IPBES Global assessment Chapter 4 - Supplementary materials

Contents

Appendix 4.1 – Supporting materials to section 1

A4.1.1 Methodology for Literature Search, Review and Analysis

General

We conducted a systematic review of the scientific literature dealing with future scenarios of biodiversity, ecosystem services, and human well-being. Peer reviewed literature was identified by using a search-based approach, which was supplemented by a screening of key review papers and their references as well as expert-based identification of missing publications.

Literature Search and Supplementation

A list of publications was retrieved using a search pattern in Web of Science as well as Scopus (see Box A1.1 and A1.2 respectively). The search pattern resulted from discussion among Chapter 4's experts, in order to cover the scope and objectives of the Chapter, including the multiple drivers, the different biomes, the diversity of life, and the different disciplines. We restricted the search to the last 10 years, as the time and human resources were limited, and to capture, in particular, the period since ca. the IPCC AR5, and CBD GBO3 and GBO4 reports.

Box A4.1.1: Search Terms in Web of Science (TS: Topic; SU: Research Area)

```
TS = (
       ( "Global*" ) AND
       ( "Future impact*" OR "Future effect*" OR "Future response*" OR
         "projecti*" OR "forecast*" OR "scenario*" ) AND
       ( "species" OR "biological" OR "ecological" OR "nature" OR
         "biodiversity" OR "natural resource*" OR "organism*" OR
         "ecosystem*" OR "ecological service*" OR "conservation" )
     )
AND
SU = (
       Agriculture OR Biodiversity & Conservation OR Biophysics OR
       Entomology OR Environmental Sciences & Ecology OR Fisheries OR
       Forestry OR Marine & Freshwater Biology OR Plant Sciences OR
       Zoology
     )
```

Box A4.1.2 - Search Terms in Scopus (ABS: abstract; KEY: keywords)

```
TITLE-ABS-KEY (
                ( "global") AND
                ( "Future impact*" OR "Future effect*" OR "Future response*" OR
                  "projecti*" OR "forecast*"  OR "scenario*" ) AND
                ( "species" OR "biological" OR "ecological" OR "nature" OR
                  "biodiversity" OR "natural resource*" OR "organism*" OR
                  "ecosystem*" OR "ecological service*" OR "conservation" )
              )
AND    SUBJAREA ( agri OR envi OR soci OR econ)
AND    DOCTYPE ( ar )
AND    PUBYEAR  >  2008
```

The search resulted in 4764 articles from Web of Science (WoS) and Scopus combined. Automatic searches from literature databases are always prone to a failure rate, due to the search pattern itself, but also to imperfect referencing of the scientific publications. The 4764 articles were distributed among the authors of the chapter to check for relevance, based on individual's area of expertise.

Papers were screened by title and abstract and excluded from the database if they did not full-fill the following criteria:
(i) The papers must present results from scenarios of future impacts on Nature, NCP, or GQL. Scenarios of drivers only are not included. For example, scenarios of climate change only, which do not address the impacts on Nature components, NCP or GQL are not considered relevant to the chapter.

(ii) The selected papers must be original research papers. They must not be a compilation, or a review, of already published scenarios. Doing so, we aimed to avoid duplication of metadata.

(iii) The selected papers must have a global scale scope: global scale scenarios, or multiple sites scenarios with a large spread of the sites across the globe, or continental scale scenarios, or ocean scale scenarios.

(iv) The selected papers must deal with scenarios which are anchored in future. For example, we did not include scenarios exploring specific management measures that do not explicit a time horizon in future.

Applying this set of criteria, 857 papers (ca. 18% of the initial set of papers) were identified as being relevant to the Chapter (see https://rkrug.github.io/IPBES_Global_Assessment_Chapter_4_LiteratureReview/ for a complete list of these references). In the process, we identified several reasons why a substantial number of papers were out of scope, in addition to the selection criteria:

- Scenarios of drivers' changes and impacts on biodiversity, ecosystem services and human well-being can be mentioned in the abstract of papers only for providing the context of the study or its rationale. For example, the importance of rainfall projections on Amazonian rainforest may be mentioned to contextualize the study but was not the substance of the paper.

- Likewise, key terms can be used in the abstract as perspectives of the study. For example, "global change" is such a key term which is often used to broaden the scope of a paper's results.

- The term "scenario" is in some studies used in place of hypothesis or assumption, or as a process-based explanation of observations.

- Many papers were not deemed relevant for the chapter because they were too local. The search hits the keyword "global" but this could also "global" drivers (e.g. global warming), but not in the sense of scenarios run at global scale or global diversity patterns. Also, global can be associated with other terms not related at all with global scale scenarios, such as "global database", or "global understanding".

Authors' expertise allowed to quickly realize that the automatic search in literature databases missed key references. We agreed on common guidelines to include additional papers in a stepwise manner, with the same purpose of transparency and objectivity.

As an entry point, key recent review papers and global assessments (e.g., IPCC AR5, GBO4) published since 2009 were identified by authors (see https://rkrug.github.io/IPBES_Global_Assessment_Chapter_4_LiteratureReview/). These review publications were screened to conduct a paper trail backward (cited papers) and forward (citing papers). Out of this, we added 29 papers which were not already referenced in our database (see https://rkrug.github.io/IPBES_Global_Assessment_Chapter_4_LiteratureReview/).

Lastly, we used authors' expertise to select additionally relevant papers that were missed by the previous steps. Particular attention was dedicated to cover three topics that were not explicit in the search patterns despite their importance to the chapter: urbanization as a driver of change, Indigenous Local Knowledge, and Good Quality of Life. This expert-judgement allowed to add another 152 papers, i.e. 11% of the total number of papers in the final literature database (see https://rkrug.github.io/IPBES_Global_Assessment_Chapter_4_LiteratureReview/).

Literature metadata analysis

In order to extract metadata from the selected papers, we developed an extensive survey form that needed to be completed for each paper. This form we set up as an excel spreadsheet with 83 variables to inform, and four categories of metadata: metadata on the article's scope and main findings, on the scenario(s) used in the article, on the model(s) used in the article, and on the indicator(s) and results(s) from the scenarios (Tables A1.1-A1.4). A comprehensive lists of pre-defined answers was specified, to facilitate extraction of the metadata, and to reduce misleading answers and errors in reporting. Wherever possible, answers to the questions were thus restricted to fixed dropdown lists
https://rkrug.github.io/IPBES_Global_Assessment_Chapter_4_LiteratureReview/).

This exercise resulted in 572 in detail reviewed papers out of the 827 identified papers.
Figures A1.1 to A1.11 provide some insight into the review process, and give some descriptive summaries of the paper. For example, the "missing" metadata informs on the current gaps in the scientific focus of the papers that would need to be filled in future research. In some cases, they can reflect the need to use standardized methods and terminology in the building and description of models, scenarios and indicators (Fig A1.7 to A1.10).

Table A4.1.1 - GENERAL_INFO

field_name	description_1	description_2
Rev	Reviewer	
bib_key	Mendeley "citation key"	
bib_comp	Complete citation as copied from Mendeley	
bib_type	Paper type	
foc_prim	FOCUS (Nature, NBP, GQL, Institutions)	Primary focus
foc_sec	FOCUS (Nature, NBP, GQL, Institutions)	Secondary focus
val_bdivm	Values of biodiversity	Valuation method
val_bdivt	Values of biodiversity	Type of values
ilk_mob	Was ILK mobilized and how?	
iplc_cons	IPLC and social groups considered	
iplc_ilk_addr	Does the paper address IPLC/ILKP issues?	
iplc_ilk_spat	Does the paper include spatial units related to IPLC/ILKP?	
Keyquestions	Key questions addressed	
Conclusions	Main conclusions/findings	
unc_exp	Uncertainty explored?	
key_paper	key paper?	
comment_general	Comments	

Table A4.1.2 - MODELS

	field_name	description_1	description_2
18	bib_key	Mendeley "citation key"	
19	mod_name	Model Name	
20	mod_key	DO NOT ENTER ANYTHING HERE	DO NOT ENTER ANYTHING HERE
21	imnat_mod_name	Model of Impact on NATURE	Model Name
22	imnat_type	Model of Impact on NATURE	Type
23	imnat_ncatcomp	Model of Impact on NATURE	Nature component
24	imnat_spex	Model of Impact on NATURE	Spatially explicit
25	imnbp_mod_name	Model of Impact on NBP	Model Name
26	imnbp_type	Model of Impact on NBP	Type
27	imnbp_component	Model of Impact on NBP	NBP Component
28	imnbp_spex	Model of Impact on NBP	Spatially explicit
29	imgql_mod_name	Model of Impact on GQL	Model Name
30	imgql_type2	Model of Impact on GQL	Type
31	imgql_component	Model of Impact on GQL	GQL Component
32	imgql_spex	Model of Impact on GQL	Spatially explicit
33	mod_link	Linkages between Models	
34	comment_model	Comments	

Table A4.1.3 - SCENARIOS

	field_name	description_1	description_2
35	bib_key	Mendeley "citation key"	
36	scen_name	Name Scenario	
37	scen_key	DO NOT ENTER ANYTHING HERE	DO NOT ENTER ANYTHING HERE
38	scen_ref	Reference to Scenario	
39	ipbes_uoa	IPBES Units of Analysis	
40	un_feedb	Feedbacks between Units of Analysis	

41	Sti	Participatory scenario/Stakeholders involved?	
42	Teleconn	Teleconnections/telecoupling represented?	
43	Sid	Scenarios Indirect Drivers	standard / archetype scenarios
44	sid_free	Scenarios Indirect Drivers	stylized / idealized scenarios
45	sdd_driver	Scenarios Direct Drivers	Driver/subdriver
46	sdd_landuse	Scenarios Direct Drivers	Scenario Land Use
47	sdd_cc	Scenarios Direct Drivers	Scenario Climate Change
48	sdd_natexp	Scenarios Direct Drivers	Scenario Natural Resource Extraction/Exploitation
49	sdd_ias	Scenarios Direct Drivers	Scenario Invasive Alien Species
50	sdd_pollution	Scenarios Direct Drivers	Scenario Pollution
51	aichi_target	Mention of CBD Aichi Targets?	Target
52	aichi_achiev	Mention of CBD Aichi Targets?	Achievement
53	sdg_target	Mention of SDGs?	Target
54	sdg_achiev	Mention of SDGs?	Achievement
55	comment_scenario	Comments	

Table A4.1.4 - INDICATORS_RESULTS

	field_name	description_1	description_2
56	bib_key	Mendeley "citation key"	
57	scen_key	Scenario Name	
58	mod_key	Model Name	
59	si_coverage	Spatial info	Coverage
60	si_region	Spatial info	Marine or Terrestrial Region
61	si_country	Spatial info	Country
62	si_spres	Spatial info	Spatial resolution
63	ti_nts	Time info	n time slices
64	ti_yts	Time info	years time slices
65	ti_steps	Time info	Time steps
66	nat_comp	Nature	Nature component
67	nat_ipbes_ind	Nature	IPBES Indicator
68	nat_indicator	Nature	Other Indicators type
69	nat_add	Nature	Other Indicators details
70	nat_trend	Nature	Trend
71	nat_trend_var	Nature	Spatial and Temporal Variability
72	nat_threshold	Nature	Thresholds
73	nbp_type	NBP/Ecosystem Services	Type of NBP/ecosystem service
74	nbp_indicator	NBP/Ecosystem Services	Indicator
75	nbp_trend	NBP/Ecosystem Services	Trend
76	nbp_trend_var	NBP/Ecosystem Services	Spatial and Temporal Variability
77	nbp_threshold	NBP/Ecosystem Services	Thresholds
78	gql_indicator	GQL/Human well-being	Indicator
79	gql_trend	GQL/Human well-being	Trend
80	gql_trend_var	GQL/Human well-being	Spatial and Temporal Variability
81	gql_threshold	GQL/Human well-being	Thresholds
82	unc_quant	Uncertainty for this Indicator Quantified	
83	comment_ind_res	Comments	

