neglecting the role of IPLC and their management practices reduces opportunities not only to achieve the conservation objectives but also to design effective incentives to integrate development and conservation.

Incentive-based conservation policies, such as the UNFCCC's REDD+, can also mobilize individual and collective behavior toward conservation-oriented actions (Ruiz-Mallén et al. 2015). Poor and small communities' livelihoods are perceived as improving when implementing REDD+ strategies and action plans although carbon sequestration and reforestation outputs are not successful at all (Holmes, Potvin, and Coomes 2017).

Chinampas substitution by greenhouses: The chinampas are highly productive, traditional wetland agricultural systems, which were able to feed most of the population in pre-hispanic times. Chinampas have been strongly substituted with plastic greenhouses for flower production. Although greenhouses are more profitable, the contribution of chinampas to ecosystem services cannot be substituted by greenhouses, as tree cover is lost, canals are filled and food is not provided (Merlin-Uribe et al. 2013). With the loss of local and traditional knowledge and practices the impacts in culture heritage and human well-being are also irreversible.

Aichi Target 4: By 2020, at the latest, Governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits.

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK) AND ["sustainable production" OR "sustainable consumption" OR ("use of natural resource*" AND impact) OR ("natural resource* use" AND impact) OR "ecological limit" OR "ecological footprint" OR "sustainable management plan" OR "fair trade"]. The search resulted in 223 entries in Scopus database accessed at UFSC, of which 130 were relevant to the topic. Additional papers were also selected from the authors' own literature database.

IPLCs offer many examples of how local economies built on ILK, including local institutions, practices, and values, can contribute to sustainable production and consumption. For example, small-scale agricultural systems, characteristic of many IPLCs, can help conciliate the goals of ensuring food security and conservation (Perfecto and Vandermeer 2010). In the same line, IPLCs' land use and territorial management plans can contribute to the use of natural resources within safe ecological limits. Many studies report IPLCs contribution to the sustainable production of natural resources, including plant and animal resources, as well as water (Schnegg and Linke 2016, Vos and Boelens 2014, Kahane et al. 2013, Andre 2012), energy (Parker et al. 2016, Pilyasov 2016, Baumert et al. 2016, Sawyer 2008) and even landscapes and ecosystems (Rebelo et al. 2011, Kimmel et al. 2010). IPLCs contributions to sustainable production have been reported in different environments, including mountains (Gratzer and Keeton 2017), pasturelands (Tessema et al. 2014), agricultural fields (Kahane et al. 2013, Schulz et al. 1994), forests (Hajjar 2015, Meyer and Miller 2015, Nuwer and Bell 2014) and environments for fisheries (Bravo-Olivas et al. 2014, Wiber et al. 2010).

The contributions of IPLCs to sustainable production and consumption follow direct and indirect pathways. When IPLCs are located in the core of productive systems using natural resources and manage them either for self-consumption or to supply external demands, their contributions are considered indirect. The use of natural resources in a sustainable way by IPLC has been documented for ligneous species in Burkina Faso (Ouédraogo et al. 2017), lianas in Brazil (Valente and Negrelle 2013), forests in Nepal (Cedamon et al. 2017), betel nut in Bangladesh (Islam and Nath 2014), edible insects in the Lake Victoria Basin (Okia et al. 2017), fisheries in New England (Tolley et al. 2015), crocodile eggs (Corey et al. 2017) and game animals in Papua New Guinea (Cuthbert 2010). Direct links with consumption are revealed when accessing issues such as fair trade and supply chains of biodiversity products (Burke 2012, Martins 2011). The analysis of supply chains of biodiversity products show how natural resources goes from production towards consumption; and discussions about fair trade deepen our understanding on social sustainability and economic sustainability of productive processes, and eventually show how consumers can be entailed in sustainable production. However, it would be desirable to add more on ecological sustainability into those frameworks.

IPLCs have also indirect contributions to sustainable production, for example as pivotal stakeholders in community-based natural resources management initiatives resulting in sustainable production (Virtanen '05). Examples of such initiatives include water management in Namibia (Schnegg and Linke 2016), pastoralism in Kyrgyzstan (Dörre 2015), forests in Nepal (Ojha 2014), cockle fishery in Ecuador (Beitl 2011), fisheries in Canada (Wiber et al. 2010), and fuelwood in South Africa (Kaschula et al. 2005). However, it is worth mentioning that a demonstrated understanding of whether such initiatives are within safe ecological limits is still limited to a few cases (e.g., Bravo-Olivas 2014 for coastal fisheries; Brown et al. 2011 and Faude et al. 2010 for forests; and Cuthbert 2010 for hunting).

The expansion of commodity frontiers driven by unsustainable consumption and production patterns exerts direct pressures both on biodiversity and on IPLCs (Orta and Finer 2010; (Moore 2000)). For example, IPLC around the world rely on small-scale farming and other uses of land and natural resources, which are often governed by customary systems of

common property (e.g. Cinner and Aswani 2007). In recent years, large-scale land acquisitions have drastically expanded around the world (De Schutter 2011) disproportionately affecting lands under common-property systems (Dell'Angelo et al. 2017). The impact of extractive industries, such as the oil industry, has also affected IPLC (Finer and Orta-Martínez 2010; Orta-Martínez and Finer 2010). In coastal areas, aquaculture expansion resulted in changes in local livelihoods of IPLC and local tenure rights (Benessaiah and Sengupta 2014; Joyce and Satterfield, 2010). Many studies have reported a lack of progress towards sustainable production when forms of production of natural resources to supply general consumption are imposed resulting in conflicts with IPLC. Examples include conflicts over the production of biofuels (Nesadurai 2013, Pilcher 2013, Amigun et al. 2011, Sawyer 2008), energy (Baumert et al. 2016, Andre 2012), mining (Ncube-Phiri et al. 2015), industrial development (Pilyasov 2016), agriculture (Kahane et al. 2013), and aquaculture (Benessaiah and Sengupta 2014, Joyce and Satterfield, 2010, Wiber et al. 2010). Conflicts with IPLCs have also been observed associated with the use of water (Vos and Boelens 2014), protected areas (Lepetu et al. 2009), forest management (Carter and Smith 2017, Grivins 2016, Ribot et al. 2010), marine resources (Rebelo et al. 2011, Thomson 2009), sports hunting (Yasuda 2011), and pastoralism (Yonas et al. 2013). The adoption of a neo-liberal agenda has been an important factor constraining progress related to sustainable production and IPLCs (Morgan and Cole-Hawthorn 2016).

IPLCs contribution to sustainable production and consumption are recognized in many situations (e.g., Bardsley and Wiseman 2016, Kahane et al. 2013, Queiroz 2011, Lane 2006), with much of this recognition depending on the understanding of local and traditional practices (Paletto et al. 2014, Kumagai and Hanazaki 2013). Several authors have studied the impact of changing livelihoods on IPLC, especially when they are presented with the pressures of external markets (Hanazaki et al. 2013, Reyes-García et al. 2005, Benz et al. 2000, Nolan and Robbins 1999).

Mamirauá Sustainable Development Reserve in the Amazon Várzea. The Mamirauá Sustainable Development Reserve (SDR) in the Amazon várzea is a co-managed initiative, involving local communities, the State and an environmental NGO that has been ongoing for more than two decades. It comprises an area of about 1,124,000 hectares and has more than 6,000 inhabitants. Successful fisheries management of pirarucu (Arapaima spp.) based on ILK and IPLC active participation was crucial to the recovery of this overexploited smallscale fishery (Castello et al. 2009). Moreover, management of this environment contributes to ecosystem services and products (Piñedo-Vasquez and Sears 2011) including other community-based forest enterprises, such as the production of logs and boards from flooded forests (Humphries et al. 2012). Since the creation of the reserve, constant efforts have been required to support this management, including flexibility when negotiating among parties (Lima and Peralta 2017), understanding of safe ecological limits (Castello et al. 2009), and social constraints on the structure of participation (Gillingham 2001). According to Lima and Peralta (2017), this kind of initiative requires sensitive agreement to embed the economy into the society and the society into the environment, which can only happen as a result of the connection between ILK and scientific knowledge. Although the SDR model seems promising, its implementation in other parts of Brazil is still limited. One unsuccessful example is found in the Areais da Ribanceira, southern coastal Brazil, where IPLC requested an SDR to conciliate sustainable use and conservation in a comparatively small area of less than 4,000 hectares and threatened by external pressures including industrial development, mining, and port expansion (Zank et al. 2011). A community project to create the SDR dates

from more than a decade ago, but a series of setbacks have occurred since then, and the area used by the local community is currently restricted to less than 30 hectares.

Aichi Target 5: By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous community" OR "indigenous peoples" or "local community" or "aboriginal") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("habitat conservation" OR "forest conservation" OR "high-biodiversity value habitat" OR "habitat loss" OR "forest loss" OR "habitat loss rate" OR "forest loss rate" OR "degradation rate" OR "fragmentation rate") OR ("Aichi Target 5"). The search was run in Web of Science yielding 121 papers of which 49 were relevant to the topic. Additional papers were also selected from the authors' own literature database. Authors considered mainly articles dated from 2010 forward, except when any subsequent article provided relevant information.

IPLCs have long been referred as contributing to the maintenance of natural habitats worldwide (Smith and Wishnie 2000). In the last decades, a growing body of literature has provided evidence that IPLC can contribute to forest conservation by slowing down global deforestation and degradation rates (Blackman et al. 2017; Nolte et al. 2013; M. Graziano Ceddia, Gunter, and Corriveau-Bourgue 2015). Forest condition is found to be improved in IPLC recognised lands even when compared with Protected Areas (PA) (Porter-Bolland et al. 2012). Although we have less information for marine habitats, there is also evidence that IPLC contribute to the conservation of, for example, coral reefs through traditional management (Williams et al. 2008; Busilacchi et al. 2013). The significant conservation success and the maintenance of sustainable livelihoods by IPLC is achieved through the integration of ILK in management practices that are aligned with conservation goals (Brooks, Waylen, and Borgerhoff Mulder 2012; Gadamus et al. 2015; Reyes-García et al. 2013). Some of these practices include the establishment of different degrees of forest and species protection (Mir and Upadhaya 2017; Camacho et al. 2012). For example, sacred forests where no extractive activities occur- are common in IPLC lands and allow the maintenance of forest cover and structure (Assefa and Hans-Rudolf 2017: McPherson et al. 2016). The existence of taboos can also protect some species (Colding and Folke 2001; Lingard et al. 2012). Other levels of protection and extraction can also contribute to the sustainable use and conservation of habitats (Guèze et al. 2015). For instance, in the Atlas Mountains, Morocco, forests under the agdal system, i.e., areas where access rights and uses of natural resources are governed by local norms, show lower degradation levels than areas outside this system (Hammi et al. 2010). Likewise, the traditional practice of selectively cutting trees or harvesting plant parts minimizes forest disturbance (Rodenburg et al. 2012). Many IPLC also limit the exploitation of resources for certain periods of time or seasons to ensure the maintenance and natural recovery of ecosystems, including forest areas, natural pastures or river sections (Camacho et al. 2012; Khan et al. 2014). Other IPLC provide examples of good practices that support forest regeneration or species dispersal as a conservation strategy. For example, in Philippines, the inhabitants of Ifugaos integrate different management techniques such as assisted natural regeneration, watershed rehabilitation, soil management, and agroforestry, to guarantee forest conservation and their subsistence (Camacho et al. 2012). IPLC also possess several techniques that limit soil erosion and land degradation, as those

developed by pastoralists along the years, such as mobility, herding, corralling, grazing reserves, and fire management (Assefa and Hans-Rudolf 2017; Seid, Kuhn, and Fikre 2016). Finally, IPLCs help maintain elements of the multifunctional landscape (i.e., agroforest and other mixed tree cover) that increase the degree of integration of PAs on multifunctional landscapes (Dewi et al. 2013). The extent to which these IPLC traditional practices can contribute to natural habitats conservation will depend upon the persistence of local management systems and the regulatory conditions that support them. More and more, traditional IPLC lifestyles are being rapidly eroded by the increasing demand of markets and modernity, which pull IPLC to participate in the market economies and adopt new systems that do not avoid habitat loss (Terborgh and Peres 2017; Michele Graziano Ceddia and Zepharovich 2017).

PAs are the primary strategy for addressing degradation, fragmentation and loss of both terrestrial and marine natural habitats at global level. Strict protected areas have often been insensible to IPLC's needs, even causing IPLC's exclusion (Lele et al. 2010) and leading to deep social conflicts, marginalization and accentuated poverty as well as to negative consequences for biological conservation (Adams and Hutton 2007; Dowie 2009). As social injustice towards IPLC perpetrated by these strict conservation approaches and the inefficiency of government agencies to exercise adequate stewardship began to be more visible at the international level, the classic conservation model has evolved towards more participatory management and inclusive conservation approaches. This conceptual change involved decentralization of power from governments to IPLC to manage natural resources through community-based conservation management initiatives (Berkes 2010), often accompained by the attribution of local and community rights over natural resources (Sayer, Margules, and Boedhihartono 2017), and the recognition of the importance of multifunctional landscapes (Dewi et al. 2013). The shift has implied a change in the conceptualization of natural habitats (Reyes-Garcia et al. 2013). Recognizing IPLC land rights have increased in mean family income and slowed down habitat loss in some areas (Michele Graziano Ceddia and Zepharovich 2017; Chen et al. 2012). In forests, for instance, sustainable management of non-timber forest products, e.g., orchids, guarantees forest conservation and generates socioeconomic benefits for IPLC (Cruz-Garcia et al. 2015). Hundreds of community forest enterprises led by IPLC in Mexico have successfully managed forests collectively by maintaining the ecosystem functions while providing opportunities for local development (Villavicencio Valdez, Hansen, and Bliss 2012). Decentralization of management, however, also leads to mixed results due to prevalent expert-based decisions and undemocratic and unequitable practices (Brown 2003; Ribot, Agrawal, and Larson 2006). Political will to enforce these practices and resist to market pressures is often missing, as it happens in the case of communal reserves that have little public visibility and, therefore, few advocates apart from the local residents who are likely to be disempowered (Terborgh and Peres 2017).

Nevertheless, policy makers have recognized the integration of ILK in conservation initiatives as useful to deter biodiversity loss (Brooks, Waylen, and Borgerhoff Mulder 2012). Such place-based knowledge, beliefs and practices also play a key role in enhancing local people's adaptive capacity to global environmental changes (Ruiz-Mallén and Corbera 2013). Nowadays, recognition of the legitimacy and effectiveness of many indigenous and other customary institutions for conservation of habitats and species have resulted into a specific type of governance of protected areas relying on IPLC. In this sense, some territories and areas conserved by IPLC (ICCAs) have been officially recognized as protected areas (Dudley 2008). Incentive-based conservation policies, such as the UNFCCC's REDD+, can also mobilize individual and collective behavior toward the formalization of conservation-oriented

actions (Ruiz-Mallén et al. 2015). Poor and small communities' livelihoods are likely to improve when implementing REDD+ strategies and action plans, although carbon sequestration and reforestation outputs may not be successful at all (Holmes, Potvin, and Coomes 2017). Unresolved contradictions between national legislation and indigenous rights as in the case of non-timber forest products in some Andean countries, for example, makes it difficult to regulate for sustainable use and management (de la Torre et al. 2011). These tensions continue feeding current debates between legal pluralism and legal centralism in conservation, in which equity and justice are central issues, and that directly influencing IPLC livelihoods and wellbeing (Boedhihartono 2017; Albano, van Dongen, and Takeda 2015).

Communities' Relations with Their Forests in Indonesia. Indonesia, as other developing countries, has a rapidly expanding economy and population growth. These processes are placing increasing pressure on natural resources and forests and causing conflicts between economic development and traditional values. Nevertheless, there are good practices among IPLC that contribute to natural conservation and preservation of local livelihoods. For example, the Baduy people of West Java are a community strongly dependent on their traditional belief systems and complex land management practices (Boedhihartono 2017). They possess strict beliefs and norms with supporting taboos that mediate their daily behavior regarding farming, extraction of natural resources, and other nature-related activities. Some of the Baduy traditions forbid changes to their landscape and the ecosystem, such as levelling the land to make houses or for irrigation, dams, and waterholes. Also, they protect sacred forests patches where they believe their ancestors reside by offering "sacred" swidden rice to maintain and reinforce their links with their ancestors. The strong maintenance of such unique traditional governance system currently allows the conservation of natural forests and biodiversity within the Baduy landscape.

Aichi Target 6: Sustainable fisheries [LA: Victoria Reyes-García, CA: Margarita Lavides, Reviewers: Nadav Gazit, Eleanor Sterling]

Target: By 2020, all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits.

Methods: The text below is based on a literature review using the following search terms as topics ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK) AND ("fisheries management" OR "sustainable fisheries" OR "Aichi Target 6"). This resulted in 300+ search results in Topic search on Web of Science, of which 130 were relevant to the topic. Additional 118 papers were also selected from the authors' own literature database.

ILK has contributed significantly to fisheries science or informed fisheries management (e.g., McMillen et al., 2014; Thornton and Scheer 2012). For example, fishers' knowledge has been used to map historical spawning grounds of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*) in the Gulf of Maine (Ames et al. 2000; Ames 2004; 2007), to understand the structure, ecology and use of seascape in south-eastern Australia (Williams & Bax 2007), to assess ecological and socioeconomic sustainability of reef fisheries in Sabah (Teh et al 2005), and to document long term reef fisheries trends in Malaysia (Teh et al 2007), in Seychelles (Daw et al 2011), in the Red Sea (Tesfamichael et al 2014), and in the Philippines (Green et al 2002; 2003; 2004; Mualil et al 2014). At the species level, fisher's ILK has also

been used to document long term changes in catch per unit effort (CPUE) of the Atlantic cod (Gadus morhua) and lumpfish (Cyclopterus lumpus) roe fishery in Newfoundland (Neis et al 1999); to reconstruct the historical distribution of local brown trout populations in Northern Sweden (Spens 2001); to describe the biology and environment of the Greenland halibut (Reinhardtius hippoglossoides) and its historical fisheries in Gulf of St. Lawrence (Camirand et al 2001); to detect changes in the abundance of yelloweye rockfish in British Columbia (Eckert et al 2017), or to assess the status and cultural importance of sawfish in Guinea-Bissau (Leeney & Poncelet 2013). Studies drawing on *ILK have also been instrumental in identifying* marine fish species at risk of extinction and the implication of such change for policy and management. Such studies have documented the declining status and history of Chinese bahaba (Bahaba taipingensis) (Sadovy & Cheung 2003); estimated trends of population decline in long-lived marine species in the Mediterranean Sea (Maynou et al 2011); revealed the local disappearance of bumphead parrotfish (Bolbometopon muricatum) in Fiji (Dulvy & Polunin 2004) and the Philippines (Lavides et al 2016; Lavides et al 2010); uncovered severe declines in the abundances and catches of Dungeness crab fishery in Pacific Canada (Ban et al 2017); assessed fish diversity changes in Mediterranean Sea (Azzurro et al 2011) and Gulf of Cadiz (Coll et al 2014); and reassessed the conservation status of Gulf grouper (Mycteroperca jordani) in Gulf of California (Saenz-Arroyo et al 2005a; 2005b), of sharks in United Arab Emirates (Jabado et al 2014), and of marine species at risk in Colombia (Castellanos-Galindo et al 2011). IPLC have also induced recovery, conservation and sustainability of marine and freshwater fisheries and ecosystems around the world, including Brazil (Campos-Silva & Perez 2016; Kalikoski & Vasconcellos 2007); Northern Australia (Phelan 2007); Torres Strait, Australia (Mulrennan 2007); North Lombok, Indonesia (Satria 2007); Mekong River, Laos (Baird 2007); Bangladesh (Sultana & Thompson 2007); Baja California, Mexico (Kuyuk et al 2007), Alaska (Fall et al 2010), Western Solomon Islands (Aswani 2011), and Belize, Mexico, Chile, Turkey, Morocco, The Gambia, Vanuatu, Indonesia, Philippines and Fiji as described in The Equator Initiative (UNDP 2017). The most ground-breaking contribution of IPLC has been in promoting ideas around "Nature's Rights" based on ILK system, which include fisheries conservation and management (Burdon 2012; Mihnea 2013; Sheehan 2014; Gordon 2017). While there is no empirical study analyzing whether the legislated Rights of Nature actually contribute to fisheries conservation and sustainability, it is argued that such ideas have leveraged positive impacts and have influenced policy at various levels, seen in examples in Ecuador, Bolivia, New Zealand, India and parts of United States (Burdon 2012; Mihnea 2013; Sheehan 2014; Gordon 2017).

IPLCs are highly reliant on marine ecosystems, and especially fisheries, for livelihood and cultural purposes (FPP 2016); IPLCs seafood consumption per capita is 15 times higher than the consumption of other populations in the same country (Cisneros-Montemayor *et al* 2016). Consequently, IPLC are disproportionately affected by unsustainable fishing practices, which particularly affect women who constitute nearly 90% of the post-harvest labor sector and are generally responsible for household food security (FPP 2016)(Cabral and Alino 2011; Babai et al. 2015). Some management policies have tried to address the issue. For example, the UNDP-GEF Equator Initiative recognizes 15 community-based marine initiatives that support sustainable fisheries while strengthening the resilience and well-being of each community (UNDP 2017). In the same line, the Ecotipping Points Project assembled a collection of 14 success stories, in which IPLCs overcame crisis to achieve sustainable fisheries, ensuring a high quality of life for everyone in the community (http://ecotippingpoints.org/index.html). A good example of a successful fishery based on ILKP is Maine's soft shell clam fishery (Berkes *et al* 2000) which integrates informal local knowledge and formal scientific information generated locally (Hanna 1998). Similarly, the mix of traditional and new knowledge of the

cai-aras and *caboclos* (two groups of mixed-race rural people of Brazil) increases the resilience of their social-ecological systems by combining adaptations from two different cultural traditions, Amerindian and European (Begossi 1998). It is important to develop locally grounded indicators that reflect local values and perspectives for human and environmental well-being (Sterling et al. 2017).

UNESCO supports the inclusion of ILK in fisheries science & management through its Coastal Marine Programme which evolved into Coastal Regions & Small Islands Platform. In 2002, UNESCO established the Local and Indigenous Knowledge Systems Programme focusing on empowering local knowledge holders in biodiversity governance by strengthening collaboration among local communities, scientists and decision-makers and enhancing knowledge transmission (Haggan et al 2007). UNDP-GEF, through The Equator Initiative supports IPLCs across the globe. In 2010, CBD established the Sustainable Ocean Initiative which empowers IPLCs for the implementation of SDG14 by providing a holistic and strategic framework through which to catalyse partnerships, build on lessons learned and knowledge gained, and facilitate improved coordination and two-way dialogue to address the capacity needs to support countries in their efforts to achieve the ABTs in marine and coastal areas including for sustainable fisheries management (CBD 2017). Local & international NGOs such as the IUCN (2016) or the ICCA Consortium are also at the forefront of supporting and recognizing the role of IPLCs not only in conservation but in sustainable fisheries. NOAA, for river herring fishery (NOAA 2013); Canadian government, for yelloweye rockfish fishery (Frid et al 2016), South Pacific Regional Environment Programme (SPREP 2017) and other government organizations across the globe are also in various stages of supporting and recognizing the role of IPLCs in sustainable fisheries. Similarly, the academe including, but not limited to, St. Mary's University through its Community Conservation Research Network (CCRN 2017); professional organizations like Ecological Society of America (Ford & Martinez 2000), and academic journals like Ecology and Society (2017) support and recognize the value and benefit of IPLCs including their knowledge-practice-belief systems for ecosystem-based management of sustainable fisheries.

Text Box: Community-based management of freshwater Arapaima fishery (Campos-Silva & Peres 2016): With growing market demand and technological innovation, large-scale commercial fishing pressure on Amazonian fish stocks has been escalating since the early 1960s. This fueled the emergence of community-based management initiatives, whereby fisherfolk began to restrict access of large commercial fishing boats into lakes near their communities. These initiatives, whenever they can be formalized, have been variously referred to as 'Fishing Accords' between subsistence and commercial fishing interests and have had a strong effect on local fisheries management. In 1993, government agencies legally sanctioned these local agreements as a formal fisheries management tool, which has since become a powerful strategy to prevent overexploitation of important fish species. Since 1999, such fishing accords, based on a strong social organization movement, paved the way to the development of a promising community-based management system focused on the exploitation of arapaima or pirarucú (Arapaima gigas, Arapaimidae), a target species of marked importance in Amazonian history and prehistory. This community-based resource management program induced stock recovery of the world's largest scaled freshwater fish, providing both food and income. Stock assessment data were analyzed over eight years and examined the effects of protected areas, community-based management, and landscape and limnological variables across 83 oxbow lakes monitored along a ~500-km section of the Juruá River of Western Brazilian Amazonia. Patterns of community management explained 71.8% of the variation in arapaima population sizes. Annual population counts showed that protected lakes on average contained 304.8 (\pm 332.5) arapaimas, compared to only 9.2 (\pm 9.8) in open-access lakes. Protected lakes have become analogous to a high-interest savings account, ensuring an average annual revenue of US\$10,601 per community and US\$1046.6 per household, greatly improving socioeconomic welfare. Arapaima management is a superb window of opportunity in harmonizing the co-delivery of sustainable resource management and poverty alleviation

Aichi Target 7: By 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "IPLC" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK) AND ("agriculture" OR "aquaculture" OR "forestry" OR "silvicultue" "Aichi Target 7") AND ("sustainability" OR "management" OR "biodiversity"). The string resulted in 656 search results in Topic search on the Core collection of the Web of Science with a subscription at The University of Texas, of which 49 were directly relevant. Additional papers were selected from the authors' own literature database.

IPLC's are important natural resource users and managers. Agroforestry (Zomer et al. 2009), community forestry (Gbedomon et al. 2016) and aquaculture initiatives (Le Gouvello et al. 2017) show particular promise for conserving local biodiversity stocks. Locally controlled resources used for local benefit also provide sustainable economic opportunities while incorporating community values into management practices (Claire and Segger 2015)(Oldekop et al. 2016). For example, restoration of a lake and wetland system in Southwestern Australia has revived traditional eel aquaculture, has had positive socio-culture impacts, and has improved local economies and aquatic habitats (Rose, Bell, and Crook 2016).

Agroforestry can add an additional dimension to IPLC livelihood strategies and provide ecosystem, social, and economic services (Boffa et al. 2000, Wiersum 2004, Padoch et al. 2002, Michon 2015). Common traditional resource management strategies include succession management, resource rotation, and multiple species management, often practiced simultaneously or sequentially (REF). With appropriate local oversight and resource use agreements, these kinds of practices can conserve local biodiversity and generate sufficient resources to maintain modest livelihoods, particularly in tandem with other sources of income such as tourism and NTFP markets (Berkes and Davidson-Hunt 2006; Gbedomon et al. 2016). Similarly, traditional agricultural practices help to conserve agrobiodiversity, maintain reservoirs of genetic diversity. and help retain ILK. Landscape scale agroecosystems with numerous small plots of highly diversified crops can lead to increased food security, improved health, and reduced risk for farmers (Johns et al. 2013). Areas associated with spiritual or sociocultural activities can provide essential refugia and habitat for diversity. In Ghana, traditionally managed sacred (conserved) forests in a patchwork agricultural landscape are more biodiverse than adjacent state managed forest reserves (Boadi et al. 2017), and forests dedicated to deities in the Himalayas reinforce strict taboos on resource extraction (Negi 2010).

IPLC management strategies respond to shifting social and economic pressures at multiple time scales and there are many contemporary obstacles to the sustainable management of natural resources. Colonial legacies and market forces have encouraged local economies based on natural resource extraction leading to environmental degradation, decreased biodiversity, and increased rates of environmental fragmentation (REF). Corruption and inept governance, particularly in developing countries, have consistently worked against interests of both conservation and IPLC resource management (REF). Climate change is an additional destabilizing complication to previously established and sustainable resource management practices (Hagerman et al. 2012; Singh et al. 2016). Therefore, IPLCs sustainable management practices must not be assumed from static data points, but must be paired with regular monitoring by internal and external socio-ecological metrics (Montova and Young 2013). Aboriginal residents along the East Alligator River in Northern Australia make tradeoffs between economically viable activities such as tourism and traditional activities such as hunting. These tradeoffs have enabled communities to retain biocultural knowledge and other traditional practices such as fishing and agriculture, and reduced hunting pressures have allowed species such as the saltwater crocodile to recover, although sport fishing has increased pressure on local fisheries (Ligtermoet 2016). Despite optimistic pilot program outcomes, there remain few IPLC controlled economic development programs (Lawler and Bullock 2017). For example, policy reforms in the Western Amazon have been promoting smallholder managed, sustainable forestry ideology for twenty years, but have failed to provide sufficient oversight to eliminate illegal logging nor provide necessary incentives for alternative economic activities at local or industrial scales in many areas (Pacheco et al. 2016). Furthermore, protected area management that does not incorporate IPLC livelihood needs can have negative environmental and local health outcomes, such as increased resource poaching and restricted traditional diets (Sylvester, Segura, and Davidson-Hunt 2016).

Community based conservation is one of the most effective means of sustainably using, managing, and conserving natural resources. Unfortunately, economic pressures often act against community organizing efforts. Degraded wetlands in Uganda provide an affordable and accessible landscape for smallholder farming and cattle ranching, activities that generate the income necessary to pay for school fees, medical care, and other economic necessities that traditional subsistence activities have not yet been able to generate (Barakagira and de Wit 2017). Interventions aimed at improving access to social services and economic institutions can have greater land-management impacts than those aimed at conservation or resource productivity alone (Bene and Friend 2011). Farmers that begin with the highest levels of species heterogeneity in their forests, farm plots, and fisheries have the greatest economic and ecological resiliency to climate instability (Altieri and Koohafkan 2008), but are often farthest from markets and other social services. Socioeconomic policies that promote crop diversification and reduce urban sprawl may be helpful in reducing some of these pressures (Biasi et al. 2017). Modifications to local pricing for natural resource commodities can improve both social and conservation indexes in areas with heterogeneous

socio-economic landscapes. Agricultural modernization often destabilizes these sustainable landscapes. Numerous economic and social pressures exist to encourage modernization at great personal expense (and debt) to traditional farmers. In Central Mexico, traditional (and sustainable) agricultural techniques are also associated with negative social stereotypes such as being lazy and uneducated (Fajardo et al. 2016). Adverse perceptions of traditional practices combined with policies that do not protect sustainable smallholder practices work against IPLC's in the absence of other institutional support. The downgrading of protected status for multiuse protected areas and the shrinking borders of Indigenous territories present an uncertain future for IPLC led resource management. In 2012 Brazil reduced the size of the Amazon National Park, an area already threatened by colonial agricultural policies and cattle ranching, by 47,080 hectares for hydropower generation (Laue and Arima 2016). In 2017, Bolivia reduced of protections in the Isiboro-Sécure National Park and Indigenous Territory, the ancestral lands of four Indigenous groups and national biodiversity hotspot, in order to build a road through the center of the park (Fernández-Llamazares et al. 2018). In the same year, The United States reduced two national monuments with important ecological and spiritual ties to five indigenous groups, for uranium mining (National Monument Review 2017). Community forestry programs still struggle with equitable access and fair governance procedures (Leisher et al. 2016). In the Wari-Maro Forest Reserve in Benin, residents complained of decreased living standards after the implementation of a participatory forestry project that neglected to consult with local community members on its goals or design (Gandji et al. 2017). In Nepal, power disparities between user groups have resulted in local elites accessing to forest resources while closing or restricting access to other community members. In these scenarios, forest conditions may improve, but at a cost to poorer members of area IPLCs (Thoms 2008). Addressing women's forest-use needs and including women in conservation efforts is another key component in effectively meeting global conservation and development goals (Wan, Colfer, and Powell 2011). This emphasizes the necessity for resource use policies that allow for traditional health and other dietary practices that may be overlooked by male-dominated governing bodies in IPLC (Colfer and Minarchek 2013).

Effective multiscalar governance is still needed to support sustainable economic and subsistence activities such as forestry, agriculture, and both fresh and marine aquaculture. With few exceptions, inland, freshwater aquaculture is excluded from discourse on biodiversity and rural development policy (Bene and Friend 2011), and both fresh and marine water resources (Le Gouvello et al. 2017) need more recognition in policy and development agendas as part of sustainable IPLC livelihoods (Ligtermoet 2016; Lynch et al. 2017). In rural IPLC managed landscapes, biodiversity, ecosystem productivity and social wellbeing are linked and must be treated as interrelated systems by policy makers and development feedbacks is still lacking in much of the world to connect IPLC's and multi-scale decision makers. Economic and environmental policies that effectively promote simultaneous social wellbeing and conservation of biodiversity is still lacking for most IPLC's (Caillon et al. 2017). When IPLC's are empowered to increase prices of locally exported, high value, goods such as timber or specialized food crops, IPLC's see more equitable benefits and livelihood improvement activities as seen in Nepal (Dhakal and Masuda 2009). Strategic market

controls, combined with resource management programs, have positive environmental outcomes (Dhakal and Masuda 2009; Johns et al. 2013). To meet the Aichi 2020 target, stronger policies protecting the rights of IPLC to collectively manage lands used for traditional livelihood practices, while also promoting the development of non-extractive economies, are essential. This includes a "right to food framework" in all protected area management, and participation by IPLCs at multiple scales of resource governance.

Aichi Target 8: By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.

Methods: The text below is based on a literature review using the following search terms as topics: ("Indigenous Community" OR "Indigenous Peoples" or "Local Community" or "Aboriginal") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("pollution" OR "pollutant" OR "contamination" OR "contaminant" OR "eutrophication" OR "excess nutrients") OR ("Aichi Target 8"). The search was run in Web of Science yielding 464 papers of which 212 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions.

Many IPLCs are limiting local and global levels of pollution through the maintenance of traditional agricultural practices with minimal use of chemical products such as pesticides or fertilizers (Altieri & Toledo 2011; Dublin & Tanaka 2012; Wezel et al. 2014; FPP et al. 201). For example, organic farming is an integral part of the food production systems of many IPLCs (Grossman 2003; Moreno-Peñaranda & Egelyng 2008; Huaman 2014). Many IPLCs apply natural pest control (Altieri 2004; Moshi & Matoju 2017), given that modern pesticides are often incongruent with traditional IPLC worldviews (Kayahara & Armstrong 2015; Morrison et al. 2015). Similarly, IPLCs traditional management practices include different remediation techniques (e.g., phytoremediation) to restore landscapes affected by pollution (Sistili et al. 2006; Pacheco et al. 2012; Sandlos & Keeling 2016). Many IPLC landscapes, including ICCAs and sacred sites, also contribute to pollution buffering and nutrient cycling (Ulrich et al. 2016; Vierros 2017). Additionally, the intimate connection that IPLCs maintain with their local ecosystems, through local observations and intergenerational transmission of ILK, puts them in a suitable position to closely monitor, map and report the expansion of pollution, e.g. in water bodies (Sardarli 2013; Bradford et al. 2017; Rosell-Melé et al. 2018). For instance, in many indigenous worldviews, water is a spiritual resource (e.g., the lifeblood of Mother Earth) that must be respected and kept clean from any pollution (Mascarenhas 2007; Collings 2012; Weir et al. 2013; Morrison et al. 2015). Given that pollution poses important threats to IPLCs' cultures and health (e.g., Orta-Martínez et al. 2007, 2017; Kelly et al. 2010; Harper et al. 2011; Huseman & Short 2012; Nilsson et al. 2013; Jiménez et al. 2015; Bradford et al. 2017), different IPLC are engaging, or even initiating community-based participatory research and monitoring of pollution and ecosystem health to collect evidence to defend against the threats towards their livelihoods (Deutsch et al. 2001; Suk et al. 2004; Berkes et al. 2007; McOliver et al. 2015; Benyei et al. 2016). For example, IPLCs in Canada contributed extensively to the creation of international conventions to reduce global levels of Persistent Organic Pollutants through the Northern Contaminants Program (Downie & Fenge 2003; Van Oostdam et al. 2005). There is well-established evidence of a rising trend towards organized IPLC resistance against polluting activities, e.g. oil extraction and mining (Orta-Martínez & Finer 2010; Veltmeyer & Bowles 2014; De La Cuadra 2015; Temper et al. 2015).

IPLC movements against pollution, including litigation to hold polluters to account, are gaining prominence all over the world (Martínez-Alier et al. 2010; 2014; 2016; Petherick 2011; Benyei et al. 2016). Mainly through global citizen action, social mobilization and capitalizing on modern technologies, the struggles of IPLCs against pollution have attracted global attention and support, helping to raise societal awareness of the magnitude of this phenomenon (Earle & Pratt 2009; Lorenzo 2011; Paulson et al. 2012; Sikor & Newell 2014; Pearce et al. 2015; Januchowski-Hartely et al. 2016).

