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The regional assessment report on
BIODIVERSITY AND ECOSYSTEM SERVICES
FOR EUROPE AND CENTRAL ASIA

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THE IPBES REGIONAL ASSESSMENT REPORT ON BIODIVERSITY AND ECOSYSTEM SERVICES FOR EUROPE AND CENTRAL ASIA

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FOREWORD

The objective of IPBES, the Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services, is to provide Governments, the private sector, and civil society with scientifically credible and independent up-to-date assessments of available knowledge to make informed decisions at the local, regional and international levels.

This regional and subregional assessment of biodiversity and ecosystem services for Europe and Central Asia has been carried out by 111 selected authors and 6 early career fellows, assisted by 149 contributing authors, primarily from this region, who have analyzed a large body of knowledge, including about 4750 scientific publications and other knowledge sources. It represents the state of knowledge about the Europe and Central Asia region and subregions. The chapters and their executive summaries were accepted, and the summary for policymakers was approved, by the 129 Member States of IPBES at the sixth session of the IPBES Plenary (18 to 24 March, 2018, Medellín, Colombia).

This report provides a critical assessment of the full range of issues facing decision makers, including the importance, status, trends and threats to biodiversity and nature's contributions to people, as well as policy and management response options. Establishing the underlying causes of the loss of biodiversity and nature's contributions to people provides policymakers with the information needed to develop appropriate response options, technologies, policies, financial incentives and behavior changes.

The assessment concludes that nature's contributions to people are critically important for a good quality of life, but are not evenly experienced by people and communities within the region, and are under threat due to the strong ongoing decline of biodiversity. While sustainability and conservation policies and actions have contributed to reversing some of the negative biodiversity trends, this

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The Regional Assessment Report on Biodiversity and Ecosystem Services for Europe and Central Asia produced by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) provides a critical analysis of the state of knowledge regarding the importance, status, and trends of biodiversity and nature's contributions to people. The assessment analyses the direct and underlying causes for the observed changes in biodiversity and in nature's contributions to people, and the impact that these changes have on the quality of life of people. The assessment, finally, identifies a mix of governance options, policies and management practices that are currently available to reduce the loss of biodiversity and of nature's contributions to people in that region. The assessment addresses terrestrial, freshwater, and coastal biodiversity and covers current status and trends, going back in time several decades, and future projections, with a focus on the 2020-2050 period.

The Summary for Policymakers of this Assessment Report was approved by the sixth session of the Plenary of IPBES (Medellín, Colombia, 18-24 March 2018) and is included in this report. The chapters and their executive summaries were accepted at this same Plenary session. The chapters are available as document IPBES/6/INF/6/Rev.1 (www.ipbes.net).

progress remains insufficient. The assessment also notes the reliance on imports of renewable resources from outside the region.

The major driver of the loss of biodiversity and ecosystem services to date has been land-use change, caused in part by production-based subsidies that led to unsustainable intensification of agricultural practices. However, the assessment notes that the impact of human-induced climate change is increasing and is likely to be one of the most important drivers in the future. The assessment also found that economic growth has, in general, not been decoupled from environmental degradation.

A continuation in past and present trends in the drivers that cause the loss of biodiversity is projected to inhibit the widespread achievement of the Sustainable Development Goals, the Aichi Biodiversity Targets and the Paris Agreement on climate change. Long-term societal transformations that focus on achieving a balanced supply of nature's contributions to people, coupled with participatory decision-making processes, are likely to be the most effective for moving towards a sustainable future.

The assessment identifies a mix of governance options, policies and management practices that is currently available to reduce the loss of biodiversity and nature's contributions to people, but recognizes that further commitment is needed to adopt and implement them. Most important is to include the conservation and sustainable use of biodiversity, and the provision of nature's contributions to people, into all sectoral policies (e.g. agriculture, energy, health, industry, transportation), plans, programmes, strategies and practices - an objective known as "mainstreaming biodiversity".

We would like as Chair and Executive Secretary of IPBES, to recognize the excellent and dedicated work of the co-chairs, Professors Markus Fischer (Switzerland) and Mark Rounsevell (UK and Germany) and of the coordinating lead authors, lead authors, review editors, fellows, contributing authors and reviewers, and to warmly thank them for their commitment, and for contributing their time freely to this important report. We would also like to thank Amor Torre-Marin Rando and André Mader, from the technical support unit located at the University of Bern, Switzerland, as well as Felice van der Plaat, coordinator of the implementation of the regional assessments, because without their dedication this report would not have been possible. We would also like to thank the Government of Switzerland for their generous support. Our thanks also go to members of the IPBES MEP and Bureau who provided guidance as part of the management committee for this report.