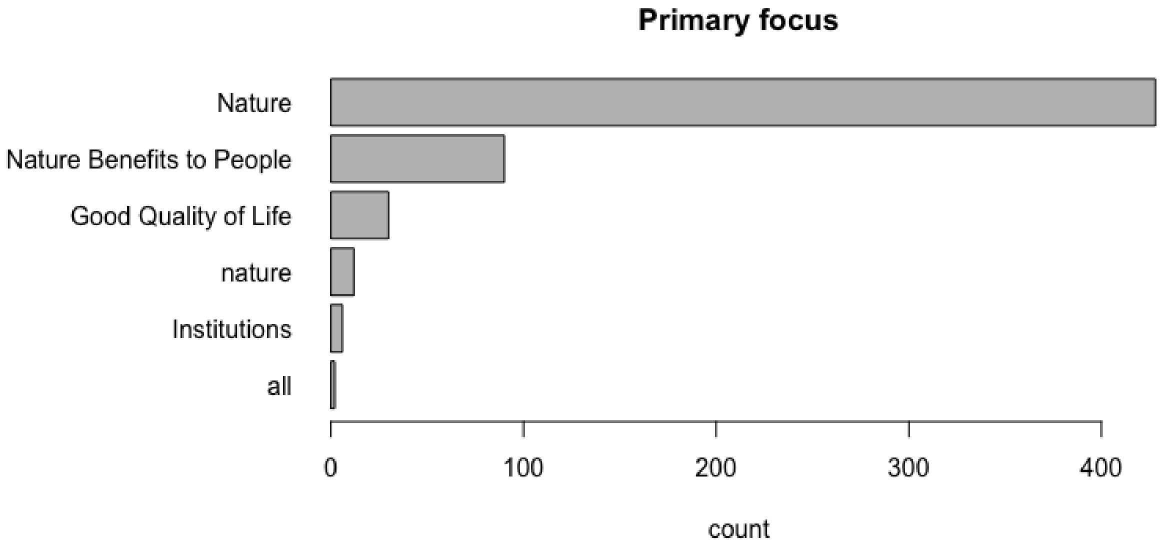


Fig A4.1.1 – Count of primary focus of the papers

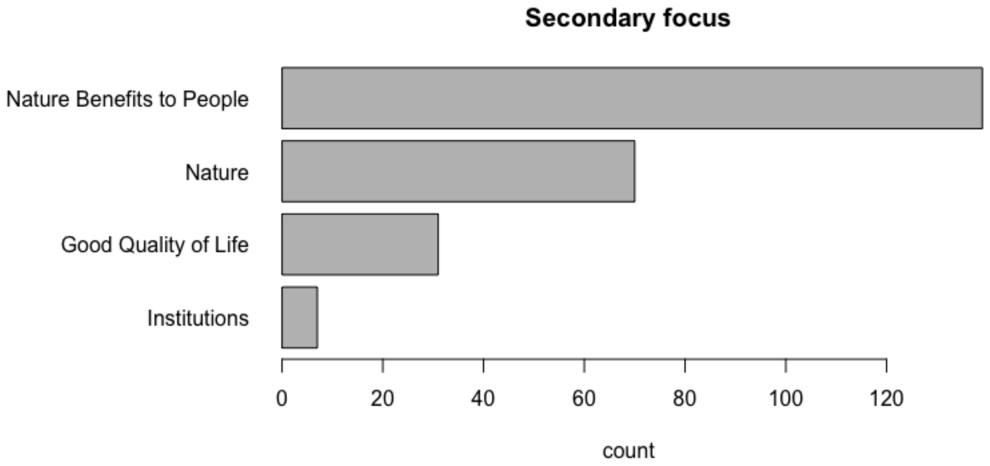


Fig A4.1.2 – Count of secondary focus of the papers

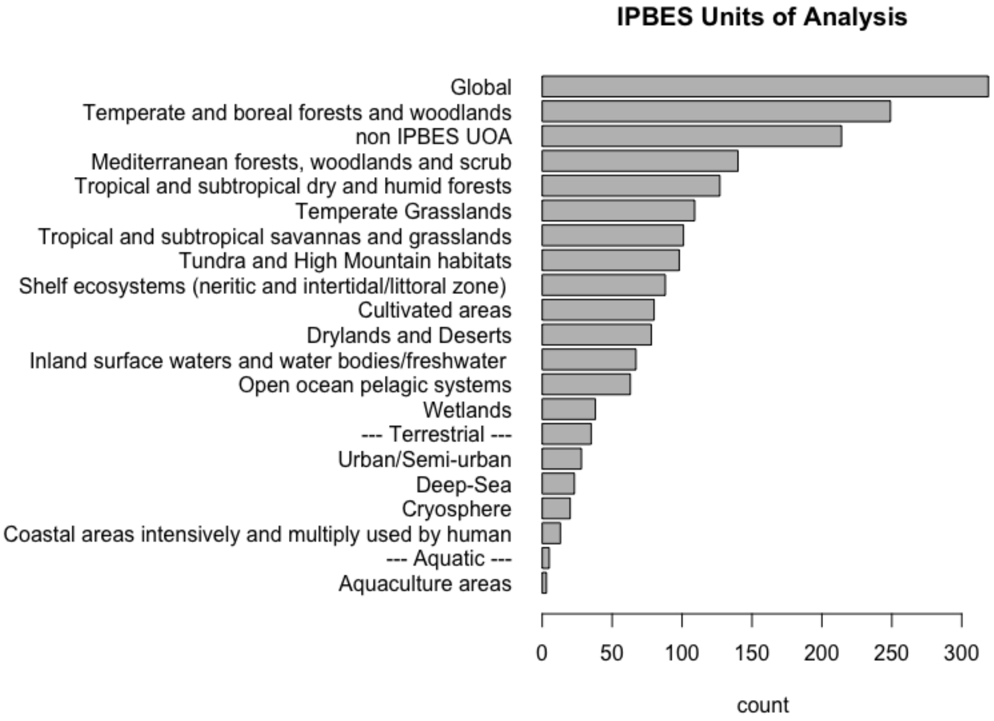


Fig A4.1.3 – Count of IPBES Units of Analysis studied in the papers

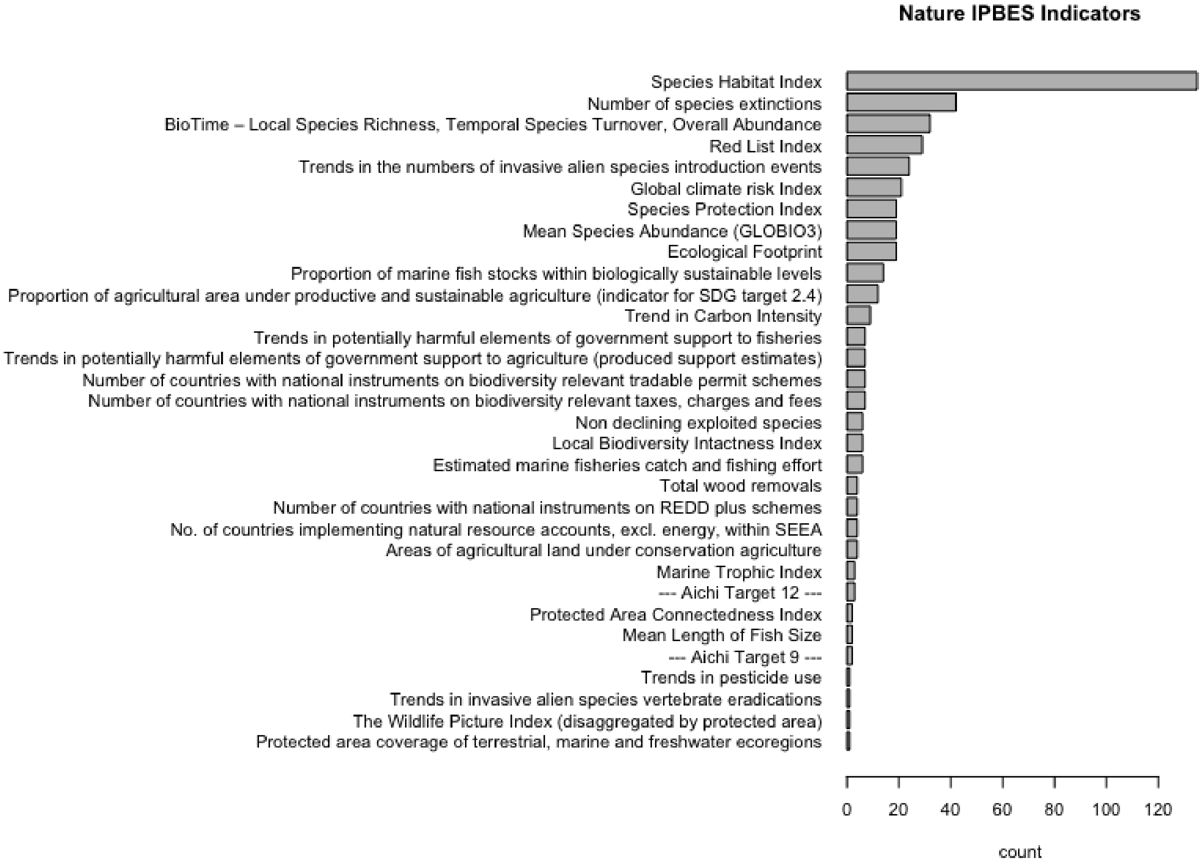


Fig A4.1.4 – Count of Nature IPBES Indicators used in the papers

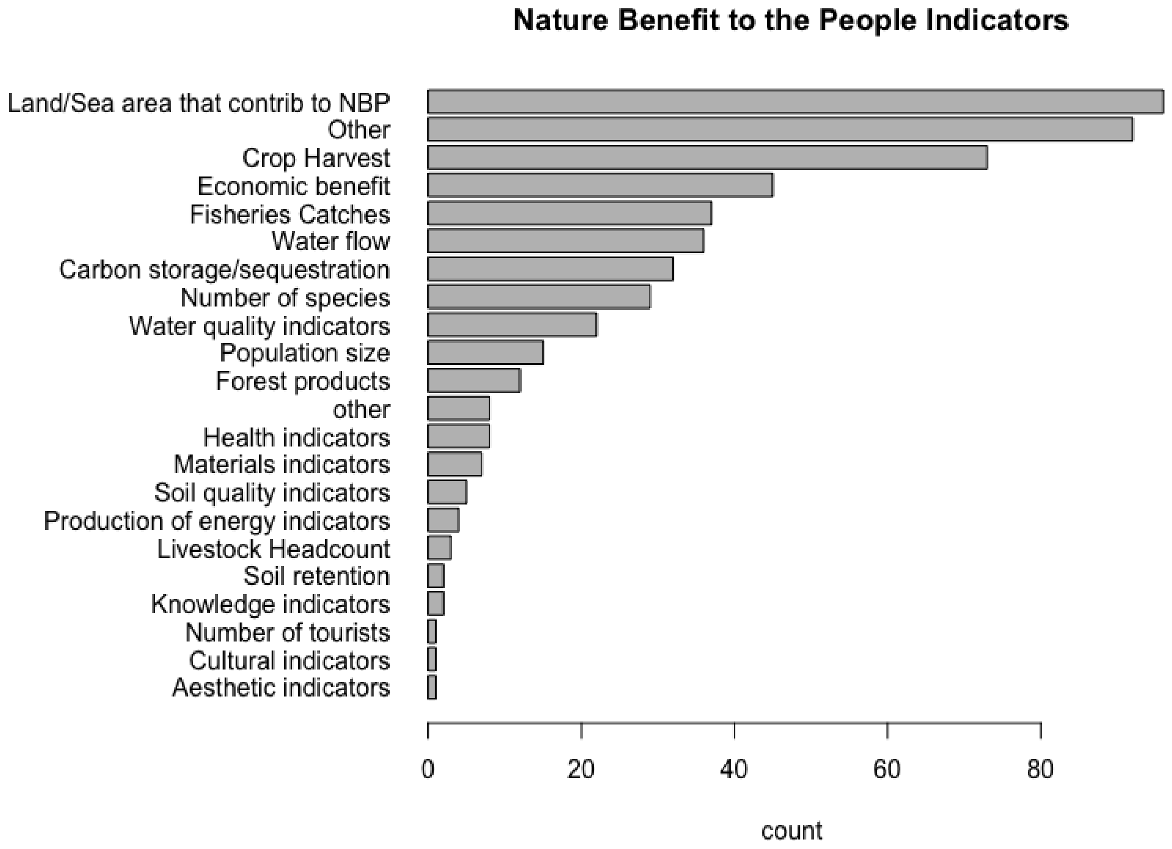


Fig A4.1.5 – Count of Nature's Contributions to People Indicators used in the papers

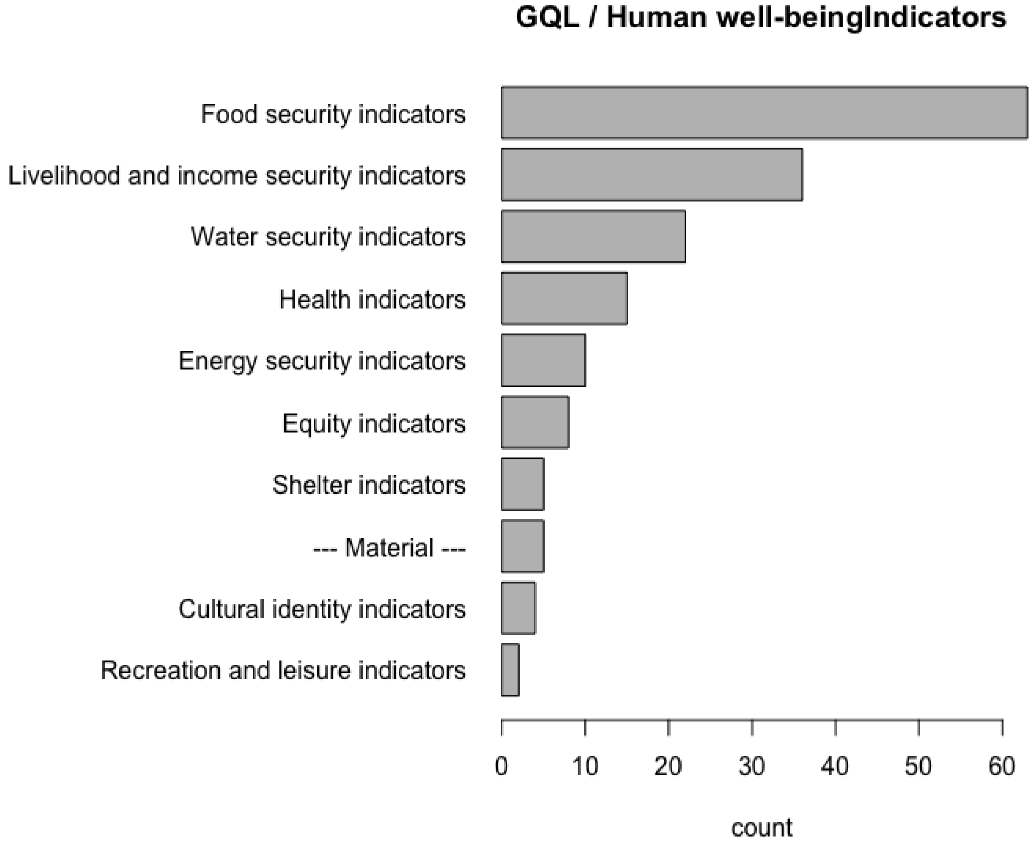


Fig A4.1.6 – Count of GQL / Human well-being indicators used in the papers.