It is well-established that IPLCs are confronted by the pressing and increasing threat of pollution (Ostertag et al. 2009; Curren et al. 2014, 2015; Santos & Nóbrega Alves 2016). Indeed, the impacts of pollution upon IPLCs have been extensively documented in many parts of the world, most notably in the Arctic (Dudarev 2012; Paunescu et al. 2013; Binnington et al. 2016; Krümmel & Gilman 2016) and the Amazon (da Silva Brabo et al. 2000; Peplow et al. 2007; Henessy 2015; Rosell-Melé et al. 2018), and to a lesser extent, in Australia (van Dam et al. 2002; Franklin et al. 2016; Williams et al. 2017) and South-East Asia (Siddiqui 2011). IPLCs are disproportionally affected by the impacts of pollution, because they rely on their immediate environments (e.g., water streams, local resources) for meeting their direct livelihood needs (Suk et al. 2004; Nguyen et al. 2009; Hoover et al. 2012; Orta-Martínez et al. 2017). Different works have shown that pollution directly affects the health and well-being of many IPLCs (Gracey & King 2009; Mapani et al. 2010; Valera et al. 2011; Dudley et al. 2015), as well as their cultural integrity (Appleyard et al. 2001; Tian et al. 2011; Pufall et al. 2011). Exposure to pollution amongst IPLCs often comes through the consumption of traditional wild foods (Curren et al. 2014; Russell et al. 2015; Ullah et al. 2016), obtained through hunting (Hlimi et al. 2012; Bordeleau et al. 2016; Lyver et al. 2017), fishing (Rivera et al. 2016; Marushka et al. 2017) and gathering (Strand et al. 2002). Pollutants accumulating in food chains have been particularly well-documented in marine mammals (Kuhnlein et al. 1995; Chan & Receveur 2000; Binnington et al. 2016), seafood (Chan et al. 1995; Donatuto et al. 2011), riverine fish (Peplow et al. 2007; Ullah et al. 2016), birds (Tsuji et al. 2007; Lyver et al. 2017) and caribou (Tracy & Kramer 2000; Ostertag et al. 2009). For instance, wildlife's geophagy can be an important route of exposure to petrogenic contamination for IPLCs living in the vicinity of oil extraction areas and relying on subsistence hunting (Doyle et al. 2010, 2012; Irvine et al. 2014; Orta-Martínez et al. 2017). Pollution of traditional wild food can lead to food insecurity and foster increased reliance on nutrient-poor alternative foods, increasing the risk of malnutrition and chronic diseases (Young et al. 1992; Howard et al. 1999; Laird et al. 2013; Nippon Foundation & Nereus Program 2017; Singh & Chan 2017). Some of the pollutants to which IPLCs are most often exposed include toxic elements such as mercury (Boischio & Henshel 2000; Chan & Receveur 2000; Maurice-Bourgoin et al. 2000; Wheatley & Wheatley 2000; Dallaire et al. 2003; Dórea et al. 2005; Kinghorn et al. 2007; Schuster et al. 2011; Webb et al. 2016; Lyver et al. 2017), lead (Chan et al. 1995; Hanning et al. 2003; Walker et al. 2006; Tsuji et al. 2008; Anticona et al. 2011; Valera et al. 2011; Udechukwu et al. 2015; Ullah et al. 2016), arsenic (Rainham 2002; Basu et al. 2006; Atkins et al. 2007; Liu et al. 2010; Sandlos & Keeling 2016) and cadmium (Chan et al. 2001; Anda et al. 2007; Haswell-Elkins et al. 2007, 2008; Charania et al. 2014; Curren et al. 2015; Primost et al. 2017), legacy Persistent Organic Pollutants such as PCB (Van Oostdam et al. 2005; Tsuji et al. 2005; Reyes et al. 2015), toxaphene (Kuhnlein et al. 1995; Van Oostdam et al. 2005; Walker et al. 2003) or DDT (Dallaire et al. 2003; Tsuji et al. 2005, 2007; Reyes et al. 2015), polycyclic aromatic hydrocarbons (Rainham 2002; Droitsch & Simieritsch 2010), as well as high levels of radiation (van Dam et al. 2002; Williams et al. 2017).

The contributions of IPLCs to the prevention and reduction of pollution are seldom recognized. Counting a few exceptions (e.g., Lyons 2004; O'Faircheallaigh 2013), IPLCs remain largely unsupported in their legal battles against polluting corporations operating in IPLC territories (MacDonald 2015; Rodríguez Goyes et al. 2017). As such, they often face enormous challenges in receiving compensation for the impacts of environmental pollution (Martínez-Alier 2014; Koh et al. 2017). It has also been suggested that the effectiveness of IPLC-led pollution prevention plans may be limited without larger scale action (Nippon Foundation & Nereus Program 2017). There is also well-established evidence of how the abandonment of certain IPLC traditional management practices often results in increasing levels of pollution (Sammul et al. 2012). Evidence of the lack of recognition of IPLC contributions to achieve this target is particularly well-documented in the case of freshwater pollution (e.g., Behrendt & Thompson 2004; Peplow et al. 2007; Jackson 2011; Morrison et al. 2015). Research has shown that IPLCs have generally been marginalized from water resource agencies in several countries (Jackson 2008; Finn & Jackson 2011; Weir 2010). IPLCs have often expressed that engagement in water management is generally limited to consultative capacity through ineffective representative processes, which undermines their capacity to defend their stakes in terms of water quality and environmental pollution (Behrendt & Thompson 2004; Hunt et al. 2009). As such, IPLCs are advocating worldwide against pollution (O'Faircheallaigh & Corbett 2005; Jackson et al. 2009; Jackson 2011). Greater engagement of IPLCs on the governance of resources (e.g., through negotiated agreements; Jackson & Barber 2015) can serve a purpose in incorporating IPLC social, spiritual and customary values in water quality and ecosystem health (King & Brown 2010; Finn & Jackson 2011; Barber & Jackson 2012), as well as ILK (Weir et al. 2013; Escott et al. 2015).

Aichi Target 9: By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.

Methods: The text below is based on a literature review using web of sciences and google scholar. The searches used the following search terms as topics: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK) AND ("invasive species" OR invasive OR "non-native species" OR "environmental invasion"). The string resulted in 65 search results in Topic search on the Core collection of the Web of Science with a subscription at UAB. We further completed the results with grey literature and references from other sources. We included in this review the most representative papers only.

The literature reports a significant role of IPLC in monitoring the presence of invasive alien species (IAS). Because of their detailed knowledge of local ecologies, IPLC are recognized as sentinels of environmental changes (Reyes-Garcia et al. 2016), with a body of knowledge which might contribute to monitor the spread of IASs and their impacts on local ecosystems. Indeed, IPLC have helped monitor different IAS in a range of ecosystems (e.g., Jevon and Shackleton 2015, Luizza et al. 2016, Sundaram et al. 2012, Uprety et al. 2012, Voggesser et al. 2013; Santo et al. 2017, Schuettler et al. 2011)(Ansell and Koenig 2011), including invasive fishes (e.g., Azzurro and Barriche 2017, Azzurro et al. 2017, Boughedir et al. 2015, Aigo and Ladio 2016) and crabs (e.g., Cosham et al. 2016) in marine environments, invasive plants (Bart 2006) and animals (Sloane 2016) in coastal wetlands, and invasive insects in Northern America (Costanza et al. 2017). Additionally, IPLC perspectives and worldviews

provide an alternative to the global discourse regarding IAS, typically framed around "warlike metaphors" (Alexander et al. 2017). As IPLC worldviews focus on other aspects such as health, reincarnation of ancestors, nature-based survival and creation (Ford 2004), they might be useful in building a different discourse around IAS and, as such, foster change in IAS management practices (Bach and Larson 2017). It is also recognized that, in some contexts IPLC livelihoods have impeded IAS expansion, while in other cases, with changes in IPLC use of the environment has resulted in the uncontrolled spread of alien species that then became invasive (Fredrickson 2006). IPLC can also provide IAS management, control, and eradication techniques targeting both IAS removal (Bart 2010, Bart and Simon 2013) and native species protection (Hall 2009). Some IPLC have adopted a mix of strategies (multifunctional approach) to control IAS in some areas and harbor them in others to secure benefits from them, such as invasive buffalo in northern Australia (Ens et al. 2015).

In many regions of the world, IPLC are directly affected by the lack of control on IASs spread (Bhatt et al. 2011, Turbelin et al. 2017). IAS may severely hinder local food production or hunting and gathering strategies and thus have negative impacts on local livelihoods, especially when added to the other challenges IPLC face, as demonstrated in a study among nomadic pastoralists in India (Duenn et al. 2017). Furthermore, studies have shown that IAS can also affect local livelihoods across multiple livelihood capital bases and across multiple assets (Rogers et al. 2017), or differently affect communities living in the same landscape (Kent and Dorward 2015). For example, IAS such as Prosopis juliflora or Lantana camara affect local livelihoods in various ways, including replacing native plant species that are normally used as fodder (Hiremath and Sundaram 2013) or creating dense thickets that block access to water points and pasture lands (Mwangi and Swallow 2005, Kaur et al. 2012) or to non-timber forest products (Jevon and Shackleton 2015). Given their negative effects on IPLC livelihoods, the IASs spread may also lead to loss in IKP systems. For example Lantana camara has been reported to replace native wild food and medicinal plants in South India, and as such threatens IKP (Harisha et al. 2016). However, the literature also reports cases in which IAS are integrated into IPLC subsistence strategies (Robinson et al. 2005, Ens et al. 2016) or into their pharmacopeia (Uma Shaanker et al. 2009, Philander 2011, Srithi et al. 2017). For example, cattail (Typha domingensis) has been commoditized and integrated into local Mexican economies, providing a revenue source for local or nonlocal peoples (Hall 2009), and Prosopis juliflora has become a key source of fuelwood in some Indian states (Sato 2013).

IPLC are increasingly being recognized and valued as key contributors towards achieving the target, mostly because of the contributions they can make in locally monitoring and understanding IAS impacts. IPLC are also involved in co-designing IAS-control experiments and management strategies (Ens et al. 2016, 2017, GIZ 2014, Saunders et al. 2007), but these initiatives would benefit in being expanded.

Lantana camara in the Biligiri Rangaswamy Temple Tiger Reserve – impacts on ecology and local knowledge. The Biligiri Rangaswamy Temple Tiger Reserve (BRT) in Karnataka (India) is part of the biodiversity rich Western Ghats. Lantana camara was first recorded in BRT in the 1930s, but it became widespread in the last decade only. The Soliga, a local tribal community living in BRT, view Lantana as having detrimental impacts on native vegetation and NTFP-dependent livelihoods by reducing access to forests and increasing human-wildlife conflicts (Murali and Setty 2001; Sundaram and Hiremath 2012). Older community members explain Lantana's spread by the cessation of traditional fire management, when BRT became a protected area (1970s). Ecological studies match with these views: evidence show that fires kill young Lantana and seeds in the soil, while dense Lantana thickets fuel intense fires that kill adult native species but not adult Lantana. Thus, fires today assist the spread of Lantana

(Sundaram et al. 2012); going back to traditional fire management would require to first reduce *Lantana* cover. However, younger Soligas, grown up in a *Lantana*-invaded forest, regard fires as ecologically detrimental. Thus, not only has *Lantana* invasion altered forest ecosystems, it has also altered the Soliga relationship with the forest, and their knowledge of traditional fire management.

Aichi Target 10: By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable coastal ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning.

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous community" OR "indigenous peoples" or "local community" or "aboriginal") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("coral reef" OR "mangrove" OR "coastal management" OR "marine protected areas"). The search was run in Web of Science yielding 188 papers of which 72 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions.

There is now clear evidence that the contribution of IPLC is essential for management and conservation of coastal zones (Aburto-Oropeza et al. 2011; Teixeira et al. 2013; Moshy and Bryceson 2016). IPLC accumulate a knowhow that can provide substantial inputs for the evaluation of biomass and biodiversity and even for the mapping of sensible zones (Levine and Feinholz 2015). The benefits also extend to the Blue Carbon ecosystem preservation, IPLC being important contributors to the management of wetlands, mangroves, or seaweedseagrass, especially when they contribute to the design of management plans (Vierros 2017). Indeed, while research shows that marine protected areas (MPA) are important for biodiversity and biomass conservation and for social-economic welfare (Mascia et al. 2017), it also shows that MPA top-down management strategies can be ineffective: by letting people out of the decision-making processes top-down management strategies create a deep frustration and a feeling of exclusion (Van Putten et al. 2016) that can result in IPLC lack of collaboration (Moshy and Bryceson 2016; Van Putten et al. 2016; Vaughan and Caldwell 2015). Co-management has emerged as an alternative bottom-up approach to resource and landscape-seascape conservation, promoting IPLC direct implication in marine management and conservation plans (Datt and Deb 2017; Vaughan and Caldwell 2015). It is important to acknowledge that land and sea management cannot be meaningfully separated in the coastal zone because of their inextricable links (Arias-González et al. 2017; Smith et al. 2017). So, MPAs and other marine conservation zones need to be considered in the context of the broader land and seascapes they support/affect. Researchers have suggested that IPLC have been foundational in recognizing and protecting these links ((Haggan, Neis, and Baird 2007; Johannes 1992), see for example the Local Marine Management Areas in the S. Pacific (Jupiter, Mangubhai, and Kingsford 2014) H. Govan et al 2009; P. Christie, et al 2007, SCA Ferse, et al 2010; M Keen and S. Mahanty 2006; A Charles et al 2016).

Fully including IPLC's in MPA management (i.e., from their design to day to day management, control and conservation) might be beneficial for conservation (Aburto-Oropeza et al. 2011). This is so because ILK can contribute to a better resource use (Siregar, Adrianto, and Madduppa 2016). For example, in Fiji, the management of coral reefs and associated resources in protected and non-protected areas is based on traditional rules, where clans differ in their vision of how and when the different resources have to be fished or harvested (Golden et al. 2014). Some case studies show that a bottom-up management of

coastal resources may result in a rise by more than 400% the fish biomass in less than a decade (Aburto-Oropeza et al. 2011), for which decentralization and co-management including IPLC have been claimed as essential for marine biodiversity and resource preservation. However, traditional management and preservation practices may be very different from one country to another, as they depend on the social structure, the cultural framework and legislative constraints. Thus, exporting the model that works on one place may be not suitable for a nearby country (Aswani, van Putten, and Minarro 2017), as each area has its own way to see the ecosystem sustainability and general rules have always to adapt to local realities (Young et al. 2014; Thornton and Scheer 2012).

The preservation of the marine natural environment and ILK in coastal zones goes beyond the need for biodiversity conservation. Mangroves and coral reefs are essential for IPLC food sovereignty and livelihood. For example, in Tonga, fish not only contributes to household subsistence, but it is also a social tool needed for the exchange of favors or simply to help those who have few or no resources (Kronen 2004). IPLC have developed particular forms of natural resource management that do not directly seek profit, but social and cultural compensation (Lauer and Aswani 2009; Walters 2004). However, the increasing monetarization of the system can lead to the loss of sense of social value with potential implications for the health of the ecosystem. Thus, in many areas traditional marine resource management is being threatened by the intensive exploitation of the system through massive tourism (coral reefs) or shrimp aquaculture (mangroves) (Arias-González et al. 2017). The increase of massive tourism implies the overexploitation of the system to feed a high number of tourists (Arias-González et al. 2017). This external pressure especially affects carnivorous but also herbivorous fishes leaving the system unbalanced from the trophic point of view and vulnerable for a deep transformation. Indeed, such overexploitation has been signaled as one of the drivers of the deep degradation of coral reefs that go from a well-structured coral-base to a simplified macroalgal-based structure (Jackson et al. 2001; Jackson et al. 2014; Hughes et al. 2003). Similarly, the industrial exploitation of aquaculture in mangroves makes a clear impoverishment of the ecosystem and a net transformation of its functioning, having a deep impact on the welfare and livelihood of IPLC, who face a confrontation between mangrove resource exploitation (artisanal fisheries, clam and crab harvesting, wood collection, etc.) and intensive shrimp aquaculture (Queiroz et al. 2013). A study conducted in Irian Java (Indonesia) shows that social-ecological links between native populations and their surrounding habitat are strengthened as a reaction to the severe restrictions caused by industrial activities based on logging (or shrimp aquaculture) (Ruitenbeek 1994).

Local perception of mangroves: a case study from NE Brazil. The development of methods to capture marine non-economic values is still incipient, but such valuation should not neglect the values assigned by the local populations (Raheem et al. 2012) as most ecosystem services produced by mangroves operate outside the market system and are integrally linked to IPLC way of life, traditions, and other values (NRC 2004). A study among fishers in the NE of Brazil found that fishers and other local people in the area use mangroves for swimming, sunbathing, social gatherings, or simply to stay there and enjoy the landscape (Queiroz et al. 2017). Fishers also maintain strong symbolic ties with the land and the sea through continuous observation and interpretation of natural cycles. In other words, the ecosystem service "spiritual/recreation/tourism" was highly valued by local communities who perceived the benefits of the mangroves beyond the monetary value typically used to evaluate ecosystem services. Moreover, the approach might, intentionally or unintentionally, lead to mangroves environmental protection, as has been shown in other settings (Walters 2004). The socio-cultural dimension of mangrove services needs to be considered by policy-makers as an

indispensable criterion for confronting the key challenges in coastal ecosystems conservation (Queiroz et al. 2017). The approach responds to the United Nations Sustainable Development Goals of improving human well-being and promoting the conservation of marine ecosystems by contributing to an improved understanding of the complex interrelationships between social and natural systems and of the multiple dimensions of ecosystem services (United Nations 2015).

Aichi Target 11: Safeguarding important areas for ecosystems and species diversity [LA: Victoria Reyes-García, CA: Sara Guadilla and Aili Pyhala; Reviewers: Fikret Berkes, Pablo Dominguez, Nadav Gazit, Eleanor Sterling]

Target: By 2020, at least 17 per cent of terrestrial and inland water areas and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscape and seascape.

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous communit*" OR "indigenous peopl*" OR "local communit*" OR "aborigin*" OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR "ILK") AND ("protected area" OR "sacred area" OR "community conserved area" OR "Aichi Target 11"). The search was run in Web of Science yielding 529 papers of which 216 were relevant to the topic. Additional bibliography was also selected from the authors' own literature database.

Many studies report on the *geographical overlap between the world's biological* hotspots and ancestral IPLC homelands (Porter-Bolland et al. 2012; Guèze et al. 2015; Kandzior 2016). Through traditional practices intricately connected to the surrounding natural areas, IPLCs have enabled the persistence of landscapes and seascapes of particular importance for biodiversity across the globe. Social norms, taboos, spiritual beliefs, and the establishment of sacred sites all exemplify means by which IPLCs have protected biophysical resources (McPherson et al. 2016; Karst 2017; Samakov and Berkes 2017) (Lopez-Maldonado and Berkes 2017). As a result, in numerous cases, Protected Areas have been designated within IPLC territories (Shen et al. 2012; Stevens 2014; Maraud and Guyot 2016; Mueller, Lima and Springer 2017). For instance, the Denesoline, a native nomadic people of northern Canada with a lifestyle linked to the caribou migratory routes, considered as sacred those areas of particular importance for caribou herds. One of those landscapes corresponds to the Thelon River watershed, which is now part of the Thelon Wildlife Sanctuary, one of the largest wildlife refuges in Canada (Holmes et al. 2016). ILK has also contributed to the persistence of areas of particular importance for biological diversity, to the point that some traditional practices beneficial for species maintenance are being incorporated into the conservation strategies of certain protected area management regimes (Kikiloi et al. 2017; Vizina and Kobei 2017). For example, the ecological knowledge held by local fishers' communities has been successfully used for monitoring the decline of emblematic species in inland water ecosystems such as the Lake Alaotra Protected Area in Madagascar (Reibelt et al. 2017). In 'Vueti Navakavu' Marine Protected Area, Fiji, knowledge of elder fishers was combined with scientific knowledge for building biodiversity inventories toward the effective recovery of marine ecosystems (Thaman et al. 2017). The potential of including ILK in scientific research for better conservation outcomes is being increasingly recognized by scholars and international organizations (Preuss and Dixon, 2012;

Millán-Rojas et al. 2016; Nash, Wong and Turvey 2016; Periago et al. 2017; Ens et al. 2016; Tengö et al. 2016).

Despite the large number of protected areas overlapping IPLC territories, *the portion of Protected Areas self-governed by IPLCs remains just 0.6% of the total registered protected areas network* (UNEP-WCMC and IUCN 2016). That said, the past few years have witnessed *a growing recognition of ICCAs* ('Indigenous Peoples' and Community Conserved territories and Areas') under conservation designations (Borrini-Feyerabend et al. 2012), both as overlapped by protected areas (Stevens et al 2016) *and as 'other effective area-based conservation measures' (OECMs)* (Jonas et al. 2017). The large extension that ICCAs occupy in terms of global surface –estimated at around 20% of the total (Kandzior 2016) – and the wide variety of habitats they cover (Bhagwat and Rutte 2006), make these territories and areas a great asset in terms of global ecosystem maintenance, to the point that some authors have argued that it may well be impossible to reach Aichi Target 11 without ICCAs (Kothari et al. 2014). While still hugely undervalued, over the past few years ICCAs have received increasing public and legal support and recognition (Jonas et al. 2012; Kothari et al 2012) (see the example of the Australian 'Indigenous Protected Areas' (Borrini-Feyerabend, Kothari and Oviedo 2004).

Although, to date, the establishment and maintenance of state-owned Protected Areas has been considered the backbone of global strategies to halt biodiversity loss (Bicknell et al. 2017), conservation experts have acknowledged that the expansion of designated sites, alone, is not enough to reduce the extinction of species (Gannon et al. 2017). An important factor that seems to influence the rate of nature preservation inside a designated site is local people's support and involvement, as well as the incorporation of ILK in its management (UNEP-WCMC and IUCN 2016). Apart from the benefits that the documentation of traditional best practices can bring to conservation (Nash, Wong and Turvey 2016; Vizina and Kobei 2017), researchers consider *IPLC engagement as the most valuable outcome of* the integration of ILK with conservation goals (Bhagwat and Rutte 2006; Berkes 2018). Developing further the concept and practice of OECMs (Jonas et al. 2017), allowing for a diversity of protected area governance types (Borrini-Feyerabend et al. 2013; Borrini-Feverabend and Hill 2015) and implementing inclusive measures such as safeguarding IPLC ownership of knowledge, respecting indigenous laws and principles (Johnson et al. 2016) Ruiz-Mallen and Corbera 2013), privileging customary management practices, and involving IPLCs as equal partners in research and monitoring (e.g., Housty et al. 2014) all appear to be essential for the future of global conservation efforts (Brooks, Waylen and Borgerhoff 2012; Hill et al. 2016; Holmes et al. 2016; Kandzior 2016).

Text box: Australian Indigenous Protected Areas

Following a global paradigm shift in protected area governance towards greater involvement of IPLCs (Davies et al. 2013), the Australian Federal Government has developed a legislation that increasingly recognizes Aboriginal rights to land and natural resources (Borrini-Feyerabend, Kothari and Oviedo 2004)(Jaireth and Smyth 2003). This recognition has resulted in 'Indigenous Protected Areas' (IPAs), defined as 'areas of land and/or sea over which the indigenous traditional owners or custodians have entered into a voluntary agreement with the Australian Government for the purposes of promoting biodiversity and cultural resource conservation' (Davies et al. 2013). Since 1997, 75 IPAs have been designated, representing 44.6% of the total area in the Australian National Reserve System (Australian Government, 2017). Apart from preserving lands of very high biodiversity significance, granting these territories with a protected status also signifies an important mechanism for integrating ILK in the conservation of Australia's natural and cultural assets (Preuss and Dixon 2012). One of the key stages in the development of an IPA is designing a management plan (Muller 2003) that aims to integrate ILK with other knowledge systems to address biodiversity conservation and threats (Davies et al. 2013).

Aichi Target 12. 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.

Methods: Initial literature searches used the following double structure: (("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR "ILK")) AND TOPIC: (("endangered species" OR "threatened species" OR "extinct*" OR "Aichi Target 12" OR "animal conservation")). Later searches used subsets of the first part and more focused secondary topics; these were followed by snowball searches using keywords suggested by the resulting literature. Where examples were too numerous to include all sources (e.g., community-based resource management impacts on threatened species, traditional ecological knowledge of threatened species), we chose more recent examples from the present century. Additional references suggested by reviewers were added when appropriated.

No database catalogs the threatened species protected by IPLC, nor have any significant meta-analyses been undertaken. An analysis of the 2012 World Database on Protected Areas found that "17% of the 4,118 threatened vertebrates are not found in a single protected area and that fully 85% are not adequately covered (i.e., to a level consistent with their likely persistence)" (Venter et al. 2014). Therefore, protection in protected areas managed by IPLC (4.6% of the total) (Juffe-Bignoli et al. 2014), must be lower. However, because IPLCs often live in areas of high biodiversity (Sobrevila 2008)(Renwick et al. 2017), they have the capacity to conserve high proportions of threatened species, even when IPLCs holdings are small (Takeuchi et al. 2017) and even disproportionately high proportions of them (Beckford et al. 2010). IPLC also support conservation and recovery of threatened species by working to reduce threats to some species. Efforts to control poaching (Lotter and Clark 2014) and reduce other sources of mortality (Gunn, Hardesty, and Butler 2010), sacred sites (Pungetti, Oviedo, and Hooke 2012) and some taboos (Colding and Folke 2001; Jones, Andriamarovololona, and Hockley 2008; Pungetti, Oviedo, and Hooke 2012) have been reported to improve status of threatened species, although this is not true in all circumstances. Traditional land management for more common species or landscape conditions may also protect threatened species (Ashenafi, Leader-Williams, and Coulson 2012; R. B. Bird et al. 2013). IPLCs community-based resource management (Noble et al. 2016)(Roe, Nelson, and Sandbrook 2009) might also benefits threatened species (Campos-Silva et al. 2017)(Naidoo et al. 2011) through changing human behaviors to benefit conservation (Nilsson et al. 2016) and a clearer understanding of underlying economic needs (Humavindu and Stage 2015 and references therein).

Where IPLCs have high capacity for conservation, they lead species-based conservation efforts to good effect. For example, IPLCs of the US, Canada, and Europe protect a variety of culturally important threatened species, including salmon (Ween and Colombi 2013), wolves (Ohlson et al. 2008), polar bear and walrus (Meek et al. 2008). Such efforts are not without conflict with non-indigenous land owners and managers (Findlay et al. 2009; Breslow 2014). IPLCs still must defend their rights to participate in threatened species conservation, (Muir and Booth 2012; Olive 2012; Olive and Rabe 2016), and defend the values they bring to that

practice (Nadasdy 2006). Nevertheless, legal rights to harvest threatened species (Brown 1994; Irvine et al. 2005) and to participate in (Sanders 2007; Canada and Environment Canada 2005; Australian Government 2015) and lead (Ohlson et al. 2008) threatened-species management ensure these efforts will continue. In areas where IPLCs active participation in threatened-species conservation is less common, its development is encouraged (Borrini-Feyerabend, Kothari, and Oviedo 2004; Ancrenaz, Dabek, and O'Neil 2007; Wilson, Edwards, and Smits 2010). IPLC skills, such as tracking, can be used to help in the conservation of threatened species (Attum et al. 2008; Dolrenry, Hazzah, and Frank 2016). Training in monitoring techniques, field technologies and increased literacy enhance conservation outcomes (Danielsen et al. 2014; Dolrenry, Hazzah, and Frank 2016; Ens et al. 2010; Benchimol, von Muhlen, and Venticinque 2017). Progress is also being made in the area of human-wildlife conflict, in conserving species that pose risks to humans and crops (Larson et al. 2016; Rastogi et al. 2012; Dolrenry, Hazzah, and Frank 2016), but important work still remains to be implemented (Jackson 2015).

Laws and policies requiring or recommending that ILK be used in management decisions are becoming more common, although ILK integration into resource management can be complex and/or contentious (Nadasdy 2006; Gerhardinger, Godoy, and Jones 2009). Recent work on threats to protected areas highlights the potential value of conservation efforts by local communities, which can detect threats where remote sensing is less effective (Schulze et al. 2018). Increasing attention is being turned to best practices in integrating ILK into threatened-species management (Gilchrist, Mallory, and Merkel 2005; Vongraven et al. 2012; McPherson et al. 2016). ILK has provided important information on historical population baselines and trends (Mallory et al. 2003; Ramstad et al. 2007; Turvey et al. 2010; Bender, Floeter, and Hanazaki 2013; Ziembicki, Woinarski, and Mackey 2013), species inventories (e.g., (Bamigboye, Tshisikhawe, and Taylor 2017)), as well as on species ecology and behavior (Nabhan 2000; Fraser et al. 2006; Ween and Colombi 2013; Voorhees et al. 2014; Wilder et al. 2016). In some cases (e.g., polar bears: (US Fish and Wildlife Service 2008; Kakekaspan et al. 2013; Vongraven et al. 2012), ILK is integrated directly into research and management plans. Where traditional practices become less common, these sources of information are dwindling (Bender, Floeter, and Hanazaki 2013; Ween and Colombi 2013).

Benefits to IPLC of threatened species conservation and recovery parallel the benefits of biodiversity, with participation in nationally recognized conservation work empowing IPLC and giving them agency in conserving their languages and cultures (Ween and Colombi 2013; Wali et al. 2017). Moreover, threatened species are often culturally significant to IPLC. For example, when India's vulture populations crashed due to widespread diclofenac poisoning, the Parsee people were forced to develop entirely new ways to dispose of the bodies of their dead (Van Dooren 2010). One study found a willingness to pay for conservation of forest elephants, although these were often inconvenient for villagers (Poufoun et al. 2016). Typically, declining species cease to play their original ecosystem roles, becoming ecologically extinct before they are completely extirpated or rendered extinct (Jackson 2015). The availability of some threatened species as food, medicine, sacrifice, or other goods and services to IPLC thus may already be greatly reduced when species are threatened (e.g., (Chiropolos 1994)). Successful recovery of such species may improve ecosystem conditions (Bottom et al. 2009) in addition to invigorating culture and, with care, economy (Coria and Calfucura 2012; Humavindu and Stage 2015). For example, locally-developed marine protected areas are becoming increasingly popular as their success in increasing local catch becomes better known (Hamilton, Potuku, and Montambault 2011). In Japan, more than 30% of MPAs are self-imposed by local fishing communities (Yagi et al. 2010), including both year-round and seasonally-closed areas. Increasingly, payments are available for conservation

work (Dolrenry, Hazzah, and Frank 2016). Payment programs for ecosystems services can extend to payments contingent on not hunting threatened species and payments to protect nests (Clements et al. 2010). Premium payments have also been proposed to conserve large species (Dinerstein et al. 2013). Ecotourism related to threatened species also provides an important source of revenue for IPLC (Humavindu and Stage 2015).

Impacts on threatened species of indigenous ecosystem management: fire and habitat mosaics in central Arnhem Land, northern Australia. On lands continuously held by aboriginal clan estate groups in northern Australia, fire is a traditional vegetation management tool (Yibarbuk et al. 2001). Traditional fire regimes are considered an obligation and serve to protect sensitive vegetation (jungle) and to increase abundance of desirable game species. The traditional fire regime results in a mosaic of vegetation types that has supported local biodiversity, including regionally and nationally threatened species that are declining elsewhere. The literature associated with this example of indigenous ecosystem management has examined early conflicts regarding the appropriateness of fire between IPLCs and government land managers (Lewis 1989), the implications of recent changes in fire regimes in the region (Russell-Smith et al. 2003), the spatial characteristics of the vegetation mosaics produced (R. Bliege Bird et al. 2008), and the impacts on such mosaics of introduced water buffalo (Trauernicht et al. 2013). However, to date, exploration of relations between threatened species, habitat, and fire has been cursory (Yibarbuk et al. 2001). As understanding of impacts of IPLC traditional land and wildlife practices grows, threatened species conservation would benefit from a larger focus on improving understanding of relations between traditional practices and non-target threatened species.

Aichi Target 13: By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socio-economically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity. Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK) AND ("crop diversity" OR "landrace diversity" OR "species diversity" OR "culturally valuable species" OR "genetic erosion" OR "Aichi Target 13"). The string resulted in 333 search results in Topic search on the Core collection of the Web of Science with a subscription at UAB, of which 91 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions).

Many studies report IPLC contribution to enhancing genetic diversity of crops (Brush 2004; Brush 2000; Gepts et al. 2012) and domesticated animals (REF). Through history, IPLC have contributed to increase crop genetic diversity both through species domestication (Khoury et al. 2016) and by contributing to diffuse crop species and varieties around the globe (Roullier, Benoit, et al. 2013; Roullier, Kambouo, et al. 2013), a process that is ongoing (Delêtre, Hodkinson, and McKey 2017; Haan et al. 2016; Altieri and Nicholls 2017). IPLC also contribute to maintain genetic and species diversity through management. For example, traditional landscape management practices contribute to maintain genetic diversity (Salick 2012; Brush 2000) including associated wild diversity (Blanckaert et al. 2007)(Nabhan 2003). Some traditional management practices that contribute to maintain crop genetic diversity include management of soil seed banks that enhance agrobiodiveristy (Pujol et al. 2007), traditional grassland management through shepherding, which provides dispersal opportunities for multiple plant species and thus offers a useful approach to restoring plant diversity in fragmented grasslands (Rico, Boehmer, and Wagner 2014; Saar et al. 2017), although with a trade-off with short-term yield (Niu et al. 2016); or mowing and burning, which create favourable environmental conditions for herbaceous plants and maintain plant species diversity through promotion of plant facilitation (Wang, Nishihiro, and Washitani 2015). Many IPLC also maintain home gardens with high landrace and species diversity contributing to in situ conservation (e.g., (Perrault-Archambault and Coomes 2008; Thomas and Caillon 2016; Galluzzi, Eyzaguirre, and Negri 2010). Finally, IPLC have also developed strategies to minimize genetic erosion. For example, seed maintenance and local exchanges are important for plant domestication, the dissemination of improved crops, and the maintenance of crop genetic diversity (Calvet-Mir and Salpeteur 2016; Nazarea 2006). The seed system is a major component of traditional management of crop genetic diversity in developing countries (Coomes et al. 2015). Seed flows (in the market or through other forms of exchange) are an important part of this system and have been conceptualized as networks through which planting material flows and genetic diversity is disseminated and conserved (Calvet-Mir et al. 2012; Abizaid, Coomes, and Perrault-Archambault 2016; Eyssartier, Ladio, and Lozada 2015; Thomas and Caillon 2016; Bonnave et al. 2016).

Crop genetic erosion is not only an ecological but also a food-security problem (Johns and Eyzaguirre 2006), and IPLC are considered to be among the most vulnerable to food insecurity (Harvey et al. 2014). Moreover, the shift in agriculture to using purchased inputs and to production for the market rather than for home consumption has largely impacted on-farm production and the maintenance of local landraces (Thomas and Caillon 2016). In situ conservation, or conserving crop genetic resources in the environments in which they occur, and use of local plant genetic resources is of prime importance for food security - as it permits IPLC to have permanent access to seed and planting material, adapted to their region, selection of improved new varieties (Maxted et al. 2002), and moreover to protect crop species against pests and introduction of new varieties which cause the loss of local varieties or genotypes, hence leading to the erosion of genetic diversity (Finetto 2010). Such conservation would complement, but not be an alternative to, national and regional gene banks (Finetto 2010).

On-farm or *in situ* conservation is increasingly being acknowledged as a relevant strategy to halt genetic erosion because it maintains evolutionary forces within and between the different components of the agricultural system (Thomas et al. 2011). The maintenance of such diversity is largely dependent on IPLC ability to directly interact with their environment (Gomez-Baggethun and Reyes-Garcia 2013) and maintain their own cultural identity (Velasquez-Milla et al. 2011). On-farm conservation continues to be threatened by the undervaluation of local management practices by some extension programs (Jacobi et al. 2017), by current legislation that is adverse to the rights to save and exchange seeds (Deibel 2013), and by the introduction of improved mass propagation methods (Jaradat 2016) and hybrid or GM seeds (e.g., (Shewayrga, Jordan, and Godwin 2008). Concrete initiatives that value IPLC contributions to the maintenance of crop genetic diversity can be found worldwide (e.g., (Wilkes 2007). For example, these initiatives include scholarly discussions about the role of biocultural refugia in Europe in safeguarding genotypes, landscape features, oral and artistic traditions and a self-organized system of rules to deal with unpredictable change (Barthel, Crumley, and Svedin 2013), or the creation of the Parque de la Papa in the Peruvian Andes, which has repatriated a thousand native potatoes from the gene bank in

Lima so as to catalyze *in situ* regeneration of lost agricultural biodiversity in the region (Graddy 2013).

Parque de la Papa: Given the limitations of ex situ agrobiodiversity conservation (Hammer 2003), the last decades have seen some efforts towards less-centralized and more on-the ground efforts that acknowledge the contribution of IPLC, women, and elderly farmers to agrobiodiversity conservation (Jarvis et al. 2000; Soleri and Smith 1999; Wilkes 1991). Some of these efforts include repatriation and in situ conservation of crop genetic diversity including neglected or underutilized crops (Hammer 2003). For example, an innovative mechanism that allow both, to maintain agrobiodiversity and to recognize IPLC roles in agrobiodiversity conservation is the repatriation of germplasm collected from centers of diversity and conserved in ex situ gene banks to communities from which they originated but from which they have since been lost (Nazarea 2006). Since 1995, the Andean NGO ANDES has undertaken efforts to return to Quechua farmers hundreds of potato landraces from the International Potato Center based in Lima to a local center in Cusco, Peru. Reintegrating these native potatoes into Quechua farming systems is complemented by initiatives to document customary laws and revive culinary traditions in the Andes. Nowadays, up to 4.000 potato varieties are growing in the Parque de la Papa managed by six Quechua communities.

Aichi Target 14: By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK AND "ecosystem service*" OR "water" OR "food security" OR "wellbeing" OR "livlehood" OR "Aichi Target 14" AND "ecosystem"). The search was supplemented with literature from the reference lists and from key organizations working on these issues.

IPLC play a key role in protecting and restoring the world's ecosystems. Estimates of IPLC contributions to protect biodiversity vary from a report documenting that over 50% of the earth's land area, including forests, savannas and other terrestrial environments, are currently under indigenous and local management through formal and informal tenure systems (RRI 2015, Oxfam et al. 2016) to another documenting that traditional indigenous territories encompass areas that hold 80 % of the planet's biodiversity (Sobrevila, 2008). These areas provide ecosystem services over multiple scales, from global (e.g., biodiversity, carbon sequestration) to local (e.g., food security, water regulation and provision, extractable goods). Forests alone, which account for about 30% of the global land area (of which ~15% is under formally recognized indigenous territory), directly contribute to the livelihoods of over 1.6 billion people (World Bank 2008, RRI 2014).