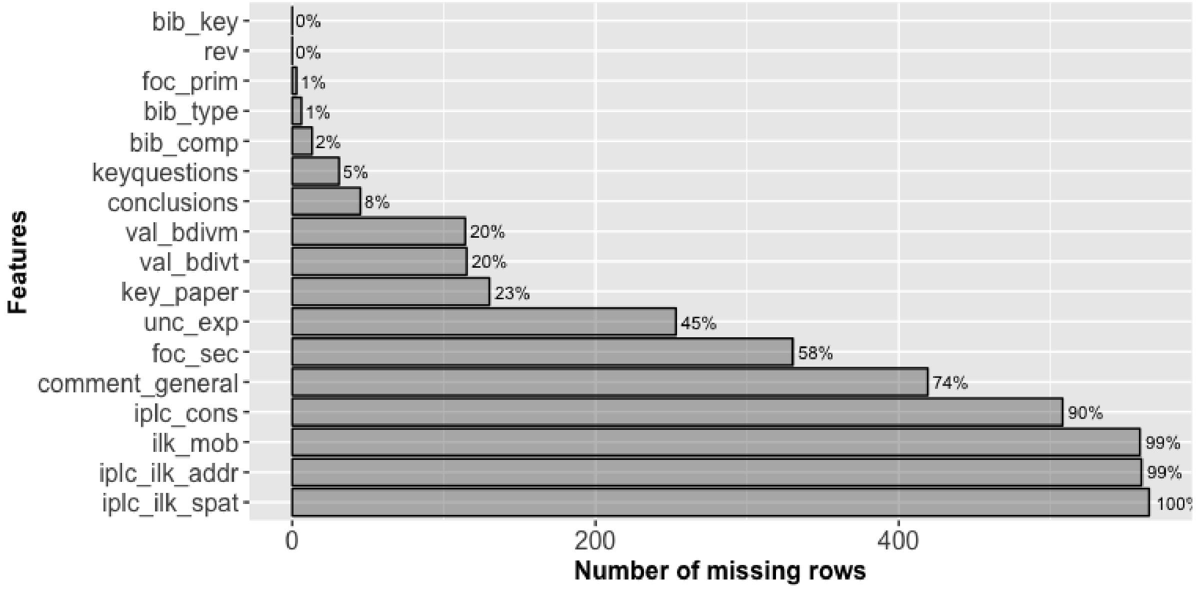


Fig A4.1.7 - missing data in GENERAL_INFO table of literature database

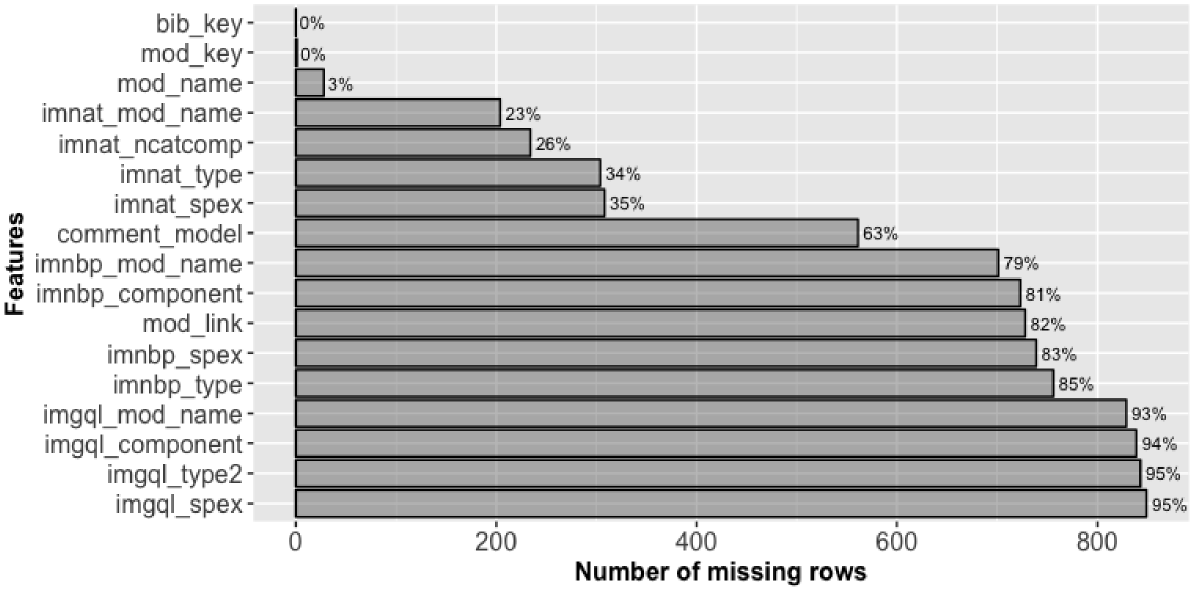


Fig A4.1.8 - missing data in MODELS table of literature database

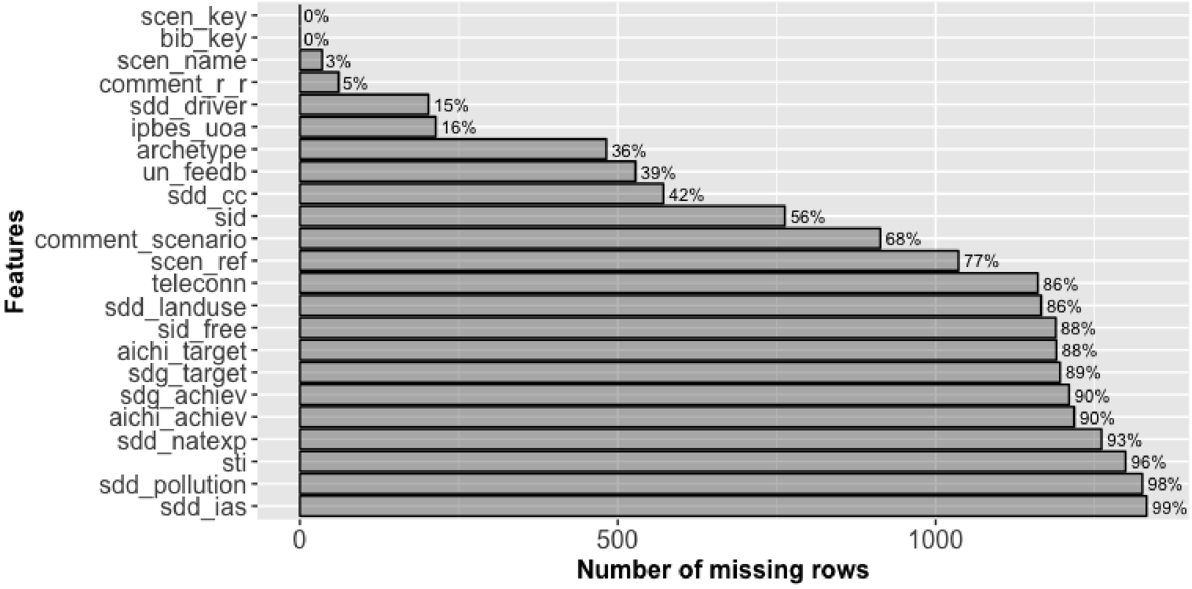


Fig A4.1.9 - missing data in SCENARIOS table of literature database

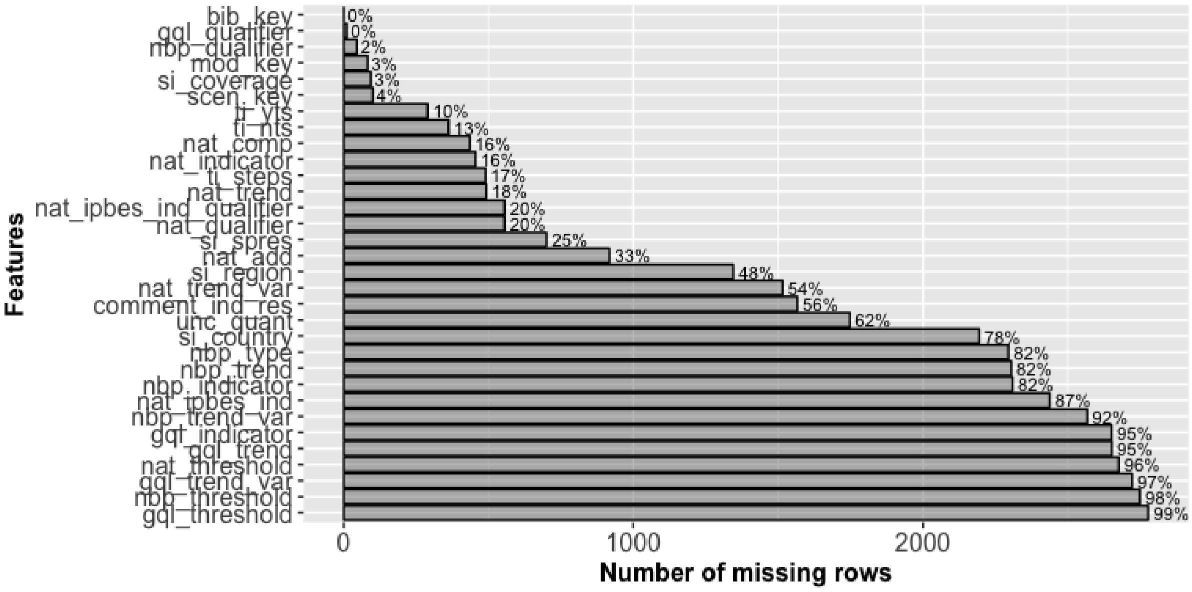


Fig A4.1.10 - missing data in INDICATORS_RESULTS table of literature database

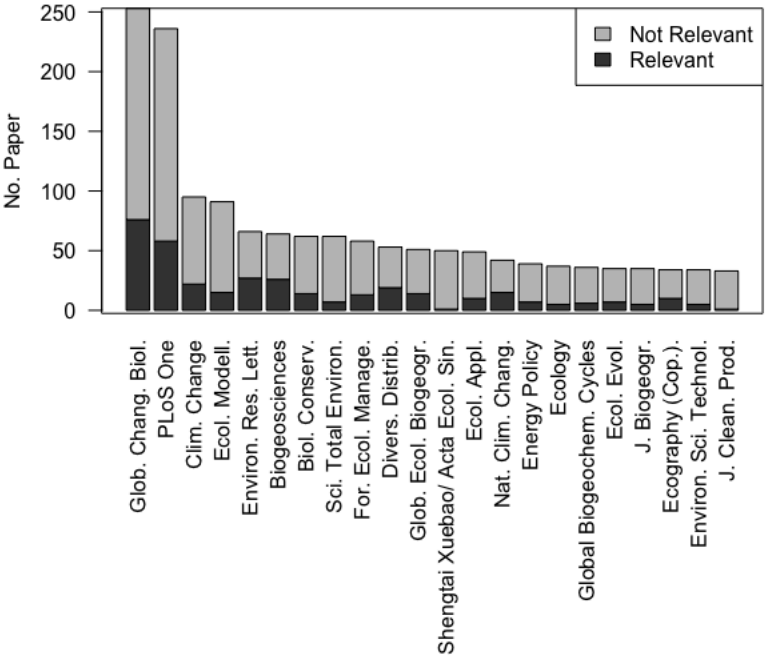


Fig A4.1.11 - Literature by Journal. Dark bars indicate relevant articles in the systematic literature review, light bars indicate not relevant papers.

A4.1.2 – Extended figures and tables to section 1

Table A4.1.5: projections of global population for each archetype (based on SRES, 2000, Raskin, 2005, (MA 2005; OECD 2012; Riahi et al. 2017; UNEP 2007)

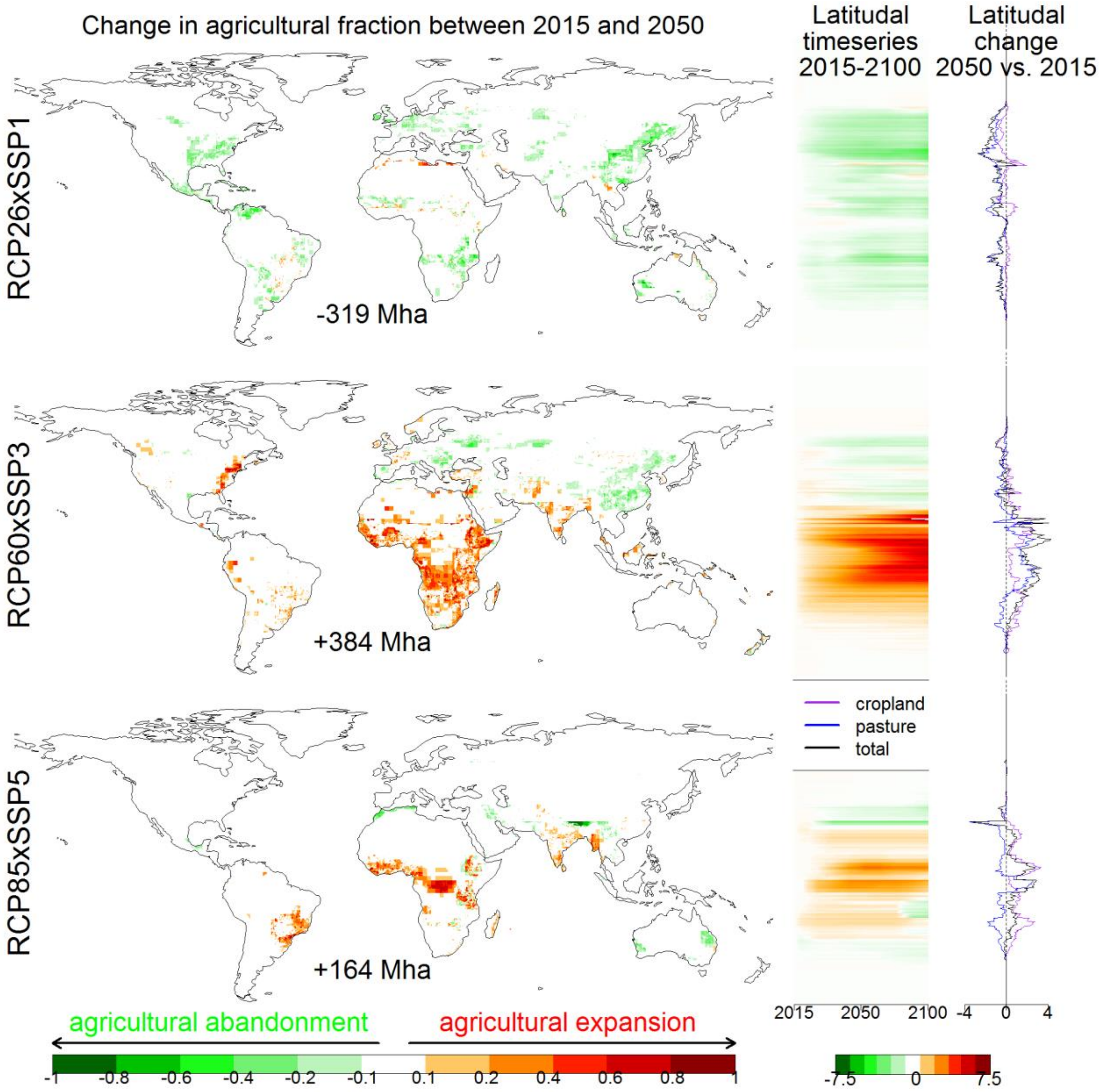
	Global Population (billion)					
	Economic Optimism	Reformed markets	Global Sustainable Development	Regional Sustainability	Regional Competition	Business as Usual
2000	6.1					
2050	8.7-9.3	8.1-8.7	7.9-9.1	9.3-9.6	9.5-11.3	7.6-10.1
2100	7.1-9.3	6.8-8.5	7.0-8.5	9.8-10.4	10.2-15.1	9.2-10.4

Table A4.1.6: Human Development Index (HDI) and Inequality Adjusted HDI, 2015 (source: (United Nations Development Prgoramme 2016)

Region	**HDI-2015**	**Inequality Adjusted HDI, 2015 (IA-HDI)**	**IA-HDI/HDI (change in %)**
Arab States	0.687	0.498	-27.5
East Asia and the Pacific	0.720	0.581	-19.3
Europe and Central Asia	0.756	0.660	-12.7
Latin America and the Caribbean	0.751	0.575	-23.4
South Asia	0.621	0.449	-27.7
Sub-Saharan Africa	0.523	0.355	-32.2
OECD	0.887	0.776	-12.6
World	**0.717**	**0.557**	**-22.3**

The Human Development Index (HDI) is a statistic constructed by combining a range of indicators thought to capture human potential and development: per capita income, education, and life expectancy. The inequality-adjusted HDI (IA-HDI) statistically adjusts the HDI to account for income inequality. The ratio of *Inequality Adjusted Human Development Index (IA-HDI) / Human Development Index (HDI)* shows that inequality is constraining the various regions from realizing their human development potential. When the world UNDP Human Development Index (HDI) is adjusted for inequality, it is 22.3% lower. Such indices are neither currently calculated in integrated assessment modelling efforts nor represented in scenarios.

Figure A4.1.12 (extended Figure 4.1.3): Projected changes in cropland and pasture area are used in the Biodiversity and Ecosystem Services Model Intercomparison Project. Note that the depicted trends are examples only and do not necessarily signify coherence in terms of direction of change in all archetypes.



Appendix 4.2 - Supporting materials to section 2

A4.2.1 The main interrelations and feedbacks between hierarchical levels that are important for biodiversity future (extended materials, Box 4.2.1)

Appendix 2 contains explanations and examples of projections at different biodiversity levels and the main feedbacks between them including the impacts of lower levels on upper levels and vice versa. The indexes of the feedbacksfigure in Chapter section 2, Box 2.1 are indicated in bold italic.

INTRAPOPULATION and INTRASPECIFIC DIVERSITY

Expected changes

Direct drivers impact on intraspecific diversity by altering the selection pressure, often resulting in loss of genetic diversity. Examples are: climate change (selection of individuals who are the best in new climatic conditions); pollution (selection of individuals who are resistant to pollution); exploitation (selective removal of certain phenotypes and genotypes, e.g., trophy hunting, large fish catch, exploitation of particular seasonal/ecological forms or local populations while others are not exploited); land use (selection of individuals who are tolerant to human presence and human-made transformations of ecosystems); disturbance (selection during floods, droughts, fires, etc.).

Very strong selection under environmental stress can lead to a genetic diversity loss if only a few individuals survive, or some local populations disappear (***GD3***). Selective harvesting can also significantly decrease the genetic diversity of exploited species. For example, so-called fisheries-induced evolution moves exploited populations toward earlier reproduction, slower growth, and increased reproductive effort. It can make fish populations more robust to overexploitation, but it can also reduce their resilience to natural fluctuations (Heino, Díaz Pauli, and Dieckmann 2015). Selective effects induced by toxic substances can decrease genetic diversity and have negative impacts on the long-term probability of survival of populations (Coutellec and Barata 2011; Rusconi et al. 2018).

The indirect causes are the changes at other biodiversity levels: reduction in species population size (***S4***), shift, reduction and fragmentation of species ranges (***S5***), changes in interspecific relations (***SD3***) and change in landscape heterogeneity (***ED3***). The expected loss of genetic diversity should be recognized as a serious threat to saving biodiversity in the future because of the importance of intrapopulation and intraspecific diversity for species survival in the changed world.

Impacts on upper levels

GD1 - Adaptation of populations to new conditions through standing genetic and phenotypic variations

Intrapopulation and intraspecific diversity underpins population fitness, stability and functioning (Cardinale, et al., 2012; Fischer et al. 2018, background A5; Tilman et al. 2014) as well as the ability to adapt and evolve in a changing environment (Hoffmann & Sgrò, 2011; Jump, Marchant, & Peñuelas, 2009; Neaves et al., 2015; Pauls et al., 2013). Variation in physiological, phenological, behavioral or morphological traits can allow species to cope with rapid climatic changes within their range and future climate changes may be met in many cases by existing genetic variations (Alfaro et al. 2014; Barrett and Schluter 2008). Genetic diversity provides material for adaptive microevolution. Thus, the loss of genetic diversity will severely limit the species' ability to adapt and evolve in a changing environment (Hoffmann & Sgrò, 2011; Jump, Marchant, & Peñuelas, 2009; Neaves et al., 2015; Pauls et al., 2013).

Incorporating intraspecific variations in projections of species' ranges has been shown to decrease predicted shifts in species distribution in response to climate change, for instance for a number of conifer and deciduous tree species in North America (Morin & Thuiller, 2009; Morin, Viner, & Chuine, 2008; Oney et al., 2013) and in the Iberian Peninsula (Benito Garzón et al., 2011). Projections that do not consider intraspecific diversity, including cryptic genetic diversity, can drastically underestimate the negative effects of global changes on biodiversity (Pauls et al. 2013). For example, the predicted losses of genetic diversity in nine montane aquatic insect species in Europe greatly exceed those at the scale of morphospecies (Bálint et al. 2011).

Metapopulation (individual populations constituting a species) considered as one of the aspects of intraspecific diversity is an important factor of species fitness, adaptability and capacity to keep up with global changes. For example, incorporation of metapopulation dynamics into a model of climate-driven changes in distribution of European mountain hare in Britain caused faster expansion on the northern range margin (leading edge) and slower contraction on the southern margin (trailing edge) (Anderson et al. 2009).

GD2 – Adaptation of populations due to phenotypic plasticity

Phenotypic plasticity allows a rapid (within the individual's lifetime) behavioral, physiological or morphological adjustment of populations to novel conditions whereas evolutionary responses require at least several generations (reviewed by Neaves et al., 2015; Pauls et al., 2013). Theoretical models show that phenotypic plasticity may promote population stability more than genetic variations (Kovach-Orr and Fussmann 2013). Incorporating phenotypic plasticity in models reduce species extinction risk as shown for trees (Benito Garzón et al., 2011; Morin & Thuiller, 2009; Morin et al., 2008; Valladares et al., 2014), but if the models assume uniformly high plasticity, simulations may underestimate the loss of species habitats (Valladares et al. 2014).

Phenotypic plasticity alone will not be sufficient when reaction norms will no longer be adaptive and microevolution on the reaction norm is needed (Gienapp et al., 2012; Gienapp et al., 2014; Visser, 2008). Moreover, phenotypic plasticity may have negative consequences for species survival in the long term because it can weaken selection pressure, and thus, slow down evolutionary adaptation (L.E. Neaves et al. 2015; Pauls et al. 2013). Evolution of phenotypic plasticity is projected as a possible response to global changes (Hoffmann et al. 2011; Pauls et al. 2013). High phenotypic plasticity will be selectively advantaged in the face of increasing climatic instability (Canale and Henry 2010), especially in regions where an increase in temporal variability is predicted.

GD3 – Adaptive evolution, "evolutionary rescue" of populations and species

The negative impacts of global changes on biodiversity may be ameliorated by evolutionary adaptation. Rapid adaptive evolution can ensure population survival in situ, what is often called "evolutionary rescue" (Gonzalez et al., 2013; Hoffmann et al., 2011; Neaves et al., 2015). Models that ignore microevolutionary processes can overestimate species extinction risk (Hof et al., 2011; Skelly et al., 2007).

The crucial question is whether species will be able to adapt fast enough. The ability of many species for rapid microevolution at the same time scale as ecological processes (Hoffmann et al., 2011; Lavergne et al., 2010; Neaves et al., 2015; Salamin et al., 2010) supports the view that rapid adaptation can occur in the face of global change. However, it is difficult to separate the results of evolutionary adaptation from phenotypic platicity in real populations (Boutin and Lane 2014; Charmantier and Gienapp 2014; Crozier and Hutchings 2014; Franks et al. 2014; Merilä and Hendry 2014; Urban, Richardson, and Freidenfelds 2014).

Some models indicate evolutionary responses that may be too slow to rescue species, for instance, for some European populations of Paris major (great tit) (Gienapp et al., 2012) and Fagus sylvatica (beech) (Jump et al., 2006). Species with low capacity to evolve and high risk of extinction are likely to be those with a small population size, low reproductive output and long generation times, a low level of genetic variations of adaptive traits, and/or low dispersal ability (Lavergne et al. 2010; L.E. Neaves et al. 2015; Salamin et al. 2010). In reality, some of these "risky" traits can be compensated. Trees, for example, are long-living organisms in relation to the speed of current global changes. However, high levels of genetic diversity and large effective population sizes of many tree species allow rapid microevolution and enhance their chances of adapting within a few generations (Petit and Hampe 2006). Spontaneous evolutionary rescue is unlikely to save small, genetically depauperate populations of organisms with long generation times, facing many simultaneous threats, as is typically the case for endangered species (Vander Wal et al., 2013).

An important feedback occurs between the rate of the evolutionary response and genetic diversity. Strong selection pressure under environmental stress will tend to reduce genetic diversity and may promote extirpation of local populations, reducing the possibility to react to future selective challenges (reviewed in Pauls et al., 2013). For example, simulations of Puerto Rican lowland lizards predict the extinction of genetic lineages that are intolerant of hot temperatures and their replacement with warm-adapted competitors (Huey et al. 2009). Another example concerns the Nothofagus pumilio tree in the montane forests of Patagonia which is expected to lose high-elevation genotypes during climate warming (Mathiasen and Premoli 2016). However, increasing spatial climate variability can also cause an increase in genetic diversity (reviewed in Neaves et al., 2015; Pauls et al., 2013) as expected, for example, for Quercus and Eucalyptus genera of trees (Kremer, Potts, and Delzon 2014) and lynx (Row et al. 2014).

All species have limits to their capacity for adaptive response to changing environments (Bell 2013; Gonzalez et al. 2013; Lavergne et al. 2010). It is important that hard physiological boundaries constrain the evolution of terrestrial organisms' tolerance to high temperatures. Analysis of thermal tolerance of hundreds of terrestrial ectotherm, endotherm and plant species showed that tolerance to heat is largely conserved across taxa, while tolerance to cold varies between and within species (Araújo et al. 2013).

Preservation of long-term evolutionary potential of biodiversity

The vital importance of preservation of existing biodiversity should not overshadow the necessity to maintain the ecological and evolutionary processes that can generate biodiversity in the future (Mace and Purvis 2008). This goal can be achieved through preservation of the evolutionary heritage of the Tree of Life that is reflected in phylogenetic diversity (Faith, 2015; Faith et al., 2010; Mace & Purvis, 2008) which is especially important because of the expected disproportionately large loss of evolutionary history in the face of global changes (Mace & Purvis, 2008; also see section *SPECIES,* below). The options for future adaptation and diversification can be maintained through preservation of phylogenetic diversity, species evolutionary distinctiveness (Faith et al., 2010; Forest et al., 2007; Mace & Purvis, 2008; Winter, Devictor, & Schweiger, 2013), areas with high speciation rates i.e. 'sources' of diversity (Condamine, Rolland, and Morlon 2013) evolutionary refugia and connectedness in landscapes (Sgrò, Lowe, and Hoffmann 2011). Furthermore, preservation of evolutionary heritage provides both present and future benefits (option values) to humanity, often in unanticipated ways (Faith, 2015; Faith et al., 2010; Faith & Richards, 2012; Forest et al., 2007).

INDIVIDUAL SPECIES

Expected changes

The impact of direct drivers (harvesting; climate change, pollution, and disturbance) on individual species lies in alteration of their physiology, behavior, functioning and population size. Other impacts come from other biodiversity levels as the results of changes in habitat characteristics (***E3, ED2***), species interactions (***SD4, SD5***), genetic diversity and species' adaptability (***GD1, GD2, GD3***). Harvesting leads to a reduction in species' population size, disturbance of population structure and extinction due to overexploitation. Overexploitation (logging, hunting, fishing, and plant gathering) along with habitat loss, is the most important negative impact on threatened and near-threatened species from the IUCN Red List (Maxwell et al., 2016). For both marine and freshwater species, overexploitation will remain a major threat in the 21-st century (Leadley et al., 2013). Climate change and pollution affect species through physiological tolerance of organisms. For the terrestrial species, temperature and humidity are critical, for marine and freshwater species, temperature, acidification and hypoxia. The expected increase of pollution can directly depress species through toxicity (Leadley et al., 2013).

Species can react to environmental changes either by adapting to new conditions in situ through standing phenotypic and genetic variations (***GD1***), phenotypic plasticity (***GD2***) and rapid adaptive evolution (***GD3***) or they can avoid new conditions by shifting their distribution. Widespread species with large populations and high fecundity have higher chances to persist and adapt in situ, whereas species with small populations and low fecundity should survive through migration (Aitken et al., 2008). Local adaptation should be expected to be a more important response than migration in highly fragmented habitats including islands (Kubisch et al., 2013). In reality, however, adaptive and spatial responses are components of the same general response (Huntley et al. 2006).

Species with narrow physiological tolerance are less likely to survive in a changing environment. Tropical species are generally more sensitive to warming because they live close to physiological temperature tolerance limits. In contrast, species at higher latitudes have broader thermal tolerance and live in climates that are currently cooler than their physiological optima, so that warming may even enhance their fitness (Deutsch et al. 2008). Thus, climatic changes could have positive effects on species in cases when a new climate will provide more resources for species, as it is expected for many populations of Pinus, Larix, Picea, Pseudostsuga genera located in cold climates in North America and Siberia (Alfaro et al. 2014), and also for many insect species.

Range shifts of terrestrial species may occur in latitudes and elevations. Marine species may respond by shifting their latitudinal and depth ranges. Species ranges are predicted to shift hundreds of km during the 21-st century (Barbet-Massin, Thuiller, and Jiguet 2012; Cheung et al. 2009; Huntley et al., 2006, 2008; Leadley et al. 2010). For European breeding birds, an average overlap between future and present ranges of 31–53%, which means that many endemic species will have no overlap between their present and future ranges (Huntley et al. 2008).

The future of biodiversity depends on species' capacity to shift their range fast enough to keep pace with climate change. In one scenario, the mean global climate velocity (the rate of movement of the climate across a landscape) during the 21st century is expected to be 0.42 km per year along the land surface (A1B scenario). The highest velocities (more than 1 km/yr) are expected in flooded grasslands, mangroves and deserts and the lowest (less than 0.1 km/yr) in mountainous biomes and temperate coniferous forest (Loarie et al. 2009). 28.8% of land areas may undergo velocities faster than 1 km/yr. In flat areas, climate velocity can exceed 8 km/yr for the highest rates of projected climate change (RCP8.5) (Settele et al. 2014). Climate velocity is low in mountainous areas because the change in temperature over short distances is large and an altitudinal range shift of 10 m is expected to translate into a 10-km latitudinal shift (Jump, Mátyás, and Peñuelas 2009). Species occupying extensive flat landscapes are particularly vulnerable because they must disperse over longer distances than species in mountainous regions (Settele et al. 2014). However, species that already occur near mountain-tops are among the most threatened by climate change because they cannot move upwards, e.g. for European mountain plants, projected habitat loss is greater for species distributed at higher elevations (Engler et al. 2011). The most vulnerable are the tropical species because climate in the tropics is predicted to leave the range of relatively small historical variability most quickly, despite the fact, that absolute changes in climatic parameters will be the greatest in the north (Mora et al. 2013). Moreover, in the tropics, latitudinal temperature gradients are largely absent and the distances to potential cool refuges are maximal (Colwell et al., 2008; Wright et al., 2009). Polar species will be also at high risk because in high latitudes (above 50◦N) these distances are large due to a great projected increase in temperature (Wright et al. 2009).

The observed average rate of range shifts of terrestrial species is 17 km per decade (Chen et al., 2011), which is enough to keep pace with moderate climate change. Obviously, not all species will be able to do this because of very different dispersal capacity. The average velocity of range expansion for plants and soil animals is less than 0,1 km/yr while for some groups of aboveground animals it is several km per year (Berg et al. 2010). Species with low dispersal capacity (many plants, freshwater mollusks, amphibians, reptiles, some birds and mammals) will not be able to keep up with climate change, while others (migratory birds, large mammals, some butterflies) have a chance of success (Settele et al. 2014). Models on the base of the trait-space-demographic approach predict that around 30% of terrestrial mammal species have potential spread rates slower than the global mean velocity of climate change (Santini et al. 2016).

Required migration rates for exploited marine fish and invertebrates are estimated in the range of 15.5 to 59 km per decade depending on a climate scenario (Cheung et al. 2011; Cheung et al. 2009; Jones and Cheung, 2015), which is comparable with those for flatland areas. Marine range shifts were observed at an average rate of 19 km/yr (Sorte, Williams, and Carlton 2010) to 72 km per decade (Burrows et al. 2014; Poloczanska et al. 2013) which is several times faster than for terrestrial species. Thus, marine species may have better chances to keep up with climate than terrestrial species, although marine species with low dispersal capacity will not have enough time. For example, recent distribution shifts of some fish and benthic invertebrates did not keep up with the isotherm movement (Poloczanska et al. 2013). Under the low emissions climate scenario, the projected rate of range shift for demersal fishes is within the range of observed rates, however, under a high emissions scenario, the projected range shift is two times higher than the observed rates (Cheung et al., 2009).

Natural geographical barriers such as coasts, mountains and deserts will limit future species shifts. Poleward shift could result in high risk for Arctic species because future range contractions from the south without significant landmass in the north could lead to their extinction, called “Arctic squeeze” (Gilg et al. 2012). Globally, 12.0% of projected spatial trajectories for climatic niches on land and 5.4% of ocean trajectories terminate in “climate sinks” (i.e. areas where climate conditions locally disappear and further migration is impossible): 6.1% of land trajectories and 5.0% of ocean trajectories end in “coastal sinks”, 5.9% and 0.4% in “internal sinks” (Burrows et al. 2014).

Along with species dispersal capacity and natural geographical barriers, future range shifts will depend on changes in species interrelations (***SD5***) and human-driven habitat loss and fragmentation (***E3***).

During the 21st century, ranges of many terrestrial (Colombo and Joly 2010; Dullinger et al. 2012; Engler et al. 2011; Huntley et al. 2008; Jetz, Wilcove, and Dobson 2007; Loyola et al. 2012; McKenney et al. 2007; Meller et al. 2015; Rondinini and Visconti 2015; Warren et al. 2013) and freshwater (Markovic et al. 2014) plants and animals are projected to contract by tens of percent as a result of changes in land use and climate However, species that keep pace with climate change or species for which climate change may be favorable, can maintain or increase their range size (Settele et al. 2014).

The joint impact of different drivers can facilitate or inhibit range alterations. For example, the climate-driven reduction in suitable habitats of red spruce in the Great Smoky Mountains National Park (USA) will be amplified by air pollution increases. Higher temperatures will cause losses of low-elevation habitats, while air pollution producing acid rain will cause loss of both low- and high-elevation habitats (Koo, Patten, and Madden 2015). Otherwise, climate-driven range shifts in mountains can be impacted by further habitat modification following natural disturbances (wind throws, fires, and avalanches) and human-induced disturbances such as recreation and land use (Lenoir et al. 2010). Unexpected range shifts in east-west directions or towards tropical latitudes and lower elevations, linked to complex abiotic and biotic interactions, have already been reported (Lenoir and Svenning 2015).

Species that fail in adaptation and migration will face the risk of extinction. Simulation of species’ extinction risks showed that the most important variables are species' range and population size that are already used in species conservation assessments (Pearson et al. 2014). However, some assessments showed that species highly and negatively exposed to future environmental changes are currently less threatened in terms of the current IUCN conservation status (Triviño et al. 2013) and areas of high concentration of species which are sensitive and vulnerable to climate change differ from those of high concentration of currently threatened species (Foden et al., 2013).

Species extinctions might occur not immediately but after substantial delays called “extinction debt” which occur when species decline to the point that they are committed to extinction, but not yet extinct (Hylander and Ehrlén 2013; Jackson and Sax 2010; Kuussaari et al. 2009). For example, models of 24 frog species in the Australian wet tropics poject that 4 of these species will face imminent extinction due to climate change by 2080 and a furterh 3 species will face delayed extinctions that will take at least a century to be realized (Fordham et al. 2016). The extinction lag times for forest plants with low rates of population turnover can be more than a century (Vellend et al. 2017) and for woody plants can be many centuries (Cronk et al. 2016). About 40% of the European high-mountain plants are predicted to be still occupying areas which will already become climatically unsuitable for them by 2100 (Dullinger et al. 2012). On the one hand, extinction debt means that long-term effects of global change can be more severe than observed now (Dullinger et al., 2012; Fordham et al., 2016; Hylander & Ehrlén, 2013; Urban, 2015) and even if we halt negative global changes today, transient eco-evolutionary dynamics would ensure centuries of further biodiversity alterations (Norberg et al., 2012). On the other hand, it gives a window of opportunity for species conservation.

Impacts on upper levels

S1 – Changes in local species composition due to alteration of species range (shift, change in area, fragmentation)

Expected species range shifts will lead to local extinctions of native species and arrivals of climatic migrants. As a result, large species turnover both in marine and terrestrial ecosystems is projected (see section *SPECIES DIVERSITY*). The impacts of species movements leading to disruption of native communities can be largely negative and of the same or greater magnitude as invasions of alien species (Sorte et al. 2010). A particular threat will represent projected changes in the distribution of pests, pathogens and disease vectors as, for instance, is predicted for climate-driven changes in the dengue mosquito Aedes aegypti in Australia (Kearney et al., 2009), avian malaria in Hawaii (Liao et al. 2015) and global shift of infectious amphibian chytrid fungus Batrachochytrium dendrobatidis into higher latitudes and altitudes (Xie, Olson, and Blaustein 2016). The expected expansion of the last threat for amphibians is reflected in the Australian national biodiversity protection plan (https://soe.environment.gov.au/theme/biodiversity/topic/2016/pest-species-and-pathogens)

S2 – Changes in local species composition due to local species extinctions and alteration of species abundance and functioning (including changes in phenology)

Disproportionate harvesting reduces primarily populations of top predators and large-sized organisms. The most serious impact is projected for marine ecosystems where continuation of unsustainable fishing will lead to shifts to alternative ecosystem states because of the loss of keystone species and top predators and a decrease in the marine trophic index (Leadley et al., 2013).