Of course, not all the lands managed by IPLC are 'intact' or 'functional' ecosystems. Similarly, not all IPLC actions are geared towards sustaining or protecting ecosystems. Moreover, because multiple problems exist with mapping the world's IPLC-managed lands (Landmark 2017) and with quantifying ecosystem services from any given ecosystem (MEA 2005), the exact contribution that IPLC make to protecting and managing ecosystems is unknown. However, multiple examples from around the world have shown that, when done carefully and with a high degree of involvement from well-organized communities, devolving control of resource management to IPLC can produce better outcomes for conservation and ecosystem service provision than private management (REFS), and in some cases, strict protected areas (Bray et al. 2008, Persha et al. 2011). Thus, IPLC managed lands can be highly effective at conserving forest ecosystems (Poteete and Ostrom 2004, Chhatre and Agrawal 2009, Persha *et al* 2011).

Community managed concessions in the Mayan forests of Guatemala and Mexico, for example, have been found to have lower deforestation rates than nearby protected areas, despite facing similar pressures from drug trafficking and cattle ranching (Bray et al. 2008). These forests provide local communities with income from timber exports, firewood, traditional foods, and so on. In Nepal, the devolvement of state forests into communities safeguarding and restoring communal forests and watersheds. These activities brought a wide range of economic, social, and environmental benefits, including reducing sedimentation, producing wood and food, regulating water, reducing natural hazards and increasing habitat for biodiversity (Paudyal et al. 2017).

IPLC also actively restore ecosystems to produce ecosystem services essential to human wellbeing. For example, in the Maradi and Zinder Regions of Niger, local communities regreened over five million hectares of land through farmer-managed natural regeneration (Sendzimir et al. 2011, Reij and Garrity 2016). In the Sahel region, restoring tree cover reduced desertification and improved water supply during the driest periods, improving local wellbeing (FAO 2015). In Ecuador, an NGO promoted restoring watershed reserves for locally important ecosystem services, especially sustained summer streamflow. Local communities adopted a created a culture of tree planting and forest restoration in a region where deforestation had been the norm only decades before, and forest change rates went from -25%/year to +3%, producing a local 'forest transition' (Wilson 2016, Wilson and Rhemtulla, 2016). In Hojancha, Costa Rica, the community organized to grow trees for timber, producing seed, improving water resources and ecotourism (Madrigal Cordero et al. 2012). Through technical assistance, financial support, and payments for environmental services, local people became empowered to serve as stewards of regrowing forests and young plantations, and forest cover increased from 8% to 54.5%. Simulating indigenous and traditional management to restore ecosystems has been effective in several national parks (Anderson and Barbour 2003), and has also proved to be successful for example in restoring plant and bird communities in Swedish oak-hazel woodlands (Hansson 2001), or restoring alluviall meadow (Jamsran, 2010).

Thus, IPLC and ILK can play an important role in increasing the effectiveness of ecosystem restoration activities (Senos et al. 2006, Uprety et al. 2012). IPLC are especially well positioned to restore and safeguard ecosystems because they know the land, and can directly benefit from restoration activities (Schaffer 2010, Wangpakapattanawong et al. 2010, Babai and Molnár 2014). IPLCs have also restored land that was overexploited and degraded by outsiders (e.g., mining, pollution, and logging) – for example, local communities restored streams following mine pollution in the US (Middleton 2001) and local Qawalangin Tribe from Alaska received funding to restore beaches affected by pollution (NOAA 2018). Modern restoration activities are increasingly using ILK as it can improve the effectiveness and ecosystem services generated from science-based restoration activities (Senos et al. 2006). Examples include using indigenous fire regimes in both Tasmania and in Washington state to restore native biodiversity and ecosystem function (Marsden-Smedley and

Kirkpatrick 2000, Storm and Shebitz 2006) and restoring native plants used for basket making in New York state (Shebitz 2005).

Lack of progress towards the target has had serious implications for IPLC. IPLC are often among the most marginalized and poorest people in a given country (Lightfoot 2016), and are often more reliant on shared or communal natural resources, such as forests (Almeida, M. 1996, Godoy et al. 2000, Angelsen et al. 2013). A study of tropical communities using data from the Poverty-Environment Network found that environmental resources contributed 28% on average to household income, with higher percentages for lower income households (Angelsen et al. 2103). Loss of access to natural resources can thus have a disproportionately negative effect on the poorest people. In developing countries, IPLCs also often have a high reliance on subsistence agriculture and are thus disproportionately vulnerable to the impacts of ecosystem service loss on local, landscape and global scales (Seaman et al. 2014). Climate change, including changes in precipitation patterns, droughts, increased storm frequency/severity, etc., and changes in local water regimes can have serious consequences for annual harvests, and thus the livelihood base, food security, and general wellbeing of IPLCs (Bandara and Cai 2014, Seaman et al. 2014).

Globally, a lack of formal titling is a major factor that limits the recognition of ecosystem services generated by ICLPs, and their ability to receive financial benefits for services restored. Although they may use or manage a big share of the world's land area, IPLC possess formal titling for only 10%, and recognized designation for another 8% of the world's land area (Oxfam 2016, RRI 2015). While not all these lands are managed in a way that maintains or enriches ecosystems and their services, the numbers indicate that IPLCs contribute a great deal to enhancing the world's natural resource base, in many cases without receiving 'formal' recognition. A lack of formal titling means that in some places IPLC are unable to access markets for environmental goods and services (Oxfam 2016, RRI 2015). Securing tenure would thus not only allow access to future services and ability to maintain services in the long term, but could also improve IPLC's opportunities to obtain monetary compensation for the ecosystem services they provide (Bark, Barber et al.).

Recognizing and valuing IPLC's contribution to producing ecosystem services is also complicated by the difficulty in measuring, quantifying, and assigning value to them (Preece et al. 2016), particularly values that are not strictly economic (e.g., heath, cultural) (Garrett et al. 2009, Bark et al. 2015, Preece et al. 2016). Even where systems are in place to compensate land managers for locally produced ecosystem services that benefit other users (for example, PES schemes), many IPLCs lack information, knowledge, or access to them. Remote or impoverished conditions, weak governance structures, or lack or critical mass/representation can all limit participation in government and other programs, and often it is the poorest of the rural poor who are unable to participate (Zbinden & Lee, 2005, Emmanuel and Blum 2015). Thus, access to compensation/ payments for the goods that others receive from this land is limited, particularly for the more marginalized groups (Zbinden & Lee, 2005, Bark et al. 2015, Emmanuel and Blum 2015). However, as mentioned above, IPLC are also well positioned to benefit from the local ES generated by their activities (Becker 2004, Garnett et al. 2009, Madrigal Cordero et al. 2012, Wilson and Rhemtulla 2016).

Local tree planting to restore watersheds in Andean Ecuador provides ES of both local and global importance. By managing and maintaining lands to meet local needs, IPLC collectively make substantial contributions to providing ecosystem services, both local and global. An example of this is watershed restoration in the Intag Valley in NW Andean Ecuador (Wilson and Rhemtulla, 2016; Wilson 2016). Here communities worked to grow and

plant trees on communal lands in watersheds reserves in an effort to overcome the summer droughts that threatened their ability to farm. An NGO played the critical role of creating communal lands and giving communities secure tenure rights to this land, and provided training to grow and plant trees. By planting trees to enhance locally important services (water, soil retention, and various local foods and 'useful' plants), local people also produced several important global services – including biodiversity protection in a biodiversity hotspot, increased carbon sequestration, and watershed protection in the upper reaches of a larger watershed. It was the production of these global services that allowed the NGO to secure funding for local services, but because this restoration was planned and executed with local people to provide services that were essential to their lives and livelihoods – a win-win scenario.

Aichi Target 15: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification.

Methods: The text below is based on a literature review using the following search terms as topics: ("Indigenous Community" OR "Indigenous Peoples" OR "Local Community" or "Aboriginal") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("carbon" OR "carbon stocks" OR "ecological restoration" OR "desertification" OR "carbon sequestration") OR ("Aichi Target 15"). The search was run in Web of Science yielding 413 papers of which 120 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions).

Through their natural resource management systems, IPLCs have made substantial contributions to conserving carbon stocks and increasing ecosystem resilience in the face of climate change (Wangpakapattanawong et al. 2010; Mijatović et al. 2012; Nakashima et al. 2012; Uprety et al. 2012; FPP & CBD 2016). There is well-established evidence, particularly from Latin America, that IPLC lands, managed under local customary institutions, are at least as effective in sequestering forest carbon as protected areas (Nepstad et al. 2006; Soares-Filho et al. 2010; Nelson & Chomitz 2011; Porter-Bolland et al. 2012; Nolte et al. 2013). This is because IPLC land management regimes (e.g., indigenous territories, reserves, ICCAs) tend to have lower deforestation rates than surrounding areas, thus avoid carbon emissions and preserve other NCPs (Adeney et al. 2009; Ricketts et al. 2010; Paneque-Gálvez et al. 2013; Vergara-Asenjo & Potvin 2014; RAISG 2016; Schleicher et al. 2017), with well-established evidence that formal land titling, strong tenure rights, and decentralized rule-making play an important role in reducing deforestation (Tucker 2004; Chhatre & Agrawal 2009; Meyfroidt & Lambin 2011; Ceddia et al. 2015; Blackman et al. 2017; Stevens et al. 2014). Overall, IPLC territories encompass some of the most carbon-rich forests in the world (Campbell et al. 2008; FAO 2010; Ricketts et al. 2010; Soares-Filho et al. 2010, 2012; RRI 2014). The carbon contained within IPLC lands of the Amazon Basin, Mesoamerica, the Democratic Republic of Congo and Indonesia has been estimated to represent more than 20% of the carbon stored aboveground in all the world's tropical forests (Walker et al. 2014).

ILK-based land management practices such as rotational farming, agroforestry, improved cropfallow systems, hedgerows, grazing enclosures, and active incorporation of mulch and manure have been deemed effective at enhancing carbon sequestration, preventing environmental degradation and combatting desertification (Albrecht & Kandji 2003; Montagnini & Nair 2004; Tschakert et al. 2007; Wangpakapattanawong et al. 2010; Cheng et al. 2011; Salick et al. 2014; Singh et al. 2014; Watanabe et al. 2014; Bhagawati et al. 2015; Hartoyo et al. 2016; Seid et al. 2016; Chirwa et al. 2017)(Coughlan 2014). IPLC practices of soil carbon enrichment, which in turn often supports higher biodiversity, are well recognized in Amazonia (Lehmann et al 2003; Glaser 2007; Junqueira et al 2010, 2016) and increasingly in other areas of the world (Sheil et al. 2012; Soloman et al 2016), and have influenced interest in biochar enhancement elsewhere (Novotny et al 2009; Soentgen et al. 2017). Similarly, IPLC fire management regimes contribute substantially to greenhouse gas abatement and ecosystem resilience in tropical and subtropical savannas and grasslands (Shaffer 2010; Welch et al. 2013; Trauernicht et al. 2015; Laris et al. 2016, 2017), with considerable long-term evidence from Australia (Murphy & Bowman 2007; Heckbert et al. 2012; Richards et al. 2012; Wilman 2015)(Bliege Bird et al. 2012; Bird et al. 2016).

There is also well-established evidence of the crucial role that IPLCs play in ecological restoration efforts that help build social-ecological resilience (Kimmerer 2000; Storm et al. 2006; Nagendra 2007; Egan et al. 2012; Lyver et al. 2016; Wehi & Lord 2017), although it is unknown what percentage of overall restoration efforts are currently led by, or feature the participation of, IPLCs. IPLCs often intermingle domestic plants in native forests to supply food and medicine, which also enhance other NCPs and can be harnessed for restoration efforts (Garibaldi & Turner 2004; Turner et al. 2007; Ford & Nigh 2015; Lee & Courtenay 2016); furthermore, management of cultural keystone species amongst IPLCs is crucial to ensure the flow of NCPs over time, and these species can serve as the basis for ecological restoration (Uprety et al. 2012; CAFF 2013; Hobbs et al. 2014; Cuerrier et al. 2015; Mustonen 2015; Fernández-Llamazares et al. 2016). In recent years, many IPLCs have increasingly taken leadership roles in diverse ecosystem restoration efforts, ranging lakes and rivers (Coombes 2007; Hormel & Norgaard 2009; Fox et al. 2017), grasslands and drylands (Pellant et al 2004; Stenseke 2009) and mangroves and reefs (Walters 2000; Selvam et al. 2003; Uychiaoco et al. 2000; Trialfhianty et al. 2017). Restoration efforts led by IPLCs can help stem the tide of species loss and landscape change, such as that caused by urbanization (Horikuchi et al 2011; Fox et al. 2017). Engagement of IPLCs in community forestry has been shown to be a useful model for restoration of degraded forests (Maikhuri et al 1997; Nagendra 2007; Paudyal et al. 2015), while co-management with IPLCs with explicit restoration goals have shown mixed success in other ecosystems (Hill & Coomes 2004; der Knaap 2013). IPLCs are also actively engaged in enrichment plantings and enhancement of secondary forests for NCPs (Paquette et al. 2009; Douterlungne et al. 2010; Suryanto & Putra 2012), as well as key participants in several large-scale forest restoration efforts, particularly in Asia, where hundreds of thousands of communities have taken part in these efforts (Yan-qiong et al. 2003; Bennett 2008; McElwee 2009; Clement et al. 2009; He & Lang 2015).

Despite having contributed little to greenhouse gas emissions (e.g., by leading carbon-neutral lifestyles and generally having a low ecological footprint; Eil 2009; Salick et al. 2014; Stewart et al. 2016), IPLCs carry a disproportionate share of the burden of global environmental changes (Green & Raygorodetsky 2010; Abate & Kronk 2013). Moreover, because they sometimes live in areas increasingly prone to the effects of climate change (e.g., high-risk areas or marginal environments) they are particularly vulnerable to its impacts (Macchi et al. 2008; Lefale 2010; Bardsley & Wiseman 2012). In view of this, there is well-established evidence that safeguarding ecosystem resilience is critical to promote IPLCs quality of life (Sangha et al. 2015; Caillon et al. 2017; Kingsley & Thomas 2017; Pascua et al. 2017; Sterling et al. 2017). The failure to restore degraded ecosystems in areas inhabited by IPLCs threatens their cultural

well-being, undermining access to important NCPs on which their livelihoods depend (Adger et al. 2005; Fa et al. 2003; Aronson et al. 2016; FPP & CBD 2016; Golden et al. 2016). Continuing species geographical shifts and local extinctions due to climate change renders urgent the need to restore and potentially relocate species and ecosystems for benefits to IPLCs (Pecl et al. 2017).

Ecological restoration has been shown to increase provision of NCPs (Benayas et al. 2009), and where participatory and attuned to local socio-economic benefits such as traditional use patterns, IPLCs gain increased access to NCPs and reduced conflicts (Gobster & Barro 2000; Eden & Tunstall 2006; Shackelford et al. 2013; Wortley et al. 2013; Baker 2017). IPLCs have also benefitted from connecting cultural revitalization to restoration projects (Anderson 1996; Long et al. 2003). Where there has been a lack of involvement of local stakeholders in restoration or conflicts over visions of the landscape, there has been less progress and benefit for IPLCs (Junker et al. 2007; Couix & Gonzalo-Turpin 2015; Heldt et al 2016). Where passive restoration is already on its way, restoration efforts can also be regarded as redundant, or at least questionable when driven by public spending (Sayer et al. 2008; Johnson et al. 2017).

Weak protection of IPLC land rights often results in ecosystem degradation (Finley-Brook 2007; Araujo et al. 2009; Naughton-Treves and Wendland 2014; Duchelle et al. 2014; Blackman et al. 2017). There is well-established evidence that land tenure insecurity is one of the main underlying drivers of deforestation at the global level (Hayes 2007; Damnyang et al. 2012; Larson & Dahal 2012; Reyes-García et al. 2012; Robinson et al. 2014; Vergara-Asenjo & Potvin 2014; Holland et al. 2017). Moreover, recognizing the customary institutions of IPLCs is a critical means for connecting IPLCs with policies promoting ecosystem restoration and carbon compensation schemes (Griffiths et al. 2004; de Koning et al. 2011; Larson et al. 2013; Sunderlin et al. 2014; Buntaine et al. 2015). Specifically, forest titles can provide access to incentive programs that pay for ecosystem services, maintenance of forest cover and carbon sequestration (Larson 2010; van Dam 2011; Awono et al. 2014; Duchelle et al. 2014; Naughton-Treves & Wendland 2014; Turnhout et al. 2017).

Initiatives engaging IPLCs in community-based carbon monitoring are also gaining prominence all over the world (Danielsen et al. 2013; Pratihast et al. 2013; Brofeldt et al. 2014; Butt et al. 2015; Hartoyo et al. 2016; McCall et al. 2016). However, up to date, the experience of forest planting and restoration projects, particularly for carbon sequestration, on IPLCs is mixed; some projects have increased smallholder incomes, diversified livelihoods, and expanded access to NCPs (Xu et al. 2007; Brown et al. 2011), while other projects have had minimal or negative impacts on IPLCs (Boyd et al. 2007; Jindal et al. 2008; Caplow et al. 2011; Reynolds 2012; Beymer-Farris & Bassett 2012; Lawlor et al. 2013). Many afforestation projects that involve IPLCs as labor or land providers have high opportunity costs and delayed and low benefits (Jindal et al. 2012; Agrawal 2014), and benefits tend to aggregate to households that are already economically better off (Thacher et al. 1997; Glomsrod et al. 2011). Overall, property rights, land availability, social organization and political networks constitute key factors for IPLC in accessing and benefiting from carbon offsets (Kerr et al. 2006; Boyd et al. 2009; Corbera & Brown, 2010; Osborne 2011).

Top-down initiatives for carbon sequestration can be detrimental to the quality of life of IPLCs and undermine ecosystem resilience, by curtailing IPLC access to, and sustainable use of, biodiversity (Phelps et al. 2010; Duchelle et al. 2014; Vijge & Gupta 2014; McElwee et al. 2016). For instance, heavily centralized REDD+ implementation has often neglected the historical, cultural and spiritual values that forests hold for IPLCs, as well as their customary

institutions (Ribot et al. 2006; Phelps et al. 2010; Gupta et al. 2012; Brugnach et al. 2014; Sunderlin et al. 2014). The literature is also mixed on the success rates of avoided deforestation or REDD+ projects (Larson et al. 2013; Awono et al. 2014; Howson & Kindon 2015). Current carbon forest standards have shown moderate success in protecting IPLC rights (Larson 2011; McDermott et al. 2012; De La Fuente & Hajjar 2013; Roe et al. 2013), but formal arrangements for participation by IPLCs in REDD+ policies at subnational levels has generally been weak, with many communities only consulted, rather than being involved in a systematic manner in all aspects of REDD+ planning (Chernela 2014)(Hall 2012; Brown 2013; Sunderlin et al. 2014; Atela et al. 2015; Holmes et al. 2017). Many IPLCs have expressed distrust of government or outsider-run REDD+ projects (Evans et al. 2014; White 2013), but local capacity to meet many of the technical requirements of REDD+ is often low among IPLCs (Cerbu et al. 2013; Asiyanbi et al. 2017; Holmes et al. 2017).

Research suggests that ILK and the cultural values of IPLCs have often remained excluded from the management of ecological restoration programs (Robertson et al. 2000; Mills 2003; Hill & Coombes 2004; Jackson et al. 2005; Wehi & Lord 2017). For instance, acknowledgement of IPLC regimes of prescribed burning is often dismissed in policy circles (Welch et al. 2013; Mistry et al. 2016), despite increasing evidence of its contribution to wildfire prevention and climate change mitigation (Cook et al. 2010; Defossé et al. 2011; Richards et al. 2012; Russell-Smith et al. 2015; Bikila et al. 2016). Because many restoration policies and carbon compensation schemes intersect with IPLC sociocultural values (e.g., what is considered as a natural baseline), active involvement of IPLCs from the beginning of the policy design has been deemed essential for success, particularly in building partnerships and avoiding value conflicts (Davenport et al. 2010; Lawlor et al. 2010; Frey & Spellerberg 2011; Robinson & Wallington 2012; Holtgren & Auer 2016; Lyver et al. 2016; Richardson & Lefroy 2016; Rose et al. 2016; Fox et al. 2017). Ensuring that restoration projects have sufficient incentives including short-term benefits like rapid NCP provision is needed for active IPLC involvement (Kessler & Laban 1994; Stone et al. 2008; Le et al. 2012; Smith et al. 2017), and there is also need for ongoing technical and financial support to maintain restored areas (Datta & Virgo 1998; Nguyen et al. 2017). Increased employment benefits of restoration are often important to IPLCs (Le et al. 2012; Nielsen-Pincus & Moseley 2012; Bendor, et al. 2015), and participation in restoration can also improve IPLC attitudes to conservation generally (Pohnan et al. 2015).

Aichi Target 16: By 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation.

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous community" OR "indigenous peoples" or "local community" or "aboriginal") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("Nagoya Protocol" OR "legislation" OR "implementation" OR "Aichi Target 16") AND ("2012" OR "2013" OR "2014" OR "2015" OR "2016" OR "2017"). The search was run in World-cat yielding a large number of publications. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions.

As it stands, 104 countries have ratified the Nagoya Protocol on Access and Benefit-Sharing (Convention on Biological Diversity 2017). The ample role of IPLCs in negotiating the

Nagoya Protocol is well documented (Teran 2016) and the western research community has by now widely speculated on the potential effects of the protocol (Rose, Quave et al. 2012; Welch, Shin and Long 2013; Burton and Evans-Illige 2014; Atanasov, Waltenberger et al. 2015; Nijar, Louafi and Welch 2017). But, while the Nagoya Protocol Implementation Fund has contributed to IPLC's involvement in drafting national legislations for the implementation of the Nagoya Protocol (GEF 2015), information is limited regarding the direct implications of the implementation of the protocol for IPLCs or in the contribution of IPLCs in its integration in national laws (Robinson and Forsyth 2015, Sanbar 2015). Moreover, the literature is largely unaccesible to IPLCs themselves

While the direct input of IPLCs in national legislation is little recognized, at the international level IPLCs have made great progress in the recognition of their rights. For example, IPLCs have had an important role in elaborating codes of conduct for research (e.g. Consortium of European Taxonomic Facilities 2015, ISE Code of Ethics 2006) and there are examples in which IPLCs have been fundamental in defining access and benefit sharing (e.g., South African San Institute 2017). International involvement has indirectly influenced national legislation in positive directions. Similarly, the recognition of the value of ILK in conservation, resource management, community health, nutrition and food security, and preservation of cultural heritage (Vandebroek, Reyes-García et al. 2011; Guèze, Paneque-Gálvez et al. 2012; Danielsen, Jensen et al. 2014a,b; Molina, Tardio et al. 2014; Reyes-García, Aceituno-Mata et al. 2014; Franco, Hidayati et al. 2015; Lins, Lima et al. 2015; Randrianarivoly, Andriamihajarivo et al. 2016; Reyes-García, Fernández-Llamazares et al. 2016; Wilder, O'Meara et al. 2016) has facilitated the incorporation of inputs from ILK in national resource management policies (Kikvidze and Tevadze 2014). Local actors are increasingly holding key positions in research focusing on ILK and resource management (Albuquerque, Cooper et al. 2012; Quiroga, Menezes and Bussmann; Hanazaki, Firme Herbst et al. 2013, Corrêa Martins, Sousa Filgueiras and Albuquerque 2014; Acharya, Apaza et al. 2015; Ochoa and Ladio 2015; Kunwar, Baral et al 2016, Hart and Salick 2017). IPLCs contributions have also been included in many aspects of the REDD+ (Reducing emissions from deforestation and forest degradation) and in sustainable management of forests and enhancement of forest carbon stocks programs in developing countries (Ecosystem Marketplace 2015).

The implementation of the Nagoya Protocol and the much broader participation of IPLCs in research and resource management have also contributed to a broad shift in international research practice. Nowadays, the contributions of IPLCs, and more importantly, their rights to fully Free Prior Informed Consent (FPIC) and participation in research at all levels are largely recognized at institutional (Balick 2016), national (Bendix, Paladines et al. 2016), and international levels (Bussmann 2013; Bussmann and Sharon 2014; Vandebroek 2016). These rights include authorship and right to benefit from any use of research results. However, there are still examples of research projects simply ignoring local, as well as international legislation, but excluding IPLCs from any access or benefits of their research.

ILK under the Nagoya Protocol – the Chácobo in Bolivia: The slow ratification of the Nagoya Protocol in Bolivia highlights both the limitations and the influence of IPLCs in this

process. Bolivia only became a party to the protocol in 2017 (Convention on Biological Diversity 2017) and, although IPLCs were involved as granters of Prior Informed Consent (PIC) in the permitting process and regarded as important to safeguard biological diversity (Acebey, Apaza et al. 2008), little legal recourse existed to actually receive access to research publications or potential benefits. The Chácobo Ethnobotany Project, started in 2013, illustrates the change in IPLC rights during the process of implementation of the Nagoya Protocol. While previous studies had already repatriated research content to the Chácobo (Paniagua-Zambrana, Bussmann et al. 2011), the Chácobo Ethnobotany Project went much further to implement the Nagoya Protocol, while still maintaining publishing peer-reviewed articles (Bussmann and Paniagua-Zambrana 2014). As such, the project served as a guideline for national legislation. In this project, indigenous Chácobo counterparts were trained in interview and botanical techniques, conducted all fieldwork, and were involved as authors in the publication of the research results (Paniagua-Zambrana, Bussmann et al. 2017). In addition, results of previous studies on the Chácobo were translated and repatriated (Paniagua-Zambrana, Bussmann et al. 2014), and new results were not only published with Chácobo authorship, but also repatriated in popular form (Paniagua-Zambrana and Bussmann 2017), with a clear indication of Chácobo rights in all publications: "any work conducted in a community is carried out under the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization to the Convention on Biological Diversity, and the right to use and authorship of any traditional knowledge all informants is maintained, and any use of this information, other than for the scientific publication does require additional prior consent of the traditional owners, as well as a consensus on access to benefits resulting from subsequent use."

Aichi Target 17: By 2020 each Party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory and updated national biodiversity strategy and action plan.

Methods: The text below is based on a literature review using the following search terms as topics: TS=(("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR "aborigin*" OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR "ILK" OR "socioecological systems" OR "co-management" OR "protected areas" OR "tribal") AND ("biodiversity strategy" OR "biodiversity action plan" OR "participatory biodiversity plan" OR "national biodiversity action plan" OR "participatory biodiversity action plan" OR "Aichi Target 17")). *Indexes=SCI-EXPANDED, SSCI, A&HCI, CPCI-S, CPCI-SSH, BKCI-S, BKCI-SSH, ESCI, CCR-EXPANDED, IC Timespan=2000-2017.* The string resulted in 21 search results in Topic search on the Core collection of the Web of Science with a subscription at UA, of which 6 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions.

There is clear consensus that the inclusion of IPLCs and ILK have the potential to yield invaluable knowledge to national biodiversity strategies and action plans (NBSAPS) (Ayesegul and Jones-Walters, 2011, Tengo et al. 2014, Sutherland et al. 2013, Armatas et al. 2016), yet IPLC input to them is still scarce. For example, in a 2008 review of the conservation literature spanning the previous 25 years, Brook and McLachlan (2008) found

that only about 0.4 percent of conservation plans included ILK. In addition, only 20 Parties reported the involvement of IPLCs in the submitted NBSAPS (18%), indicating that only a minority of Parties has so far developed adequate participatory approaches (Adenle, Stevens, and Bridgewater 2015). The same is true for the national reporting processes (Forest People's Programme et al 2016). Ollerer, Molnár and Biró (2017) the national policy reports for the convention on biological diversity to understand the extent to which ILK and IPLC's were 1) included in the plans, and 2) recognized for their actual or potential contributions. The study concluded that 49% of the countries (covering 43% of the global terrestrial surface) do not mention ILK/TEK and/or IPLCs in relation to the maintenance/conservation of wild animal and plant populations and semi-natural ecosystems, even though many of these countries have significant areas of land under formal or informal IPLC management (REF). Barriers to ILK inclusion into conservation plans include bridging epistemological and ontological differences between knowledge systems (Löfmarck and Lidskog, 2017), low academic recognition of ILK (Farwig et al. 2017), and issues of scale and power (Beck et al. 2017). For these reasons, recent work has focused on the creation of frameworks and approaches to include ILK into conservation plans (Tengo et al. 2014, Sutherland et al. 2013). These frameworks attempt to create a methodology to incorporate input across diverse knowledge and value systems and promote collaboration and equitable knowledge exchange. Since there is a great potential for co-optation and coercion of ILK into Western-style governance and knowledge systems (Stevenson 2006, White 2006, Ross et al. 2010, Johnson et al. 2014, Whyte et al. 2016), these frameworks strive to keep ILK and the social systems that support them intact while creating a synergistic, multi-contextual outcome that is beneficial to all stakeholders. With the exception of IPBES, however, there have been no applications of these specific frameworks found in the literature.

The impact of achieving the target on IPLC varies greatly and is largely dependent on whether the territories are managed or co-managed by IPLC. In areas where territories are managed solely by the IPLC, progress (or lack of progress) on the target might have minimal impact to the IPLC. Rather, in these instances, ILK preservation is vital (Maffi 2005, Wehi et al. 2009a, Wehi et al. 2009b, Loh and Harmon 2014, Nabhan et al. 2016, Wilder et al. 2016). In situations where the land is co-managed and ILK is incorporated into management plans, IPLCs are often positively impacted and conservation efforts are greatly improved (Berkes et al. 1995, Gadgil et al. 2000, Borrini-Feyerabend et al. 2004, Rozzi et al. 2006, Berkes 2018). These results support the assertion by Forest People's Programme et al. (2016) that IPLCs are essential partners for achieving tangible outcomes at the local and national level and should be fully involved in the development, updating and/or revision of NBSAPs. Unfortunately, the engagement of IPLCs in the NBSAP process is not yet receiving sufficient attention and IPLC are not receiving the potential benefits of involvement in NBSAPS.

The extent to which IPLC are recognized, valued, and benefit from contributing to the target can be difficult to assess (Marques et al. 2016). The retroactive inclusion of IPLC into an existing biodiversity plan can highlight inequities and instances where the plans have been detrimental to IPLC (Galbraith et al. 2017). Conversely, the recognition of the value of ILK and the inclusion of IPLC in the formulation of management plans can bring great benefit to IPLC. Shimada (2015) provides an example of how multi-level natural resource management of semi-natural grassland in Tarōji, Nara, Japan successfully recognized and valued input from the local community. The local community benefitted because the multi-level natural resource management plan reversed the trend of decreasing traditional management practices and benefitted from the shared cost of management with the government at the local and prefecture level. In the Noto Peninsula of Japan, the incorporation of IPLC input into a

biodiversity plan to preserve rice paddy fields has been greatly beneficial to the IPLC (Chen 2016). The management plan based on multi-stakeholder input coupled sustainable tourism with the reinvigoration of traditional/cultural practices to achieve agricultural landscape conservation.

Brazil exemplifies a participative process which recognizes and values IPLC input in the elaboration of the National Biodiversity Strategy for 2020. IPLC input into the NBSAP is of particular importance in Brazil because 13.3% of Brazil's national lands are either quilombo or indigenous territories. Until 2010, Brazil's NBSAP consisted of a legal framework for the conservation and sustainable use of biodiversity and was published in a website which served as a national biodiversity clearing-house mechanism. In 2010, Brazil adopted a new approach aiming to compile these instruments into a single document. This new approach initiated an extensive participatory process to update the NBSAP and seek a consensus on the definition of National Biodiversity Goals for 2020. As part of this process, new regulation was produced with input from IPLC to strengthen the protection of the rights of IPLC. The participatory process specifically recognized and valued the ILK of local and indigenous women and solicited contributions from IPLC women's practices and knowledge in the processes of proposing, planning, constructing, decision making and implementation of policies, programs, and actions to preserve biodiversity. In addition, Brazil's NBSAP has developed an action plan for the increase in global, national and local biodiversity benefits through strengthening local supply chains based on regular access to Brazilian medicinal plants, promoting the improvement of IPLC's quality of life.

Aichi Target 18: By 2020, the traditional knowledge, innovations and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the Convention with the full and effective participation of indigenous and local communities, at all relevant levels.

Methods: The text below is based on a literature review using the following search terms as topics: TITLE-ABS-KEY ("indigenous community" OR "indigenous peoples" OR "local community" OR "aboriginal" OR "traditional ecological knowledge" OR "indigenous knowledge" AND "traditional management" OR "indigenous management" OR "biodiversity conservation" OR "biodiversity use" AND "inclusion" OR "respect" OR "participation" OR "integration"). The search was run in Scopus yielding 187 papers of which 89 were relevant to the topic. Additional papers were also selected from the authors' own literature database and from reviewer's suggestions.

Scientists began to seek inputs from IPLC and to consider ILK relevant for conservation in the 1980s. The move was driven by research highlighting ILK potential value for biodiversity conservation (Gadgil, Berkes, and Folke 1993; Khan, Khumbongmayum, and Tripathi 2008; Hegde et al. 2017; Irakiza et al. 2016; Shen et al. 2012; Assefa and Hans-Rudolf 2017; Conrad and Hilchey 2011; Porter-Bolland et al. 2012; Berkes, Colding, and Folke 2000), the trans-nationalization of the indigenous rights movement (Reyes-Garcia 2015; Benyei et al. 2017), the perception that survival of IPLC and ILK were threatened by globalization (Brundtland Report, 1987), and the realization that biological and cultural erosion could be intertwined (Zent 2009; Zent and Zent 2007; Maffi 2005). At a political level, the importance of integrating ILK into biodiversity conservation efforts was strongly acknowledged at the

1992 CBD (Reyes-Garcia 2015). Since then, researchers working on resource management, from fisheries (Sekhar 2004) to forestry (Cheveau et al. 2008) and in all regions of the world (e.g., (Ferroni, Foglia, and Cioffi 2015; Apostolopoulou, Drakou, and Pediaditi 2012; Hernandez-Morcillo et al. 2014) for the EU; (Sibanda and Omwega 1996; Marie et al. 2009) for Southern Africa; or (Daniels, Chandran, and Gadgil 1993; Vaz and Agama 2013) for South and South-east Asia), have called for effective integration and participation of IPLCs in conservation initiatives.

It is argued that integrating ILK in conservation efforts, and doing so following a knowledge co-creation approach, not only improves IPLC's engagement in conservation initiatives (Grainger 2003; Carpenter 1998; Andrade and Rhodes 2012), but it also benefits IPLC in many ways (see(Reyes-Garcia 2015)). For example, the approach can potentially help halt ILK erosion (Zent 1999, 2001; Zent and Lopez-Zent 2004; Reyes-Garcia et al. 2013; Shen and Tan 2012; Ford, Smit, and Wandel 2006; Alessa et al. 2008; Fernández-Llamazares et al. 2016; Fernandez-Llamazares et al. 2015; Gomez-Baggethun et al. 2010; Reyes-García et al. 2013)), strengthen IPLC's collective action capacity, land/resource rights, health, religious freedom, self-determination, intangible heritage protection, and control over how ILK is used (Cil and Jones-Walters 2011; Chitakira, Torquebiau, and Ferguson 2012; Baral and Stern 2010).

Integrating IPLC and ILK into conservation initiatives has been done through a variety of approaches including Integrated Conservation-Development Projects, Participatory Monitoring Projects, Technical Resource Centers, Transfrontier Conservation Areas, Other Effective Area-Based Conservation Measures (OECMs) or co-management regimes (Sanjayan, Shen, and Jansen 1997; Danielsen et al. 2000; Joseph 1997; Hanks 2003; Berkes 2004; Berkes 2007; Kothari, Camill, and Brown 2013; Ruiz-Mallén and Corbera 2013). However, researchers and IPLC have contested the real "participatory" nature of some of these projects and approaches (e.g., (Dressler et al. 2010; Khadka and Nepal 2010; Turreira-García et al. 2018; Sterling et al. 2017) as well as the real benefits for IPLC and even for conservation itself (West 2006; Büscher et al. 2017; Nadasdy 1999).

IPLC have also led conservation and ILK revitalization initiatives, such as sacred natural sites, and Indigenous and Community Conserved Areas (ICCAs), community-based radio programs, pedagogical materials, or codes of conduct to safeguard their cultural heritage (see the Tkarihwai'e:ri Code of Ethical Conduct or the Akwé:Kon Voluntary Guidelines, developed by IPLC as part of the CBD) (see (Nelson 2008; Brooks, Waylen, and Mulder 2013; Gavin et al. 2015; Berdej and Armitage 2016; Nilsson et al. 2016). Overall, these IPLC-led initiatives have enhanced IPLC's role as environmental managers and increased pressure to be included in environmental policy fora (for example the Ad Hoc Working Group on Article 8J, the Indigenous Peoples' Biodiversity Network, the World Council of Indigenous Peoples, the COICA or the World Conference on Indigenous Peoples).

Global citizen action (including IPLC environmental activism), social mobilization, and the use of modern technologies and social media have provided opportunities to IPLC from around the world to engage in communication and recognize similarities in their historical experiences and contemporaneous fights (Martin 2003; Earle and Pratt 2009; Lorenzo 2011; Salman and de Theije 2011; Sikor and Newell 2014; Benyei et al. 2017). ILPC have also formed alliances with advocacy groups (e.g., IWGIA, Cultural Survival, Survival International, ISE) that have led to the transformation of local disputes into international claims and to the emergence of a transnational indigenous movement (Hodgson 2002), which

has resulted not only in the inclusion of IPLC as political actors, but also gave visibility to the existence of their knowledge systems. Consequently, IPLC are increasingly active in global environmental fora and in a range of international organizations, such as the International Indigenous Forum on Biodiversity or the Indigenous Environmental Network (Schroeder 2010; Wallbott 2014; Nasiritousi, Hjerpe, and Linnér 2016), propelling a growing recognition of ILK in environmental negotiations and intergovernmental processes (Tengö et al. 2014) as well as discussions on IPLC intellectual property rights and the need to protect (in a legal sense) and preserve (in a practical sense) ILK (see (Zent 2009; Lakshmi Poorna, Mymoon, and Hariharan 2014) for further discussion). However, despite the positive trend, ILK is not yet fully respected and integrated in the implementations of the CBD. A preliminary analysis of the inclusion/presence of ILK in CBD country reports revealed that 49% of the signatory countries (covering 43% of the global terrestrial surface) do not mention ILK and/or IPLCs in relation to the maintenance/conservation of wild animal and plant populations and seminatural ecosystems (Ollerer et al. 2017). Moreover, IPLC continue to remain politically marginalized parties in most of the policy-making stages (Corson 2012), often dependant on opportunities provided by policy-maker/project-designers for participation (Harada 2003).

Knowledge databases and registers. IPLCs' knowledge databases and registers emerged before the signing of the CBD and are well recognized ways in which IPLCs contribute to revitalize ILK and contribute towards Target 18. In some cases, the initiatives are fully developed by IPLCs. This is the case of the Makivik Corporation database (Canada). Since the James Bay and Northern Quebec Land Claims Agreement was signed (1975), the Inuits of Nunavik have taken over the task of management, development and control of their territory and resources. In order to complete this task, the western-based scientific reports were found to be lacking local and culturally sensitive information, and so there was a need to include Inuit perspectives. The Makivik Corporation then created the Research Department and a program that aimed at collecting all land use and ecological knowledge data for the entire region using interviews with Inuit in each of the fourteen communities of Nunavik. The knowledge gathered is reviewed through a process of community verification and continual updating of the information. However, Inuit ILK has not been made publicly available due to the lack of intellectual property right protection mechanisms, although it is used to inform decision-making at the local level (Alexander et al. 2004). In some other cases, the initiatives are initially developed by research institutions and NGO's, although they can become autonomous. This is the case of the People's Biodiversity Registers, which aim at providing a record of local knowledge that would help with its revitalization and protection. During 1996 and1998, 52 documents were prepared by coalitions of NGO's, researchers and locals (approached individually and in public meetings) in village clusters distributed through India. Copies of each PBR were left to be held locally by panchayats (local level elected councils), educational institutions, and district level "biodiversity cells" that configured a diverse stakeholder group that would be in charge of updating and validating periodically the information from the registers. Also, these biodiversity cells (later called Biodiversity Management Committees) hold legal ownership rights over the content of the registers. In this way, befit sharing rights can be enforced, since these local committees are the ones in charge of controlling access and collecting (with the *panchayats*) the fees generated from the use of this knowledge by bioprospectors, as according to the Biological Diversity Act and Rules (Gadgil et al. 2000; Downes and Laird 1999)

Aichi Target 19: By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.

Methods: This summary statement is based on a literature review using the following search terms as topics: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR "aborigin*" OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR "ILK") AND ("biological diversity knowledge" OR "biodiversity knowledge" OR "cultural values" OR "culturally valuable species" OR "knowledge loss" OR "knowledge transfer" OR "technology" OR "Aichi Target 19"). The string resulted in 299 search results in Topic search on the Core collection of the Web of Science with a subscription at University of Oxford, of which 97 were pertinent. Additional references were selected from the author's own literature database.

IPLC's diverse wealth of cultural and ecological expertise has informed almost all colonization, development, and conservation initiatives, though rarely acknowledged or accounted for in its entirety (Mann 2012). While IPLCs have been sharing their expertise and know-how with the global knowledge community in myriad undocumented ways, here we only review the ways in which ILK has been increasingly occurring in research projects undertaken by IPLCs or in collaboration with outsider researchers and under the aegis of a variety of ecological, sociocultural, and policy/development disciplines (Fung and Wong 2017; Sillitoe and Marzano 2009).

Technological cross-fertilization has long occurred, with IPLC biodiversity-sustaining technology and know-how being adopted and adapted to wider use and vice-versa (Berkes et al. 2000; Lynch et al. 2010; Jasmine et al. 2016). Recent technology and knowledge sharing is occurring in the use of drones (Paneque-Galvez et al. 2017), and in community mapping (Assumma and Ventura 2014; Heckenberg 2016) and counter-mapping (McLain et al. 2017); cloud computing (Valencia Perez et al. 2015) and other information and communication technology (Bazilchuk 2008; Coleman 2015) applications for local biodiversity conservation, including citizen-science and knowledge networks initiatives (Bortolotto et al. 2017; Wyndham et al. 2016) and projects to return control over biodiversity and landscape narratives to heritage owners (Bolhassan et al. 2014; Cairney et al. 2017; Thompson 1999), including royalty and other monetary negotiations (Gomes Souza et al. 2017; Robinson et al. 2016). In addition to data-points, know-how, and ILK, IPLC education systems and traditional institutions for knowledge transfer are beginning to be recognized and valued in research and policy (Fernández-Llamazares and Cabeza 2017; Kawharu et al. 2017; Walsh et al. 2013; Wuryaningrat et al. 2017), as well as the value of diversity in knowledge systems, including gendered (Fillmore et al. 2014; Wirf et al. 2008), age-class (Bayne et al. 2015), and intra- (Saynes-Vasquez et al. 2016) and intercultural differences (Reyes-García et al. 2016). ILK has been particularly prominent in the domain of climate-change studies, giving insights (Baztan et al. 2017), provoking dialogue (Bone et al. 2011; Pareek and Trivedi 2011) and highlighting adaptability through cultural values and institutions (Klenke et al. 2017; Walshe and Argumedo 2016).

Progress towards achieving Aichi 19 might increase the visibility of IPLCs' key and pervasive contributions towards biodiversity maintenance, and should raise their status as co-equal partners in the production, interpretation, and application of biodiversity-protection measures

(Tengö et al. 2014). One overarching lesson from the literature on IPLC and biodiversity knowledge is that ideology (Gorman and Vemuri 2012; Oviedo and Puschkarsky 2012), social organization (Elands et al. 2015), cultural/spiritual values (Cocks et al. 2012; Daye and Healey 2015; Oleson et al. 2015; Thondhlana and Shackleton 2015), politics (Wartmann et al. 2016), and ontology (Clarke 2016) play a big part in structuring local ecological relations. Industrialized societies can learn sustainable living from IPLCs (Kaltenborn 2017; Soh and Omar 2012; Masferrer-Dodas et al. 2012). Conversely, from an IPLC perspective, negotiating ideological difference imposed from outside can be onerous and damaging (Islam and Berkes 2016; Lee 2016; Lemkuhl 2017; Turner et al. 2008), and land rights are crucially important (Schmidt and Peterson 2009; Thondhlana et al. 2016). IPLC are particularly vulnerable to lack of progress towards Aichi 19 in that their economies and identities are often inextricably connected to local land and waterscapes (Fox et al. 2017) and they have been historically disadvantaged in terms of information access and equal participation in decision-making (Smith 1999). De-colonization of curricula, museums, and libraries are steps towards reducing historical power-information imbalances (Bangsbo 2008; Falkowski et al. 2015; Gordillo Sanchez 2017; Ladio and Molares 2013; Pulla 2017; Zolotareva 2015).

Achieving Aichi 19 can strengthen sustainable local foodways (Kamal et al. 2015; Turner and Turner 2007; Turreira et al. 2015), secure land tenure, health and wellbeing (Catarino et al. 2016; Jack et al. 2010; Lah et al. 2015; Phondani et al. 2013), and ecological resilience (do Vale 2007; Leonard et al. 2013). Ecotourism projects increasingly recognize the need for community control over biodiversity-reliant development (Karst 2017; Mendoza-Ramos and Zeppel 2011; Rommens 2017). The valuation of biodiversity in an ecosystem services paradigm is beginning to include more local cultural values (Afentina et al. 2017; Sangha and Russell-Smith 2017) or to raise a flag as to problems created for IPLCs (Preece et al. 2016). IPLC involvement in environmental impact assessments (Nakamura 2008), species (Housty et al. 2014; Gichuki and Terer 2001) and water/land management planning (Flood and McAvoy 2007; Harmsworth et al. 2016; LaFlamme 2007) are increasingly *de rigeur*.

CONECT-e (Compartiendo el CONocimiento ECologico Tradicional, www.conecte.es) is a Wikipedia-like citizen-science platform to promote the conservation and sharing of traditional knowledge in Spain. The platform is for anyone interested in gathering and sharing ILK, and has dedicated sections on wild plants, landraces, and ecosystems (already active), local indicators of climate change (forthcoming), and animals and minerals (planned). Registered users from any geographical location can enter information about any of these topics by adding text or through a map. Non-registered users can access verified information, for which CONECT-e contributes to expand ILK exchanges that typically happen within limited geographical areas. By November 2017, the platform (launched in March 2017) had about 150,000 views and 350 registered users, who contributed information on 2747 wild plant species and 421 landraces. In addition to its potential to cover spatial and thematic gaps in the documentation of Spanish traditional knowledge, CONECT-e addresses ILK protection and promotion in innovative ways. For example, CONECT-e promotes the validation of knowledge within the same knowledge system (Tengö et al 2014) through a two-step process. First, all data entered in CONECT-e are reviewed by a multidisciplinary expert group including scientists and traditional knowledge experts who ensure that the data actually captures ILK. Second, data entered are subject to a peer-to-peer validation process, according to which participants can "like" or "agree" with information posted in the platform. CONECT-e also addresses ILK misappropriation issues, a big concern of the civil society organizations. All the content of the platform is protected under a copy left license, which guarantees non-exclusion by allowing reproduction and exchange. Active participation of

IPLC in this project can be difficult due to the technological gaps between ILK holders and the digital environment, which is why the project was framed under a citizen-science approach (encouraging the general population to collect and share their elders' ILK). Moreover, this cannot be considered a bottom-up project since the initiators of the project were academics. However, the case study reflects the result of the literature review in that we rarely find IPLC-led projects but rather projects in which IPLC participate in greater or lesser extent. Considering that the information obtained with CONECT-e will contribute to complete Spanish ILK inventories (Pardo de Santayana et al. 2014), this case is a good example of how to include ILK in national biodiversity conservation communication strategies and advance towards Aichi Target 19.

Aichi Target 20: Mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011-2020 [LA: Victoria Reyes-García; CA: Diego Pacheco] Methods: The text below is based on an extensive literature review at decisions of the Convention of Biological Diversity and related documents since 2010 (when it was approved the Strategy for Resource Mobilization-SRM for achieving the Aichi Biodiversity Target at the CBD).

The CBD COP11 agreed doubling total biodiversity-related international financial resource flow to developing countries by 2015 to fund the twenty Aichi Targets (CBD 2012). This decision included financing through the replenishment of the main financial mechanism for the implementation of the CBD, the Global Environment Facility (GEF) (UN 1992), as well as other funds (CBD 2012). Countries are requested to identify resource needs, to develop their national resource mobilization strategies, and to provide information on their own financing of the goals through the Financial Reporting Framework-FRF (CBD 2014). The last COP13 expressed concerns regarding the insufficient information gathered from the FRF and of the limited progress towards achieving the overall financing targets (CBD 2016). The CBD Invites Parties to include IPLC in planning and implementing national biodiversity strategies and action plans, and thereby to contribute to the achievement of the Strategic Plan for Biodiversity 2011-2020 (UNEP 2012. UNEP/CBD/COP/DEC/XI/2). However, no specific procedures have been developed to fill this gap (Pacheco 2014). Moreover, there is an important gap regarding the needs of countries to fulfill their National Biodiversity Strategy Action Plan-NBSAP and the resources given to them. Countries generally do not specify how much is given to IPLCs and what are their direct needs (CBD 2016).

Although there are limited data on mobilisation of financial resources, especially in relation to domestic funding for biodiversity initiatives (CBD 2014), it is recognized that for small organizations in general and IPLC in particular it is difficult to access the financial mechanisms set in place to contribute to achieve the Aichi Targets (FPP, IIFB, and CBD 2016). To date, GEF has supported 160 full- and medium-sized projects involving IPLC (FPP, IIFB, and CBD 2016). Despite an overall positive trend, with a growing number of GEF-funded projects involving IPLC over the past years (CBD 2016), in 2015 only about 15% of the GEF Small Grants Programme (GEF-SGP), a scheme which enables GEF to partner with IPLC globally, involved indigenous peoples (GEF 2015). The global assessment of the High-Level Panel on Global Assessment of Resources for implementing the Strategic Plan for Biodiversity 2011-2020 estimated that between US\$150 billion and US\$440 billion per year would be required to achieve the Aichi targets, while only US\$4,2 billion has programmed by the GEF between 1991 and 2014, and only US\$228 million has been

financed to IPLC in last years (CBD 2016). Others sources of funds include the IFAD Indigenous Peoples' Assistance Facility-IPAF, which funds projects designed and implemented by IPLC, and the CBD LifeWeb Initiative, launched at COP9 to help bridge the funding gap for achieving Aichi Target 11 (FPP, IIFB, and CBD 2016).

A process has been initiated to estimate the contribution of IPLC to the conservation and sustainable use of biological diversity. In 2008, the COP9 approved the Strategy for Resource Mobilization-SRM to assist Parties in establishing national targets and action for enhancing international financial flows and domestic funding for biological diversity (CBD 2008), and in 2012 COP-11 decided to adopt targets for resource mobilization. However, the framework of the CDB only recognized the contribution of the private and the public sectors, largely ignoring the contribution of IPLC (Pacheco 2014). In this context, Bolivia proposed to include in the SRM the contribution of IPLCs collective action in achieving the Aichi Targets, a proposal that was adopted in COP11 (CBD 2012). Furthermore, measurement of the collective action was included in the FRF for the mobilization of financial resources oriented towards achieving the Aichi Targets (CBD 2016). The methodology for measuring the COP13 (CBD 2016). It has been said that with a relatively modest increase in financial resources and support, IPLCs contributions to achieve Aichi Targets through collective action could be even greater (FPP, IIFB, and CBD 2016).

Since the recognition of collective action, IPLC are recognized in the achievement of the Aichi target 20. Local users' efforts are often very important for a country's effectiveness to protect biodiversity, although these contributions are made in-kind rather than as monetary contributions. The recently proposed methodology by the CBD offers tools to assess these contributions quantitatively (e.g., impact on rates, extent, direction of environmental change) and qualitatively (e.g., impact of formal and informal rules regarding resource use and management) (CBD 2014). Additional modules could be developed and integrated to allow the estimation of different types of value (e.g., cultural, economic, social). IPLC noted that from the perspective of governments, assessing collective actions may require investments (CBD 2015). A strong argument for further and continued investment in local initiatives is that the outcomes often serve multiple policy objectives, including community development, environmental recovery and cultural wellbeing, while being highly cost-effective. However, IPLC will also benefit, for example through the strengthening of public policies regarding indigenous rights, poverty reduction, food security and food sovereignty, maintenance of biodiversity and ecosystem services and functions, cultural heritage and other aspects of sustainability. It was also indicated that it is necessary to create a list of non-monetary indicators, non-monetary units, and non-monetary evaluation as a key component of next phase of CBD and implementation (CBD 2015).

Conceptual and Methodological Framework for Evaluating the Contribution of Collective Action to Biodiversity Conservation. The policy relevance of this framework rests on resolution XI/4, paragraph 23, taken during COP11 of the Convention on Biological Diversity, which requested the development of an approach to assess the contribution of IPLC collective action to biodiversity conservation. Following this statement in resolution XI/4 and the guidelines for the Fifth National Reports of the CBD, the proposed conceptual framework and methodology aims at supporting countries to assess and report the contribution of collection action for biodiversity for the implementation of the Strategic Plan for Biodiversity for 2011-2020, including the development of country-specific frameworks for the mobilization of financial resources that consider the contribution of IPLC to the national strategy for biodiversity conservation. The methodology links geospatial analysis - the analysis of environmental change at different scales- with the analysis of institutional arrangements that examine the underlying mechanisms of local individual and collective action to protect biodiversity and ecosystems. From this methodology different indicators can be generated to evaluate the relationship between collective action and biodiversity conservation, with respect to resource mobilization (CBD 2014).

S3.4 Methods for literature search for assessment of progress towards SDGs

The intent of the literature review for the section on SDGs was to ensure that the relationships between Nature and Nature's Contribution to People (NCP) and the achievement of the goal and associated targets were well-supported in the literature. Because the establishment of the SDGs is relatively recent, there are virtually no analyses that directly assess the status of Nature and NCP in achieving SDGs or their targets. Assessment of trends in aspects of Nature and NCP and contribute to progress towards the SDGs was assessed either through the authors' knowledge of the literature on the subject or, for SDGs 2, 11, 13 and 14, through comprehensive searches of the peer-reviewed and grey literature (see Table S3.4 for search terms). For the latter, only articles from the last 10 years were included and only the first 25 relevant articles were assessed. Where a search returned many less relevant papers, papers were sorted on number of citations. In many cases, additional, targeted searches were needed to identify relevant papers on topics relevant to individual targets. Supplementary searches are required for SDG 14.

SDG target	Search Terms	
SDG 2.1	meta-analy* AND food security AND wild foods	
	literature review AND food security AND wild foods	
	systematic review AND food security AND wild foods	
	systematic review AND food security AND biodiversity	
	meta-analy* AND food security AND bushmeat	
	systematic review AND food security AND bush meat	
	literature review AND food security AND bush meat	
SDG 2.2	systematic review AND malnutriton AND biodiveristy	
	meta-analy* AND malnutrition AND biodiversity	
	literature review AND malnutition AND biodiversity	
	systematic review AND malnutriton AND Ecosystem services	
	literature review AND nutrition AND ecosystem services	
	meta-analy* AND malnutrition AND Ecosystem services	
	meta-analy* AND nutrition AND crop diversity	
	literature review AND nutrition AND Crop diversity	
	systematic review AND nutrition AND Crop diversity	
SDG 2.3	systematic review AND small-scale AND agricultural productivity	

Table S3.4 Search terms for literature search for assessment of progress towards SDGs.

	systematic review AND small-scale AND sustainable agriculture		
	systematic review AND small-scale AND sustainable farm*		
	literature review AND small scale AND sustainable fish		
	meta-analy* AND small-scale AND agricultural productivity		
	meta-analy* AND small scale AND sustainable fish*		
	meta-analy* AND small scale AND sustainable fish*		
	systematic review AND small scale AND sustainable fish*		
	literature review AND small scale AND agricultural productivity		
	literature review AND small-scale AND agricultural productivity		
	meta-analy* AND small-scale AND sustainable agriculture		
	meta-analy* AND small-scale AND sustainable agriculture		
	meta-analy* AND small-scale AND sustainable farm*		
	literature review AND small scale AND sustainable farm*		
SDG 2.4	systematic review AND food production AND pollinat*		
	literature review AND food production AND pollinat*		
	meta-analy* AND food production AND pollinat*		
	meta-analy* AND food production AND pollinat*		
	systematic review AND food production AND soil		
	literature review AND food production AND freshwater		
	tematic review AND food production AND freshwater		
	meta-analysis AND food production AND freshwater		
	meta-analysis AND food production AND freshwater quality		
	meta-analy* AND food production AND freshwater quality		
	literature review AND food production AND freshwater quality		
	systematic review AND food production AND freshwater quality		
	systematic review AND food production AND freshwater quantity		
	literature review AND food production AND freshwater quantity		
SDG 2.5	meta-analy* AND genetic diversity AND seeds		
500 2.5	literature review AND genetic diversity AND seeds		
	literature review AND genetic diversity AND cultivated		
	meta-analysis AND genetic diversity AND cultivated plants		
	meta-analy* AND genetic diversity AND cultivated plant		
	meta-analy* AND genetic diversity AND seed bank		
	systematic review AND genetic diversity AND seed bank		
	literature review AND genetic diversity AND seed bank		
	literature review AND genetic diversity AND plant bank		
	meta-analysis AND genetic diversity AND plant bank		
	meta-analysis AND genetic diversity AND plant bank		
	systematic review AND genetic diversity AND Plant bank		
	systematic review AND genetic diversity AND domestic		
	literature review AND genetic diversity AND domestic animal		
	eta-analysis AND genetic diversity AND domestic animals		

	meta-analy* AND genetic diversity AND domestic animal		
	literature review AND genetic diversity AND farmed animals		
	systematic review AND genetic diversity AND farmed animals		
SDG 14.1	(meta-analysis OR literature review OR systematic review) AND (marine OR ocean OR coastal) AND (pollution OR "run-off" OR nutrients OR "water quality" OR plastic OR "marine debris")		
SDG 14.2	(meta-analysis OR literature review OR systematic review) AND (marine OR ocean OR coastal) AND (sustainable management OR marine protected area OR fisheries management)		
SDG 14.3	(meta-analysis OR literature review OR systematic review) AND (ocean acidification)		
SDG 14.4	(meta-analysis OR literature review OR systematic review) AND (global) AND (fishing OR fisheries) AND (status OR trends)		
SDG 14.5	(meta-analysis OR literature review OR systematic review) AND (marine resource* OR ocean resources OR coastal resource OR fisheries OR aquaculture) AND (tourism OR benefits OR economic benefits OR jobs OR revenue OR livelihoods) AND sustainable		
SDG 14.6	14.6 (meta-analysis OR literature review OR systematic review) AND (marine resource* OR ocean resources OR coastal resource OR fisheries OR aquaculture) AND (subsidies OR overcapacity OR ille fishing)		
SDG 14.7	(meta-analysis OR literature review OR systematic review) AND (marine resource* OR ocean resources OR coastal resource OR fisheries OR aquaculture) AND (tourism OR benefits OR economic benefits OR jobs OR revenue OR livelihoods) AND sustainable		

S3.5 Further information on progress to the Sustainable Development Goals

SDG2.2

Alternative farming methods and systems to support progress to this target.

Intercropping farming methods that co-plant dicots and gramineous species, such as in peanut-maize, wheat-chickpea or guava-sorghum mixtures, can naturally biofortify these crops, increasing their iron and zinc content --two important micronutrients that are often deficient in peoples' diets (Zuo and Zhang 2009). Associations between crop roots and arbuscular mycorrhizal fungi in soils, can also increase the zinc content of crops (Lehmann et al 2014). Many non- and under-utilized crops, which have not been the focus of intensive breeding programs, offer also important potential sources of diversity and nutrition in local diets (Frison et al. 2006). For example, traditionally cultivated minor millets in India (Bergamini et al. 2013), amaranth in the Andes (Padulosi et al. 2014) or wild leafy greens in Africa (Uusiku et al. 2010) are higher in essential micronutrients than exotic commercially

grown crop and are well-adapted to local environmental conditions. Efforts to domesticate and improve access to high nutrient foods, as well as incentives to encourage subsistence or commercial production of them could help to increase demand and consumption of these micro-nutrient rich species (Bvenura et al. 2017).

Diversified farming systems that promote mixed crop production and wild species can yield benefits for farmers' diets (Bellon et al. 2016). A review of the literature by Jones (2017) has shown that increasing agricultural biodiversity in small-holding farming systems has a small, but consistent positive impact on household and individual diets in low- and middle-income countries (Jones 2017).

SDG2.3

Increasing smallholder farmer access to improved crop varieties, high quality seed and inputs

Increasing smallholder farmer access to improved crop varieties, high quality seed and inputs led to dramatic yield increases during the Green Revolution from the 1920s-1960s (and its negative environmental outcomes, see target 2.4). To be successful they must be adapted to the smallholder contexts of rural Africa and Asia (Pingali 2012). In these landscapes, poor soils and arid climates hamper the use of the classic 'high-yielding' varieties that were developed for more productive regions that require high fertilizer and water inputs (Pingali 2012). Instead, a second generation of crop breeding and practices, targeted to these more challenging agroclimatic conditions and improved soil management techniques to unlock the potential of biodiverse soils (del Mar Alguacil et al. 2010) will be needed to boost yields and incomes. These also need to be conceived and developed in ways that aim to avoid the negative environmental trades offs of water pollution, eutrophication, erosion and biodiversity loss that accompanied the rise of industrialized agriculture.

Many smallholder farmers lack access to improved and quality controlled seed, which can lead to high pre- and post-harvesting losses (Barnard and Calitz 2011) due to vulnerabilities to climate, pest and diseases (Diaz et al. 1998), thereby lowering incomes. The research and technological investment that has gone into improving many of our staple crops (e.g. rice, wheat, corn, potato) has not been invested in many crops that support livelihoods of smallholder farmers in Africa, Asia or South America. Crops such as yams, taro, sorghum, millets, cassava, which are the mainstay of households in many countries, lack proper development and control of seeds and germplasm to improve and maintain quality. Traditional breeding for select genetic traits in cultivars can improve crop harvests by selecting for traits such as hardiness, timing of seed set, seed size and number amongst others. The genetic diversity within crop species can also be drawn upon to breed crops for improved quality (e.g. taste, appearance, cooking properties) increasing their market value (e.g. Calingacion et al. 2014). Maintaining high levels of both wild and cultivated genetic biodiversity is important to increase production across smallholder farms (see also target 2.5).

Improving 'green water' retention

Soil organic matter is also critical to waterholding capacity as it supports the community of soil micro-organisms that maintain soil structure (Quinteros-Ramos et al. 1993, Lehmann and Kleber 2015). The use of organic production techniques can have the added benefit of increasing farmer income through prices premiums and lower cost:benefit ratios for organically certified products (IPES-Food 2016). While not necessarily advantageous in all landscapes, Niggli (2015) showed through meta-analysis that organic agricultural practices on disadvantaged sites and climate conditions can have equal or even higher productivity

than conventional agriculture in subsistence farming in sub-Saharan Africa. While organic farming tends to have lower production in developed countries, organic polycultures can improve production dramatically in many developing country contexts (IPES 2016).

SDG2.4

Agroecological practices

Agroecological practices include amongst others intercropping, crop rotation, riparian buffers, non-cropped vegetation, and diversified intensification as well as conservation or notillage agriculture (Atwood et al. 2017). Sustainable intensification practices, including their transferrability potential, have been evaluated at multiple spatial scales in specific socioecological conditions and contexts in which these may contribute to both improved agroecology and food security (Sietz et al. 2012, Sietz et al. 2017). Systems of Crop Intensification (SCI) are being developed for wheat, millets, pulses, sugarcane, legumes and beans (Abraham et al. 2014).

Integrated pest management (IPM) which relies on mechanical (e.g. weeding), biological (e.g mycorrhizal fungi) and (bio)chemical interventions which support diverse soil communities that prevent fungal outbreaks and other pathogens can be used to reduce pesticide use (Silverio et al 2009, Correia et al. 2013). However concerted adoption of IPM, particularly in developing countries, remains low due to high knowledge and time commitments required (Parsa et al. 2014).

Increasing agrobiodiversity within farming fields and farming landscapes plays an important role in promoting production and resilience of farming systems (Isbel et al. 2017). A review of diversified farming systems found that these practices improve soil quality and water holding capacity, crop uptake of nutrients, sequester carbon, support substantially more biodiversity and have greater resistance and resilience to climate change (Kremen and Miles 2012). Growing multiple varietals of a crop together can provide buffering effect and insurance against crop failure under environmental stress (Di Falco et al. 2010), rebuild soils and protect against pest damage (Hajjar et al. 2010). Management of soil symbionts, such as mycorrhizal fungi, can increase the absorption capacity in mineral elements P, N, Zn, Cu (Quintero-Ramos et al. 1993, Cardoso and Kuyper 2006) and is able to fix up to 90% of P and 40% of N deficiencies (Mäder et al. 2000) reducing need for external inputs. The maintenance of native vegetation and wild habitats within the agricultural matrix is also important for sustaining wild biodiversity, providing pollination and pest control services and boosting yields (Kremen et al. 2004, Bianchi et al. 2006, Garbaldi et al. 2014). The volume of production of pollinator dependent crops has increased by 300 per cent over the last five decades, making livelihoods increasingly dependent on the provision of pollination (IPBES Pollinator Assessment 2016), however wild pollinator populations are declining in North America and North Western Europe. Recent modelling analysis of the US landscape suggest that 39% of pollinator-dependent crops may experience pollinator shortages based on lack of adequate habitat within the farmland matrix (Koh et al. 2016). In general, mixed crop and livestock farming exhibit more sustainable practices, and these practices can be practiced and beneficial in both small-scale and large-scale farming systems (Rudel et al. 2016). Trends however in these practices are mixed or limited showing poor progress to this target (Table 3.8).

Composition and configuration of farmed landscapes have also been shown to influence pest dynamics (Bianchi et al. 2006). Patchy landscapes with high levels of non-crop habitat harbour larger populations of natural pest enemies, greater parasitism near forest edges and

lower pest pressure in crop habitats (Bianchi et al. 2006, 2008). While some invertebrates are pests, many invertebrates positively affect ecosystem services provisioning (e.g. decomposers, pollinators, pest predators) and are also highly responsive to pesticides use and to climate change (Prather et al. 2013). Measures to encourage, or at least not target, beneficial insects need to be adopted. For example, increasing the proportion of natural habitat in the surrounding landscape and allowing some weeds in fields has been shown to significantly buffer wild bees from the effects of pesticides (Park et al. 2015), and help support viable populations of pollinators (Nicholls and Altieri 2013, Kremen and McGonigle 2015, Tschumi 2018).

Reducing fertilizer inputs

N2O emissions rates increase in a non-linear manner with N-application (van Groening et al. 2013, Gerber et al. 2016), suggesting that at moderate N-applications N2O yield efficiencies are maximized (van Groening et al. 2010). The energy intensity can also be reduced by switching from conventional tillage systems (412-760 MJ/ha) to no-till systems (80-284 Mj/ha) due to lower fossil fuels use for machinery (Verma 2015) or by incorporating biomass-biofuel technologies into farm production. For example herbaceous biomass buffers can increase carbon storage and be harvested for electricity generation to power on-farm work (Ferrarini et al. 2014) while use of rice straw and husks for bioenergy can offset the costs and fuel-needed to operate irrigation pumps in off-seasons (Chitawo et al. 2017), Integration and management of soil symbionts can also make a significant contribution to reducing such dependence and waste on fertilizers. Practices favouring mycorrhizal symbionts can facilitate bio-fertilization (reviewed in Jordan et al. 2000), improve soil structure, sequester carbon and help to improve water-holding capacity (see target 2.3) to cope with harsher climatic conditions.

SDG3.3

Relationship to N and NCP for specific diseases

Malaria

Malaria is multifaceted and has many contributing factors, but ecological factors play a major role in driving transmission dynamics (Kar et al. 2014). Despite widespread and concerted efforts, malaria continues to be one of the leading causes of global disease burden (Murray et al. 2010). There were approximately 207 million cases of malaria in 2012 and an estimated 627,000 deaths. Malaria mortality rates have declined by 42% globally since 2000, and by almost half in Africa. Most deaths still occur among children living in Africa, but since 2000, malaria mortality rates among children in Africa have been reduced by an estimated 54% (WHO 2014). Malaria is a newly recognised cause of adult global disease burden, making up 22.6% in adults over 15 years old (Murray et al. 2010).

Forest loss, habitat fragmentation and modification, and the accompanying loss of plant diversity have been shown to affect the risk of malaria transmission through changes in mosquito behaviour, abundance, survival and distribution (Vittor et al. 2006; Yasuoka and Levins 2007). Deforestation and resulting development and human settlement can create breeding sites for malaria-transmitting mosquitoes, but there are regional differences. Warmer microclimates often speed up mosquito reproduction rates, as well as development times of the pathogen in the mosquito (Giles 1999). In the wake of deforestation, malaria increased as a result of higher density of the mosquito species *Anopheles gambiae* and *A. arabiensis* in the Sahara, and of *A. funestus* and *A. gambiae* in sub-Saharan regions (Yasuoka and Levins 2007). Changes in biodiversity due to deforestation have also been demonstrated

to have adverse effects on the risk of malaria (increasing numbers of *A. darlingi* and increased biting rates) in the Brazilian and Peruvian Amazon (Vittor et al. 2009).

Mosquito species show significant variation in their preferred habitats and feeding behavior, both of which affect malaria control efforts (Morgan et al. 2013) and relationships between biodiversity loss and disease prevalence. In some parts of Asia, deforestation appears to be linked to lower malaria incidence when malaria is transmitted by forest preferring mosquitoes such as *Anopheles dirus* in Thailand. In other parts of Thailand and elsewhere in Southeast Asia, deforestation led to increased malaria transmission by *A. minimus* in Thailand and India; *A. philippinensis, A. annualaris* and *A. varuna* in India; and, *A. culicifacies* in Nepal and Sri Lanka (Kar et al. 2014). Human factors, such as immune status, migration patterns, and treatment of disease, also play important roles in malaria incidence and continued transmission (Molyneux 1998).

Lyme disease

When landscape changes are fragmented, there can be biodiversity loss and changes in the composition of the animal host community, leading to changes in the dynamics of some tickborne diseases, such as Lyme disease in the northeastern U.S. (LoGiudice et al. 2003). The rates of infection of ticks by the pathogen associated with Lyme disease increases as animal community composition is reduced. Increased species richness of non-passerine birds, which are less competent reservoir hosts than passerines, was associated with decreased West Nile virus infection rates in mosquitoes and a decreased number of human cases (Ezenwa, Godsey et al. 2006). The loss or extinction of large predators due to hunting and land-use change can increase the population of a particular vector or host. This can result in an increased transmission of infectious disease to humans. Changing species composition in small fragments and conservation units remaining around the Atlantic Forest have resulted in growing cases in south eastern Brazil, as also seen with Lyme disease in the United States (Meira et al. 2013).

Ebola

The first documented outbreak of Ebola Virus Disease (EVD) was in 1976. Since that time, human populations in Africa have increased dramatically. Bats are suspected as one of the natural hosts of the Ebola virus although evidence remains largely unconfirmed and epidemiological links between fruit bats and human index-cases are limited (Leendertz et al. 2016 and Science 2017). Several species of fruit bats and a small number of other bat species can carry antibodies against Ebola virus and/or test positive for its viral RNA (Olson 2012). Great apes, duikers and pigs also appear to be susceptible to EVD, with some outbreaks traced back to hunters consuming ape and duiker carcasses that were likely infected from other animals or fruit bats (Rouquet 2005).

Current evidence for causality between environmental change and EVD outbreaks is still largely circumstantial but considerable land cover and population changes across Africa over the last 30 years have increased the likelihood of contact between people and wildlife, and thus the likelihood of zoonotic disease emergence (Hansen et al. 2013; Whitmee et al. 2015 and references therein). Wallace et al. also argue that the region's "policy-driven phase change in agroecology," in particular to oil palm, is a disturbance that could be increasing human and fruit bat contact in the dry season, when EVD outbreaks often occur (Wallace et al. 2014). Fruit bats are social animals that often congregate in large groups. Shifting resource or habitat availability could dramatically alter human disease risk by altering migratory patterns, group size, and connectivity along with other life history traits that are associated with zoonotic infections in bats (Plowright, 2011). Fruit bats and land cover change following development have also been linked to human outbreaks of Nipah virus in Malaysia (Pulliam et al. 2012), as well as to Hendra virus in Australia (Plowright et al, 2011).

Schistosomiasis

Schistosomiasis affects hundreds of millions of people every year, leading to malnutrition, stunting, anaemia, loss of worker productivity and poor school performance (King 2010). River fragmentation and biodiversity loss has led to an increase in the number of freshwater snails that act as vectors of schistosomiasis, which has in turn may lead to the extensive proliferation of the disease (Myers et al. 2009). Eutrophication and overfishing can also contribute to an abundance of the snail hosts which act as the intermediate vector of schistosomiasis (Myers et al. 2009). Schistosomiasis has also been identified as a co-factor in the spread and progression of HIV/AIDS in places where both diseases are endemic (Secor 2012) as a result of damage and inflammation in male and female genital tracts due to urogenital schistosomiasis (Bustinduy et al. 2014).

Rabies

Vultures provide a number of ecological, economic, and cultural services to humanity – the most important of which is likely their role as a scavenger of carrion and other organic refuse - removing dead and decomposing carrion from the environment quickly and efficiently, especially compared to less specialist facultative scavengers (Buechley & Şekercioğlu 2016). Currently, 14 of 23 (61%) vulture species worldwide are threatened with extinction, and the most rapid declines have occurred in the vulture-rich regions of Asia and Africa (Ogada et al. 2012). Key drivers of decline are poisoning by the veterinary drug diclofenac, and human persecution. In Kenva, the absence of vultures at carcasses correlated with longer decomposition times, increased numbers of mammals at carcasses (primarily hyenas and jackals), and increased direct contact between mammals at carcasses (Ogada et al., 2012a, 2012b). Increased contact among facultative scavengers is expected to increase the potential for disease transmission between themselves and ultimately to humans. South Asia provides an alarming example of this. In Asia, *Gyps* vultures have declined by >95% due to poisoning by the veterinary drug diclofenac, which was banned by regional governments in 2006 (Ogada et al. 2012). During this same time period, feral dog numbers increased by 7 million, despite widespread sterilization programs (Markandya et al., 2008). This increase in dogs resulted in 39 million dog bites from 1992 to 2003, causing an estimated 48,000 human rabies mortalities in India (Markandya et al., 2008).

SDG11.6

Waste management strategies

Accumulation of solid waste *in situ* contaminates the soil and water and has direct and indirect effects on surrounding ecosystems, with negative repercussions on NCP related to drinking water and food sources. Similarly, the burning of waste by residents as an alternative to municipal removal and disposal presents environmental hazards to air, soil, and water quality. Where urban centers lack the requisite governance, infrastructure, and resources to operate a conventional waste collection, NCP such as urban soil-based biodegradation processes can be useful for neutralizing certain forms of waste (Vauramo and Setälä 2010). Built infrastructures help to neutralize human-generated waste products, but can also entail reliance on energy-intensive systems (Rashidi et al. 2015; target 13.2). Municipalities may face technical constraints to implementing conventional waste management systems and may

need to continue relying on local sanitation practices based on socio-cultural systems (Andersson et al. 2016).

Another waste management strategy involves a reduction in the generation of solid waste byproducts through local food production. To that end, urban agriculture (UA) contributes by reducing reliance on external food supplies and the energy and material costs associated with transport, temperature- control, and packaging. Potential UA cultivation spaces include not only peri-urban fields and community gardens, but also backyards and rooftops (Andersson et al. 2007; Barthel et al. 2010). Additionally, UA can also have ecosystems benefits such as diminished greenhouse gas emissions and urban heat island effects, while concurrently increasing food security (Goldstein et al. 2016; Sukhdev 2013). At least fourteen major urban centers in developing regions of Africa and Asia generate a majority of their food energy needs in this way (Moustier 2007). It has been noted that "decentralized, self-organized foodproducing communities" have historically tended to generate surplus production through sustainable land-use practices (Barthel and Isendahl 2013).