Different responses of species on climate change will lead to mismatches in phenology and disruption of spatial association between species (Berg et al. 2010). Alteration of species phenology may result in temporal mismatch in trophic interactions and in mutualistic interactions including plant-pollinator relations. Disturbance of plant-soil microorganism and soil microbe-microbe relationships may also lead to large impacts (Classen et al. 2015). Species in higher trophic positions can be more sensitive to changing temperatures and thus, climate change can cause a decline in carnivore abundance or exacerbate predation with further trophic cascades (Zarnetske et al., 2012). The replacement of cold-resistant and ice-associated species with more warm tolerant species will rearrange the Arctic and alpine biocenoses and food webs. In the Arctic, changes in plankton abundance can propagate to higher levels of the marine food web and even to terrestrial ecosystems through birds and mammals linking marine and terrestrial ecosystems (Stempniewicz et al., 2007). Climate-driven alteration of competitive relationships can cause changes in communities. For example, extirpation of arctic char and brown trout and extension of pike distribution are predicted in Swedish lakes under projected increase in temperature (Hein et al., 2013; 2012). In boreal forests, climate-driven increase in fire frequency may depress fire-sensitive species, for instance, dispersal ability decrease of evergreen conifers will increase the prevalence of deciduous hardwoods (Tautenhahn et al. 2016). Climate change may depress species' resistance to pests and diseases and lead to outbreaks with cascading effects along food chains (Alfaro et al. 2014; Edeline et al. 2016).

Pollution, eutrophication and acidification influence species performance, change community structure and generally decrease species richness. Oligotrophic species can be outcompeted by more nitrogen/phosphorus-loving or acid-tolerant species. In eutrophic habitats, invasive species will probably increase in abundance and richness at the cost of native species (Leadley et al., 2013).

Size-selective impacts of different direct drivers (e.g. temperature, pathogens, harvesting) may change the architecture of food webs and alter the whole ecosystem sensitivity to climate variation (Edeline et al. 2016). Reduction in body size is predicted for marine and freshwater taxa as a result of expected increase in water temperature (Cheung et al., 2013; Lurgi, López, & Montoya, 2012), although the adaptive value of such changes has little evidence (Teplitsky and Millien 2014).

S3 – Changes in ecosystem structure and functioning due to changes in key species abundance and functioning

When alterations of range and abundance occur in key or habitat-forming species, they can have pervasive effects that propagate through entire communities (Pecl et al. 2017; Zarnetske et al. 2012). For example, in marine ecosystems, the climate-driven loss of dominant habitat-forming species can result in community phase shifts. Decrease in tropical fish herbivory leads to shifting from coral to macroalgal dominance. Increase in temperate urchin grazing leads to a replacement of algal forests to 'barren' ecosystems. Such tropicalization of temperate marine communities could become a global phenomenon (Vergés et al. 2014). In terrestrial ecosystems, abrupt climate change impacts on trees that play a key role in ecosystem functioning may have profound consequences for forest ecosystems as a whole.

Feedbacks on lower levels

S4 – Changes in genetic diversity due to changes in population size

Reduction in population size leads to the loss of genetic diversity, increased inbreeding, decrease in population fitness and further decline in population size. This feedback is known as an "extinction vortex" (Brook, Sodhi, & Bradshaw, 2008; Frankham, 2010; Frankham et al., 2014) (some authors understand "extinction vortex" as a result of only demographic stochasticity). If effective population size falls below a threshold value (about 1000 reproducing individuals) then the population cannot maintain genetic variations in the long term (Frankham et al., 2014). The loss of genetic diversity reduces the chances of evolutionary rescue (***GD1***). Many populations of conservation concern have small effective population sizes, and thus, are prone to high rates of inbreeding, lack of adaptive capacity and evolutionary response (Skelly et al. 2007).

S5 – Changes in genetic diversity due to alteration in species' range (shift, change in area, fragmentation) and dispersal ability

Expected range reduction will lead to the genetic diversity loss because of decrease in effective population size and extinction of local populations. Range fragmentation leads to smaller populations with lower genetic variability and breaks species metapopulation structure and gene flow undermining adaptive capacity of small isolated populations (Neaves et al., 2015; Pauls et al., 2013). A meta-analysis of studies of woody plant species shows that habitat fragmentation is associated with a substantial decrease in population genetic diversity (Vranckx et al. 2012). Increase in connectivity among populations may enhance levels of local genetic diversity and thus, increase population fitness and adaptability (Frankham, 2015; Whiteley et al., 2015) but also may have negative effects as outbreeding depression, the loss of local adaptations and reduction in genetic differentiation between local populations (Neaves et al., 2015). For example, a projected threefold increase in ice-free areas in the Antarctic Peninsula by the end of the century could expand connectivity and gene flow between populations of native species. As a result, some genetic lineages may outcompete others which in the long term could cause the loss of genetic diversity and homogenization of populations (Lee et al. 2017). Range shift by itself can lead to reduction of genetic diversity because only part of the original genetic variation moves to a newly colonized habitat, and because of genetic drift and strong selection pressure in small founder populations (Arenas et al., 2012; Cobben et al., 2011; Hill, Griffiths, & Thomas, 2011; Neaves et al., 2015). Overall genetic diversity is predicted to be lost if core populations become extinct before gene flow restores diversity in newly established populations (Arenas et al. 2012; L E Neaves et al. 2015).

A significant loss of genetic diversity due to range reduction and disappearance of local populations is projected, for example, for brown seaweeds in European coastal ecosystems (Neiva et al. 2015), 27 species of circumpolar northern plants (Alsos et al. 2012), the Nothofagus pumilio tree in the montane forests of Patagonia (Mathiasen and Premoli 2016), European montane insects (Bálint et al., 2011; Habel et al., 2011; Taubmann et al., 2011), geckos in Australia (Duckett and Stow 2013), caribou in North America (Yannic et al. 2014). It is important, that genetic diversity is projected to be lost more quickly than the species' range because of uneven distribution within the range and a projected loss of cryptic evolutionary lineages (Bálint et al. 2011; Habel et al. 2011; L.E. Neaves et al. 2015; Taubmann et al. 2011). However, in some cases, the loss of genetic diversity due to changes in species'range is expected to be insignificant (Kramer et al. 2010; Pfenninger, Bálint, and Pauls 2012).

The rate of the loss of genetic diversity is expected to be higher for species with low dispersal ability. Among plants, short-distance-dispersed herbs are expected to enter an extinction vortex rapidly, while long-distance-dispersed woody species may be less vulnerable to the loss of genetic diversity during range alteration (Alsos et al. 2012; Kremer et al. 2012).

Thus, projections neglecting a possible loss of intraspecific diversity can underestimate negative effects of global changes on biodiversity because diversity losses could greatly exceed those at the scale of morphospecies (Bálint et al. 2011; Habel et al. 2011; L.E. Neaves et al. 2015; Pauls et al. 2013; Taubmann et al. 2011) and at the same time can overestimate of a population's adaptability under rapid environmental change (Schiffers et al., 2014).

SPECIES DIVERSITY

Expected changes

The changes in species abundance (***S2***) and shifts of species' ranges (***S1***) are expected to be the main causes of future alteration of species diversity and species composition. Human-driven introduction of alien species can be considered as a direct impact on species diversity and composition. Alien species are expected to be the major cause of biodiversity loss (Pimm et al. 2014). Species invasions can radically disrupt native species composition and interspecific relationship. The potential areas at the highest risk of terrestrial invasions are located in Western Europe, Eastern United States, Central America, the eastern coast of Australia, and some Indonesian islands (Bellard, Genovesi, & Jeschke, 2016). The highest levels of alien plant invasions are projected for arable and abandoned lands as well as urban areas (Chytry et al. 2012). Human-modified freshwater ecosystems are also highly vulnerable to alien invasions (Johnson et al., 2008).

Changes in species richness should be expected different at different spatial scales. Species richness of vascular plants is expected to decline at the global scale because species extinctions will outnumber speciation. An increase in species richness can be expected at the regional scale if the number of new non-native species exceeds the number of locally extinct native species, and at the local scale, different changes can occur depending on local processes (Vellend et al. 2017).

Globally, from 7.9% (Urban, 2015) to 10% (Maclean and Wilson 2011) of terrestrial and aquatic plant and animal species are predicted to become extinct due to climate change during the 21st century. An average loss of 3.4% of terrestrial local diversity because of land use is predicted under a business-as-usual scenario (MESSAGE8.5) and an average increase in species richness up to 1.9% under the scenario with climate mitigation through carbon markets, crop improvements and diet shifts (MiniCAM 4.5) (Newbold et al. 2015). The range of estimates of projected extinctions for different taxonomic groups is quite large - from 0% to more than 50% (Bellard et al., 2012; Urban, 2015)). Extinction risks for exploited marine fish and invertebrates are projected as 4% - 7% which is, on average, lower than projections for terrestrial species due to expected greater freedom of movement in the sea (Cheung et al., 2009). Projections of changes in species abundance due to climate change, land use and pollution are of tens of percent (Alkemade et al., 2009; Leadley et al., 2013; Visconti et al., 2016). The real effects of climate change might far exceed the current predictions because the mean observed extinction risk is systematically higher than the mean predicted risk (Maclean and Wilson 2011). Moreover, many undescribed species may become extinct without our knowledge (Mora et al. 2011; Pimm et al. 2014).

The highest climate-driven extinction risks for terrestrial and aquatic plants and animals are projected for South America, Australia and New Zealand (Urban, 2015). 86% of terrestrial and 83% of freshwater ecoregions from the “Global 200” list will be exposed to extreme climatic conditions. The tropical and subtropical ecoregions in Africa and South America, and the mangroves will face extreme conditions first (Beaumont et al. 2011). High loss of terrestrial species is expected in the Amazon, sub-Saharan Africa because of the predicted doubling of human population size and rapid reduction in the extent of natural vegetation, and in insular Southeast Asia because of the highest rates of deforestation (Visconti et al., 2016). Agricultural expansion can cause the loss of 30% of species richness and 31% of species abundances (for mammals, birds, amphibians and reptiles) in the Amazon and Afrotropics, and the loss of about 7% of species richness and 13% of abundance in India, Eastern Europe and the Afromontane region (Kehoe et al. 2017). Other projections predict biodiversity gain under a business-as-usual scenario in Europe and North America while Southeast Asia and especially sub-Saharan Africa will suffer the greatest biodiversity loss (Newbold et al. 2015). The high-risk regions are also the areas rich in top-mountain specialists, such as the Andes for mammals (Lawler et al. 2009; Schloss, Nunez, and Lawler 2012) and the Himalayas for birds (Jetz et al. 2007). The biomes projected to lose the most vascular plant species are warm mixed forest, savannahs, shrub, tropical forest, and tropical woodlands (Detlef P van Vuuren, Sala, and Pereira 2006). Island biodiversity is also seriously threatened by rising sea levels and the particularly severe adverse effects of climate change and species invasions (Courchamp et al. 2014). For marine species the highest extinctions are expected in the tropics, sub-polar regions and semi-enclosed seas, e.g., the Mediterranean and Red Seas (Cheung et al., 2009; Jones & Cheung, 2015). For coastal biogeographic provinces, disproportionately high risk is expected in the tropics and subtropics due to the coincidence of high intrinsic extinction risk for vertebrate and invertebrate taxa, rapid climate shifts and elevated human activity (Finnegan et al. 2015).

Biological differences between species may significantly increase or reduce their vulnerability to climate change and largely change the regional distribution of future risks. Areas with a large concentration of species with the highest sensitivity and lowest adaptive capacity to climate change differ from areas greatly exposed to climate change. Thus, exposure-based assessments may over- or underestimate climate change impacts (W. b Foden et al. 2013).

Biodiversity hotspots are projected to be highly vulnerable to global changes during the 21st century. Hotspots may experience an average loss of 31% of their area because of climate change, primarily at low latitudes. Also climate change might negatively influence 25% of endemic plant and vertebrate animal species per hotspot on average (Bellard et al., 2014). Other projections show average potential risk of 11.6% for endemic species (Malcolm et al., 2006). The highest climate-driven extinction risks are predicted for the Cape Floristic Region, the Caribbean, Indo-Burma, Mediterranean Basin, Southwest Australia, and Tropical Andes (Malcolm et al., 2006). Three biodiversity hotspots – the Atlantic forest, the Cape Floristic Region and Polynesia–Micronesia - are particularly vulnerable to global changes under the impact of three drivers – climate change, land use and invasive species (Bellard et al., 2014). Insular biodiversity hotspots are expected to be threatened world-wide by invasive species and sea level rise. Between 6% and 19% of islands would be entirely submerged when 1–6 m of sea level rise. The most significant loss of insular habitat is expected in the Caribbean islands, the Philippines and Sundaland, representing a potential threat for 300 endemic species (Bellard, Leclerc, & Courchamp, 2013). Some previously overlooked biodiversity hotspots could completely disappear, for example, it is projected for the Atlantic Coastal Plain lichen biodiversity hotspot due to sea level rise (Lendemer and Allen 2014). A projected increase in N deposition is also an important threat for biodiversity hotspots (Bleeker et al. 2014).

Projected local species extinctions are extremely different, with some areas with no losses and others facing nearly complete loss of current species (Bellard et al., 2012; Leadley et al., 2010; 2013). Local changes in species richness and abundance can range from positive to highly negative (Settele et al. 2014). The expected poleward and upward species shifts can lead to a local increase in species richness. For example, an increase in species richness in high latitudes while a decrease in low latitudes is predicted for exploited marine fish and invertebrates (Jones and Cheung 2015), for marine mammals (Kaschner et al. 2011) including deep-water cetaceans (Whitehead, McGill, and Worm 2008), an increase in α–diversity is predicted for plants in French Alps (Thuiller et al. 2014) as well as increase in plant species richness is generally expected at the regional scale due to invasions of non-native species (Vellend et al. 2017). However, such local and temporal increase in species richness will be accompanied by disruption of community structure. For example, the expected threefold increase in ice-free area and habitat connectivity in the Antarctic Peninsula can have destabilizing impacts on ecological communities and can enhance biotic homogenization via the spread of invasive species and extinction of less-competitive native species (Lee et al. 2017).

Major alteration of species composition is expected as a result of species shifts and local changes in species abundance. Species turnover of over 60% of the present diversity is predicted for exploited marine fish and invertebrates under high-range climate change scenario (Cheung et al., 2009) and 25% - 38% turnover is predicted for birds, mammals, and amphibians in the Western Hemisphere under B1 and A2 scenarios, respectively (Lawler et al., 2009). For breeding birds across the Important Bird Area (IBA) network in sub-Saharan Africa, species turnover is predicted to be more than 50% at 42% of IBAs (Hole et al., 2009). The general directions of changes in species composition both in marine and terrestrial communities are expected to be tropicalization, thermophilization (Gottfried et al. 2012; Vergés et al. 2014) and switching from polar to more temperate species in the north (Philippart et al. 2011).

The emergence of novel ecological communities is expected as a result of the unprecedented speed of global changes and the local formation of non-analogous conditions. Novel climates will arise in tropical and subtropical regions (Williams and Jackson 2007), in the North American Great Plains and temperate forests, the Amazon, South American grasslands, Australia, boreal Asia and Africa (Ordonez, Williams, and Svenning 2016). Novel combinations of temperature, precipitation, nitrogen deposition and human population are predicted in East Africa, the Arabian Peninsula, India, China, and Australia (Radeloff et al. 2015). In Antarctica, the physical expansion of ice-free areas can lead to extinctions in terrestrial native biodiversity (Lee et al. 2017) and formation of novel communities.

Heterogeneity in species responses to global changes could disrupt existing communities and create new no-analog communities, where species co-occur in historically unknown combinations (Blois et al., 2013; Gilman et al., 2010; Lurgi et al., 2012; Ordonez et al., 2016). The structure of novel communities is expected to shift towards generalists and smaller size species (Lurgi et al. 2012). The combined impacts of species extinctions and invasions will make novel communities highly homogeneous. Tightly coevolved interactions such as mutualism and parasitism have a great risk of disappearance (Blois et al. 2013). It can also be assumed that novel communities will be less stable than native communities, because interspecific relations do not have a long history of co-adaptation (Gilman et al. 2010).

Impacts on upper levels

SD1 – Weakening and destabilization of ecosystem functioning due to loss of local species diversity

Since it has been proved that species diversity is the structural base determining the magnitude and stability of ecosystem functioning (Cardinale, et al., 2012; Fischer et al. 2018, background A5; Tilman et al. 2014), it should be expected that the loss of native species diversity will lead to weakening and destabilization of ecosystem functioning.

SD2 – Biotic homogenization as a result of species shift, local species extinctions and invasions

Projections for terrestrial and aquatic biodiversity predict disproportionate loss of evolutionary and functionally distinct species, rare and endemic species that will reduce functional (Buisson et al., 2013; Thuiller et al., 2014)) and phylogenetic diversity (González-Orozco et al., 2016; Thuiller et al., 2011; Vamosi & Wilson, 2008; Zhang et al., 2015)). In some cases, expected species range shift can lead simultaneously both to a severe decline in functional diversity within communities, and to an increase in functional similarity among communities as predicted for fish species in French streams (Buisson et al. 2013). In other cases, for example, for plant species in the French Alps, a paradox of gaining α-diversity but losing β-diversity is expected (Thuiller et al. 2014). Generally, invasions of alien species and a decline in native specialist species and disproportional loss of taxonomic, phylogenetic of functional diversity will lead to biotic homogenization of ecological communities and landscapes reducing the differences between communities and their uniqueness (Clavel, Julliard, and Devictor 2011; Lee et al. 2017; Olden 2006; Thuiller et al. 2014). On a global scale, human-mediated species dispersal is causing a breakdown of biogeographic patterns and in the future spatial distribution of species may become more homogenous and will be determined primarily by climate and socioeconomic factors (Capinha et al., 2015).

Feedbacks on lower levels

SD3 – Changes in selection pressure because of alteration of species composition and interspecific relations (including effects of alien species invasions)

Changes in species composition alter selection pressure and affect genetic diversity. For example, a reduction in pollinator abundance may lead to selection favoring self-fertilization in plant populations, leading to a decrease in genetic diversity (reviewed by Neaves et al., 2015). Pest invasions may catastrophically decrease genetic diversity of target species. Species invasions may result in hybridization, out-breeding depression and a decrease in genetic diversity. However, hybridization may also introduce new genetic variations which can facilitate adaptation to changing conditions (Hoffmann & Sgro, 2011). Generally, environmental changes provoke an eco-evolutionary response, that integrates ecological and evolutionary responses of species interacting within communities (Fussmann, Loreau, and Abrams 2007; Lavergne et al. 2010; Norberg et al. 2012).

SD4 – Species extinctions as a result of cascading effects of alteration of species composition

Alteration of species composition and interspecific relations may lead to cascading co-extinctions called “chains of extinction” (Bellard et al., 2012; Brook et al., 2008; Gilman et al., 2010). The loss or depression of key species, including top predators and pollinators, as well as invasions and reinforcement of pests and pathogens can destroy species relationships most strongly, lead to cascade alteration of community and secondary species extinctions (Bellard et al., 2012; Edeline et al., 2016; Estes et al., 2011; Tylianakis et al. , 2008). In many cases, such biotic stresses are likely to be more severe than maladaptation caused by global changes (Aitken et al., 2008) Analyses of a large amount of local species extinctions and extinction risks showed that only a small part of extinctions is directly caused by climate change or anthropogenic drivers. Instead, many studies implicate species interactions and extinction of associated species as an important proximate extinction cause (Bellard et al., 2012; Cahill et al., 2012). Failure to incorporate species interactions limits the ability to predict responses of species to climate change and the probability of local extinctions (Gilman et al. 2010; Tylianakis et al. 2008).

SD5 – Impact of alteration of species composition on species capacity to track climate change

Changes in species composition and interspecific interactions may have important consequences for species range dynamics. Competition may prevent species from tracking their climatic niche or enhance expansion rates. (Lavergne et al. 2010; Svenning et al. 2014). In some cases, competitive release at the rear (lower) margin of species distribution can reverse range shifts in the opposite direction (Lenoir et al. 2010). Predators can directly limit prey range or, conversely, improve the prey distribution if specialist predators stabilize prey population (Holt and Barfield 2009). Mutualistic interactions, such as plant-pollinator relations, are expected to slow down tracking environmental change because of their lower effective colonization rate (Lavergne et al. 2010). The interplay between species relations which impact species dispersal in opposing directions makes accurate predictions of range shifts difficult. For example, long-distance dispersal of seeds of the North American black cherry by birds can have both positive and negative effects on seed establishment because both specific soil pathogens and symbiotic mutualists such as arbuscular and ectomycorrhizal fungi are absent away from adult trees (Van Der Putten, Macel, and Visser 2010). Incorporating species interactions in species distribution models can both slow down climate tracking (Urban, Zarnetske, & Skelly, 2013) and expand predicted range (Bateman et al., 2012).

ECOSYSTEMS

Expected changes

The impact of direct drivers on ecosystems lies in change of climatic conditions, changes in area and fragmentation due to land use and use of water bodies, impact of pollutants, destruction due to harvesting, habitat-forming activity of alien species and destructions due to different kinds of disturbances (fires, floods, earthquakes, etc.). Cumulatively, urban and agricultural expansion, oil, gas, coal, solar, wind, biofuels and mining development place at risk 20% of the remaining global natural lands (Oakleaf et al. 2015). Habitat loss remains a more important negative driver of biodiversity changes than climate change for many species. In the 2000–2050, land-use change is expected to contribute 7%–13% to the global plant species diversity loss, while climate change is expected to contribute 2%–4% under different scenarios. Globally, by 2100, the loss of 3.4% of local species diversity due to land use is predicted under a business-as-usual scenario (MESSAGE8.5) (Newbold et al. 2015). Habitat loss due to agricultural expansion may lead to the loss of 30% of species richness and 31% of species abundances (for mammals, birds, amphibians and reptiles) in the Amazon and Afrotropics, and to the loss of 7% of species richness and 13% of abundance in India, Eastern Europe and the Afromontane region (Kehoe et al. 2017). On islands in the Southeast Asian and the Pacific region, sea-level rise due to climate change will lead to loss from 3% to 32% of coastal areas and secondary habitat loss caused by the displacement of human due to sea-level rise can lead to an equal or even higher range loss than primary effects of sea-level rise (Wetzel et al. 2012). However, after 2050, climate change may become increasingly important (van Vuuren, Sala, & Pereira, 2006). Aquaculture and deep-sea mining will become increasingly important factors of freshwater and marine habitat loss (Leadley et al., 2013).

Other impacts on ecosystems are results of changes in species in key species abundance and functioning (***S3***) and of alteration of local species diversity (***SD1***).

The main predicted changes in marine, freshwater and terrestrial ecosystems are climate-driven shifts, decrease in biodiversity and substantial changes in ecosystems' structure and functioning (Global Assessment, Chapter 4, sections 2.2, 2.3, 2.4).

Impacts on upper levels

E1 – The contribution of individual ecosystems to the total landscape/seascape ecosystem functioning

Integral landscape or regional ecosystem functioning depends on the state and functions of all ecosystems and habitats. Maintenance of landscape multifunctionality requires preservation of landscape-level habitat diversity (Bolliger and Kienast 2010; van der Plas et al. 2016).

E2 – Disappearance of the most vulnerable ecosystems in landscapes/seascapes and regions

Different ecosystems, habitats and communities within a landscape or a region vary greatly in the risk of collapse and extinction, as it was, for example, assessed for the Mojave and Sonoran deserts (Comer et al. 2012). Thus, disappearance of the most vulnerable ecosystems, habitats and communities and decrease in landscape/regional diversity of ecosystems/habitats should be expected as a probable result of climatic or human impacts. This, in turn, will decrease landscape heterogeneity and increase biotic homogenization and its consequences (***ED2, ED3***).

Feedback on lower levels

E3 – reduction in species population size, reduction in, and fragmentation of species' ranges, disruption of population structure because of habitat loss and fragmentation

Habitat loss and fragmentation increases the risk of species extinction under global changes. Fragmentation of both terrestrial and aquatic ecosystems leads to disruption of species metapopulations, isolation and extinction of local populations, violation of life cycles, breaking of migration routes. Smaller habitat patches sustain smaller populations, which are likely to fall into "extinction vortex" (***S4***). Habitat fragmentation leads to genetic diversity loss and decreases species adaptability (***S5).*** Habitat destruction and fragmentation are expected to reduce the possibilities of species to survive in suitable microclimatic refugia and to hamper species climatic migrations (Hof et al., 2011; Jump & Peñuelas, 2005; Scheffers et al., 2014; Settele et al., 2014). In the Western Hemisphere, the ability of mammals to keep pace with projected climate change will be seriously limited due to habitat fragmentation. High intensities of human land use may prevent mammals' migration at wide areas where species dispersal velocities could allow them to keep pace with climate change in undisturbed habitats (Schloss et al., 2012). A model of 14 European trees predicts a considerable time lag in their range shifts due to habitat fragmentation (Meier et al., 2012).

The synergistic effect enhancing the negative impact of both climate change and habitat loss is observed for existing species (Mantyka-Pringle et al. 2015) and is projected for future species distributions. For example, for the critically endangered mountain tapir in Equador the effect of climate change by 2050 is predicted as a net range reduction from 4% to 37.86% (RCP 4.5 and 8.5 scenarios), while the synergistic effect of both climate change and habitat loss will lead to a range reduction of 19.90% to 44.65% (Ortega-Andrade et al., 2015).

DIVERSITY OF ECOSYSTEMS

Expected changes

Changes in individual ecosystems (***E2***) and biotic homogenization due to species invasions, shifts and local extinctions (***SD2***) are expected as the main causes of future alteration of diversity of ecosystems and habitats. Expansion of human made types of ecosystem (urban and agricultural ecosystems, specific ecosystems related to aquaculture, technical systems, etc.).

Impacts on upper levels

ED1 – Weakening and destabilization of the total landscape/seascape functioning because of loss of ecosystem/habitat diversity

The projected biotic homogenization and the loss of diversity of local communities may reduce the variability of biological responses on disturbances across individual communities. This will increase landscape vulnerability to climate- and human-driven impacts and compromise the potential for landscape- and regional-level buffering.

Homogenization may also decrease landscape resistance to future species invasions because it was shown that spatial heterogeneity reduces the expansion of invasive species (Olden 2006)

Shrinking or disappearance of the most vulnerable ecosystems and habitats will reduce their regional or landscape diversity, which may have negative consequences for large-scale ecosystem multifunctionality (van der Plas et al. 2016).

Feedback on lower levels

ED2 – The influence of landscape heterogeneity on local species persistence

Habitat heterogeneity and microclimatic variability which provides microrefugia amidst unfavorable conditions, increase the species ability to survive under climate change (Ackerly et al. 2010; Lenoir et al. 2013; Moritz and Agudo 2013; Randin et al. 2009)). Microrefugia may be located within species' ranges, especially for species with wider distributions, or may be reached over much shorter distances than those implied by range shifts (HOF et al. 2011). In the tropics, microhabitats reduce maximum temperatures by 3.58 C that may have crucial importance under expected increase in extreme weather events (Scheffers et al., 2014). In England, microclimatic heterogeneity reduces extirpation risk for climate-threatened and range-declining species by 22% for plants and by 9% for insects (Suggitt et al. 2018). Incorporating microclimatic variability into species distribution models predicts a greater local species persistence (Lenoir et al. 2013; Randin et al. 2009; Willis and Bhagwat 2009) as predicted for alpine and nival species in the Swiss Alps (Randin et al. 2009). However, mocroclimate buffering may be significantly reduced by habitat fragmentation. For example, in the Atlantic forests of Brazil, temperature buffering effect of forests is reduced near edges up to 20 m inside the forest and 12% of the remaining forests have altered microclimate conditions because of fragmentation (Ewers and Banks-Leite 2013).

ED3 – The influence of landscape heterogeneity on genetic diversity and evolution

Landscape and habitat heterogeneity produces spatial variations of environmental conditions that require local adaptations. This spatial pattern influences intraspecific diversity, adaptive capacity of populations and species and a probability of evolutionary rescue. For example, a model of a plant population adapting to an increasing temperature in habitats with spatial heterogeneity in soil conditions shows that the probability of evolutionary rescue is maximized at some intermediate dispersal distance providing maintenance of local adaptation to soil conditions. The discrepancy between intraspecific gene flow and habitat heterogeneity may substantially reduce projected species' range and the likelihood of evolutionary rescue (Schiffers et al., 2012). A species can sustain faster environmental shifts, develop a wider range and greater local adaptation when a spatial environmental variation is not excessively high (Duputié et al., 2012). Moreover, rapid adaptation is favored by a good match between the coarseness of the trait's genetic architecture (many loci of small effects versus few loci of large effects) and the coarseness of the landscape (the abruptness of transitions in environmental conditions (Schiffers et al., 2014). In the long-term, highly uniform habitats and biotic homogenization may compromise the potential for future speciation because of the limited spatial variability in species diversity and composition (Olden 2006).

A4.2.2 Supporting information, Figure 4.2.2 (terrestrial ecosystems)

Table A4.2.1: Short overview over impact of major drivers on IPBES Units of Analyses

Unit of Analysis	Driver	Future Impacts	Source
Tropical and subtropical dry and humid forests	Climate change	Warmer temperatures and enhanced water stress in some regions will likely result in deterioration of humid forest (that is: reduced fractional cover, vegetation C stocks, and NPP) especially at the dry distribution end.	(Aragão et al. 2014; Malhi et al. 2009; Nobre et al. 2016)(Huntingford et al. 2013)
		Species expected to be most vulnerable are those at today's dry end of humid forest region, and at higher altitudes. Lowlands may lose species if temperatures rise above species' heat tolerance, and there is the possibility of novel communities of heat tolerant species to be emerging. Suitable area for mammals has been estimated to be reduced, with large variability depending on the different species.	(Krupnick 2013; Nobre et al. 2016)(R K Colwell et al. 2008)(Brodie 2016)
	CO_2	CO_2 fertilisation and increased water use efficiency compensates at least to some degree detrimental climate effects of climate change on forest carbon balance and growth. In an intercomparison of one impact model driven by 22 GCMs, forest "die-back" was observed only in very few cases. The enhancement of C-cycle variables tends to saturate or, begins to decline at the end of the 21st century. Increasing occurrence of lianas in humid forests has been linked to CO_2 effects, but lianas (and their impacts on tree growth and mortality) are not included in global ecosystem models.	(Aragão et al. 2014; Malhi et al. 2009)(Gumpenberger et al. 2010)(Rammig et al. 2010)(Nobre et al. 2016)(Poulter et al. 2010)(Duran and Gianoli 2013; Schnitzer and Bongers 2011)
		Forest fires might increase in future, especially in dry years, and contribute to ecosystem deterioration, unless controlled	(Aragão et al. 2014; Malhi et al. 2009)
	Land-use change	Future deforestation will reduce expanse of humid forest, the degree depends on the scenario. Considerable loss of plants, invertebrates, mammals, reptiles birds or beetles has been found not only in response to deforestation but also to selective logging and other disturbances, such as fire; harvesting for tropical timber may be a particular threat for certain tree species or genera (e.g., dipterocarps). Scenarios of reforestation and avoided deforestation contribute to reduced C-emissions, and can contribute to halt species loss. Birds, invertebrates, mammals and reptiles show a decline in their probability of presence with declining forest cover (especially pronounced in forest specialists or narrow-ranged birds).	(Davies-Barnard et al. 2015; Malhi et al. 2009; Poulter et al. 2010)(Heubes et al. 2011; Krupnick 2013)(Barlow et al. 2016)(Bird et al. 2012)(Newbold et al. 2014)