Air pollution review

McDonald (2009) notes that air pollution originating from urban sources affects humans and nature, at local to regional scales, and in some cases even traversing hemispheres via global circulation patterns of the atmosphere (see SM). This public health impact is correlated with gradually increasing levels of fine PM exposure worldwide over the period 2000-2012, to which the majority of the global population (approximately 59-63%) has been exposed mostly in urban settings (Shepherd et al. 2016). There is wide disparity between different regions of the world. Annual mean levels of fine particulate matter (ranging in size from 2.5-10 micrograms) shows a global average of 37 micrograms as of 2012, with a wide degree of variability to either side of this mean value. Most cities in Asia have much higher values, with cities of Central/South Asia with levels of around 65 micrograms, and cities of East/Southeast Asia were measured at 46 micrograms. These levels are 4-6 times higher than Europe and North America on average, and 8-12 times higher than Australia/New Zealand on average (UNSD 2017).

SDG11.7

Public participation, access, and space availability

It is often taken for granted that urban public green spaces provide public participation but with little data (Fors et al. 2015). On the societal level, extending access to green spaces to these segments of the population connects with environmental justice objectives to promote more balanced access to ecosystem services that are disproportionately enjoyed by better off segments of society (Gómez-Baggethun et al. 2013). Studies of Stockholm, Sweden, and Cape Town, South Africa, among other cities, provide examples of using green space in a manner consistent with environmental justice objectives in urban areas. Such spaces should be of sufficient size to resist disappearance from construction or complete ecological degradation (Ernstson 2012).

On the spatial level, growing cities confront problems of space availability that impact upon human residents and nature, particularly green spaces. This is particularly true for cities in countries that have compact designs, with little area available for vegetation due to the configuration of buildings and roads (Jim 2013). This condition may also apply to both redeveloped and newly developed urban areas built quickly and without concerns for the environment (Jim 2013). To ameliorate this situation, Jim (2013) and others advocate for "urban greening" measures to identify suitable sites for planting vegetation, and providing necessary management and governance of these spaces. In the context of ongoing urbanization, such disparate green spaces (forests and parks, gardens, large trees, etc.), comprise an "urban green infrastructure" (UGI), that is highly important for allowing humannature interactions (Vierikko et al. 2016), and which requires active involvement by citizens on all levels of governance (Bujis et al. 2017).

Urban development poses a challenge in terms of available space and the quality of those spaces. Where private or public development reduce greenspace, greenspace losses need to offset by increases elsewhere and the quality of remaining areas needs to be increased (Haaland and van den Bosch 2015). Parks and protected areas can meet both human needs and meet ecosystem viability thresholds if sufficiently large in extent (>50 hectares) and adequately connected with other such areas. Where such large extents can be set aside, biodiversity will benefit significantly (Beninde et al., 2105). To that end, relatively low-cost restoration techniques using vacant lots could also greatly benefit low-income areas (Anderson et al. 2017).

Communities may also benefit from greenspace with respect to local ecological knowledge which tends to be lost in more affluent communities (Barthel et al. 2010; Pilgrim et al. 2008). Such knowledge preservation can boost adaptive capacity and resilience of communities (Buchmann 2009).

Figure S3.1. Limits and alternatives for global food security, showing (a) food production against agriculture-induced impacts, with green shading indicating a safe space where global food demands are met; and alternative scenarios of ecological (b) and continued conventional (c) intensification. Conventional intensification is expected to move systems towards the right, with increased impacts on ecosystems services and the environment. Even if conventional intensification moved systems into safe space above minimum global food needs, there remains little room for manoeuvre close to maximum attainable yields, posing increased risks under further environmental change. As systems move towards the right-hand boundary of the safe space, maximum attainable food production is expected to decrease due to degraded ecosystem services. Furthermore, negative impacts on the environment, biodiversity, and other benefits are expected to increase in this direction. A complementary strategy is to widen safe space by dampening demands for food products, such that minimum global needs for agricultural products are lowered. Source: Bommarco et al. (2014).

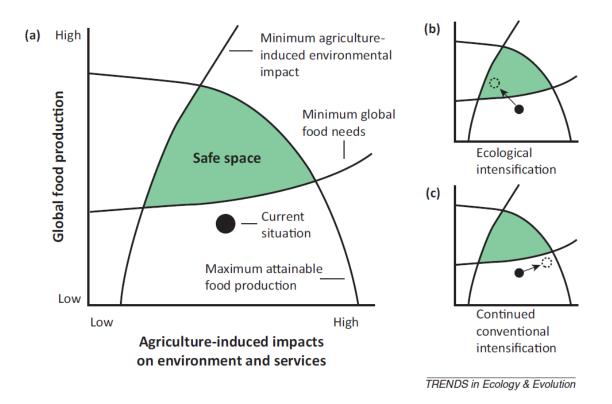


Figure S3.2. Benefits to health from exposure to natural environments. Source: Bowler et al. (2010).

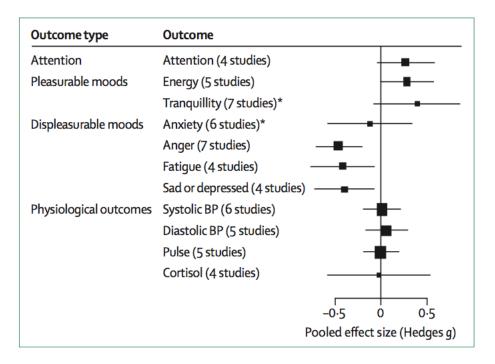


Figure S3.3. Challenges and solutions for promoting education for sustainable development. Source: UN Environment (2018).

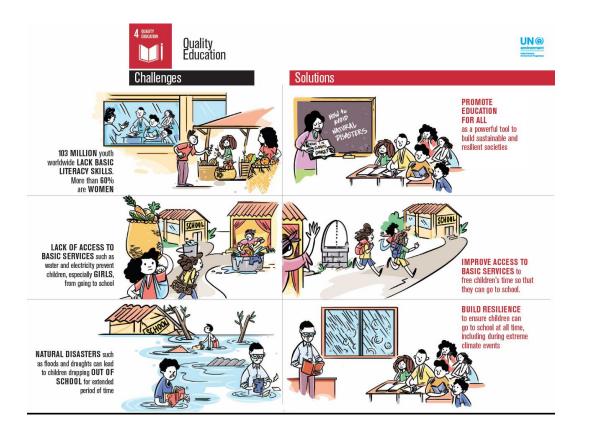


Figure S3.4. Interactions between inequality and the biosphere in social-ecological

systems. Inequalities exist both in society and nature. The biosphere, which underpins all of society, directly impacts inequality through environmental shocks or gradual environmental change. Inequality has an effect on the biosphere through human actions, which can be shaped by an individual's perceptions and sense of fairness, as well as their aspirations. Beyond the individual, inequality can influence the actions of groups or institutions by shaping cooperation in sustaining the local common, or through market concentration. The pathways of interaction that are highlighted here represent a sub-set of possible interactions, and a starting point for further research. Source: Hamann et al. (2018).

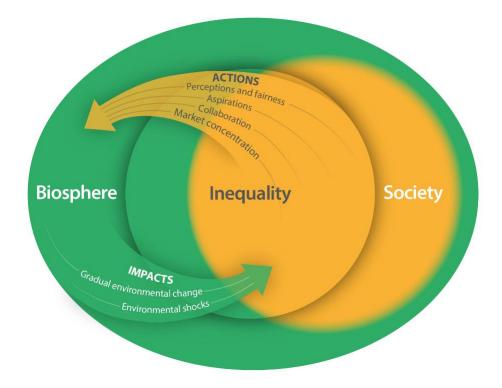


Table S3.5. Mechanisms linking biodiversity change and human health at differentlevels. Source: Pongsiri et al. (2009).

Level of diversity	Aspect of biodiversity undergoing change	Possible mechanism leading to human health effect	
Genetic	Gene frequencies within populations of pathogens or hosts	Change in pathogen virulence or host resistance	
Microbial	Composition of microbial communities in the external environment or within the host	Change in pathogen exposure or virulence; change in host immune response and allergic sensitization; expansion of range through anthropogenic transport	
Vector species	Abundance, diversity, composition, and geographic range of vectors	Change in host-vector contact rates; change in contact between infected vectors and humans; expansion of range through anthropogenic movement	
Host species	Diversity, composition, and range of host species	Change in host-pathogen contact rates; change in compe- tent host-vector contact rates; change in pathogen preva- lence; expansion of range through anthropogenic transport	
Community (interacting species including preda- tors, competitors, etc.)	Host density and contact with pathogen; host susceptibility to infection	Change in pathogen prevalence; change in human-pathoge contact rates	
Habitat structure	Structure, complexity, and diversity of vegetation	Change in vector abundance and composition; change in host composition and distribution; change in host-pathoger contact rates; change in vector-host contact rates; change in infected vector-human contact rates; change in host- human contact rates	

S3.6 Quantitative analysis of progress towards the Sustainable Development Goals

For SDGs 6, 14 and 15, all indicators listed in Table 3.5 were extrapolated to 2030 using the same methods as outlined in S3.2. The same criteria for selecting indicators to extrapolate

were used, except that the data had to date back to at least 1994, in order to avoid extrapolating into the future across a longer time period than we had historic trends for. Table S3.6 lists the indicators, their time series, results of the extrapolations and progress category.

Table S3.6 Indicators used in the quantitative analysis of progress towards SDGs 6, 14 and 15, the sampling dates, and projected trends. Official SDG indicators are indicated with an asterisk. Spatial coverage is scored as poor (1-2 continents, or 3-4 continents and <10 countries), moderate (3-4 continents and \geq 10 countries, or \geq 5 continents and <20 countries), or good (or \geq 5 continents and \geq 20 countries).

SDG	SDG Target	Indicator name	Sampling dates	Projected trend to 2030
6	6.6	Wetland Extent Trends Index	1970-2015	Significant decrease
14	14.1	Red List Index (impacts of pollution)	1988-2016	Significant decrease
14	14.4	Proportion of fish stocks in safe biological limits*	1974-2013	Non-significant decrease
14	14.4	Marine Stewardship Council certified fisheries (tonnes)	1999-2016	Significant increase
14	14.4	Red List Index (impacts of fisheries)	1988-2016	Significant decrease
14	14.5	Percentage of marine and coastal areas covered by protected areas*	1990-2016	Significant increase
14	14.5	Percentage of marine Key Biodiversity Areas covered by protected areas	1980-2017	Significant increase
15	15.1	Percentage of freshwater Key Biodiversity Areas covered by protected areas*	1980-2017	Significant increase
15	15.1	Percentage of terrestrial Key Biodiversity Areas covered by protected areas*	1980-2016	Significant increase
15	15.1	Red List Index (impacts of utilisation)	1986-2016	Significant decrease
15	15.1	Wild Bird Index (habitat specialists)	1968-2014	Significant decrease
15	15.2	Area of forest under sustainable management: total FSC and PEFC forest management certification (million ha)	2000-2016	Significant increase
15	15.2	Area of tree cover loss (ha)	2001-2016	Significant increase
15	15.4	Percentage of mountain Key Biodiversity Areas covered by protected areas*	1980-2017	Significant increase
15	15.5	Red List Index*	1994-2016	Significant decrease
15	15.5	Area of tree cover loss (ha)	2001-2016	Significant increase
15	15.5	Climatic Impact Index for birds	1980-2010	Significant increase
15	15.5	Living Planet Index	1970-2012	Significant decrease
15	15.5	Percentage of terrestrial areas covered by protected areas	1990-2016	Significant increase
15	15.5	Percentage of terrestrial ecoregions covered by protected areas	1911-2012	Significant increase
15	15.5	Number of protected area management effectiveness assessments	1990-2013	Significant increase
15	15.7	Red List Index (impacts of utilisation)	1986-2016	Significant decrease
15	15.7	Red List Index (species used for food and medicine)	1986-2016	Significant decrease
15	15.8	Number of invasive alien species introductions	1500-2012	Significant increase
15	15.8	Percentage of countries with invasive alien species legislation	1967-2009	No significant change
15	15.8	Red List Index (impacts of invasive alien species)	1988-2016	Significant decrease

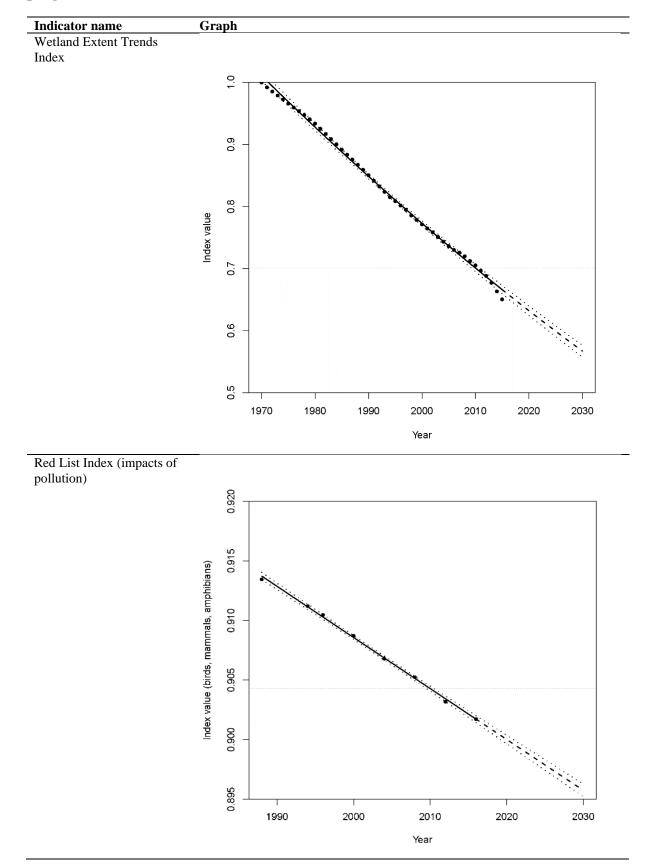
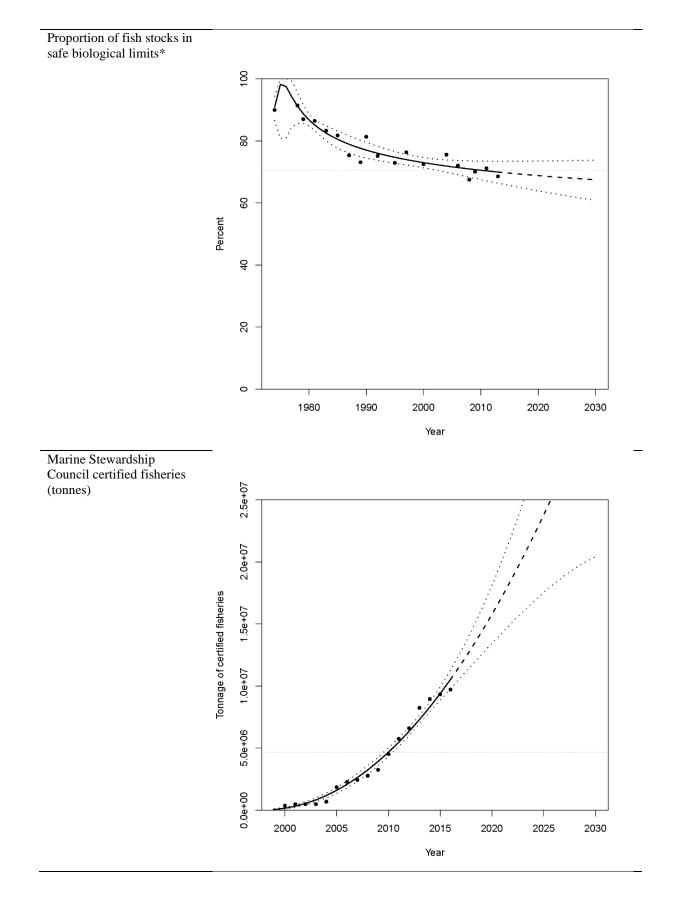
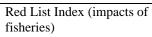
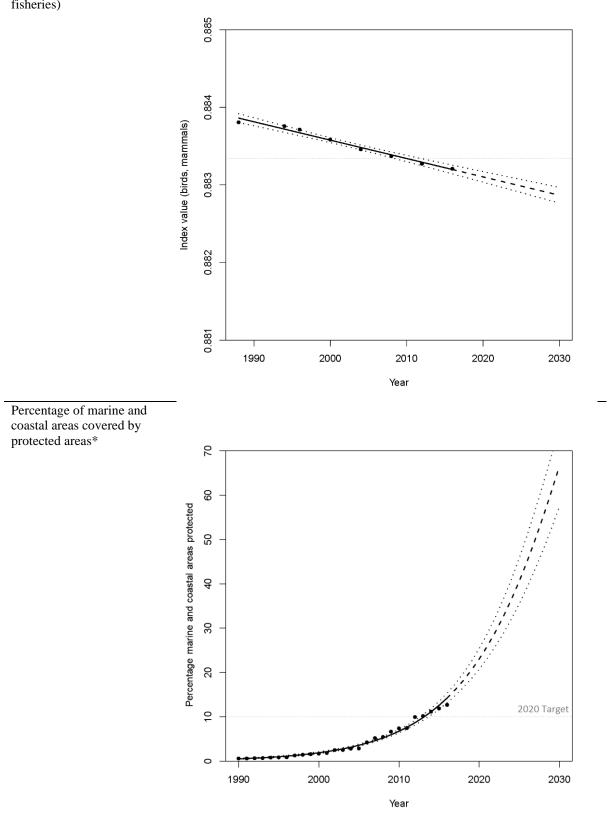
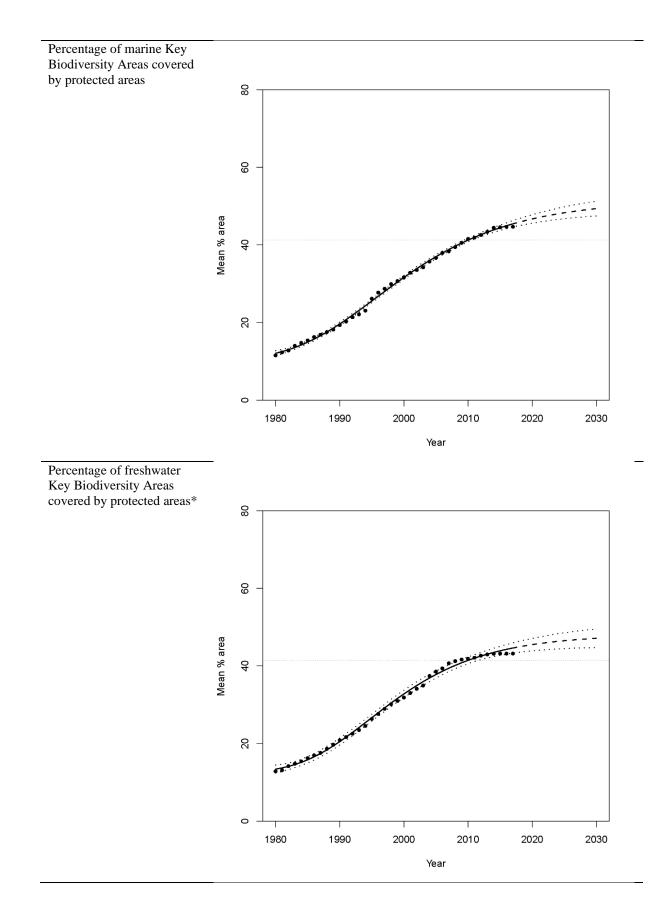


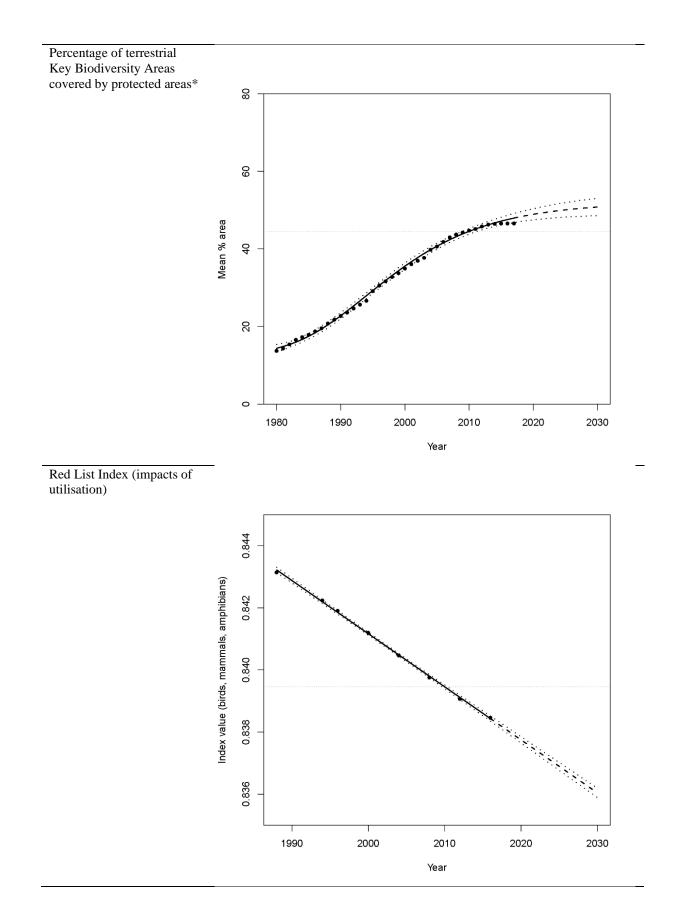
Table S3.7 Graphs for indicator extrapolations used in the quantitative analysis of progress towards SDGs 6, 14 and 15.



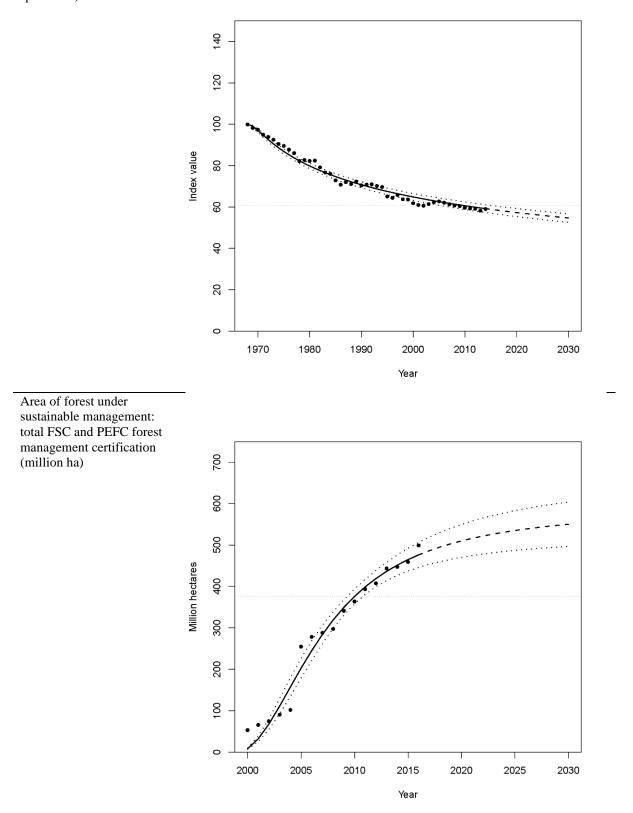


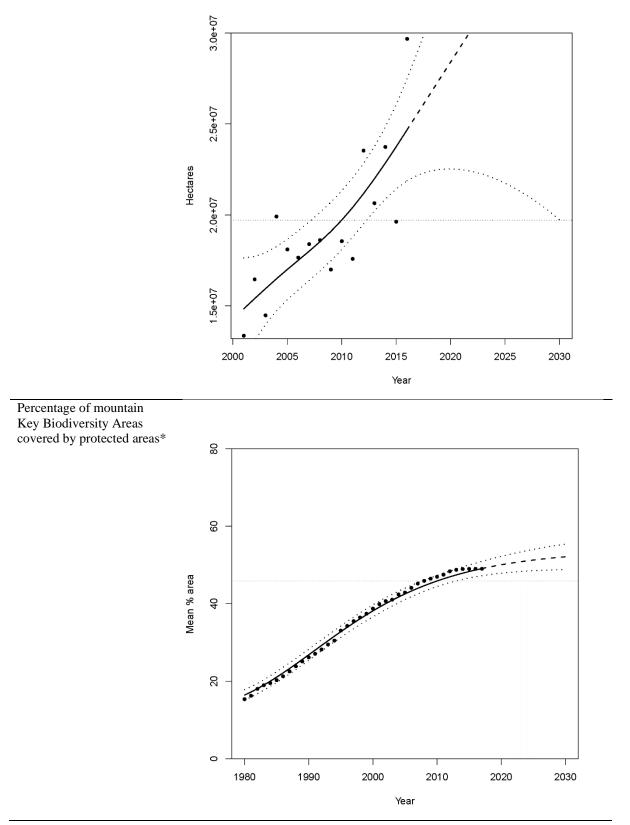


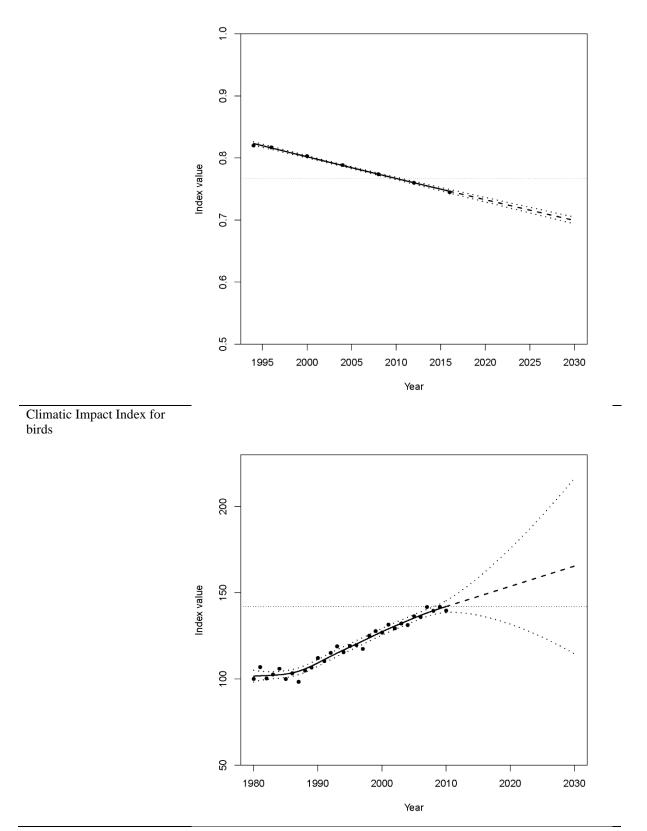


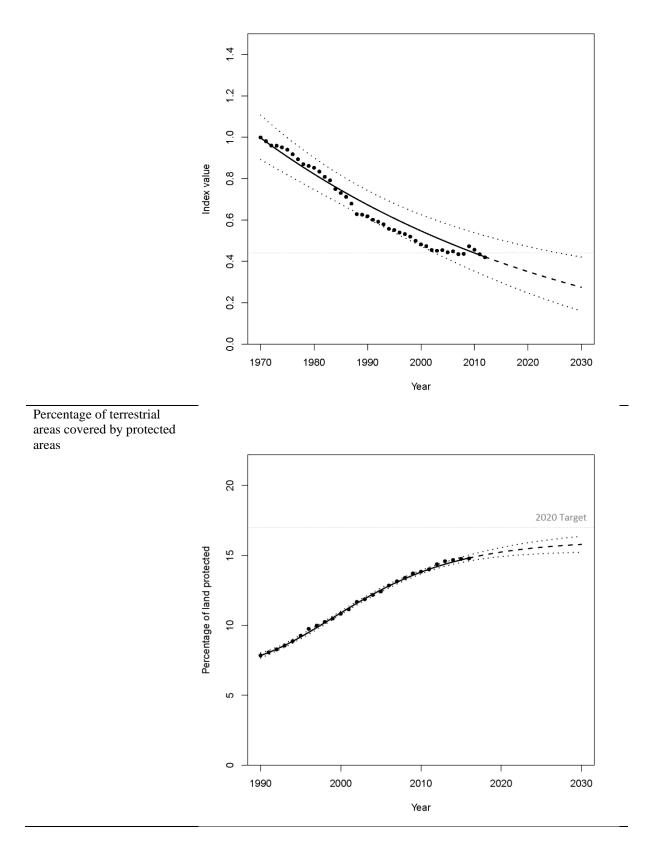


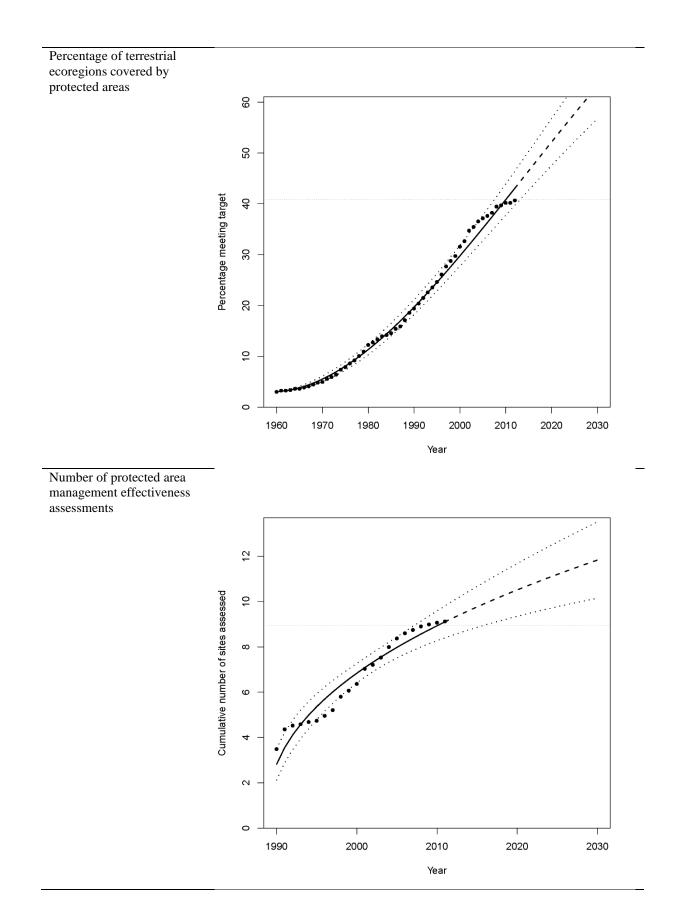
Wild Bird Index (habitat specialists)



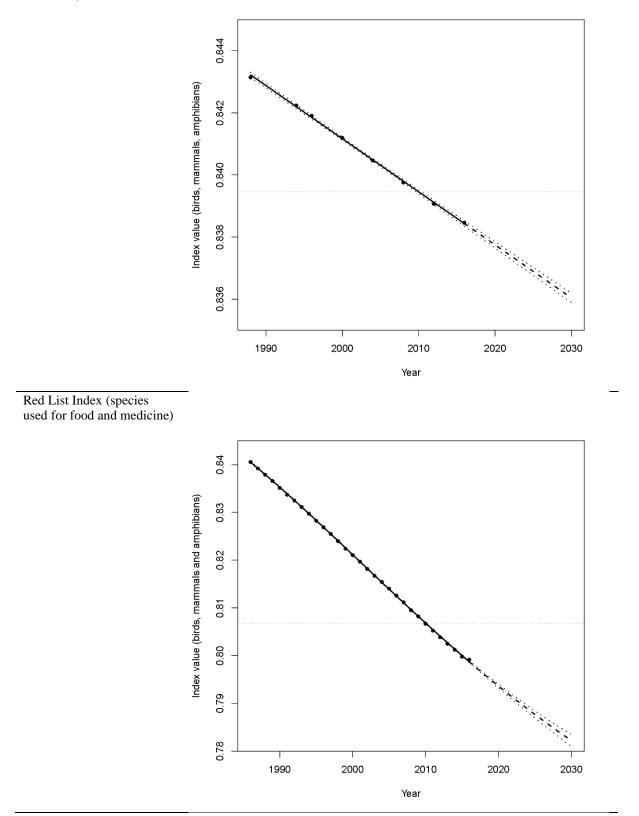




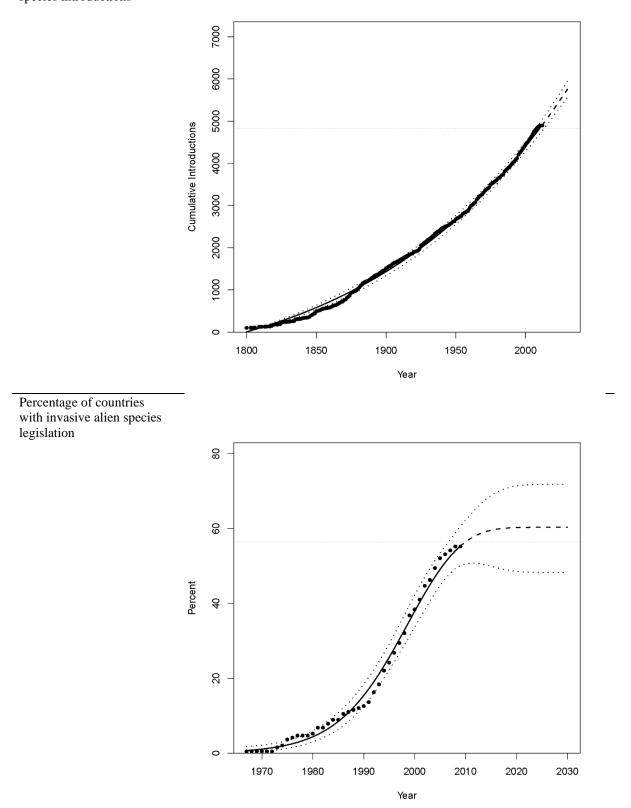




Red List Index (impacts of utilisation)

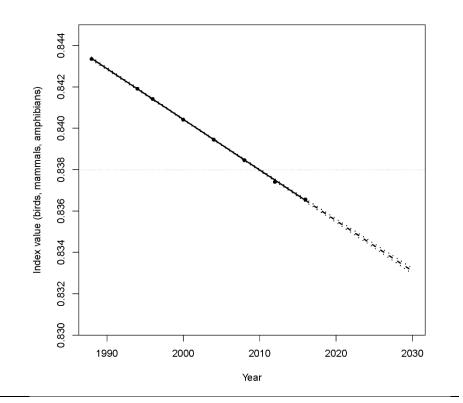


Number of invasive alien species introductions



190

Red List Index (impacts of invasive alien species)



S3.7 Extended review of the SDGs and Indigenous Peoples and Local Communities

SDG1: End poverty in all its forms everywhere

Methods: The text is based on a literature review using the following strings of terms: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK) AND ("natural resource*" OR biodiversity OR ecosystem service* OR "forest product*" OR "forest good*") AND alternatively, according to subtarget: 1.1-1.2) (poverty OR poor OR multidimensional OR income OR wealth) AND (reduction OR mitigation OR alleviation) AND); 1.3) ("social protection" OR "social program*" OR "cash-transfer*" OR CCT OR antipoverty OR "social polic*" OR "social securit*" OR "cash-payment*"); 1.4) (rights OR access) AND (equal* OR inequal*); 1.5) (poverty OR poor OR vulnerab*) AND ("climate change" OR "extreme event*" OR disaster* OR shock* OR resilien*). Using the Topic search on the Core collection of Web of Science (USP subscription), each subtarget string resulted in the overall number of articles (relevant to the topic between brackets): 1.1. & 1.2) 120(80); 1.3) 12(2); 1.4) 43(20); 1.5) 193(103). Additional papers were selected from the authors' own databases and from reviewer's suggestions.

IPLC have a threefold contribution to poverty eradication. First, IPLC are the main actors, with different participation levels, in the so-called win-win initiatives (or triple benefit -Brockington and Duffy 2011) aimed at biodiversity conservation and climate mitigation while improving IPLC's well-being or income level (Roe 2008). Interventions such as community-based natural resources management, ecotourism, integrated strategies for environmental restoration (e.g., Brown et al. 2011; El Bagouri 2007; Adhikari, Di Falco, and Lovett 2004; Ahenkan and Boon 2010; Chirenje 2017; Campos-Silva and Peres 2016), and carbon sequestration or REDD+ projects (e.g., Dulal, Shah, and Sapkota 2012) have been implemented with a major justification being to tackle poverty among IPLC mainly by NGOs (Romero-Brito, Buckley, and Byrne 2016; Adams 2013), but also through government actions (e.g., to restore habitat and provide firewood to the poor - Nepal, Nepal, and Berrens 2017). Second, IPLC traditional institutions (e.g., taboos - Cinner, Fuentes, and Randriamahazo 2009; Undurraga et al. 2014) ILK and management practices (e.g., diversification) may in some cases help mitigate the effects of poverty and exposure to shocks by reducing vulnerabilities (Aryal, Cockfield, and Maraseni 2014), and to adapt to natural disasters and global changes (Ingty 2017; Parraguez-Vergara, Barton, and Raposo-Quintana 2016). Thus, scholars have argued that IPLC long-term adaptation to environmental stresses (Ingty 2017) provide them with the toolbox that allow themselves (Boillat and Berkes 2013; Boissière et al. 2013) or in conjunction with scientific knowledge (Armatas et al. 2016; Aswani and Ruddle 2013) to engage in planning (Bele, Sonwa, and Tiani 2013; Bennett, Kadfak, and Dearden 2016) or in government-related interventions (e.g., infrastructure, governance models, early warning systems) (Batista et al. 2014; Baudoin et al. 2016; Boyer-Villemaire et al. 2014) to build their resilience and adaptive capacity to withstand natural hazards and climate change. Third, interventions among IPLCs have contributed to highlight the debate on whether narrow poverty definitions based solely on monetary indicators (e.g., SD1 estimate of income lower than \$1.25) are adequate (Alkire 2002; Fukuda-Parr 2016). Instead, IPLCs often: have different understandings of what

poverty, wealth, or well-being are (Chambers 2005); live in remote locations and rely substantially on non-monetary sources of wild natural resources for food, shelter and healing (Angelsen, Jagger, Babigumira, Belcher, Hogarth, Bauch, Börner, et al. 2014; Robinson 2016), and face multiple deprivations (Olsson et al. 2014). SDGs advanced from earlier Millennium Development Goals in how poverty is conceptualised (Gratzer and Keeton 2017), with target 1.2 explicitly referring to multidimensional poverty, a current widespread view. Contradictorily, most international evaluations and national accounts still report only monetary dimensions of poverty (Laderchi, Saith, and Stewart 2003). Given that conservation and development interventions have occasionally coincided with the loss of access to land and resources (e.g., Asquith, Rios, and Smith 2002), income (e.g., L'Roe and Naughton-Treves 2014), traditional livelihoods and culture (Mbaiwa, Ngwenya, and Kgathi 2008), alternative approaches to monetary assessments of poverty have been devised for understanding and guiding policy-making (Bridgewater, Regnier, and Garcia 2015) and environmental policy frameworks (e.g., in REDD+ safeguards Arhin 2014) addressed to IPLCs. Some of these frameworks include the Sustainable Livelihoods (Bebbington 2000; Ferrol-Schulte et al. 2013) and the capabilities framework (Sen 1999).