		Declining crop yields in response to climate change could lead to additional deforestation and conversion to agricultural land, unless crop demand is not met by other ways such as through trade or changes in diet. However, trade for food commodities can also foster tropical deforestation if agriculture and pasture production has competitive advantage on the world market.	(Lapola et al. 2011)(Schmitz et al. 2015)
		Deforestation in humid tropical forests might amplify future decreasing precipitation in tropical regions and lead to a regional warming through reduced evapotranspiration; avoiding deforestation has therefore an important incentive through impacts on regional climate.	(Alkama and Cescatti 2016; Bright et al. 2017; Davidson et al. 2012)(Perugini et al. 2017)
		Replacement of natural tropical forest systems by commodity tree crops such as oil palm reduces biodiversity, but also has additional regional health impacts by emissions contributing to surface O_3 and SOA formation; substances that are also relevant for climate change.	(Hewitt et al. 2009)(Silva et al. 2016) Silva et al. 2016, (Hewitt et al. 2016)
Tropical and subtropical savannas and grasslands	Climate change	Impacts of only climate change on savanna extend is inconclusive and emerge differently between continents, likely because of the variable interplay between factors that shape savannas. Fire plays a very important role in simulated future spread of savannas but representation in ecosystem models of other crucially interacting factors, such as soil type and chemistry, or grazing and browsing is mostly absent.	(Staver, Archibald, and Levin 2011; Zeng et al. 2014)(Moncrieff et al. 2016)(Lehsten et al. 2016)(Veenendaal et al. 2015)(Midgley and Bond 2015)(Pachzelt et al. 2015)(Scheiter et al. 2015)(Heubes et al. 2011)
	CO_2	Across a range of climate change and CO_2 scenarios, rising atmospheric CO_2 has a profound effect on future modelled savanna vegetation, which is projected to shift towards wood dominated regions in response to CO_2 fertilisation fostering the C3 photosynthetic type. These results are enhanced by feedbacks with simulated decreased fire spread as woody cover increases. Altered seasonality and amount of precipitation may have a modulating impact on the fire- CO_2 interplay.	(Moncrieff et al. 2014, 2016; Scheiter et al. 2015)(Lehmann et al. 2014; Lehsten et al. 2009) (Knorr, Arneth, and Jiang 2016)
	Land-use change	Conversion of savannah-type landscape into cropland and pastures (such as has taken place in the Cerrado or Chaco regions of southern America) will continue to exist; arguments have been put forward that parts of African savannas or woodlands might follow a similar agricultural development pathway. Land-use change projections resulted in a decline in grassy biome habitat loss in parts of sub-Saharan Africa by 2070, but with regional differences reflecting	(Aleman et al. 2016, 2017; Searchinger et al. 2015)(Scheiter and Savadogo 2016)

		differences in the underlying socioeconomic assumptions. It was also argued that some types of land use (grazing, wood harvest, fire management) can maintain open savanna over wooded state in future.	
		Assumptions about large agricultural technological and knowledge advances support projections of crop and pasture areas being stable or even declining in future despite of population growth or dietary changes. For Africa, such a scenario led to even an increase in average mean species abundance by 2050. A study that explored a stringent global forest conservation policy under a RCP 2.6 framework that mimics a REDD scenario led to expansion of agricultural land into what is currently grasslands and savannas by 2050 and 2100, with associated loss of carbon. Based on present-day ecosystem state only a small percentage area of African humid savanna could be considered for food or bioenergy production when sustainability criteria or plausible loss of vertebrate diversity are also considered.	(Alkemade et al. 2013)(Popp et al. 2014)(Searchinger et al. 2015)
		Human population growth as well as different degrees of urbanization were found to have equally large effects on projected burnt area than different climate and CO2 scenarios. A large degree of warming combined with relatively low population growth and rapid urbanisation was simulated to result in burned area increase in some savanna regions.	(Knorr et al. 2016)
Drylands and desserts	Climate Change	Climate change is projected to result in dryland expansion in many regions, the degree depends on the rate and magnitude of warming and drying in climate change projections. A substantial fraction of the area now classified as temperate drylands might fall into the sub-tropical category in future, averaged across different climate models at the end of the 21st century. Biophysical surface exchange processes are projected to amplify future increases of the aridity index.	(Huang et al. 2016, 2017; Schlaepfer et al. 2017)(Feng and Fu 2013)(Berg et al. 2016)
	CO_2	Enhanced water use efficiency can lead to wood vegetation expanding into arid grasslands, and grasslands expanding into desserts	(Moncrieff et al. 2016; Scheiter and Savadogo 2016)
	Land-use change	Simulation experiments that assess future land-use change impacts (in context also of climate change and CO2) for global dryland ecosystems are absent. Climate change alone is expected to reduce dryland area crop yields and runoff. Being fragile ecosystems that have large pressure due to large human population density, land use change is expected to enhance degradation and desertification, and loss of biodiversity. As species richness emerged as an important predictor variable in ecosystem multifunctionality relationships found in dryland ecosystems, maintaining richness, for instance through sustainable land-use, will be important to dampen impacts of climate change.	(Huang et al. 2017)(Maestre et al. 2016)(Maestre et al. 2012)
Temperate grasslands	Climate Change	A recent cross-continental synthesis of grassland plot studies, mostly located in temperate climate zones, confirmed the positive correlation of species richness and above-ground productivity at site and plot-scales in grasslands, while biomass accumulation was negatively correlated with species richness. Even though productivity in often enhanced with precipitation, meta-analyses of experimental study plots that are located to large degree in the temperate regions found increase in aboveground biomass, especially in forbs, in response to addition of water. Effects on	(Grace et al. 2016)(DeMalach, Zaady, and Kadmon 2017; Stevens et al. 2015)(Lin et al. 2013)(Barnett and Facey 2016)

		plant biodiversity were not significant. Altered amount or seasonality of precipitation was found to also affect anthropods, soft-bodied species being expected to be vulnerable to reductions in water availability. Simulation studies on the future impact of global change on the diversity and functioning of the global temperate grassland biome are largely absent. In one study, applying an A2 climate change scenario, temperate humid grasslands increase slightly by 2050 in their extent, while steppe remained constant.	
	CO_2	Whether or not increasing CO_2 enhances above ground biomass seems connected with the seasonality and balance between summer and spring/autumn rainfall. Soil texture, and related available soil water capacity, notably affects grassland response to changes in precipitation, temperature and CO_2. Although elevated CO_2 has been found to lead to shifts in species composition towards a higher fraction of legumes, this effect diminished in experiments in experimental plots that were also grazed.	(Hovenden, Newton, and Wills 2014; Obermeier et al. 2017)(Weng and Luo 2008)(Newton et al. 2014)(Soussana 2013)
	Land-use change	It is still open whether the globally rapidly increasing demand for meat products will be predominantly met by an increase of industrial livestock production, exerting pressure on cropland extension, rather than rangelands. In scenarios of different degrees of production-intensification, varying assumptions on grazing intensity had little effects on mean species abundance (MSA) by 2050 in the northern US, increased MSA (at different degrees) in northern Asia and had contrasting effects in Europe.	(Alkemade et al. 2013)
Mediterranean forests, woodlands and scrub	Climate change	Global future climate scenarios for Mediterranean areas project warming in all seasons, and declining precipitation. For different emissions scenarios global and regional climate models project expansion (and/or shift) of current climatically suitable areas in the European Mediterranean and Chile, stability in California and a contraction in South Africa and Australia. Existing evidence is conflicting whether or not the accounting for dynamic vegetation feedbacks in climate projections amplifies or dampens climate change. Occurrence of episodic events such as fire and drought, and how these affect fuel moisture and fuel amount are important components that shape Mediterranean vegetation but are usually not considered in future projections. Scenarios tend to identify an increase in fire frequency related to hotter and drier conditions, but at the same time, can a reduced plant density and growth due warming lead to reduced fire spread in more sparse vegetation.	(Polade et al. 2014)(Klausmeyer and Shaw 2009)(Barredo, Caudullo, and Dosio 2016)(Alo and Anagnostou 2017; Wramneby, Smith, and Samuelsson 2010)(Wu et al. 2015)(Batllori et al. 2013, 2017)(Turco et al. 2014)
	CO_2	Similar to savannas, increasing CO_2 may lead to an increase of woody over grassy vegetation in the fire-dependent Mediterranean ecosystem and reduce burned area due to reduced spread.	(Midgley and Bond 2015) (Wu et al. 2015)
	Land-use change	All Mediterranean-type regions globally are characterised by either high human population density, substantial agriculture, or both. Statistical relationship that show an increase of threatened mammal and plant richness with population therefore could indicate a potential threat, depending how socio- economic conditions develop in future, and also how people will affect fire regimes	(Underwood et al. 2009)(Syphard et al. 2009, 2013)
	Climate change	Boreal tree species are projected to migrate in response to warming into regions dominated by tundra in response to climate warming, unless constrained by edaphic features. Temperate species are projected to grow in regions	(Arneth et al. 2016; Davies-Barnard et al. 2015; Falloon et al. 2012) (Gauthier et al.

Temperate and boreal forests and woodlands		currently occupied by southern boreal forest. In general, deciduous trees will grow in regions currently dominated by conifers, such simulated vegetation changes are supported by observations of deciduous-conifer shifts already occurring now in some regions. Combining climate variables in a stylized experiments such that variable-combinations do not have present-day analogues can also lead to no-analogue vegetation, as demonstrated in one study for northern Eurasia. Vegetation shifts will also affect diversity of other species as was demonstrated for birds. Forest futures will depend on how stand-replacing disturbances will interact with climate changes. In particular fire, insects, drought and other extreme weather events, and their interactions are well documented to lead to tree mortality in temperate and boreal forests. Fragmentation in boreal forest appears to affect plant species diversity less than in tropical forests because large-scale disturbance is an intrinsic feature of the ecosystem. Overall, disturbances appear to enhance indices for biodiversity, while decreasing ecosystem functionality that underlies a range of services (e.g., carbon pool size). However, since process-based vegetation models at present do only account for fire explicitly as a direct means of mortality projections into the future are speculative. Climate change is projected to have conflicting impact on the abundance of boreal breeding birds, some species decreasing, some increasing, but with large uncertainties not only due to the climate projections but also due to ignoring additional factors (dispersal, habitat availability).	2015)(Reu et al. 2014)(Stralberg et al. 2015)(Allen, Breshears, and McDowell 2015; Kautz et al. 2017; Pawson et al. 2013; Thom and Seidl 2016)(Millar and Stephenson 2015)
	CO_2	The response of vegetation and soil carbon storage to projected warming and changes in forest composition are very uncertain, and depend on the balance between increasing uptake in vegetation as the growing season lengthens and CO_2 increases, and stimulation of microbial decay. For forest growing on permafrost soils simulation results depend not only on the degree of warming but also on whether or not microbial heat production, thermokarst formation, snow insulation, C-N interactions or fire are considered. Changes in vegetation will reduce albedo in northern regions, increase evapotranspiration and enhance BVOC emissions.	Arneth et al. 2016; Davies-Barnard et al. 2015; Falloon et al. 2012)(Schaefer et al. 2014) (Arneth et al. 2010)
	Land-use change	The temperate forest region has seen in recent years large areas of regrowth, due to land becoming available in the wake of the green revolution, and from active reforestation and afforestation efforts. These trends might well continue, both in response to projected wood demand, but also since in response to the Paris COP21 climate agreement afforestation and reforestation is considered a viable, cost-effective mitigation strategy, but –like for natural forest shift in response to warming– trade-offs with a range of other ecosystem processes will develop. In scenarios that are globally dominated by large afforestation/reforestation efforts, land-use change is projected to increase forest area	(Naudts et al. 2016)(d'Annunzio et al. 2015)(Popp et al. 2016)(Davies-Barnard et al. 2015)
Tundra and high mountain habitats	Climate change	Climate models project that warming for the Arctic tundra is likely to continue at about double the global rate (IPCC WGI 2013). The Arctic is also projected to have among the largest increases in precipitation globally, although there is high uncertainty in these projections (IPCC WGI 2013). Large C losses are projected from carbon-rich soils, especially in permafrost regions, depending on the degree of warming. These C losses will feedback to climate warming.	(Koven et al. 2011; Schuur et al. 2008, 2009)

	Polar bears have received considerable attention because populations have been rapidly declining in several regions with climate change as one of the likely drivers (Settele et al. 2014) contributing to making polar bears an iconic species for climate change impacts on biodiversity. Climate change had projected negative effects on polar bear populations via its effects on sea ice, and that these were much more important determinants of bear population dynamics than being hunted or other interactions with people. A shift from seal to bird nest predation by polar bears as a mechanism of adaptation to climate change is projected to be insufficient to halt the decline of bear populations. Increases in shrub dominance due changes due to climate warming, are projected to have uncertain or widely varying effects on tundra plant species, ground squirrels and birds. For example, many more bird species are projected to profit from than decline due to modest increases in shrubs, while large shifts in shrub dominance are projected to have negative effects on most bird species.	(Atwood et al. 2016; Dey et al. 2017; Mod and Luoto 2016; Thompson et al. 2014; Wauchope et al. 2017)
	Models of plant functional types and vegetation response to climate project a shrinking of the area of tundra globally (Settele et al. 2014) due to increases in shrub dominance and boreal forest encroachment. Multi-model comparisons using dynamic vegetation models show that Arctic tundra ecosystems are generally projected to continue to sequester carbon throughout most of the 21st century. But there is much higher uncertainty than previously recognized with some models indicating a shift to very large carbon sources by the end of the century, and that the bulk of this uncertainty arises from differences between the vegetation models rather than from differences in climate models or greenhouse gas emissions scenarios. A model recently developed specifically for Arctic tundra ecosystems suggests that the paradigm of increasing shrub domination due to climate change should also be nuanced, because this response appears to depend heavily on concurrent changes in precipitation as well as the dynamics of local thawing of the permafrost.	(Mod and Luoto 2016)(Gang et al. 2017; Ito, Nishina, and Noda 2016; van der Kolk et al. 2016; Nishina et al. 2015)
	Settele et al. (Settele et al. 2014) found that most plants and animals can potentially move quickly enough to remain in favourable climates in mountains for all projected climate warming scenarios. This means that species that move upward to adapt to climate change typically will have their area of distribution rapidly compressed (Carlson et al. 2014, Gang et al. 2016), potentially facing extinction due to the complete loss of favorable environmental conditions (Ramirez-Amezcua et al. 2016). In contrast, an important mechanism working in favor of the persistence of high mountain species, and one that is increasingly being accounted for in future projections, is that strong environmental gradients create many opportunities for climate refugia where species can potentially persist for long periods of time (Settele et al. 2014, Niskanen et al. 2017).	
Land-use change	In contrast to climate change, land use change is projected to be very low in Arctic tundra systems. Land use pressures in high mountain areas are dominated by grazing, farming or forestry, and these activities typically have been and are projected to be substantially lower than land use impacts in lower altitude areas. In the European Alps where grazing and forestry in high mountain areas was traditionally relatively intense, the legacy effects of abandonment of management are projected to dominate biodiversity and ecosystem dynamics compared to climate change over this century except in the most extreme climate warming scenarios.	(Hurtt et al. 2011; Krause et al. 2017)(van Asselen and Verburg 2013; Tasser, Leitinger, and Tappeiner 2017)

A4.2.3 Figures and tables to support the Biodiversity and ecosystem services in the Shared Socio-economic Pathway scenarios Box 4.2.5 in section 4.2

This appendix provides additional information about the outcomes of the BES-SIM model comparison with a particular focus on ecosystem carbon storage in terrestrial systems, which is one of the best studied regulating NCP in the literature. Several models and types of models have been used to project the impacts of future climate change, rising CO_2 concentrations and land use change on ecosystem carbon storage.

The majority of models participating in the BES-SIM model intercomparison found that calculated changes in ecosystem carbon pools (= climate regulation NCP; Figure A2.1) increased globally and regionally. Land-use change continues to be the dominant driver (reducing ecosystem carbon pools) compared to climate change, even when including increasing atmospheric CO_2 concentration (which enhances ecosystem carbon pools) in the climate change simulations (Table A2.2).

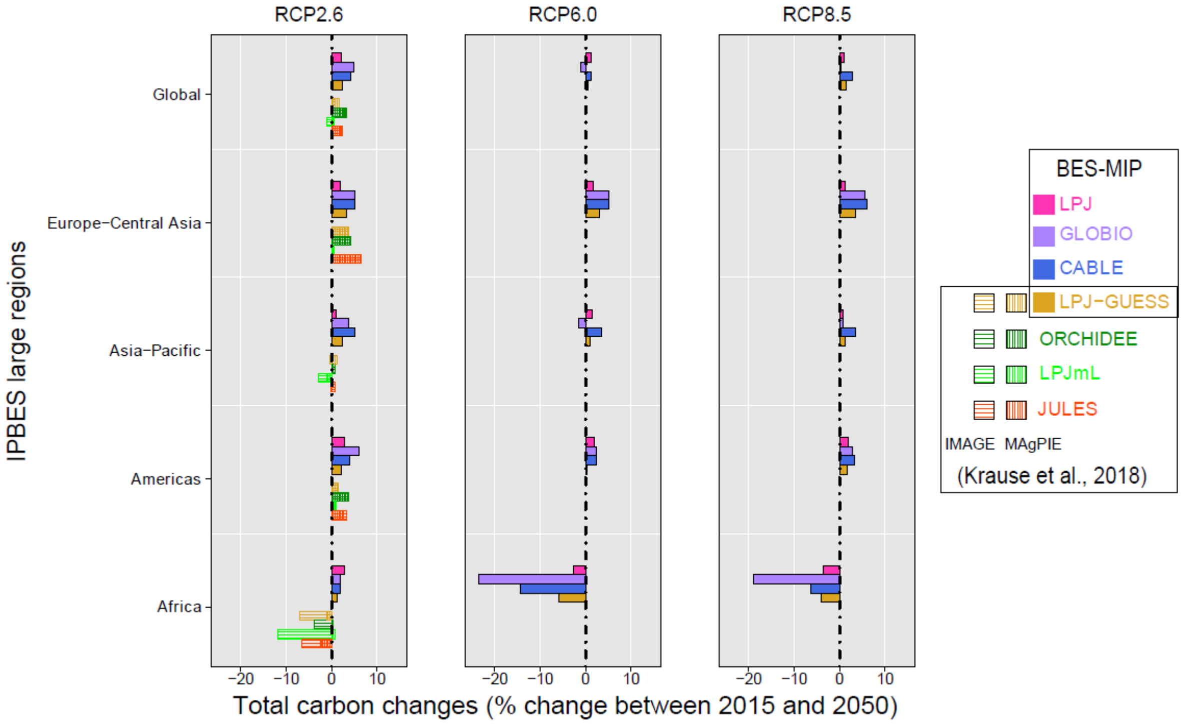


Figure A2.1: Impacts of projected climate and land use change total carbon stocks (% change between 2015 and 2050) for three scnearios: SSP1/RCP2.6 (low GHG emissions, left panel), SSP3/RCP6.0 (high GHG emissions, middle panel) and SSP5/RCP8.5 (very high GHG emissions, right) scenarios in the 5 IPBES regions. Bars indicate the model results for the three scenarios (filled bars, 4 models : LPJ, GLOBIO, CABLE and LPJ-GUESS). BES-SIM results are compared with as similar study carried out by Krause et al. (2018) (hatched bars, 4 models : LPJ-GUESS, ORCHIDEE, LPJmL and JULES) using one integrated assessment model (horizontal hatching, IMAGE) and one land-use model (vertical hatching, MAgPIE) implementing baseline RCP2.6 scenario. Absolute values are shown in Table A4.2.2.

Across Africa, models simulated a C loss — in some cases substantial — for the 'regional competition' and 'economic optimism' scenarios (SSP3/RCP6 and SSP5/RCP8.5 scenarios, respectively) and a small gain for RCP2.6/SSP1. The results for the can be completed by and and compared with the RCP2.6/SSP2 model intercomparison of Krause et al (2018), taking their “baseline” RCP 2.6 land-use change scenario which was provided by the IMAGE and MagPIE land-use change models (van Vuuren et al. 2017)(Doelman et al. 2018). The SSP1 scenario has lower population growth, more environmental awareness, reduced meat consumption and higher livestock efficiency leading to reduced pasture requirements compared to SSP2. In SSP1/RCP2.6, climate policy is added leading to lower deforestation and restoration of degraded forests. This combination leads to reduced agricultural land use similar to other studies where increased agricultural efficiency was implemented in scenarios to achieve afforestation (Krause et al., 2017; LUC4C-ADAFF scenario).

Table A4.2.2: (A) Total projected ecosystem carbon changes (2050-2015, PgC) for the four ecosystem models used in BES-SIM models (rows) calculated for (columns):

- *LU = land use change only (derived from the associated SSP scenario)*
- *CC = climate change + rising CO_2 concentrations only (derived from the associated RCP greenhouse gas emissions scenario), and*
- *LU&CC = combination of both land use and climate change + CO_2 (LU&CC).*

Analyses were done for each of the three SSP/RCP combinations (SSP1/RCP2.6, RCP6.0/SSP3, 8.5) used in the BES-SIM study. Note that the scenarios are referred to by the associated RCP numbers.

(B) Relative contributions of climate change + CO_2, land-use change and present-day CO_2 emissions impact on projected total carbon changes (for example, for the SSP3/RCP6.0 scenario the CABLE model simulates that accounting for land use change dampens total carbon changes by 80% compared to CC only scenario). In all models and scenarios, the contribution of land use (LU) was greater in magnitude than the one of climate change + CO_2 (CC). Contribution of 1) climate change+CO2, 2) Land Use change and 3) Background Climate are calculated as % compared to CC simulations: 1) (LU&CC-CC)/CC, 2) (LU&CC-LU)/CC and 3) (LU&CC – (LU&CC-LU) – (LU&CC-CC))/CC.

A)

GLOBAL Total Carbon changes (2050-2015, PgC)	**LU&CC**			**LU**			**CC**		
	2.6	**6.0**	**8.5**	**2.6**	**6.0**	**8.5**	**2.6**	**6.0**	**8.5**
CABLE	83,1	22,2	55,3	68,4	0,19	17	103,2	110,8	129,8
LPJ-GUESS	42,2	9,7	24,6	32,9	2,15	4,5	68,5	68,3	84,9
LPJ	40,4	23,5	16,8	40,6	20,3	16,2	48,3	55	50,3
GLOBIO	64,1	-14,9	3,9	/	/	/	/	/	/

B)

Contribution	**CC**			**LU**			**Background climate/LU&CC state**		
Compared to CC simulation	**2.6**	**6.0**	**8.5**	**2.6**	**6.0**	**8.5**	**2.6**	**6.0**	**8.5**
CABLE	**14,2%**	**19,9%**	**29,5%**	**-19,5%**	**-80,0%**	**-57,4%**	**85,8%**	**80,1%**	**70,5%**
LPJ-GUESS	**13,6%**	**11,1%**	**23,7%**	**-38,4%**	**-85,8%**	**-71,0%**	**86,4%**	**88,9%**	**76,3%**
LPJ	**-0,4%**	**5,8%**	**1,2%**	**-16,4%**	**-57,3%**	**-66,6%**	**100,4%**	**94,2%**	**98**

Appendix 4.3 – Supporting materials to section 3

n.a.

Appendix 4.4 – Supporting materials to section 4

A4.4.1 Extended information, Figure 4.4.1

Table A4.4.1. GQL Indicators in Archetypes or families of scenarios from previous global environmental assessments

	Business as Usual	**Economic Optimism**	**Reformed Markets**	**Regional competition**	**Global Sustainable Development**	**Regional Sustainability**
Main objective of the Scenario	*Not defined*	*Economic Growth*	*Various goals*	*Security*	*Global sustainability*	*Local sustainability*
MATERIAL GQL						
Food security	**-**	**-**	**+**	**- -**	**++**	**+**
Water security	**-**	**-**	**+**	**- -**	**++**	**+**
Energy security	**-**	**-**	**+**	**- -**	**++**	**+**
Shelter	**-**	**-**	**+**	**- -**	**++**	**+**
Livelihood and income security	**+**	**++**	**++**	**- -**	**+++**	**+**
Health	**-**	**+**	**++**	**- - -**	**+++**	**+**
Non-Material GQL						
Good Social relationships	**-**	**-**	**+**	**- - -**	**+++**	**++**
Equity	**-**	**-**	**+**	**- - -**	**++**	**+**
Cultural identity	**-**	**- -**	**+**	**- - -**	**++**	**+++**
Personal and physical security	**-**	**-**	**+**	**- - -**	**++**	**+**
Freedom of choice and action	**+**	**+**	**+**	**- - -**	**++**	**+**
Knowledge and education	**+**	**+**	**++**	**- -**	**+++**	**+**
Spirituality	**-**	**- -**	**+**	**- - -**	**+++**	**++**
Recreation and leisure	**+**	**+**	**++**	**- -**	**+++**	**++**
Examples						
SSPs	*SSP2*	*SSP5*		*SSP3/SSP4*	*SSP1*	
SRES	*B2*	*A1F1*		*A2*	*B1 (A1T)*	*B2*
GEO3/GEO4		*Market First*	*Policy First*	*Security First*	*Sustainability First*	
Global Scenario Group		*Conventional World*	*Policy Reform*	*Barbarization*	*New Sustainability Paradigm*	*Eco-communalism*
Millennium Eco-System Assessment			*Global Orchestration*	*Order from Strength*	*Techno-garden*	*Adapting Mosaic*

Notes: (+): Increase; (-): reduction

Source: Elaborated by the authors based on IPBES *Deliverable* 3c *(Methodological assessment of scenarios and models of biodiversity and ecosystem services)*, 2016, Chapter 6, Table 6.3

This table shows future global trends for the GQL dimensions based on the narratives of the various archetype scenarios, taken from previous global environmental assessments including IPBES Deliverable 3c (Methodological assessment of scenarios and models of biodiversity and ecosystem services), 2016, Chapter 6, Table 6.3. The positive or negative trends are assigned based on the authors' expert knowledge and their interpretation of the implications the archetype scenarios have on the considered GQL components. The sources for the narratives of the archetype scenarios or families of scenarios and their citations are: Shared Socio-economic Pathways (ONeill 2016, Dellink 2017, Samir 2017), Intergovernmental Panel on Climate Change Special Report on Emissions Scenarios (Nakićenoić and Swart, 2000), Global Environmental Outlook 3 (UNEP 2002), Global Environmental Outlook 4 UNEP 2007), Global Scenario Group (Gallopin 1997, Raskin et al. 1998, Raskin et al. 2002), and Millennium Ecosystem Assessment (MA 2005).

Appendix 4.5: Supporting material section 4.5

Table A4.5.1: number of papers from the systematic literature review that report interactions between models to link drivers, Nature, NCP and GQL components. The information flow can be 1-way (e.g, forcing from Nature to NCP) or 2-ways (with feedbacks).

Link	Mode	Frequency
Nature-NCP	1-way	23
	2-ways	15
Nature-GQL	1-way	7
	2-ways	4
NBP-GQL	1-way	8
	2-ways	3
Drivers-GQL	2-ways	9
Drivers-Nature	2-ways	99
Drivers-NCP	2-ways	24

Table A4.5.2 Typology and some examples of feedbacks across N, NCP and GQL that lead to regime shifts

	Feedbacks	Description	References
Biophysical & Ecosystems	Albedo effect in northern latitudes	Declining snow cover in response to warming results in increased absorption of solar energy that enhances local and regional warming of the surface.	Lenton et al. 2008, Vuille et al. 2008, Lenton 2013, Leadley et al. 2014, Bright et al. 2017; L Perugini et al. 2017
	Evapotranspiration (ET)	Reduced ET due to climate change (or deforestation) feeds back on surface humidity, formation of regional cloud or rainfall (which could also enhance forest vulnerability to fire and drought).	Avissar and Werth 2005, Ray et al. 2006, Lenton 2013, Leadley et al. 2014, (Devaraju, Bala, and Modak 2015; Quesada, Devaraju, et al. 2017; Quesada, Arneth, and de Noblet-Ducoudre 2017)
	Warming feedback from tropical deforestation	Both observational and modelling studies over recent years have demonstrated a clear warming impact from tropical forest deforestation (or from climate-related forest decline) on regional surface temperatures. Declining tropical forests in response to climate change, emphasized by replacing by crops or pastures, results in reduced	Alkama and Cescatti, 2016, Perugini et al., 2017

		evaporative cooling that feeds back to local surface temperature.	
	Greenhouse gases (GHG)	Changes in CO2, temperature and/or precipitation alters the ratio of (vegetation and soil) uptake and release of GHG, which amplifies or dampens the original forcing. Can be enhanced if response include a shift in vegetation dynamics (e.g., forest dieback or reduction of peatlands). Vegetation dieback due to warming or deforestation releases the stored GHG in vegetation and soils into the atmosphere and feeds back to climate change.	Lenton et al. 2008, Davidson et al. 2012, Achard et al. 2014, Pearson et al. 2017, Lenton 2013, Leadley et al. 2014, (Arneth et al. 2010)(Stocker et al. 2013)
	Fire feedback	Fire facilitates the growth of vegetation which is also fire-prone. If e.g., fire frequency is reduced due to atmospheric CO2 (increased shrubiness) local climate change or human intervention, a further reduced spread of fire can result.	Leadley et al. 2010, 2014, Vergara and Scholz 2011, Davidson et al. 2012
Socio-economic	Weath and natural resource extraction	Infrastructure used for extraction and use of natural resources generates wealth, which amplifies technological development and further extraction of resources. As the demand of a natural resource intensifies, its economic value increases. To seek monetary profits, exploitation increases as well. As long as the demand is high economic value and exploitation keeps increasing.	Walker et al. 2009, Leadley et al. 2010, 2014, Cinner et al. 2011
	Instability (limited resources trap)	Instability,conflicts and social and political unrest, leads to increased degradation of natural resources through poor governance, limited resource availability and could amplify instability.	Reuveny 2007, Leadley et al. 2010, Foresight 2011, Leadley et al. 2014
	Gilded Trap (Reinforcing feedback between social and ecological systems)	A social driver like market demand increase the value of natural resources with increasing scarcity of the resource. For instance in coastal ecosystems, large predatory fishes are of high monetary value. Overfishing leads to their depletion, new global markets develop for their prey species in turn (like sea urchins and lobsters) which leads to further depletion of marine resources..	Steneck et al. 2011
	Poverty (marginal resources) traps	Exploitation of natural resources gets aggravated with increasing climate variability and uncertainty. The enhanced rate of exploitation is further enhanced by increasing poverty, thus creating a vicious cycle of poverty and environmental degradation.	Reynolds et al. 2007, Foresight 2011, Leadley et al. 2014
	GQL oriented feedbacks	Changes in value systems and lifestyle, sense of nature and loss of ILK can be side effects of globalization, unawareness and intrusion of market economy that ultimately impacts the GQL which in turn leads to more exploitation of natural resources.	Mbaiwa 2011, Reyes-García et al. 2013, van der Hoeven et al. 2013, Uniyal et al. 2003, Hubacek et al. (2009)

Policy oriented feedbacks	Investment on environmental education influence long term value system/perception of people to appreciate NCP	Muhamad et al. 2014, Ward et al. 2010, Chankrajang and Muttarak 2017, Slavoljub et al 2015
Groundwater Management	High cumulative wealth leads to low institutional compliance, social norm violation, groundwater depletion and income inequality. Pathways to groundwater conservation are controlled by tipping points when compliance becomes highly sensitive to cultural values and enforcement powers. Increasing the number of social rule followers can prevent non-compliant behaviours, amplify the spread of social norms at tipping point and enhance the stability of the state of groundwater conservation. Once compliance is backed up by social norms, little effort in monitoring and enforcement will be required to uphold sustainable conservation targets.	Castilla-Rho et al., 2017
Migration feedback	Migrants, including migrants due to environmental change like storm surge, flood or drought, most likely choose urban or developed areas as their destinations, resulting in (sub)urban sprawl which can increase degradation of the environmental quality of the destination area. With the passage of time the sub(urban) slums turn uninhabitable leading to further migration.	McDonald et al. (2008), Seto et al. (2012)

Table A4.5.3 *summarizes examples for the regime shifts caused by combined multiple/cross-sectional interactions.*

Regime shift	Description	References
Coral reef transformation	Combination of factors like overfishing of herbivorous fish, increased nutrient enrichment, natural hazards like hurricanes and pathogen attacks degrades coral reefs. When coral growth is reduced, there is increased colonization by algae which in turn inhibits coral recruitment.	Hughes 1994, Hoegh-Guldberg et al.2007, Jackson 2008, Jessen et al. 2013, Roth et al. 2015
Intertidal to permanently flooded transition	The surface elevation of a coastal ecosystem cannot keep pace with eustatic sea level rise, and intertidal system transform into permanently flooded system. Sediment and soil accretion and erosion, tectonic movements, strong waves and storm surges, or coastal subsidence can change coastal ecosystems into permanently flooded system.	Gilman et al. 2008, McLeod et al. 2010, Leadley et al. 2014
Long-lasting soil degradation	Excessive grazing, deforestation and poor agricultural practices contribute to severe degradation of soil quality. Loss of soil quality and productivity enhances grazing pressure. Once degraded, recovery is difficult.	Reynolds et al. 2007, Leadley et al. 2014
Marine fisheries collapse	Volume of marine fish stocks collapses when fishing exceeds the threshold limit. Algae and invertebrates dominate the marine ecosystem when marine resources are overexploited	Jackson 2008, Worm et al. 2009, Leadley et al. 2014 , Jessen et al. 2013, Roth et al. 2015

	along with eutrophication and other factors that change the community structure. Restoration is difficult and slow as the system gets severely transformed and degraded.	
Long supply chain of NCP by telecoupling	Off-site alteration of ecosystems (spillover effect) through changes in resources use in another area by telecoupling	Liu 2014, Gasparri et al. 2016, Sun et al. 2017, (Bird, Zanchi, and Pena 2013; Melillo et al. 2009)
Forest transition: Combined socio-ecological feedback and socio-economic dynamics	Forest transition pathways are driven by two forces that influence landuse decisions: 1) a negative socio-ecological feedback that force to slow down or even reversal of land conversion as the flow of good and services from natural ecosystems decline, and 2) exogenous socio-economic dynamics influence by economic modernization, global trade, market access and global awareness about environmental conservation ideas.	Rudel et al. 2005, Lambin and Meyfroidt 2010

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