Indigenous peoples are considered to be among the (income) poorest of the world's poor (Hall and Patrinos 2012; Macdonald 2012) and IPLCs are recognized as the most vulnerable to climate change (Brown et al. 2013; Change 2007). Moreover, poverty is often higher in rural remote areas (Sunderlin et al. 2005) and prioritized areas for biodiversity conservation (Fisher and Christopher 2006), where both most IPLC live. As remote rural inhabitants rely substantially on natural resources, higher monetary income may affect IPLC livelihoods, while also impacting biodiversity in multiple ways (Godoy et al. 2005). For instance, while higher monetary income tends to reduce dependency on natural resources, it can increase absolute levels of resource extraction (Godoy et al. 2010); thus, poverty alleviation through increased cash income does not necessarily take pressure off natural resources (Angelsen, Jagger, Babigumira, Belcher, Hogarth, Bauch, Boerner, et al. 2014). In particular, clearing forest for agricultural land (deforestation) is associated with higher household income and assets (Babigumira et al. 2014). Enterprise-based conservation and development initiatives, but also national poverty reduction strategies through conditional and unconditional cash transfers, may therefore impact people's livelihoods and natural resource use in unanticipated ways. Several IPLCs have been reached by government-led cash transfers, an increasingly common element of government policies, but consequences to their well-being and to biodiversity conservation are still poorly documented (Zavaleta et al. 2017; Barber et al. 2015; Bauchet et al. 2018; Piperata, McSweeney, and Murrieta 2016). The evidence regarding integrated conservation and poverty alleviation initiatives have been mixed, and sometimes poorly quantified (Romero-Brito, Buckley, and Byrne 2016; Charnley and Poe 2007). Restricting IPLC rights on forest products harvest and trade have precluded opportunities for income generation (e.g., Scheba and Mustalahti 2015; Mbaiwa, Ngwenya, and Kgathi 2008; Jagger et al. 2014), or lowered cash returns (e.g., Katikiro 2016). Community forestry and freshwater fisheries management have occasionally facilitated both income generation and conservation (Campos-Silva and Peres 2016; Charnley and Poe 2007; Michon et al. 2007), whereas in others, regulations and local power dynamics resulted in no benefits to most people (e.g., Thondhlana, Shackleton, and Blignaut 2015), or lowered resource access to the poorer (Lund and Treue 2008; Adhikari, Di Falco, and Lovett 2004). Moreover, while promoting income generation, some initiatives have produced both ecological and socioeconomic negative impacts, such as depopulation of harvested species, decline in resource quality and landscape impacts (Kusters et al. 2006), social conflicts (Arnold and Pérez 2001), weakened cooperation (Rizek and Morsello 2012) 2012) and tradeoffs in time investments between income earning and subsistence activities (Fisher and Dechaineux 1998). For REDD+ projects, material benefits (jobs and income) have been modest (Lawlor et al. 2013); while some projects promoted tenure rights gains (Lawlor et al. 2013), others have failed to guarantee property and participation rights (Ludwig 2012; Duchelle et al. 2017). REDD+ success depends highly on IPLC being active players in planning and implementation (West 2016), which is seldom the case. A similar pattern is found with tourism: poverty alleviation success depends on land rights, revenue sharing, and proper business training, which are rare (Nelson 2012; Spenceley and Meyer 2012; Snyder and Sulle 2011). Despite not addressing poverty issues, historically, protected areas have also compromised IPLC rights and access to land or resources, inflicting material and psychological harm (McElwee 2010; Brockington and Igoe 2006). Recent studies, however, have also found evidence of poverty alleviation linked to protected areas (Baird 2014; Canavire-Bacarreza and Hanauer 2013).

Government and non-government development projects have frequently neglected IPLC rights and knowledge, and have not adequately addressed asymmetric relationships and inequities in their access to economic and political opportunities (Reyes-Garcia et al. 2010). Government-led poverty-alleviation programs are not necessarily adapted to IPLCs and are sometimes culturally and even linguistically inaccessible to indigenous families (Bauchet et al. 2018; Zavaleta et al. 2017). Several studies have recognized that IPLCs can play important roles in managing natural resources sustainably (e.g., Oviedo et al. 2016; Magcale-Macandog et al. 2014; Aswani and Ruddle 2013; Ruiz-Mallen and Corbera 2013; Keenan 2015) and in recovering from shocks and adapting to changes, such as those caused and intensified by climate change (Negi et al. 2017; Armatas et al. 2016; Lunga and Musarurwa 2016; Mapfumo, Mtambanengwe, and Chikowo 2016; Pyhala et al. 2016; Reid et al. 2014; Boillat and Berkes 2013; Boissière et al. 2013; Bardsley and Wiseman 2012; Bone et al. 2011; McSweeney and Coomes 2011). Guided by ILK, IPLC perceptions on the availability and quality of natural resources can help identify whether changes merit response (Alessa et al. 2008), facilitating collective responses to shocks, and the maintenance of long-term resilience of social-ecological systems (e.g., Gomez-Baggethun et al. 2012; Takasaki 2011). As part of this, ILK may improve the responses to climatic shocks at the local scale, since projections derived from regional- or global-scale models cannot yet be reliably downscaled (Bridges and McClatchey 2009).

SDG2: End hunger, achieve food security, improve nutrition and promote sustainable agriculture

Methods: The text is based on a literature review using the following strings of terms: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK) AND ("natural resource*" OR biodiversity OR ecosystem service* OR "forest product*" OR "forest good*") AND alternatively, according to subtarget: 2.1) (hunger OR "nutritious food" OR "sufficient food"); 2.2) ("malnutrition" OR "wasting" OR "stunting"); 2.3) "small-scale" AND (agricultural productivity OR sustainable agriculture OR sustainable OR sustainable farm* OR sustainable fish*); 2.4) "food production" AND (pollinat* OR soil OR freshwater quality OR freshwater quantity); 2.5) "genetic diversity" AND (seeds OR cultivated plants OR farmed animals OR domesticated). Using the Topic search on the Core collection of Web of Science (UAB subscription), each subtarget string resulted in the overall number of articles (relevant to the topic between brackets): 2.1) 19(17); 2.2) 35(33); 2.3) 34(30); 2.4) 16(15); 2.5) 59(50). Additional papers were selected from the authors' own databases and from reviewer's suggestions.

Through history, IPLCs have developed a variety of systems to achieve local food security through sustainable use of their environment. This has been assessed, for example, in research showing that the diets of hunter-gatherers are diverse and highly nutritious (Berbesque et al. 2014; Crittenden and Schnorr 2017) and research showing that traditional farming systems that exploit biodiversification, soil management, and water harvesting or decentralized water management have helped IPLCs to achieve food security with sustainable agricultural production (Altieri and Nicholls 2017; Bjornlund and Bjornlund 2010). Sustainable and longstanding production systems in forests have also played a vital role in IPLCs' food security (Takahashi and Liang 2016), with many examples of sustainable land use and crop production through farm forestry (Appiah and Pappinen 2010), sustainable forest management (Boscolo, van Dijk, and Savenije 2010), agroforestry (Leimona and Noordwijk 2017, Jana et al 2014, Weerahewa e al 2012; Holmes et al 2017, Parihaar et. al. 2015) and sustainable collection of medicinal and edible wild plants (Ciftcioglu 2015). In many parts of the world, fisheries and aquaculture also provide nutritional security for many IPLCs (Ali et al. 2017). Some authors have argued that the consumption of insects, common in Latin American and African countries, could also contribute to ensure food security (Costa-Neto 2015). Finally, commercial use of local native plants is also suggested to enhance food security (Oren et al. 2017; Edwards-Jones et al 2008) with multiple co-benefits like social cohesion (Hinrichs, 2000), reducing agricultural inputs (Alexander et al., 2017), promoting health (Provenza et al., 2015), and contributing to biodiversity conservation (Cardinale et al., 2012).

Many IPLCs directly depend on natural resources for their livelihoods for which the degradation of local natural resources has a direct impact on their nutrition. For example, changes in climatic variables could compromise local food security (Altieri and Nicholls 2017), including through extreme climate events with food transport vulnerability, loss of hunting grounds, or access routes (Ford 2008). Malnutrition and under nourishment among under 5 is major problem among some IPLCs, specially after they loss access to their lands and traditional livelihoods (Babatunde 2011, Dutt and Pant 2003)(Anticona and Sebastian 2014)(Ferreira et. al 2013)(Gracey 2007). It is well documented that the diets of many IPLCs are rapidly changing (Kuhnlein 2009; Kuhnlein and Receveur 1996)(Crittenden and Schnorr 2017; Bailey and Headland 1991; Bailey et al. 1989); dietary transitions being linked to market integration and commodification of food systems. IPLCs dietary transitions are resulting in a move away from traditional foods towards more processed foods, higher in fat, added sugar, and salt (Kuhnlein 2009; Kuhnlein and Receveur 1996; Popkin 2004), leading to increasing rates of overweight, obesity and associated chronic diseases, or what is known as "hidden hunger" (Popkin 2004)(Ganry, Egal, and Taylor 2011)(Damman et al 2008)(Kuhnlein et al 2006, Saibul et al. 2009).

Scientists now recognize that many food production systems developed by IPLCs could contribute to sustainable food production (Altieri and Nicholls 2017) (Winowiecki et. al. 2014, Pauli et. al 2016), and that the inclusion of highly valuable but presently under-valorised crops and species in agricultural systems could contribute to sustainable agriculture (Kahane et al. 2013). So, there are proposals to rescue aspects of traditional management systems to increase the productivity, sustainability and resilience of agricultural production (Altieri and Nicholls 2017), or to use agroforestry for both environmental restoration and

income generation (Brown et al. 2011; Leakey 2017). Some studies have also found that community-based management induces rapid recovery of a high-value tropical freshwater fishery while ensuring food security (Campos-Silva and Peres 2016). However, it is also acknowledged that the success of programs for sustainable resource management remains dependent on issues such as rights and access allocation, corruption, lack of local financial, intellectual and innovative capacity and centralized governance (Ferrol-Schulte et al. 2013), for which policies to fight hunger need a more comprehensive approach, addressing not only technical measures, but also tackling power asymmetries that reduce access to land and other resources for IPLCs (Francescon 2006; Beckh et al. 2015) or raising investment capital and support through organisational infrastructure (Godfray et a, 2010).

SDG 3: Ensure healthy lives and promote wellbeing for all at all ages

Methods: The text below is based on a literature review using the following search terms as topics: ((neonatal mortality OR infant mortality OR preventable child* death) AND (forest* OR ecosystem OR biodiversity OR wildlife OR *water)) OR (disease* AND human health AND (forest OR ecosystem OR biodiversity OR wildlife)) OR (("mental health" OR wellbeing or wellbeing) AND (green space OR park* OR ecosystem OR forest OR biodiversity OR wildlife)) OR (disease* AND (water quality OR air quality)) AND (*"indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional management" OR "indigenous management" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK). The string resulted in 308 search results in Topic search on the Core collection of the Web of Science with a subscription at UAB, of which 75 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions.*

While most contemporary peoples have plural medical systems, traditional medicine continues to play an important role among IPLCs (Chekole 2017; Kamatenesi-Mugisha et al. 2007; Paniagua-Zambrana et al. 2015; Tolossa et al. 2013; Cartaxo et al. 2010; P. A. Cox 2004; Moura-Costa et al. 2012; Padalia et al. 2015). For example, according to WHO, 80% of the African population uses traditional medicine to meet their primary health care needs (Cunningham 1993), Leaman 2015(WHO 2013) while the global long distance demand for traditional medicine is actually growing placing many threats on resources locally (Schippman et al 2006). Limited access to other healthcare systems, due for example to the relative isolation, small healthcare workforce, few providers, relatively large out-of-pocket expenses, poor transport networks makes traditional medicine the only treatment option in certain some communities (Paniagua-Zambrana et al. 2015; Tolossa et al. 2013; Hamilton and Aumeeruddy-Thomas 2013). However, research has also shown that traditional medicine can be the preferred treatment option even when other healthcare systems are accessible (Padalia et al. 2015; Hamilton and Aumeeruddy-Thomas 2013). Medicine-related ILK has contributed to the discovery of active principles for drug development to treat noncommunicable diseases (NCDs) and infectious diseases, including AIDS, neglected tropical diseases, hepatitis, and water-borne diseases (Cartaxo et al. 2010; P. A. Cox 2004; Moura-Costa et al. 2012; Padalia et al. 2015; Tolossa et al. 2013; Rullas et al. 2004; Paul Alan Cox 1993; Johnson et al. 2008), often without recognizing or compensating IPLCs and giving rise to conflicts over unfair appropriation of ILK Richerzhazen 2010 (Nelliyat 2017).

Research has shown higher rates of mortality and morbidity among indigenous peoples than among their non-indigenous counterparts (Hernandez et al. 2017; I. Anderson et al. 2016; Coimbra et al. 2013; Hurtado et al. 2005). For example, indigenous peoples experience

disproportionately high prevalence of malaria (Coimbra et al. 2013), tuberculosis (ref), hepatitis (Hurtado et al. 2005), and HIV/AIDS (ref). IPLCs are also more likely to experience disability (ref). Nutritional transitions (or the shift in dietary patterns from high fiber, healthy local food to energy-dense, imported food with low nutritional value) has resulted in a high prevalence and incidence of central obesity and diabetes and poor nutrition among many IPLCs (e.g., (McDermott et al. 2009; Port Lourenco et al. 2008; Rosinger et al. 2013; Corsi et al. 2008) as well as high rates of alcohol use and tobacco smoking (Kirmayer et al. 2000; Natera et al. 2002; Wolsko et al. 2007). Poor health conditions among indigenous women provide a poor intra-uterine environment and contribute to high perinatal morbidity and future disability (McDermott et al. 2009). Among Aboriginals in Australia and First Nations in Canada, suicide rates, particularly among the young, are considerably higher among indigenous than non-indigenous peoples, largely linked to cultural discontinuity and oppression (Kirmayer et al. 2000; Phillips 2009; Silviken 2009; Cheung et al. 2012)

Given IPLCs direct dependence on the environment to cover their material (e.g., water, food, shelter and medicines) and cultural needs (e.g., spiritual beliefs and worldviews), environmental factors, including climate change, chemical contamination, and land use changes threaten to jeopardize the achievement of SDG3 among IPLCs (Aparicio-Effen et al. 2016; Dudley et al. 2015; Genthe et al. 2013; N. E. Anderson et al. 2015; Bradford et al. 2016). High levels of land and water chemical pollution have been reported in areas inhabited by IPLCs (Orta-Martínez et al. 2007; Gonzalez-Merizalde et al. 2016; Oestreicher et al. 2017) where alternative sources of water and food are limited. Climate change, chemical contamination, and land use change can also affect the availability of clean water (McOliver et al. 2015; Mboweni and de Crom 2016). Poor water quality is a risk for the spread of several water-borne infectious diseases (Miller et al. 2010) (target 3.3), one of the main causes of infant mortality (Pratt et al. 1992) and a factor for non-communicable diseases, such as cancer. Limited access to water can also reduce crop yields and limit access to game (Medeiros et al. 2017; Negi et al. 2017; M. H. Rahman and Alam 2016). Climate change and land use change can also contribute to an increase in the incidence of infectious diseases including neglected tropical diseases, malaria, and HIV, for example, by expanding the geographic range of some vectors, shortening their reproduction cycles, and by increasing their viral transmission capacity, thus effectively increasing risk of infection (Mengel et al. 2014; Alexander et al. 2015; Hahn et al. 2014; Corno and de Walque 2012; Patz et al. 2003; Xu et al. 2017). Finally, poor environmental conditions can also lead to decreased contact with nature, decreasing overall IPLC's wellbeing (Willox et al. 2012; Johnston et al. 2007; Cocks et al. 2012; Fulford et al. 2015; Mboweni and de Crom 2016).

Some researchers have suggested the need to create new indicators related to indigenous health that are socially and culturally sensitive and that adopt a more holistic and integrated approaches that address the causes of inequalities both inside and outside the health sector (Hernandez et al. 2017). In this line, many countries now recognize the need to develop a cohesive and integrative approach to health care that allows governments, health care practitioners and those who use health care services to access traditional medicine in a safe, respectful, cost-efficient and effective manner, and have started to establish regulations to meet this goal (WHO 2013). This is particularly relevant with regard to birth attendance, where traditional practices have great significance and cultural value. Several studies also highlight the importance and benefits of including IPLCs in the development of policies to effectively achieve healthy lives and promote wellbeing (Incayawar and Maldonado-Bouchard 2009; McOliver et al. 2015; Suk et al. 2004; Wambrauw and Morgan 2016). ILK can aid in the development of local strategies to cope with environmental factors that might put at risk IPLC's health (Negi et al. 2017; M. H. Rahman and Alam 2016), and there exists a

handful of community-based interventions aimed at controlling infectious diseases in a sustainable, environmentally friendly way (Andersson et al. 2015; Arunachalam et al. 2012; Ledogar et al. 2017).

Assessing wellbeing. Improving people's wellbeing is a main challenge for governments, human development agencies, and researchers (Alkire 2002; Hagerty et al. 2001; Max-Neef et al. 1993; T. Rahman et al. 2011), for which several indicators have been developed to aid public policy to improve wellbeing (Dluhy and Swatrz 2006; Førnes 2007; Hagerty et al. 2001). Since attributes that define the several dimensions of wellbeing change across cultures and historical periods (Costanza et al. 2007; Swain and Hollar 2003)Frønes 2007; Hagerty et al. 2001; Max-Neef et al. 1993), many indicators might not be applicable for all societies. Several authors have stressed the importance of the involvement of IPLC and their views in defining wellbeing (Malkina-Pykh and Pykh 2008) McMhom 2002;Swain and Hollar 2003). Research in Kodagu, Karnataka (India) analysing the correspondence between Human Development Index, as an indicator adopted by governments to assess wellbeing, and the elements defined by local people as important in their wellbeing, (Zorondo-Rodriguez et al. 2014) found that the list of local means included access to basic facilities and many issues related to agriculture and natural resources management as elements locally defining wellbeing. The findings suggest an important gap between current indicators of wellbeing considered by public policies and local definitions. This type of research provides insights for a set of plausible local indicators useful to achieve a balance between top-down and bottomup approaches for the local public policies.

SDG 6: Clean Water and Sanitation

Methods: The text below is based on a literature review using the following search terms as topics: ("Indigenous Community" OR "Indigenous Peoples" or "Local Community" or "Aboriginal") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("water quality" OR "water use efficiency" OR "water stress" OR "water supply" OR "integrated water management" OR "integrated water resources management" OR "water sanitation" OR "sustainable management of water" OR "water availability") OR ("Sustainable Development Goal 6" OR "SDG 6" OR "SDG6"). The search was run in Web of Science yielding 264 papers of which 87 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions.

There is well-established evidence that IPLCs have developed complex customary institutions for governing and managing freshwater resources in sustainable ways (Adams et al. 1997; Cooper & Jackson 2008; Jackson & Altman 2009; Weir et al. 2013; Boelens 2014; Tharakan 2015; Strauch et al. 2016). Many studies have shown the strong cultural and spiritual ties between IPLCs and freshwater bodies (e.g., lakes, rivers and lagoons), which are deeply rooted in cultural beliefs and social practices, and are thus at the basis of IPLC customary institutions for water management (Lorente Fernández 2006; Nash 2007; Alessa et al. 2008; Jackson & Altman 2009; King and Brown 2010; Barber & Jackson 2011; McGregor 2012; Anderson et al. 2013; Dallmann et al. 2013; Jaravani et al. 2017). Many IPLCs consider water as a living and sentient being (Anderson et al. 2013; Jiménez et al. 2014) and as such, they often carry a strong responsibility to safeguard it (Finn & Jackson 2011; Mooney & Tan 2012; Singh 2006; Toussaint 2008). Similarly, proximity to specific water bodies is intimately connected to the cultural identity of many IPLCs (Jackson et al. 2005; Alessa et al. 2008; Mooney & Tan 2012). Thus, many sacred groves are located at the

catchment areas of important rivers and streams, ensuring water availability for thousands of people (Dudley & Stolton 2003; Jeeva et al. 2006; Ray & Ramachandra 2015). Sacred wetlands managed by IPLCs provide essential NCPs, such as water filtration or stream flow regulation (Verschuuren 2006; Ndlovu & Manjeru 2014; Moges et al. 2016; Singh & Dwivedi 2016).

ILK-based water management systems are as diverse as the social and ecological contexts they emanate from and include time-honoured practices such as rainwater harvesting (Baguma & Loiskandl 2010; Van Meter et al. 2014; Widiyanti & Dittmann 2014; Oweis 2016), small-scale sand dams (Lasage et al. 2008, 2015), water tanks (Ariza-Montobbio et al. 2007; Reyes-García et al. 2011), traditional water purification methods (Katuwal & Bohara 2011; Mwabi et al. 2013; Opare 2017), forestry-based groundwater recharge (Becker & Ghimire 2003; Camacho et al. 2016; Strauch et al. 2017; Everard et al. 2018), and complex systems of river zonation (e.g., Tagal System in Malaysia; Halim et al. 2013; AIPP 2015). Additionally, several water-smart agricultural practices are prominent in different IPLC cultures (Bridges & McClatchey 2009; Ziegler et al. 2011; Nakashima et al. 2012; Nicol et al. 2015; Nyamadzawo et al. 2015; Lee & Courtenay 2016)(Lansing 2006; Brooks, Reyes-García, and Burnside 2018). These multifunctional and holistic systems of water resource management, often referred to as indigenous water cultures (McLean 2017), have been deemed effective at simultaneously ensuring water availability and conservation of biodiversity (Palanisami & Meinzen-Dick 2001; Ariza-Montobbio et al. 2007; Reyes-García et al. 2011; Hughey & Booth 2012) (Lansing 2006; Brooks, Reyes-García, and Burnside 2018).

The strong cultural connections that IPLCs maintain with their freshwater bodies have also allowed them to closely monitor water availability and quality (Nare et al. 2006; Alessa et al. 2008; Sardarli 2013; Bradford et al. 2017). Different IPLC groups have started programs for community-based monitoring of water availability and quality (Deutsch et al. 2001; Benyei et al. 2017), although evidence on the effectiveness of these initiatives is still largely lacking. In recent years, many IPLCs have also engaged, or even initiated, restoration efforts in rivers (Hormel & Norgaard 2009; Fox et al. 2017), lakes (Coombes 2007) and wetlands (Olima et al. 2015; Henwood et al. 2016), positively contributing to improve water security.

There is well-established evidence that water insecurity disproportionately impacts IPLCs (Kuruppu 2009; Jiménez et al. 2014; Lam et al. 2017; Medeiros et al. 2017), resulting in multidimensional consequences for their good quality of life, including multiple adverse health, economic and sociocultural burdens (Jin & Martin 2003; Boelens & Seemann 2014; Hanrahan et al. 2014; Daley et al. 2015; Sarkar et al. 2015; Henessy & Bressler 2016). Research shows that IPLCs have systematically lower access to clean water supplies than other segments of the population of many countries all over the world (McGinnis & Davis 2001; Ring & Brown 2002; Baillie et al. 2004; Hossain & Helao 2008; Rivas 2012; Jiménez et al. 2014; Bradford et al. 2016; Hanrahan 2017; Mercer & Hanrahan 2017). Lack of access to safe drinking water and secure sanitation has been associated with high prevalence of several infectious diseases amongst IPLCs at the global level (Gracey et al. 1997; Currie et al. 2001; Lin et al. 2004; Chambers et al. 2008; Henessy et al. 2008; Ahmed et al. 2011; Alessa et al. 2011; Dudarev et al. 2013; Stigler-Granados et al. 2014; Anuar et al. 2016; Han et al. 2016).

Given that access to pipeline water is low amongst many IPLCs (Rivas 2012; Hanrahan 2017), they generally show a high reliance on healthy freshwater environments to ensure their

water supplies (Langton 2002; Finn & Jackson 2011). As such, environmental pollution poses important threats upon the water resources on which IPLCs depend (Orta-Martínez et al. 2007; Kelly et al. 2010; Huseman & Short 2012; Dudarev et al. 2013; Bradford et al. 2017). Similarly, climate change exacerbates ongoing threats to the water supplies of IPLCs (White et al. 2007; Alessa et al. 2008, 2011; Dussias 2009; Nakashima et al. 2012; Ford et al. 2014), particularly in communities relying on glacier-fed water reservoirs (Barnett et al. 2005; Bradley et al. 2006) or on aquifers impacted by saline intrusion due to rising sea levels (Tsosie 2007; Mortreux & Barnett 2009). Along these lines, IPLCs are also some of the most vulnerable group to the impact of large-scale water resource development projects (Gellert & Lynch 2003; Barbera-Hernández 2005; King & Brown 2010; Finn & Jackson 2011), including dams and irrigations plans (Finer & Jenkins 2012; Winemiller et al. 2016; Dell'Angelo et al. 2017).

IPLCs have often been excluded from water decision-making bodies (Jackson 2008, 2011; Jackson & Altman 2009; Weir 2010; Finn & Jackson 2011; Hanrahan 2017). Water resource agencies have generally overlooked the cultural and spiritual values underpinning water use by IPLCs (Singh 2006; Bark & Jacobs 2009; Osborn 2009; Tipa 2009; Barber & Jackson 2012). In several countries, a narrow conceptualization of IPLC water rights limits their ability to sustainably manage water resources according to traditional responsibilities (Durette 2010; Tan & Jackson 2013). For instance, environmental flow allocations are often used as a surrogate for the protection of IPLC interests in water resource planning (e.g., Department of Water 2006), with little respect towards IPLC customary rights over water (Finn & Jackson 2011; Bark et al. 2012; Jiménez et al. 2015). Low participation of IPLCs in water management bodies has often fuelled water conflicts, as well as disagreement over the most culturally-appropriate policy options to ensure availability and sustainable management of water (Boelens & Doornbos 2001; Boelens & Hoogendam 2001; Trawick 2003; Jiménez et al. 2015). While domestic laws tend to assimilate water to a property or a commodity, IPLCs often conceive it as an entitlement (Alessa et al. 2008; Demos 2015; Humphreys 2016).

IPLCs are increasingly expressing their cultural values and aspirations in water management plans developed through participatory processes (Alessa et al. 2011; Hoffmann et al. 2012; Davies et al. 2013; von der Porten & de Loë 2013, 2014). If interventions aimed at improving the role of indigenous water management systems are to be effective, water resource planners need to consider not only technical but also sociocultural factors (Gleick 2000; Pahl-Wostl et al. 2007; Reyes-García et al. 2011; Daley et al. 2015; Dobbs et al. 2015; Jaravani et al. 2016), including greater respect towards ILK and IPLC cultural values (Tipa 2009; Hughey & Booth 2012; Mooney & Tan 2012; Nikolakis et al. 2013; MacIean & The Bana Yarralji Bubu Inc. 2015; Henwood et al. 2016). Collaborations involving integration of scientific knowledge and ILK, as well as the development of biocultural indicators, are particularly well-documented in Australia (Liedloff et al. 2013; Jackson et al. 2014; Jackson & Douglas 2015; Dobbs et al. 2015), New Zealand (Townsend et al. 2004; Harmsworth et al. 2011, 2016; Hughey & Booth 2012) and Canada (McGregor 2012; Castleden et al. 2017; Shrubsole et al. 2017). Yet, the present levels of engagement of IPLCs in water resource planning at national or international levels remains low (Jollands & Harmsworth 2007; Memon & Kirk 2012), under-resourced (Jackson et al. 2009; Escott et al. 2015; Shrubsole et al. 2017) and uncoordinated (Jackson & Altman 2009; Te Aho 2010; Hoverman & Ayre 2012).

The development of partnerships optimizing IPLC participation offers substantial opportunities for stronger IPLC engagement in water planning and management (Tinoco et al. 2014; Escott et al. 2015; Jackson & Barber 2015). Enhanced participation of IPLCs in water resource agencies (e.g., through negotiated agreements; Jackson & Barber 2015) can advance the recognition of the social, spiritual and customary values of IPLCs in water management (King & Brown 2010; Finn & Jackson 2011; Barber & Jackson 2012; Dobbs et al. 2015; Harmsworth et al. 2016), including ILK (Weir et al. 2013; Escott et al. 2015). For example, the Government of New Zealand approved in 2017 a pioneering law granting legal personhood to the Whanganui River, providing an innovative legal forum in which to implement Maori cultural and spiritual values in relation to freshwater (Archer 2013; Caillon et al. 2017; Rodgers 2017; Sanders 2017; Strack 2017). By recognizing the holistic character of the river as "an indivisible and living whole", the law enshrines the traditional worldview of the Maori indigenous peoples regarding the river as an ancestor (Morris & Ruru 2010; Hutchinson 2014; Garrick et al. 2017). Moreover, the law establishes a regime of water comanagement, led by the authorities of the Whanganui River Iwi indigenous community as its guardians, explicitly respecting the traditional customary institutions for river governance of the Maori (Tanasescu 2015; Rodgers 2017).

SDG 11: Make cities inclusive, safe, resilient and sustainable

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous community" OR "indigenous peoples" OR "local community" OR "aboriginal" OR "traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("cities" OR "urban") AND ("resilient" OR "sustainable" OR "safe" OR "inclusive" OR "Sustainable Development Goal 11"). The search was run in Web of Science yielding 241 papers of which 49 were relevant to the topic. Additional papers were also selected from the authors' literature database and based on reviewers' suggestions.

IPLCs play a strong role in enhancing urban sustainability, through more efficient water and energy consumption, waste reduction and management, urban carbon footprint reduction and climate adaptation and resilience (e.g., Mihelcic et al., 2007; Cosmi et al., 2016; Schoor et al., 2015) (Hurlimann et al., 2014; Andersson and Barthel, 2016). These contributions come from IPLCs migrating from the countryside to urban hinterlands and informal settlements and/or from NGOs applying ILK and tools from rural communities to urban areas. For example, Bunting et al. (2010) described the relevance of ILK for maintaining the aquaculture wetlands involved in recycling wastewater in Kolkata (India). ILK transfer and generation are also key elements for sustainable urban agriculture and the stewardship of multiple ecosystem services (cf., Barthel et al., 2005; Andersson et al., 2007; Barthel et al., 2010; Langemeyer et al., 2017). In a study in Kano (Nigeria), the integration of ILK alongside scientific knowledge was found crucial for the maintenance of urban soils fertility (Maconachie, 2012), while in a study in Hanoi (Vietnam), the use of pesticides in urban agriculture was found to be reduced among individuals with experience in traditional agriculture (Prain et al., 2007). IPLCs also contribute to social-ecological resilience and a sustained flow of ecosystem services in urban contexts under changes (Hurlimann et al., 2014; Andersson and Barthel, 2016). For example, social-ecological memories stored in urban agriculture and water management practices have crucially determined urban food security during (historic) periods of energy scarcity and shocks to urban food supply lines (Barthel & Isendahl, 2013), as shown in examples from European cities during World Wars I and II (Barthel et al., 2015) and Havana (Cuba), after the end of the Soviet Union (Altieri et al., 1999). Yet, researchers have also argued that

IPLCs alone are not sufficient to create critical urban resilience, underscoring the need for functioning institutions supporting IPLCs (Walters, 2015).

IPLCs make cities safer by improving disaster risk detection and management. For instance, several scholars have shown the importance of integrating ILK into community-based risk assessment and management programs (e.g., informal settlements in Cape Town (South Africa) (Zweig, 2017), into community-based adaptation (Carmin, Anguelovski, and Roberts 2012), and into post-disaster reconstruction (e.g., in the Chilean locality of Dichato, prone to earthquakes and tsunamis; Arriagada-Sickinger et al., 2016). With rapid urbanization and more IPLCs moving to urban areas, the fact that many cities are not achieving cultural integration leads to increasingly negative effects for IPLCs, including negative health effects as demonstrated through several studies in cities in Australia, New Zealand and Canada (e.g., Askew et al., 2017; Darroch and Giles, 2017; Henwood et al., 2017; Waa et al., 2017). Often IPLCs find employment in the informal economy and live in poor conditions, for instance in slums, with limited access to essential services such as water or sanitation. The areas where IPLCs live tend to have unsafe housing, poor hygiene conditions and weak connections to public transportation, and also face greater risks from the impacts of natural disasters. This social and physical marginalization of IPLCs in the urban setting increases their overall vulnerability.

Inclusiveness of ILK issues in urban management and institutions requires higher consideration of IPLCs' values and practices. While ILK is growingly being included in health care programs (Vance et al., 2016; Munns et al., 2016), its inclusion in other fields might have further positive effects on IPLC livelihoods. For instance, these positive effects are demonstrated by case studies on the participation of IPLCs in climate adaptation planning and implementation in Quito (Ecuador) (Chu et al., 2016) and on the needs of IPLC economic inclusion in urban areas of Pakistan (Sher et al., 2017). Fostering bottom-up identification of hazards and vulnerabilities increases IPLC's awareness and empowerment for enhancing their resilience to hazardous events, as shown through an example of flood risk reduction in Warri (Nigeria) (Odemerho, 2014).

IPLC knowledge is increasingly valued in sustainable urban planning and design as a way for better tailoring goals and strategies according to 'real needs' (Rey-Perez et al., 2017). This is especially true in the context of building ecological (Bunting et al., 2010) and social (Young et al., 2017) resilience of cities, a context in which scholars have demanded the adaptation of urban space and design principles to the needs and habits of IPLCs (cf. Rozzi, 2012; Colding et al., 2013). For example, Stuart & Thompson-Fawcett (2010) collected examples on how traditional Māori knowledge could feed into the creation of sustainable urban design in New Zealand (reviewed by Thompson, 2013). Instead, Li et al. (2016) used cognitive mapping and GPS tracking for classifying community space in a historical site in China to ensure its sustainable touristic use and development. Although there is still need to integrate IPLC views and knowledge into modern institutions and management systems (e.g., Jiusto & Hersh, 2009; Mahmood et al., 2013; Ryttä et al., 2017), some effective strategies on how to do so are now emerging (Kyttä et al., 2013; Kyttä et al., 2016; Samuelsson et al., 2018).

Urban-adapted Farmer Field Schools: Urbanization poses global scale environmental pressures. The reconnection to nature by city inhabitants is critical for understanding the multiple ecosystem services nature provides to us, our critical dependence on healthy ecosystems, and the need for a global environmental stewardship agenda (*cf.* Andersson et al., 2014; Bendt et al., 2013, Camps-Calvet et al., 2016; Hassan et al., 2005, Gómez-

Baggethun et al., 2013; Langemeyer et al., 2017). In this context, ILK can counteract on the 'extinction of experience' (*cf.* Miller, 2005) and steer progress towards more resilient cities by helping recover the link between biological and cultural diversity (Colding & Barthel, 2013; Rozzi, 2012). In developing countries, the loss of ILK is correlated to the intensification of pesticide uses in urban horticultural production. For instance, Prain et al. (2007) showed that the transfer of ILK to urban's farmers could help counteracting unsustainable urban horticultural practices, and increase the awareness of urban farmers towards environmental issues and, at the same time, improve their livelihoods. In some cities (e.g., Lima in Peru; Laguna and Manila in the Philippines), Farmer Field Schools have been introduced to facilitate the transfer of such knowledge. These Farmer Field Schools are nonformal, on-site education modules delivered by experienced farmers aimed to transfer and adapt ILK of integrated crop management to urban production sites.

SDG 12: Ensure sustainable consumption and production patterns

Methods: This literature review was conducted by linking (AND) the following two topics in a Web of Science search. 1) "sustainable consumption" OR "sustainable production" OR "responsible production" OR "responsible consumption" OR "Sustainable Development Goal 12" OR "SDG 12" OR "SDG12" OR "consumption and production patterns" OR "food waste" OR "household consumption" OR "environmental impact"; 2) "indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR TEK OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK. This search yielded 231 articles and book chapters of which 70 were considered in greater depth for this review. These publications led to the inclusion of further published literature in this review.

The existing body of academic research on IPLCs and responsible production and consumption is particularly illuminating on three pivotal issues that not only strongly effect IPLC but can also be identified as posing obstacles for sustainable development in general: 1) The heterogeneity of people with regards to drivers and consequences of resource use expansion linked to unsustainable production and consumption; 2) power disparities and their role in the appropriation of natural resources, including via the appropriation of ILK; and 3) the shortcomings in existing mechanisms for sustainable management of resources.

The irresponsible or unsustainable production and consumption that SDG 12 seeks to tackle by 2030 triggers long-term global environmental degradation and destruction (Steffen et al., 2015b). Even if the term Anthropocene may suggest otherwise, the exponential growth in resource use and generation of emissions is not driven by a homogenous 'humanity' nor are the environmental burdens experienced equally by all (Pichler et al., 2017). Through their low degree of involvement with mass production and consumption, IPLCs are not a driving force of the global environmental change from which they disproportionally suffer (Chance and Andreeva, 1995; Martinez-Alier, 2014; Smith and Rhiney, 2016; Tsosie, 2007).

Social inequalities and regional differences in driving unsustainable production and consumption are the result of (and additionally serve to uphold) existing power relations (Biermann et al., 2015; Pichler et al., 2017). Therein, the enclosure and commodification of resources is a dominant tool that is frequently employed in the dispossession of IPLCs from their land (Brad et al., 2015; Kelly and Peluso, 2015; Peluso, 2005). As the resource frontier is continuously expanded for economic growth and increased production and consumption, encroachment on IPLC land has become widespread (e.g., Finer et al., 2008; Pichler, 2013),

commonly threatening livelihoods (Bunker, 1984; Gerber, 2011; Larsen et al., 2014; Mingorría et al., 2014). In this economic model, the power of IPLC to determine resource use is severely restricted: IPLCs may be violently evicted from land or their protest over (intended) land use is brutally silenced (Devine and Ojeda, 2017; Watts and Vidal, 2017), their legal claim to land often relies on customary rights (Benda-Beckmann and Benda-Beckmann, 2010) that are disregarded in favor of newly acquired land titles (Li, 2010, 2001; Vandergeest, 1996). In negotiations with large corporations in particular, IPLCs commonly find themselves at a significant disadvantage (Kirsch, 2002; Rodríguez, 2016; St-Laurent and Le Billon, 2015).

Notwithstanding, the appropriation of IPLCs' knowledge is considered pivotal in attaining more sustainable management of resources (e.g., Fearnside, 1999; Gadgil et al., 1993; Johannes et al., 2000; Véron, 2001), as well as in the adaptation to global environmental change (Bardsley, 2018). Published research has focused very strongly on integrating ILK into the existing capitalist system of production and consumption (Donovan and Puri, 2004; Ilori et al., 1997; Kahane et al., 2013; Sarkar, 2013; Usher, 2000) with its reliance on growth through the appropriation of resources and labor (Moore, 2015). By largely disregarding them, this body of research (inadvertently) proposes to use ILK to reinforce existing power relations (Nadasdy, 1999). Integrating ILK into production and consumption may endanger any sustainability benefits. The increased commercialization of traditional resource use practices increases the associated monetary income along with the environmental impact (Sierra, 1999) while commercialization itself challenges the ability of local communities to maintain traditional practices (Mulyoutami et al., 2009). Pastoralism, for example, constitutes an opportunity to make use of marginal land unsuitable for crop production and improve diets of the local communities (Tessema et al., 2014), while large-scale animal husbandry is one of the most environmentally destructive agricultural practices, and diets rich in animal products are detrimental to human health (Kastner et al., 2012).

Despite the inherent unsustainability of the current resource-use trajectory, existing tools for sustainable resource management typically propose the integration of IPLC claims and thus the further consolidation of these patterns. Rather than interpreting the (non-monetary) preferences of IPLCs (Avc1 et al., 2010; Dongoske et al., 2015; Martinez-Alier, 2009) in terms of possible alternative resource use futures (White, 2006), their integration into planning and managing extractive projects is recommended (Fernandez-Gimenez et al., 2006; O'Faircheallaigh, 2007). In this manner, environmental impact assessments and even certification schemes for sustainable production have often left IPLCs' interests weakened vis-à-vis their corporate counterparts (e.g., Pichler, 2013). In order to avoid the detrimental impact of current patterns of production and consumption, it appears necessary to remove mechanisms conducive to the expansion of the resource frontier (Fearnside, 1999) rather than championing destruction-and-recompensation as community engagement (Kepore and Imbun, 2011). In general, the focus on co-management of resources between IPLCs and the state or corporations (Véron, 2001) presupposes resource extraction. Alternatives, commonly proposed and championed by IPLCs, that challenge the extraction proposition (e.g. 'leave it in the ground'; Benedikter et al., 2016; Bozzi, 2015; Broad and Cavanagh, 2015; Martin, 2015; Piggot, 2017) are strongly contested and much less commonly applied. With regard to sustainable production and consumption, greater consideration is needed of alternative visions of what it means to prosper and to live well (buen vivir) rather than in material abundance (Kothari et al., 2014; Radcliffe, 2012; Zimmerer, 2015).

Current levels of global resource use and associated environmental impact demonstrate that the overarching goal for responsible production and consumption must be to appropriate less resources (Akenji and Bengtsson, 2014). In curbing further expansion of the resource use frontier, especially for large-scale extractive projects, safe-guarding IPLC rights can be instrumental (Bebbington, 2009). The UN could focus on safeguarding the implementation of its Declaration on the Rights on Indigenous people (Pichler, 2013) to reach SDG 12. Vested interests in the further extraction of resources, however, have demonstrated how contested the protection of people and the environment might be (Le Billon, 2001)

For research relating sustainable production and consumption to IPLC, the need for approaches conducive to the integration of insights across disciplines, methods, and levels of scale is apparent. While quantitative, macro-level studies reveal the environmental pressures associated with current patterns of production and consumption (e.g., Friis et al., 2015; Güneralp et al., 2013; Steffen et al., 2015a), detailed, qualitative local case studies illuminate the link between drivers, practices, and impacts (e.g., Bryan, 2011; Chambers, 1994; Kottusch and Schaffartzik, 2017; Sharma and Thakur, 2017).

SDG 13: Climate Action. Combat climate change and its impacts

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous community" OR "indigenous peoples" or "local community" or "aboriginal") OR ("traditional ecological knowledge" OR "indigenous knowledge" OR "traditional management" OR "indigenous management") AND ("natural resources" OR "biodiversity" OR "ecosystem" OR "ecosystem service") AND ("climate" OR "trend" OR "impact") AND ("mitigation" OR "adaptation" OR "hazard" OR "disaster" OR "environmental shock"). The search was run in Web of Science yielding 95 papers of which 70 were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions).

It is well established that IPLCs have scarcely contributed to GHG emissions (Heckbert et al., 2012; Russell-Smith et al., 2013)(Stewart, Anda, and Harper 2016; Salick et al. 2014), and largely contributed to mitigate climate change impacts (Campbell, 2011, Gabay et al. 2017; Lunga & Musarurwa, 2016). For example, fire-based landscape management techniques used by IPLC reduce unwanted fires (Hill et al., 2003; Mistry et al., 2016)(Trauernicht et al. 2015), consequently diminishing CO2 emissions and increasing carbon sequestration in soil and vegetation (Mutuo et al., 2005; Robinson et al., 2014; Siedenburg et al., 2016; Vierros, 2017). There is an increasing recognition of the role of IKP as an alternative source of knowledge for mitigating and adapting to climate change (Altieri & Nicholls, 2017; Chanza & De Wit, 2016; Eicken, 2010; Magni, 2017; Pearce et al., 2015)(Ignatowski and Rosales 2013; Nakashima et al. 2012; Green and Raygorodetsky 2010). Through history, IPLCs have created strategies that have allowed them to deal with the impacts of extreme weather events and to adapt to new climatic conditions (Boillat & Berkes, 2013; Hiwasaki et al., 2015; Palframan, 2015; Turner and Spalding, 2013; Grau-Satorras et al., 2016). For example, agricultural systems promoting crop biodiversity and intercropping have helped IPLCs to guarantee agricultural production (Ajlouni et al., 2010; Chivenge et al., 2016) despite risks associated to pests or adverse climate change impacts (Altieri & Toledo, 2005; Altieri & Koohafkan, 2008; Dey & Sarkar, 2011; Walshe & Argumedo, 2016).

It is also well acknowledged that climate change disproportionately impacts societies with a higher degree of direct dependency to natural resources (Dudley et al., 2015; Pettersson,

2009; Savo et al., 2016), for which IPLCs are among the groups most affected (Jiao & Moinuddin, 2016; McNeeley & Shulski, 2011; Salick et al., 2014; Scott et al., 2010)(Bardsley and Wiseman 2012). For example, researchers have documented how IPLCs are affected by socioeconomic impacts of climate change such as impacts of unexpected extreme rainfall events (Baird et al., 2014; Joshi et al., 2013), floods (Cai et al., 2017), loss of crops due to droughts (Kalanda-Joshua et al., 2011; Swe et al., 2015), disappearance of pastures (He & Richards, 2015; Wu et al., 2015)(Hopping, Yangzong, and Klein 2016), local extinction of plants with medicinal properties (Klein et al., 2014; Mapfumo et al., 2016), changes in animal behaviour patterns (Pringle & Conway, 2012)(Pecl et al. 2017), or the appearance of pests and invasive alien species (Shijin & Dahe, 2015; Shukla et al., 2016). While in the past IKP had allowed IPCLs to understand weather variability and change, IKP might now be less accurate as weather becomes increasingly unpredictable (Cai et al., 2017; Konchar et al., 2015)(Weatherhead, Gearheard, and Barry 2010; Valdivia et al. 2010; Turner and Clifton 2009). This gap affects agricultural productivity (Crona et al., 2013; Velempini et al., 2016), cattle survival (Postigo, 2014; Zampaligré et al., 2014), and even human health (Kassie et al., 2013). The failure of IKP to detect, interpret and respond to change generates a feeling of insecurity and defencelessness that can reduce IPLCs' resilience and exacerbate their vulnerability (Mercer et al., 2010; Simelton et al., 2013)(McNeeley and Shulski 2011).

The potential of combining IKP and scientific knowledge to understand climate change impacts and to design successful climate adaptation policies is increasingly acknowledged (Tengö et al. 2014)(Alessa et al., 2016; Altieri & Nicholls, 2017; Boillat & Berkes, 2013; Ingty, 2017; Austin et al., 2017; Hiwasaki et al., 2014; Kasali, 2011; Mantyka-Pringle et al., 2017). However, scarce and unsuccessful efforts are being done to make IPLCs aware of the scientific approaches used to counteract climate change impacts (Inamara & Thomas, 2017; Shukla et al., 2016; Fernández-Llamazares et al., 2015)(Alexander et al. 2011). One example of an initiative aiming to integrate IKP in climate policies includes integration in Disaster Risk Reduction (DRR) plans to cope with environmental catastrophes such as floods (Ton et al., 2017) or droughts (Masinde, 2015). Another example is the recognition that fired-based landscape management practices can contribute to carbon sequestration, a strategy that has allowed some Australian indigenous groups to enter the carbon market (Jackson et al., 2017; Robinson et al., 2016)(Russell-Smith et al. 2015). Unfortunately, examples of integration of IKP with science and policy are still rare (Seijo et al., 2105) and some of the practices based in IKP are still condemned in some countries (Mistry et al., 2016)(Welch et al. 2013). Moreover, many governments continue to overlook IPLC views and knowledge when developing climate change adaptation and mitigation policies and measures (Blair et al., 2014; Bruegger et al., 2014; Eakin et al., 2012; Jacobi et al., 2017; Muzaffar et al., 2011; Oviedo et al., 2016).

Tsimane' ethnoclimatological knowledge. The Tsimane' are an indigenous society of contemporary hunter-gatherers and small-scale agriculturalists living in the rainforests of the Bolivian Amazon. Their intricate relationship with their local ecosystems has resulted in detailed bodies of ILK, including a great deal ethnoclimatological knowledge, which are at the basis of their subsistence practices (Reyes-Garcia et al. 2003; Reyes-García et al. 2013; Reyes-García et al. 2018). This knowledge is independent from the scientific discourse on anthropogenic climate change, which remains still largely inaccessible to the Tsimane' (Fernández-Llamazares et al. 2015a). Yet, despite being unaware of the scientific construct of climate change, the Tsimane' are frontline observers of climate change and report a number of local climate anomalies that are robustly associated with scientific records (Fernández-Llamazares et al. 2017). The Tsimane' rely on a set of more than 40 ethnoclimatic indicators to track local changes in climate (Fernández-Llamazares et al. 2015a). For example, the

fructification time of the peach palm (*Bactris gasipaes*), a sacred plant in the Tsimane' cosmology that marks the beginning of the annual harvesting cycle, is reported to be changing, making it increasingly difficult to plan harvesting activities (Fernández-Llamazares et al. 2017a).

Research amongst the Tsimane' has shown not only the immense potential that ILK has to contribute to better understanding of local climate change in data-deficient regions (Fernández-Llamazares et al. 2017a), but also to inform culturally-sensitive adaptation planning and decision-making (Fernández-Llamazares et al. 2016; Ruiz-Mallén, Fernández-Llamazares, and Reves-García 2017), given that interpretations of climate change are deeply rooted on Tsimane' cultural beliefs (Fernández-Llamazares et al. 2015a). For example, the use of the term 'tsäqui'' (or great danger) is recurrent in myths providing supernatural explanations of climate change (Fernández-Llamazares et al. 2017a). A number of Tsimane' stories, largely referred to as the eschatological myths, show human attitudes towards nature that could potentially wipe out humans from Earth (Huanca 2008). Similarly, the Tsimane' often teach culturally-inappropriate behaviours through traditional stories about forest spirits that need to be revered. As such, they largely associate ecological changes with punishments by the spirits in response to disrespectful conducts (i.e., for not respecting certain established cultural norms; (Fernández-Llamazares, Díaz-Reviriego, and Reves-García 2017; Luz et al. 2017). Such local norms and institutions, based on the continuous transmission of ILK, are essential for the endurance of the long-term sustainable management of natural resources in the face of climate change (Fernandez-Llamazares et al. 2015; Fernández-Llamazares et al. 2016).

SDG 14: Conserve and sustainably use the oceans, seas and marine resources for sustainable development [LA: Victoria Reyes-García, CA: Shankar Aswani, Reviewers: Nadav Gazit, Eleanor Sterling]

Methods: The text below is based on a literature review using the following search terms as topics: ("indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR "indigenous management" OR ILK) AND ("marine resourc*" OR "ocean resourc*" OR "coastal resourc*" OR "fisheries" OR "aquaculture" OR "SDG 14"). The string resulted in 755 search results in Topic search on the Core collection of the Web of Science with a subscription at UAB, of which we reviewed 58 that were relevant to the topic. Additional papers were also selected from the authors' own literature database and based on reviewers' suggestions.

IPLC have long history of interacting with the oceans and, in many occasions, sustainably managing coastal and marine resources (Lotze and Milewski 2004; Spanier et al. 2015; Thornton and Mamontova 2017; Johannes 1978; Cordell 1989; Lepofsky and Caldwell 2013). Many IPLC also have a deep knowledge of marine ecology, which can provide critical information regarding changes in marine resources over short and long-term scales (McGreer and Frid 2017; Savo, Morton, and Lepofsky 2017; Salomon, Tanape, and Huntington 2007), and an empirical understanding of the behavior and abundance of target species and on how these influence and are influenced by fishing strategies (Spanier et al. 2015; Johannes, Freeman, and Hamilton 2000; Silvano et al. 2006; Drew 2005; Morales, Lepofsky, and Berkes 2017; Groesbeck et al. 2014). Such knowledge can inform marine resources management, especially where other data are not available (Johannes, Freeman, and Hamilton 2000) and thus play an important role in defining marine resource management strategies that aim to include both IPLC and government representatives (Johannes, Freeman, and Hamilton

2000; Pitcher 2001; Aswani and Lauer 2006; Aswani and Ruddle 2013; Thornton and Scheer 2012). However, the literature also acknowledges that traditional management regimes can also result in intense exploitation and even to high levels of illegal fishing (e.g. (Andreu-Cazenave, Subida, and Fernandez 2017; Islam and Haque 2004; Ratner 2006), especially when affected by external pressures (Ruddle 1998; Turner et al. 2013). In the same line, researchers have also warned against the uncritical use of ILK on marine resources, as contemporary bodies of IKP might not always be adapted to rapidly changing ecosystems (Turvey et al. 2010) or might be highly eroded (Turner et al. 2013).

The continued degradation of marine ecosystems, along with the services they provide, *affects the many IPLC who are dependent on the oceans, seas, and marine resources for their livelihoods. Such effects not only relate to their food security* (de Lara and Corral 2017; McGreer and Frid 2017; Robards and Greenberg 2007; Watts et al. 2017), *but also for their social and spiritual integrity* (McCarthy et al. 2014). Research shows that, in addition to marine resources depletion, IPLC face restrictions in their traditional governance of natural resource use, including restrictions on their fishing (Thornton and Mamontova 2017) and tenure rights (Joyce and Satterfield 2010), or "blue" or "coastal grab," a term used to refer to the enclosure, appropriation and dispossession of marine resources and coastal land by outside interests (i.e., the state, tourist operators) (Hill 2017; Bavinck et al. 2017). Although it is well established that IPLC often value marine resources beyond their economic value (Queiroz et al. 2017), little emphasis in the literature is placed on the social and cultural consequences of overfishing and marine biodiversity loss. An important finding of the scant research on this topic is that the removal of cultural marine keystone species jeopardizes cultural integrity (McCarthy et al. 2014; Thornton and Hebert 2015; Turner et al. 2013).

IPLC have an *important role to play in developing alternative approaches to sustainably* managing marine resources (Jones, Rigg, and Pinkerton 2017; Johnson et al. 2016) although this is not always recognized. Best-practice environmental policy suggests co-management of marine resources as a means of achieving sustainable development (Cinner et al. 2006)Gaymer et al. 2014). Thus, marine conservation strategies that include -in one way or another- IPLC are being implemented in many parts of the world, such as Fiji (Thaman et al. 2017), the Pacific Island region (Ruddle 1998)Jupiter et al. 2014), Chile (Gelcich et al. 2015), Indonesia (Mangubhai et al. 2012), Spain (de Lara and Corral 2017), and the Philippines (Alcala and Russ 2006). Such approaches include Voluntary Guidelines for Securing Sustainable Small-Scale Fisheries in the Context of Food Security and Poverty Eradication (SSF-Guidelines) (Singleton et al. 2017), Community Supported Fishery programs (Godwin et al. 2017), or Community-based management (Aswani, Albert, and Love 2017). Community-based management has also been proposed as a viable alternative for sustainably managing coral reefs (Cinner et al. 2006) or mangroves (Datta, Chattopadhyay, and Guha 2012). Current knowledge, however, does not allow us to clearly establish either the social (e.g., (Gelcich et al. 2006) or the conservation impact of co-managed marine areas (Datta, Chattopadhyay, and Guha 2012), although some studies suggest ecological impacts are positive (Cinner et al. 2006; Cinner and Aswani 2007). Differently from what environmental policy suggests, in many areas, traditional fishing techniques have been criminalized from the state perspective, and their practice is consider illegible (Deur et al. 2015; Langdon 2007; von der Porten et al. 2016; Jones, Rigg, and Pinkerton 2017). For example, it is illegal to transplant Pacific herring in Alaska, even though the Tlingit have used this strategy to introduce and enhance spawning areas for centuries (Thornton, Deur, and Kitka 2015).

Text box: Vueti Navakavu

In recent years, Fiji's approach of combining traditional systems of community-based coastal management and modern management systems has become a successful blueprint for marine conservation, particularly the Locally Managed Marine Area (LMMA) network model (Aswani, Albert, and Love 2017). Thus, since the establishment of Vueti Navakavu, a locally managed marine area in Fiji in 2002, nearly 300 mollusk species, including gastropods, bivalves, and cephalopods, are either being seen for the first time in over 40 years or are clearly increasing in abundance and/or size class. Most of the species for which a particularly dramatic increase in abundance has been observed are of economic, cultural, and ecological importance. The results show that sustained effective marine conservation can, in general, lead to the recovery of seriously degraded fisheries and, in particular, of tropical mollusk fauna (Thaman et al. 2017). Moreover, results show that this is done without eliminating traditional fishing or livelihoods, in contrast to some MPAs which follow a 'fortress conservation' model. Conservation practitioners have imported the Fiji LMMA model to the Solomon Islands and Vanuatu, although researchers argue that different socio-political situations and historical particulars do not warrant the success of such upscaling (Aswani, Albert, and Love 2017). Given institutional and contextual social-cultural variability, more thoughtful, systematic, synthetic and detailed historical, cultural, socioeconomic, human ecological, and marine science research will be required to understand the role of IPLCs in a changing world.

SDG 15: Life on Land [LA: Victoria Reyes-García; CA: Sébastien Boillat, Chinwe Ifejika Speranza; Reviewer: Peter Larsen, Amanda Sigouin, Eleanor Sterling]

Target: Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss

Methods: we performed a literature review through the Web of Science database (Thomson Reuters), using the combination of search terms related to IPLC (*"indigenous communit*" OR "indigenous people\$" OR "local communit*" OR aborigin* OR "traditional ecological knowledge" OR "TEK" OR "indigenous knowledge" OR "traditional management" OR <i>"indigenous management" OR ILK*) and search terms related to the Aichi Target 15, obtaining between 246 and 3698 search results (Table 1). For each search, we chose the 50 most relevant results according to the database, and added most cited papers on the topics as well as papers from the authors' own literature database.

Search terms group a		Search terms group b	Search results
"indigenous communit*" OR	AND	"forest conservation" OR "forest governance"	736
"indigenous people\$" OR "local		OR "forest management"	
communit*" OR aborigin* OR		"deforestation"	559
"traditional ecological knowledge"		"forest" AND "commons"	63
OR "TEK" OR "indigenous		"sacred" AND ("forest*" OR "grove*" OR	76
knowledge" OR "traditional		"ground*")	
management" OR "indigenous		("benefit*" OR "use" OR "ecosystem service*")	1807
management" OR ILK		"biodivers*" OR "biological diversity"	2506
		"desertification" OR "land degradation" OR	246
		"soil erosion" OR "soil degradation"	

With 22% of the world's land surface held by IPLC and 80% of biodiversity found there (FAO 2017), IPLCs potentially play a substantial role in governing and managing forests, land, and biodiversity. The often long-lasting relationship between IPLCs and terrestrial ecosystems has led to a co-evolution of social and ecological components that has enhanced adaptive capacity, resilience and sustainability (Berkes and Folke 1998; Folke 2006; Berkes, Colding, and Folke 2000). In this context, IPLCs may benefit from and contribute to the maintenance and enhancement of land-based ecosystems (Comberti et al. 2015). For example, IPLCs have long-term interactions with forests (Mir and Upadhava 2017; Xu and Melick 2007). They have developed values and social norms to manage forests (Lawler and Bullock 2017; Ouma, Stadel, and Okalo 2016). Their management practices focus on ecological processes (Herrmann and Torri 2009), multiple use (V M Toledo et al. 2003), agroforestry (Suyanto et al. 2005), sustainable logging and hunting (Roopsind et al. 2017), fire management (Mistry et al. 2016), protection and management of culturally significant trees (Stara, Tsiakiris, and Wong 2015; Genin and Simenel 2011; Turner et al. 2009), and long-term monitoring (Long and Zhou 2001; Olivero et al. 2016). They also have an extensive **body of knowledge** on species (Campos and Ehringhaus 2003) and forest types (Long and Zhou 2001; Yuliana, Sriyati, and Sanjaya 2017). IPLCs often govern and manage forests collectively (Agrawal 2003; Nagendra and Ostrom 2012) using mixed individual and common access rights (Genin and Simenel 2011) MacLean et al 2013). Collectively-managed forests tend to store more carbon (Chhatre and Agrawal 2009). Giving land titles to IPLC tends to protect forests from large-scale conversion into other land uses (Blackman et al. 2017; Chhatre et al. 2012; Ceddia, Gunter, and Corriveau-Bourque 2015; Nepstad et al. 2006). In many instances, forests that have cultural and religious significance for IPLCs have been shown to be more diverse, denser and they harbor larger and older trees, than non-sacred forests (Ormsby 2013; Aerts et al. 2016; Kokou, Adjossou, and Kokutse 2008; Borona 2014; Frascaroli et al. 2016; Daye and Healey 2015; Aniah and Yelfaanibe 2016; Barre, Grant, and Draper 2009; Harpet, Navarro, and Ramanankirahina 2008; Rao et al. 2011; Stara, Tsiakiris, and Wong 2015; Salick et al. 2007; Shen et al. 2012). Forests are essential to the livelihoods of IPLCs living within and around forested ecosystems. Between 1-1.5 billion people benefit from forests in the form of employment, forest products, and direct or indirect contributions to livelihoods and incomes (Agrawal et al. 2013). Besides commercial and self-used timber and fuelwood, IPLCs also use a wide range of non-timber forest products (NTFPs) for both self-consumption and commercial purposes (Kibria et al. 2017; Padoch et al. 2008; Cruz-Garcia et al. 2015). Forests also provide regulating ecosystem services such as groundwater recharge (Castro et al. 2015), enhance rainfall (Jackson and Naughton-Treves 2012), and protect soil against erosion, wind and floods. Cultural ecosystem services, defined as non-material benefits that people obtain from ecosystems, reflect the particular importance of forests for IPLCs (Afentina et al. 2017).

There is evidence of a globally relevant **spatial overlap between high biodiversity and high linguistic diversity** (Gorenflo et al. 2012; Maffi 2005)(Oviedo, Maffi, and Larsen 2000). Underlying reasons for this link are complex and may differ from one area to another (Gorenflo et al. 2012)(Sutherland 2003; Nettle 1999). There is also evidence that IPLCs have developed livelihoods that **actively maintain biological diversity** (Victor Manuel Toledo 2002; Janis B Alcorn 1993). For example, IPLCs' low intensity logging and hunting (Putz et al. 2012) and multiple use practices (Roopsind et al. 2017) are compatible with the existence of a large number of wild species. Shifting cultivation systems with long fallows also favor biodiversity (van Vliet et al. 2012; Bhagawati et al. 2015), as do crop cultivation in agroforestry systems (Madsa' Juarez-Lopez, Velazquez-Rosas, and Lopez-Binnquist 2017) Barthel et al. 2013). IPLCs have shaped **cultural landscapes** (Walter and Hamilton 2014;

Chazdon et al. 2009) which are multifunctional (Ramirez-Gomez et al. 2016). ILK can also play a key role in **monitoring biodiversity** (Danielsen et al. 2014; Lyver et al. 2017; Polfus et al. 2016) Jupp et al. 2016), especially by providing long-term perspectives of environmental change (Fraser et al. 2006). Approaches that consider ILK and scientific knowledge as complementary are particularly promising to monitor biodiversity (Tengö et al. 2014; Tengo et al. 2017; Chazdon et al. 2009). IPLCs can become key allies to achieve biodiversity conservation in **protected areas** (Ens et al. 2016; Schwartzman and Zimmerman 2005; Andrade and Rhodes 2012). There are, however, many examples of conservation actions affecting IPLCs negatively (Cavanagh and Benjaminsen 2014; Holmes and Cavanagh 2016; West, Igoe, and Brockington 2006; Brockington and Igoe 2006). Therefore, balanced power relation and the recognition of ILK are crucial to achieve conservation in fair terms (Martin, McGuire, and Sullivan 2013; Martin et al. 2016).

IPLCs directly benefit from biodiversity, for example through the use of wild plants in **diet and medicinal** purposes (J B Alcorn 1995; Singh et al. 2014). Biodiversity can have a **spiritual importance** to IPLCs (Torri and Herrmann 2011)Pascua et al 2017). Biodiversity also makes cultural landscapes and agroecosystems more **resilient** to climate change (Altieri and Nicholls 2017; Ingty 2017). Furthermore, non-subtractive uses of biodiversity can provide **additional income** to IPLCs through carbon offsetting (Renwick et al. 2014), ecotourism (Sakata and Prideaux 2013; Gonzalez et al. 2008) and intellectual property rights on biodiversity use (Efferth et al. 2016). Yet the equitable sharing of these benefits remain a challenge in practice (De Jonge 2011; Suiseeya 2014).

There is relatively little literature on how IPLCs can contribute to combat desertification and land degradation (Sendzimir et al. 2011). IPLCs benefit from ecosystem services provided by resilient lands (Sigwela et al. 2017) and are particularly vulnerable to land degradation (Ellis-Jones 1999). The largest body of literature addresses the participation of IPLCs in combating land degradation in relation with externally supported projects and the need to establish effective participation and knowledge co-production schemes (Raymond et al. 2010; Mark S. Reed 2008; Thomas and Turkelboom 2008; d'Aquino and Bah 2013; Stringer, Scrieciu, and Reed 2009; Oba, Sjaastad, and Roba 2008; Roba and Oba 2008; M S Reed et al. 2013). IPLCs have also contributed to fight desertification and soil erosion through endogenous initiatives, some of them rooted in a long-term relation with their environment. This includes plant selection for resistance to drought (Gaur and Gaur 2004), keeping spiritually relevant patches of forest to halt soil erosion (Yuan and Liu 2009), the construction and maintenance of traditional irrigation systems (Ashraf, Majeed, and Saeed 2016; Ostrom 1990; Saldías, Speelman, and Van Huylenbroeck 2013), traditional knowledge on soil types and conditions (Barrera-Bassols, Zinck, and Van Ranst 2006) and terrace construction (Boillat et al. 2004). IPLCs can play a key role in monitoring land degradation and soil conditions (Roba and Oba 2009; Forsyth 1996) and in land rehabilitation (Yirdaw, Tigabu, and Monge 2017).

Text box: Multiple land uses of Quechua-speaking communities of the Tunari (Bolivia)

Links between cultural and biological diversity have been highlighted at global and continental scale (Gorenflo et al. 2012). At local scale, biological diversity exists within a single language area. At that scale, these links exist through ILK dimensions (Berkes 2008): environmental management practices, local knowledge of ecosystems, local institutions governing natural resources, and world views. For example, the Quechua-speaking communities of the Tunari (Bolivia) privilege integrated and diversified use of their territory through mixed agriculture, pastoralism, small-scale forestry and off-farm labor that leads to a

high habitat heterogeneity. These multiple uses are regulated by local institutions based on mixed individual and collective forms of land access (Boillat, Mathez-Stiefel, and Rist 2013). They are also embedded into a relational worldview that interprets risk management actions as part of an ideal of maximizing relations with different people and ecosystems (Boillat and Berkes 2013), personifying ecosystems through place names which are considered living beings with whom one must build respectful relationships (Boillat et al. 2013). Though these linkages persist in people's narratives, their reproduction is threatened by urbanization and conservation approaches that tend to consider people and nature as separate realms (Boillat, Mathez-Stiefel, and Rist 2013).

S3.8 Methods for literature search for assessment of progress towards other conventions related to nature and nature's contributions to people.

For each of the conventions considered, the most recent Strategic Plan and Vision was examined to identify the relevant goals, targets and objectives. We assessed progress at the level of Goals, although it was often necessary to search for literature at the level of targets or objectives. Relevant search terms were identified using key words in each Goal. A series of search strings (Table S3.8) were created with the aim of addressing as many of the key terms as possible while avoiding general terms that would bring up non-relevant literature. We used "Publish or Perish" software (and, for CITES, Google Scholar) to generate an initial list of literature, limiting the results to 1000 for the period 2005-2017, and searching title words only. The list of 1000 was exported to Excel where all duplicate titles were deleted and the first 100 items to review were selected using the 'GSRank' field. For the First Order Draft, we selected 5-8 articles per Goal from this list that appeared to be most relevant based on their titles. For the Second Order Draft we will assess the full list of relevant articles. PDF versions of the articles were obtained to confirm relevance by first reading the abstract. If the abstract did not appear highly relevant, the document was replaced. Text relating directly to progress of the target, goal or convention as a whole was extracted. On the basis of the information identified, progress towards each goal was scored as 'on track to exceed the target', 'on track to achieve the target', 'progress towards the target but at an insufficient rate', 'no significant overall progress', or 'moving away from target'. Information relating to reasons for variation in progress or to knowledge gaps was extracted to inform sections 3.5 and 3.8 respectively.

Table S3.8. Search terms used for literature search for assessment of progress towards other biodiversity and ecosystem service-related conventions.

Con- vention	Search string	No. hits	
CITES	'CITES' AND 'Convention' AND 'Enforcement' (In title only)	2	2

	'CITES' AND 'Convention' AND 'Enforcement' (anywhere in text)	998	23
	'CITES' AND 'Convention' AND 'operation' AND 'implementation success' AND 'financial' AND 'resources'	997	0
		100	
	'CITES' AND 'Convention' AND 'financial' AND 'resources'	0	6
	'Financing' AND 'trade' AND 'convention' AND 'endangered' AND 'species'	998	20
	'CITES' AND 'Convention' AND 'contribution' AND 'biodiversity' AND 'loss''	992	6
	'CITES' AND 'convention' AND 'coherence', AND 'multilateral' AND 'instruments' AND 'other' AND 'conventions' AND 'Aichi' AND 'Targets' OR 'SDG' AND 'mutually' AND 'supportive'	999	13
CMS	'Migratory' AND 'species' AND 'convention'	100 0	25
	'underlying' AND 'causes' AND 'decline' AND 'Migratory' AND 'species'	100 0	12
	'underlying' AND 'causes' AND 'migratory' AND 'species' AND 'decline' AND 'Convention' AND 'Migratory' AND 'Species'	993	21
WHC	'success' AND 'world heritage sites' AND 'biodiversity conservation'	100 0	30
	'natural' AND 'world' AND 'heritage' AND 'sites' AND 'status'	988	33
Ramsar Goal 1	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'wetlands' OR 'wetland drivers' OR 'wetland degradation' OR 'wetland loss'	462	18
	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'wetland benefits' OR 'wetland ecosystem services' OR 'wetland policy' OR 'wetland strategy'	673	24
	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'wetland use' OR 'wetland management' OR 'guidelines for wetland management' OR 'invasive species in wetlands' OR 'invasion pathways in wetlands'	539	19
Ramsar Goal 2	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'Ramsar Site Network'	8	2
	'Ramsar Site Network' AND 'wetland connectivity' OR 'ecoregions' OR 'trans-boundary sites' OR 'transboundary sites'	8	3
Ramsar Goal 3	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'wetland inventories' OR 'national wetland inventories'	5	1

	'Integrated resource management' AND 'wetlands' OR 'coastal wetlands' OR 'river basin' OR 'degraded wetlands'	266	12
	'Wetlands' AND 'indigenous knowledge' OR 'traditional knowledge' OR 'indigenous practices' OR 'co-management' OR 'restoration' OR 'climate change' OR 'biodiversity' OR 'biodiversity conservation'	759	22
Ramsar Goal 4	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'Ramsar Regional Initiatives'	3	1
	'Wetlands conservation' AND 'scientific guidelines' OR 'technical methodologies' OR 'capacity development' OR 'cooperation' OR 'international cooperation' OR 'awareness raising'	13	4
IPPC Goal A	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'Pest spread' OR 'global pest spread' OR 'agricultural pests'	215	19
IPPC Goal B	'Plant pests' AND 'biodiversity' OR 'biodiversity conservation' OR 'forests' OR 'agriculture' OR 'food security'	847	26
IPPC Goal C	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'phytosanitary measures' OR 'non-tariff barriers to trade' OR 'NTBs' OR 'non-tariff measures' OR 'NTMs'	552	15
IPPC Goal D	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'phytosanitary capacity' OR 'phytosanitary capacity building'	14	3
UNCCD Goal 1	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'desertification' OR 'land degradation' OR 'deforestation'	191	23
	'Desertification' OR 'land degradation' OR 'deforestation' AND 'human dimensions' OR 'affected populations' OR 'affected communities'	365	16
UNCCD Goal 2	'Meta-analysis' OR 'metaanalysis' OR 'review' OR 'systematic review' OR 'synthesis' AND 'combating desertification' OR 'reversing land degradation' OR 'combating deforestation'	65	6
UNCCD Goal 3	'Desertification' OR 'land degradation' OR 'deforestation' AND 'climate change' OR 'biodiversity conservation' OR 'biodiversity'	243	9
UNCCD Goal 4	'Desertification' OR 'land degradation' OR 'deforestation' AND 'international cooperation' OR 'cooperation' OR 'policy' OR 'strategy'	657	12

S3.9 Coordination between the CBD and other MEAs.

To support achievement of the Aichi Targets, the CBD cooperates and coordinates with several other conventions, organizations and processes including all the biodiversity-related conventions and

several MEAs. Specific Memoranda of Understanding, Memoranda of Cooperation and Joint Work Programmes exist between conventions specifying the details of each collaboration (Gomar et al., 2014; CBD 2018a). Two formal liaison groups have been formed with the secretariats of the Rio Conventions (Joint Liaison Group: CBD, UNCCD and UNFCCC) and those of the biodiversityrelated conventions (Biodiversity Liaison Group: CBD, CITES, CMS, Ramsar, IPPC, WHC and ITPGRAF) (CBD 2018b). In a move to improve synergies between conventions and to synchronize between them, several Conventions in the Biodiversity Liaison Group have not only endorsed the Aichi Biodiversity Targets, but also incorporated the most relevant ones into their own strategies and objectives (UNEP-WCMC 2015).

S3.10 The International Treaty on Plant Genetic Resources for Food and Agriculture (ITPGRFA)

Concluded in 2001 under the auspices of FAO, the ITPGRFA entered into force on 29 June 2004, and, as of October 2018, has 145 Parties. It is a legally binding instrument that targets the conservation and sustainable use of plant genetic resources for food and agriculture (PGRFA) and fair and equitable benefit-sharing, in harmony with the CBD, for sustainable agriculture and food security (ITPGRFA Art. 1). The Treaty does not have a strategic plan (although FAO's Commission on Genetic Resources for Food and Agriculture, which has a wider remit, does), and a multi-year programme of work is still under consideration, to be adopted possibly in 2019.

The Treaty's core is its Multilateral System (MLS) of access and benefit-sharing (ABS). The MLS aimed to respond to the specificities of agricultural biodiversity and the 'public good' nature of PGRFA and basic scientific research in general (Cooper et al. 1994; Halewood et al. 2013), for which the CBD bilateral system of exchanges was considered unsuitable (Chiarolla et al. 2013). PGRFA exchange is indispensable for the continuation of agricultural research, as well as for the adaptation of key crops to the new conditions brought about by climate change, and plant pests and diseases. Moreover, when it comes to crop genetic resources, all countries are interdependent and identification of the country of origin is often difficult, given the millennia of agricultural history (ITPGRFA Preamble). The MLS is aimed at facilitating access to, and exchange of, collections of a specified list of crops considered vital for food security and agricultural research (ITPGRFA Annex I), that are under the management and control of Parties, in the public domain, and in international agricultural research centres. It also institutionalizes the sharing of benefits arising from the utilization of these resources (ITPGRFA Arts 10-13; Tsioumani 2018). Areas of conflict may arise with regard to the implementation of the ITPGRFA and the CBD Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization, with regard, for instance, to crop genetic resources that do not belong to Annex I or to crop wild relatives. These issues are expected to be addressed through cooperation among the Secretariats and mutually supportive national legislation.

The benefits to be shared can be monetary or non-monetary. The latter include exchange of information, access to and transfer of technology, capacity building, and facilitated access to the resources, which is recognized as a benefit in itself. Accumulation of monetary benefits arising from commercialization of products developed on the basis of MLS material is

achieved through payments by the users of material, according to the provisions of the standard Material Transfer Agreement (SMTA), which is a standardized private law contract signed by providers and users (ITPGRFA Governing Body Resolution 2/2006; SMTA Arts 6.7 and 6.11). Such payments, together with voluntary donations, are directed to the Treaty's Benefit-sharing Fund, which allocates funds, under the direction of the ITPGRFA Governing Body, primarily to farmers, especially in developing countries, and countries with economies in transition, who conserve and sustainably utilize plant genetic resources for food and agriculture (ITPGRFA Art. 13.3). Importantly, the Treaty recognizes the 'enormous contribution that the local and indigenous communities and farmers of all regions of the world, particularly those in the centres of origin and crop diversity, have made and will continue to make for the conservation and development of plant genetic resources which constitute the basis of food and agriculture production throughout the world' (ITPGRFA Art. 9.1). It acknowledges that Parties are responsible for realizing farmers' rights and that Parties should, as appropriate and subject to national legislation, take measures to protect and promote farmers' rights, including the protection of traditional knowledge, the right to participate equitably in sharing benefits arising from the utilization of PGRFA, and the right to participate in national-level decision-making (ITPGRFA Art. 9.2). Furthermore, the Treaty provides guidance to countries regarding measures and activities to be undertaken to promote the conservation and sustainable use of PGRFA, highlighting the importance of adopting a complementary approach between in situ and ex situ conservation, maintaining diverse farming systems and implementing participatory approaches to plant breeding (ITPGRFA Arts 5-6).

With regard to implementation, the Treaty has been successful in facilitating millions of exchanges of PGRFA, mainly to enable public agricultural research; and in providing support to build the capacities required for PGRFA utilization (ITPGRFA 2017a; Tsioumani 2018). A set of challenges have however arisen with regard to the ability of the MLS to generate and share monetary benefits (Frison et al. 2011). The Benefit-sharing Fund has been operating mostly on the basis of donor country voluntary contributions (ITPGRFA 2013; Moeller and Stannard 2013). The only user-based payment realized since the Treaty's entry into force took place in June 2018. It concerns the payment of USD 119,083 by a Dutch company, equalling 0.77% of seed sales of ten vegetable varieties which used germplasm made available according to the terms of the SMTA. To address this challenge, in 2013 the Governing Body established an intersessional process aiming to 'enhance the functioning the Multilateral System'. While negotiations are ongoing, the Working Group has discussed a possible way forward involving elaborating a subscription system for access to PGRFA in the MLS. Compliance-related challenges also persist with regard to material to be included in the MLS, with some Parties failing to notify the Secretariat accordingly and thus not providing facilitated access to MLS material.

With PGRFA management being at the intersection between food security, agriculture and the environment (ITPGRFA Preamble), the International Treaty provides an effective policy response to the global challenges of crop diversity loss, the need for sustainable food production and climate change adaptation. Its effective implementation therefore contributes

to several SDGs, particularly Goals 2 and 15, but also indirectly to others such as Goals 1, 12, 13, and 17 (ITPGRFA 2017b).

S3.11 The United Nations Convention on the Law of the Sea

International law as reflected in the UNCLOS, in force since 1994, provides for rights and obligations of States Parties with regard to the use of the world's oceans and their resources, and the protection of the marine and coastal environment. UNCLOS also contains provisions that reflect nations' jurisdiction within different maritime zones and contains restrictions and requirements related to the use and management of marine environmental resources. It includes provisions relating to the conservation and sustainable use of marine living resources (e.g. Articles 61 and 62 and 116 to 129), the conservation of stocks within the exclusive economic zone of two or more coastal states (e.g. Article 63), and the conservation and appropriate management of highly migratory species (e.g. Article 64), marine mammals (e.g. Articles 65 and 120) and anadromous, catadromous and sedentary species stocks (e.g. Articles 66 to 68). UNCLOS is supplemented by a 1994 implementing agreement relating to Deep Seabed Mining and the 1995 agreement for the Implementation of the Provisions of UNCLOS relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, also known as the UN Fish Stocks Agreement (UNFSA).

The UNFSA, in force since 2001 is considered to be the most important legally binding global instrument adopted for the long-term conservation, management and sustainable use of fisheries resources. The UN Division on Ocean Affairs and Law of the Sea deals mainly with implementation of the UNFSA on highly migratory and straddling fish stocks, and the Food and Agriculture Organization (FAO) on fisheries advice and statistics within and beyond national jurisdiction under the FAO Code of Conduct for responsible fishing. Since the adoption of the Code, there has been significant progress in the monitoring and reporting of the status and catch statistics of several fish stocks; however, there are several countries that either do not report their catches or that produce data that are not considered totally reliable (FAO, 2016). The most important regulatory bodies for the UNFSA are the Regional Management Organizations or Arrangements (RFMO/As), a subset of the c.50 Regional Fishery Bodies (RFBs), which conduct monitoring, collect fisheries statistics, assess resources, and make management (http://www.fao.org/figis/geoserver/factsheets/rfbs.html). Each RFMO has its own goals regarding conservation and management of stocks or populations.

S3.12 Polar agreements and cooperative arrangements

CCAMLR, in force since 1982 is an international treaty with 25 members and a further 11 acceding countries. It is considered to be the ocean counterpart for the Antarctic Treaty, which was signed in 1959 to ensure, in the interest of mankind, that the Antarctic will be used exclusively for peaceful purposes and will not become the subject of international discord. CCAMLR originated in response to increasing commercial interest in Antarctic krill resources, a keystone component of the Antarctic ecosystem affecting all the food web, and a history of over-exploitation of several other marine resources in the Southern Ocean. It agrees

on a set of conservation measures that determine the use of Antarctic marine living resources based on the best available scientific information. CCAMLR applies to all Antarctic populations of finfish, molluscs, crustacean and seabirds found south of the Antarctic Polar Front (the CCAMLR Area, covering c.10% of the world's ocean) and excludes whales and seals, which are managed by other conventions. CCAMLR is overseen by a regulatory body (the Commission on CAMLR) and by a scientific advising body (the Scientific Committee on CCAMLR). The Commission is regarded as a model for regional cooperation using an ecosystem-based approach that allows sustainable harvesting ("rational use") and maintains scientific research and monitoring programmes to address risks to commercially exploited fish stocks in the Southern Ocean. The overarching objective of CCAMLR is the "conservation of Antarctic living resources", and for the purposes of the convention, "conservation" refers to their rational use. The 33 articles of CCAMLR provide the rights and obligations of the parties to achieve this goal.

The Conservation for the Arctic Flora and Fauna (CAFF) is the biodiversity working group of the Arctic Council, consisting of National Representatives from the eight Arctic Council Member States, and representatives of Indigenous Peoples' organizations and other Arctic Council observer countries and organizations. The CAFF program was inaugurated in 1992 with the vision of being a "distinct forum for scientists, indigenous peoples and conservation managers ...to exchange data and information on issues such as shared species and habitats and to collaborate, as appropriate for more effective research, sustainable utilization and conservation" (CAFF, 1998). Through research and monitoring, CAFF's goals are the conservation and sustainable use of Arctic biodiversity and habitats and the provision of data on the status and trends of the Arctic's living resources to pertinent governments and residents. CAFF provides a series of recommendations on actions related to climate change, ecosystem-based management, biodiversity issues within main economic activities such as gas development, shipping, fishing, tourism and mining, on the protection of important areas and on the reduction of stressors on biodiversity and migratory species (CAFF, 1998).

CAFF and the CBD, along with the Arctic Biodiversity Observation Network of the Group on Earth Observations Biodiversity Observation Network have endorsed the Circumpolar Biodiversity Monitoring Program, an international network of scientists, governments, Indigenous organizations and conservation groups aiming to monitor living resources in the marine, freshwater, terrestrial and coastal resources of the Arctic. This program has contributed to the Arctic Biodiversity Assessment, with information on status and trends of terrestrial, freshwater and marine ecosystems and Arctic mammals, birds, fishes, amphibians and reptiles, invertebrates and parasites (CAFF, 2013, 2017). The program will develop regular State of the Arctic Biodiversity Reports. These reports contain status and trends on defined Focal Ecosystem Components which have the potential to be regarded as Arctic indicators of biodiversity changes. Further reports will describe existing monitoring status and advise on future monitoring and collaboration with CAFF (Christensen et al. 2018).

S3.13 Methods for cross-cutting synthesis across goals and targets

To structure a crosscutting synthesis, we identified links between the 20 individual Aichi Targets, 17 SDGs and 42 SDG targets (here referred to collectively as 'targets') to the different components of Nature and Nature's Contributions to People (NCPs) in the IPBES conceptual framework (CF) using a combination of an expert panel exercise and a Delphi approach. The main goals of this exercise were to use the links identified:

- as a basis for identifying meaningful clusters of targets for summarizing and synthesizing information,
- for visualizing the context of the Aichi Targets and SDS in the form of infographics,
- to explore synergies and trade-offs between targets
- to enhance the internal coherence of the whole Global Assessment

To identify the links, we used a two-phase elicitation process to complete a matrix of each Aichi Target, SDG and SDG target against the CF elements for Nature and NCP. In the first phase, the matrix was filled with initial scores by 31 experts (authors of the IPBES Global Assessment) at the IPBES Global Assessment second author meeting (Cape Town, Sep 2017). Four panels of 4-8 experts completed cells in one of four subsets of the matrix (Aichi Targets vs Nature, Aichi Targets vs NCPs, SDGs vs Nature, SDGs vs NCPs), following the same set of detailed instructions. Specifically, the panels addressing Nature were asked: "What impact would the process of successfully achieving these targets have on the quantity (total area, range, extent) and/or quality (ecological state/condition, degradedness, species diversity) of these ecosystems?", and the panels addressing NCPs were asked "What impact would the process of successfully achieving these targets have on the supply/availability of the NCPs?"

The relationships were scored as:

- 0: no clear relationship (or the relationship is very weak and very indirect);
- 1: positive relationship (quantity and/or quality ecosystems will increase with successful target implementation);
- 2: negative relationship (quantity and/or quality ecosystems will decrease with successful target implementation);
- 3: there is a (direct or indirect) relationship, but its direction is unclear, ambiguous, or dual (e.g. a U-shaped relationship), the relationship varies geographically or is pathway-dependent;
- 11, 22, 33: double digits were used to indicate particularly strong relationships.

Each panel went line by line through their matrix subset, discussing scores until reaching consensus, taking note of the key discussion points, and going back to modify previous scores as necessary to ensure consistency.

In the second phase, the initial scores given by the panels were cross-validated in a Delphi process by a smaller team of four Chapter 3 authors. In this phase we started with scoring the SDG targets, distributing the 42 targets between the four experts. Each expert then scored their targets starting out from the goal-level scores from the first phase to (a) look for indications in the target text for

pathways on how the target "should be" achieved (considering all feasible/logical pathways equally probable where such indications were lacking), and (b) focussing on general/major trends at a global level and avoiding the documentation of "exceptions" (which tend to capture attention easily and appear more significant then they are).

Furthermore, for each target two short "interpretations" were created (for Nature and NCPs separately), which explained and documented the logic for the scores assigned, starting out from the definitions of the CF elements and the language of the target. In a next round every row of scores was reviewed by a second expert, making comments on the interpretetions and the individual scores. Disagreements were resolved by the whole team. Where it was necessary, "column-wise" interpretations were also added to the individual Nature/NCP elements, and these were also discussed and agreed. This whole exercise created a detailed shared understanding, and a high level of coherence in the scores given. Based on this shared understanding, the goal-level SDG scores and the Aichi scores from the first phase were also reviewed and revised where necessary, thus improving the coherence and consistency of the scores and the underlying interpretations. The proposed interpretations and score changes were also discussed until consensus was achieved. This second Delphi phase of the scoring process was carried out in Oct-Dec 2017 in Google Spreadsheets, making use of the commenting and collaboration facilities of this environment. Altogether, 39% of the scores were left completely unchanged, and in a further 37% of the cell values the intensity of the score was reduced by a single step (e.g. from "11" to "1" or from "1" to "0"). Intensity was increased in 9% of the cells, and in 15% of the cells a more thorough revision took place. Interestingly, all the negative scores ("2", "22": 2% of the cells) from the first phase disappeared (turned to "0" or "3") during the crosschecking process, which means that all the negative influences published in the literature are pathway or location specific, and thus covered by "3"s and "33"s. The final matrix is shown in Table S3.8.

To generate summary clusters from the matrix we applied an agglomerative hierarchical clustering approach in four steps: (1) the matrix scores were recoded to linear scale(s), (2) inter-target distance matrices were computed, (3) clustering algorithms were applied, and (4) consensus clusters were identified based on the ensemble of the outputs. In steps (1)-(3) there were multiple valid design choices, which resulted in 30 slightly different clustering outputs. These outputs were then reanalysed in step (4) establishing a more robust and reliable consensus structure.

In terms of recoding, the following 3 options were considered:

- *partial*: just focussing on the "strength" of the relationships: "0" --> 0; "1", "3" --> 1; "11", "33" --> 2,
- *two-dimensional*: recoding each score to two variables: one being strength just as above, and the other was "direction", computed this way: "33", "3" --> -1; "0" --> 0; "1", "11" --> 1,
- *flat*: combining the two variables above to a single variable as "strength * direction": "33" -->
 -2, "3" --> -1; "0" --> 0; "1" --> 1; "11" --> 2.

For the distance metric we considered the *Euclidean* and the *Manhattan* metrics, and we also considered five clustering algorithms: the *single link*, *complete link*, *average*, *Ward* (D2), and *McQuitty* algorithms (Legendre and Legendre, 2012, Murtagh and Legendre, 2014). All the computations were performed in R (v3.4.3, R Core Team, 2017), using *hclust* in the *stats* package of core R.

This process generated 30 classification outputs, which exhibited many common patterns. To generate the "cross-cutting themes" based on the clustering outputs we started out from a specific clustering (two-dimensional coding with Euclidean distance and the Ward method; see Figure S3.1) and a subjective cutoff threshold resulting in 10 clusters. In a subsequent expert examination two of these clusters were considered to be two heterogeneous and out of the main scope (as they grouped targets that were too distantly related to Nature & NCPs), and one cluster was split into two parts. For the thus remaining 9 clusters 1-3 core Aichi targets and SDG goals were identified (SDG targets were excluded), which formed the core of the new cross-cutting themes. Then for each theme, further related targets (Aichi targets, SDG goals and targets) were identified based on the ensemble of 30 classification outputs. Further targets that were associated to the core ATs and SDGs of each theme in more than 50% of the cases were also added to each theme, creating some overlap between the various themes (Table S3.7 and Fig. 3.23).

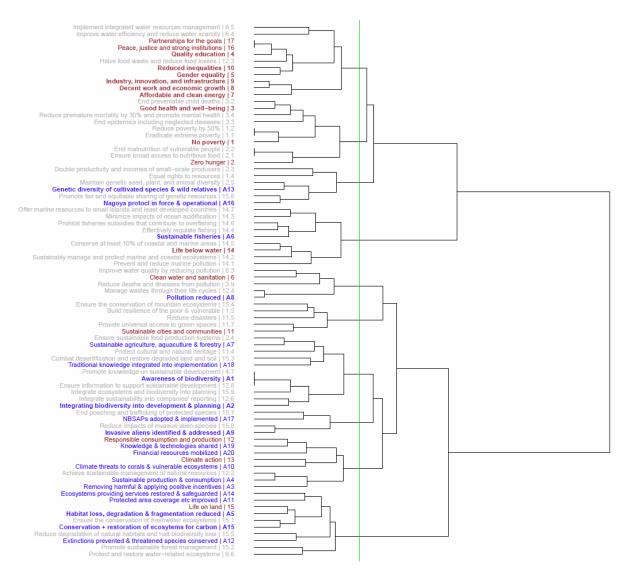


Figure S3.1: The clustering output, cutting thresholds, and core targets (in **bold**) which were selected for the generation of the cross-cutting themes.

Table S3.9. The composition of the nine overarching themes (containing all Aichi Targets, SDG goals, and the 42 selected SDG targets) identified during the clustering exercise. The targets in bold are those that were identified as core targets for each theme (indicated by bold links in Fig 3.23).

1 Terrestrial and freshwater conservation and restoration
A05 Habitat loss, degradation & fragmentation reduced
A15 Conservation + restoration of ecosytems for carbon
A07 Sustainable agriculture, aquacutlure & forestry
A11 Protected area coverage etc improved
A12 Extinctions prevented & threatened species conserved
A14 Ecosystems providing services restored & safeguarded
S02.4 Ensure sustainable food production systems
S06.6 Protect and restore water-related ecosystems
S15 Life on land
S15.1 Ensure the conservation of freshwater ecosystems
S15.2 Promote sustainable forest management
S15.3 Combat desertification and restore degraded land and soil
S15.5 Reduce degradation of natural habitats and halt biodiversity loss
2 Marine conservation and sustainable use
A06 Sustainable fisheries
S14 Life below water
S06 Clean water and sanitation
S14.1 Prevent and reduce marine pollution
S14.2 Sustainably manage and protect marine and coastal ecosystems
S14.3 Minimize impacts of ocean acidification
S14.4 Effectively regulate fishing
S14.5 Conserve at least 10% of coastal and marine areas
S14.6 Prohibit fisheries subsidies that contribute to overfishing
S14.7 Offer marine resources to small islands and least developed countries
3 Sustaining genetic resource diversity
A13 Genetic diversity of cultivated species & wild relatives
A16 Nagoya protocl in force & operational
S01.4 Equal rights to resources
S02.3 Double productivity and incomes of small-scale producers
S02.5 Maintain genetic seed, plant, and animal diversity
S15.6 Promote fair and equitable sharing of genetic resources
4 Addressing pollution
A08 Pollution reduced
S03.9 Reduce deaths and illnesses from pollution
S06 Clean water and sanitation
S06.3 Improve water quality by reducing pollution
S12.4 Manage wastes through their life cycles
5 Addressing invasive alien species

A09 Invasive aliens identified & addressed
A17 NBSAPs adopted & implemented
S15.7 End poaching and trafficking of protected species
S15.8 Reduce impacts of invasive alien species
6 Addressing poverty, hunger and health
S01 No poverty
S03 Good health and well-being
S01.1 Eradicate extreme poverty
S01.2 Reduce poverty by 50%
S02 Zero hunger
S02.1 Ensure broad access to nutritious food
S02.2 End malnutrition of vulnerable people
S03.2 End preventable child deaths
S03.3 End epidemics including neglected diseases
S03.4 Reduce premature mortality by 30% and promote mental health
7 Sustainable economic production
S07 Affordable and clean energy
S08 Decent work and economic growth
S09 Industry, innovation, and infrastructure
S04 Quality education
S05 Gender equality
S06.4 Improve water efficiency and reduce water scarcity
S06.5 Implement integrated water resources management
S10 Reduced inequalities
S12.3 Halve food waste and reduce food losses
S16 Peace, justice and strong institutions
S17 Partnerships for the goals
8 Ensuring equity and education
S04 Quality education
S05 Gender equality
S10 Reduced inequalities
S16 Peace, justice and strong institutions
S17 Partnerships for the goals
S06.4 Improve water efficiency and reduce water scarcity
S06.5 Implement integrated water resources management
S07 Affordable and clean energy
S08 Decent work and economic growth
S09 Industry, innovation, and infrastructure
S12.3 Halve food waste and reduce food losses
9 Mainstreaming biodiversity
A01 Awareness of biodiversity
A02 Integrating biodiversity into development & planning
A07 Sustainable agriculture, aquacutlure & forestry
A18 Traditional knowledge integrated into implementation
S02.4 Ensure sustainable food production systems

S04.7 Promote knowledge on sustainable development
S11.4 Protect cultural and natural heritage
S12.6 Integrate sustainability into companies' reporting
S12.8 Ensure information to support sustainable development
S15.3 Combat desertification and restore degraded land and soil
S15.9 Integrate ecosystems and biodiversity into planning

Table S3.10 Relationships between targets (Aichi Targets [A1–A20], SDG goals [1–17], and selected SDG targets [1.1–15.9], see short names in Table S3.9) and the components of Nature (N1: Tropical and subtropical dry and humid forests; N2: Temperate and boreal forests and woodlands; N3: Mediterranean forests, woodlands and scrub; N4: Tundra and high mountain habitats; N5: Tropical and subtropical savannas and grasslands; N6: Temperate grasslands; N7: Deserts and xeric shrublands; N8: Wetlands – peatlands, mires, bogs; N9: Urban/semi-urban; N10: Cultivated areas; N11: Cryosphere; N12: Aquaculture areas; N13: Inland surface waters and water bodies/freshwater; N14: Shelf ecosystems (neritic and intertidal/littoral zone); N15: Open ocean pelagic systems (euphotic zone); N16: Deep sea; N17: Coastal areas intensively and multiply used by humans). The scores used in the table are: 0: no relationship; 1, 11: positive relationship; 2,22: negative relationship; 3, 33: ambiguous relationship; double digits indicate strong relationships (see also the explanations in the text). An annotated version of this table is available online at https://goo.gl/D94hRL.

	N01	N02	N03	N04	N05	90N	N07	N08	60N	N10	N11	N12	N13	N14	N15	N16	N17
	Z	Z	Z	Z	Z	Z	Z	Z	Z	Z		Z	Z	Z	Z	Z	Z
A1	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1
A2	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1
A3	1	1	1	1	1	1	1	1	1	1	0	1	1	11	11	11	1
A4	11	11	11	11	11	11	1	11	1	1	0	1	11	11	11	11	11
A5	11	11	11	1	11	1	1	11	0	3	0	3	1	0	0	0	0
A6	0	0	0	0	0	0	0	0	0	0	0	3	11	11	11	11	0
A7	11	11	11	0	1	1	0	1	1	11	0	11	11	1	1	0	1
A8	1	1	1	1	1	1	1	1	11	11	1	11	11	11	11	11	11
A9	1	1	1	1	1	1	1	1	0	1	0	1	11	11	0	0	0
A10	1	1	1	1	1	1	1	1	0	0	11	1	1	11	11	11	1
A11	11	11	11	11	11	11	1	11	3	3	0	3	11	11	11	11	0
A12	11	1	11	1	11	1	1	11	0	0	1	0	11	11	1	1	0
A13	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0
A14	11	11	11	11	11	11	1	11	1	3	0	3	11	11	11	1	1
A15	11	11	11	11	11	11	11	11	0	3	0	3	11	11	0	0	0
A16	1	0	1	0	0	0	0	0	0	1	0	1	0	0	0	0	0
A17	1	1	1	1	1	1	1	1	0	0	0	0	1	1	0	0	0
A18	11	0	1	11	1	1	1	0	0	1	0	1	1	1	0	0	0
A19	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
A20	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1	0	0	0	0	0	0	0	0	3	3	0	3	3	3	0	0	3
1.1	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	3

	N01	N02	N03	N04	N05	90N	N07	N08	60N	N10	N11	N12	N13	N14	N15	N16	N17
1.2	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	3
1.4	3	0	0	0	3	0	0	0	0	0	0	0	3	3	3	0	0
1.5	1	0	0	0	1	0	0	1	1	0	0	0	1	1	0	0	0
2	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	3
2.1	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	3
2.2	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	3
2.3	3	0	3	0	3	3	0	0	3	3	0	3	3	3	3	0	0
2.4	1	1	1	0	1	1	0	1	1	1	0	1	1	1	1	0	1
2.5	0	0	0	0	0	0	0	0	0	3	0	3	0	0	0	0	0
3	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	3
3.2	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	3
3.3	0	0	0	0	0	0	0	11	3	3	0	3	11	0	0	0	3
3.4	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	3
3.9	0	0	0	0	0	0	0	11	11 0	11	0	1	11	1	0	0	1
4	0	0	0	0	0	0	0	0		0	0	0	0	0	0	0	0
4.7 5	1 0	1 0	1 0	1 0	1 0	1 0	1 0	1 0	1 0	1 0	0 0	1 0	1 0	1 0	1 0	1 0	1 0
5 6	1	0	0	0	0	0	0	11	1	3	0	0 1	11	0 1	0 1	1	1
6.3	0	0	0	0	0	0	0	11	11	11	0	11	11	11	11	11	11
6.4	0	0	0	0	0	0	0	3	1	3	0	1	3	0	0	0	1
6.5	1	1	1	0	0	0	3	1	3	3	0	1	1	1	0	0	0
6.6	11	1	1	1	0	0	0	11	0	3	0	1	11	1	0	0	0
7	0	0	0	0	0	0	0	0	1	3	0	0	0	0	0	0	0
8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
11	1	0	0	0	1	0	0	1	11	0	0	0	1	1	0	0	1
11.4	11	1	1	1	11	1	1	1	0	1	0	1	1	11	1	0	1
11.5	1	1	1	0	1	0	0	11	11	0	0	0	11	11	0	0	11
11.7	0	0	0	0	0	0	0	0	11	0	0	0	0	0	0	0	1
12	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
12.2	11	11	11	11	11	11	1	11	1	1	0	1	11	11	11	11	11
12.3	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0
12.4	1	1	1	1	1	1	1	1	11	11	1	11	11	11	11	11	11
12.6	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1
12.8	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1
13	1	1	1	1	1	1	1	1	1	1	11	1	1	1	3	3	1
14 14 1	0	0	0	0	0	0	0	0	0	0	0	1	0	11	11	11	1
14.1 14.2	0	0 0	0 0	0 0	0 0	0 0	0 0	1 0	1 0	1 0	0	11	1	11 11	11 11	11	11
14.2 14.3	0 0	0	0	0	0	0	0	0	0	0	0 0	11 11	0 0	11 11	11 11	11 11	11 1
14.5 14.4	0	0	0	0	0	0	0	0	0	0	0	0	0	11 11	11 11	11	1 0
14.4	0	0	0	0	0	0	0	0	0	0	0	0	0	11	11	11	0
14.5 14.6	0	0	0	0	0	0	0	0	0	0	0	0	0	11	11	11	0
14.7	0	0	0	0	0	0	0	0	1	0	0	11	0	11	11	11	11
15	11	1	11	1	1	1	1	1	0	3	0	3	11	1	0	0	0
1.7	11	11	11	11	11	11	1	11	3	3	0	3	11	1	0	0	0
			11	1	0	0	0	0	0	3	0	0	1	0	0	0	0
15.1		11		-				1	1	3	0	1	11				
15.1 15.2	11	11 1		1	11	1	11	1	1	5	0	1	11	1	0	0	0
15.1 15.2 15.3	11 11	1	11	1 11	11 0	1 0	11 0								0	0 0	0 0
15.1 15.2	11			1 11 1	11 0 11	1 0 1	11 0 1	0 11	1 0 0	3 0 0	0 0 1	1 0 0	11 11 11	1 1 0		0 0 0	0 0 0

	N01	N02	N03	N04	N05	90N	N07	N08	000	N10	N11	N12	N13	N14	N15	N16	N17
15.7	11	1	1	1	11	1	1	1	0	0	0	0	1	1	0	0	0
15.8	1	1	1	1	1	1	1	1	0	1	0	1	11	1	0	0	0
15.9	1	1	1	1	1	1	1	1	1	1	0	1	1	1	0	0	0
16	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
17	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table S3.11 Relationships between targets (Aichi Targets [A1–A20], SDG goals [1–17], and selected SDG targets [1.1–15.9], see short names in Table S3.9) and NCPs (C1: Habitat creation and maintenance; C2: Pollination and seed dispersal; C3: Regulation of air quality; C4: Regulation of climate; C5: Regulation of ocean acidification; C6: Regulation of freshwater quantity, flow and timing; C7: Regulation of freshwater and coastal water quality; C8: Formation, protection and decontamination of soils; C9: Regulation of hazards and extreme events; C10: Regulation of organisms detrimental to humans; C11: Energy; C12: Food and feed; C13: Materials and assistance; C14: Medicinal, biochemical and genetic resources; C15: Learning and inspiration; C16: Physical and psychological experiences; C17: Supporting identities; C18: Maintenance of options). The scores used in the table are: 0: no relationship; 1, 11: positive relationship; 2,22: negative relationship; 3, 33: ambiguous relationship; double digits indicate strong relationship (see also the explanations in the text). An annotated version of this table is available online at https://goo.gl/D94hRL.

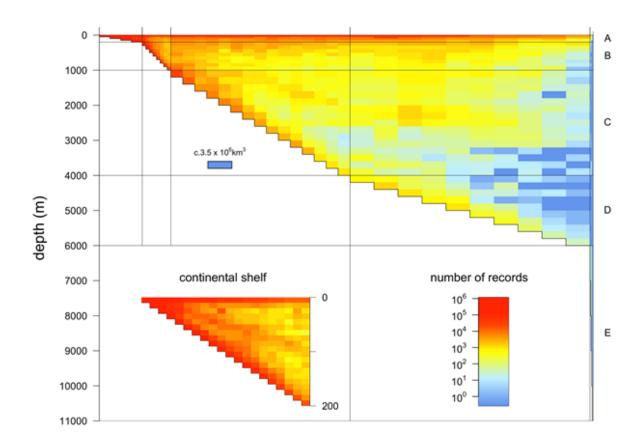
	C01	C02	C03	C04	C05	C06	C07	C08	C09	C10	C11	C12	C13	C14	C15	C16	C17	C18
A1	1	1	1	1	1	1	1	1	1	1	3	1	1	1	11	1	1	1
A2	1	1	1	1	1	1	1	1	0	0	3	1	1	1	1	1	0	11
A3	11	1	0	1	1	1	1	1	0	0	3	3	3	3	0	0	3	1
A4	1	1	0	1	1	1	1	1	0	1	3	3	3	3	1	1	0	1
A5	11	11	0	11	11	1	1	11	1	11	3	3	3	3	1	11	1	11
A6	11	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0	1	11
A7	1	1	0	0	0	1	11	11	1	1	3	1	3	1	1	1	1	11
A8	1	1	1	0	0	0	11	11	0	1	0	1	1	1	0	1	0	1
A9	1	1	0	0	0	0	1	0	0	11	0	1	1	1	3	3	0	1
A10	1	1	1	1	1	3	3	3	1	1	3	3	3	3	0	0	0	11
A11	11	11	1	1	1	1	1	1	1	1	3	3	3	3	1	11	1	11
A12	1	11	0	0	0	1	1	1	1	1	3	3	3	11	1	11	1	11
A13	0	0	0	0	0	0	0	0	0	1	0	1	1	11	1	1	1	11
A14	11	11	11	11	11	11	11	11	11	11	1	3	3	1	1	11	1	11
A15	11	11	1	11	11	1	1	1	1	11	3	3	3	3	1	11	1	11
A16	0	0	0	0	0	0	0	0	0	0	0	0	0	11	1	1	1	11
A17	1	1	0	0	0	0	0	0	0	1	3	3	3	3	0	1	0	11
A18	1	1	0	0	0	0	0	1	1	1	3	1	1	11	1	11	11	11
A19	0	0	1	1	1	0	1	1	0	0	3	3	3	1	1	1	0	0
A20	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
1	0	0	0	0	0	0	0	0	0	0	3	3	3	3	0	1	3	0
1.1	0	0	0	0	0	0	0	0	0	0	3	3	3	3	0	1	3	0
1.2	0	0	0	0	0	0	0	0	0	0	3	3	3	3	0	1	3	0
1.4	0	0	0	0	0	0	0	0	0	0	1	3	3	1	1	1	1	3
1.5	0	0	0	0	0	1	0	0	11	1	0	0	0	1	1	1	1	1

	C01	C02	C03	C04	C05	C06	C07	C08	C09	C10	C11	C12	C13	C14	C15	C16	C17	C18
2	0	3	0	0	0	3	0	3	0	3	3	11	3	1	1	1	1	1
2.1	0	3	0	0	0	3	0	3	0	3	3	11	3	0	0	0	0	0
2.2	0	3	0	0	0	3	0	3	0	3	3	11	3	0	0	0	0	0
2.3	0	1	0	0	0	3	0	1	0	1	3	11	3	1	1	1	1	1
2.4	1	1	0	0	0	1	1	11	11	1	3	1	3	1	1	1	1	11
2.5	0	0	0	0	0	0	0	0	0	1	0	1	1	11	1	1	1	11
3	0	0	0	0	0	0	1	0	0	1	3	3	3	1	0	1	0	0
3.2	0	0	0	0	0	0	0	0	0	1	3	3	3	1	0	0	0	0
3.3	0	0	0	0	0	1	1	0	0	1	3	3	3	1	0	0	0	0
3.4	0	0	0	0	0	0	1	0	0	0	3	3	3	1	1	11	1	0
3.9	1	1	11	0	0	0	11	11	0	1	3	3	3	1	0	1	0	1
4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
4.7	1	1	1	1	1	1	1	1	1	1	3	1	1	1	11	1	1	1
5	0	0	0	0	0	0	0	0	0	0	0	3	3	3	0	0	0	0
6	1	Õ	ů 0	ů 0	Õ	1	11	1	Õ	1	Õ	1	1	0	Õ	1	Õ	1
6.3	1	0	1	0	0	1	11	1	0	11	0	1	1	0	0	1	0	1
6.4	0	0	0	0	0	0	1	0	0	0	0	1	1	0	0	1	0	0
6.5	0	0	0	0	0	0	11	0	0	0	3	1	1	0	0	1	0	0
6.6	1	1	0	1	1	11	11	1	1	1	3	3	3	3	1	1	1	1
7	0	0	3	3	3	0	3	3	0	0	33	3	3	0	0	0	0	0
8	0	0	3	3	3	0	3	3	0	0	3	3	3	0	0	1	0	3
9	0	0	3	3	3	0	3	3	0	0	3	3	3	0	1	1	0	3
10	0	0	0	0	0	0	0	0	0	0	3	3	3	0	0	1	0	0
11	0	0	1	0	0	1	1	1	1	1	0	0	0	0	11	11	0	0
11.4	1	1	0	0	0	1	1	1	0	1	0	0	0	0	11	11	1	1
11.5	1	0	0	0	0	11	1	1	11	1	0	0	0	0	0	0	0	1
11.7	0	0	11	0	0	1	1	1	1	11	0	0	0	0	11	11	1	0
11.7	0	0	0	1	1	0	1	1	0	0	3	3	3	3	1	0	0	1
12.2	1	1	0	∎ 1	∎ 1	1	1	1	1	1	3	3	3	3	1	1	1	1
12.2	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0
12.3	1	1	1	0	0	0	11	1	0	1	0	1	1	1	0	1	0	1
12.4	0	0	1	1	1	1	1	1	0	0	3	1	1	0	0	0	0	1
12.0	1	1		1	1	1	1	1	1	1	3	1		1	0 11	1	1	1
12.0			1	1 11	1 11	1 3	1 3	1 3		1	3 3	0	1 0	1 0	0	1 0	1 0	1
13 14	1 11	1 0	1 0	0	1	3 0	3 0	3 0	1 0	0	3 0	3	3	0 3	1	1	1	1 11
14 14.1	1	0	0	0	1 0			1	0	0	3	3	3	3		1 11	1	11
14.1	1 11	0	0	0	0	1 0	1 1	1 0	1	0	5 0	3	3	3	1 1	1	1	11
14.2 14.3	1																	
14.3 14.4		0	0	0	11	0	0	0	0	0	0	1	1	1	1	1	1	11
	11	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0	1	11
14.5	11	0	0	0	0	0	0	0	0	0	0	3	3	3	1	1	1	11
14.6	11	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0	33	11
14.7	1	0	0	0	0	0	1	0	0	0	0	1	1	1	1	1	11	1
15	11	11	0	1	1	1	1	1	1	1	3	3	3	3	1	11	1	11
15.1	11	11	1	1	1	1	1	1	1	11	3	3	3	3	1	11	1	11
15.2	11	1	1	11	11	1	1	1	1	1	3	3	3	3	1	11	1	11
15.3	1	1	0	0	0	1	1	11	1	1	1	11	11	1	1	1	1	11
15.4	1	1	0	0	0	11	1	1	1	1	0	0	0	1	1	1	1	1
15.5	11	11	0	0	0	1	1	1	1	11	3	3	3	3	1	11	1	11
15.6	0	0	0	0	0	0	0	0	0	0	0	3	3	11	1	1	1	11
15.7	0	1	0	0	0	0	0	0	0	3	0	3	3	3	3	3	33	1
15.8	1	11	0	0	0	0	1	0	0	11	0	1	1	1	3	3	1	1
15.9	1	1	1	1	1	1	1	1	1	1	3	1	1	1	1	1	1	11

	C01	C02	C03	C04	C05	C06	C07	C08	C09	C10	C11	C12	C13	C14	C15	C16	C17	C18
16	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
17	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

S3.14 Additional information on knowledge gaps

Figure S3.5. Global distribution of ~20 million OBIS records across depth zones of the ocean. (A) Continental shelf (0-200 m), (B) Mesopelagic continental slope (200-1000), (C) Bathypelagic continental slope (1000-4000), (D) Abyssal plain (4000-6000), (E) Hadal (>6000) zones. The inset shows the continental shelf zone in more detail. Source: updated from Webb et al. (2010) with ~7 million records and Appeltans et al. (2016) with ~19 million records. OBIS: Ocean Biogeographic Information System (www.iobis.org).



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