This regional assessment provides invaluable information for policymakers in Europe and Central Asia to make informed decisions regarding the conservation and sustainable use of biodiversity, the promotion of access to genetic resources, and the fair and equitable sharing of benefits arising from their use. It also provides valuable information for the ongoing IPBES global assessment, to be released in May 2019 and is expected to inform discussions regarding the post-2020 global biodiversity framework under the Convention on Biological Diversity, as well as to inform action on implementing the 2030 Agenda for Sustainable Development and the Sustainable Development Goals.

Sir Robert T. Watson
Chair of IPBES

Anne Larigauderie
Executive Secretary of IPBES

STATEMENTS FROM KEY PARTNERS

The Sustainable Development Goals aim to "leave no one behind". If we don't protect and value biodiversity, we will never achieve this goal. When we erode biodiversity, we impact food, water, forests and livelihoods. But to tackle any challenge head on, we need to get the science right and this is why UN Environment is proud to support this series of assessments. Investing in the science of biodiversity and indigenous knowledge, means investing in people and the future we want.

Erik Solheim

Executive Director,
United Nations Environment Programme
(UNEP)

Biodiversity is the living fabric of our planet - the source of our present and our future. It is essential to helping us all adapt to the changes we face over the coming years. UNESCO, both as a UN partner of IPBES and as the host of the IPBES Technical Support Unit on Indigenous and Local Knowledge, has always been committed to supporting harmony between people and nature through its programmes and networks. These four regional reports are critical to understanding the role of human activities in biodiversity loss and its conservation, and our capacity to collectively implementing solutions to address the challenges ahead.

Audrey Azoulay

Director-General,
United Nations Educational,
Scientific and Cultural Organization (UNESCO)

"The regional assessments demonstrate once again that biodiversity is among the earth's most important resources. Biodiversity is also key to food security and nutrition. The maintenance of biological diversity is important for food production and for the conservation of the ecological foundations on which rural livelihoods depend. Biodiversity is under serious threat in many regions of the world and it is time for policy-makers to take action at national, regional and global levels."

José Graziano da Silva
Director-General,
Food and Agriculture Organization of the
United Nations (FAO)

"Tools like these four regional assessments provide scientific evidence for better decision making and a path we can take forward to achieve the Sustainable Development Goals and harness nature's power for our collective sustainable future. The world has lost over 130 million hectares of rainforests since 1990 and we lose dozens of species every day, pushing the Earth's ecological system to its limit. Biodiversity and the ecosystem services it supports are not only the foundation for our life on Earth, but critical to the livelihoods and well-being of people everywhere."

Achim Steiner
Administrator,
United Nations Development Programme
(UNDP)

ACKNOWLEDGEMENTS

Three years of intense work have resulted in the IPBES Regional Assessment of Biodiversity and Ecosystem Services for Europe and Central Asia. This has been a pressured, but also highly enjoyable process thanks to everyone who has contributed to this important collective effort. As Co-Chairs we were privileged to work with so many outstanding experts from Europe and Central Asia. We warmly acknowledge the first-rate intellectual contributions and immense investment of time by all of our authors, review editors, and early-career fellows, as well as the various types of support from their respective funding organizations, institutions and Governments, which facilitated their participation. This Assessment would also not have been possible without the generous commitment of several other key individuals. We would first like to recognize the invaluable contributions of the two members of our Technical Support Unit (TSU), Amor Torre-Marin Rando and André Mader, whose enthusiasm, professionalism and dedication facilitated every aspect of the process. We also thank Eva Spehn of the Swiss Biodiversity Forum for helping to get us started by acting as a provisional TSU. We thank the IPBES Chair, Bob Watson, and the IPBES Executive Secretary, Anne Larigauderie, for their wise guidance and enormous support to the assessment team, with the other members of our assessment management committee, Senka Barudanovic, Ruslan Novitsky, Marie Stenseke as well as Felice van der Plaat, from the IPBES secretariat, and other members of the IPBES Multidisciplinary Expert Panel (MEP) and Bureau. We thank the IPBES secretariat as a whole for excellent support and collaboration throughout the process.

In addition, we thank all expert reviewers from Governments, stakeholders and the scientific community for their time, perspectives and valuable comments, which markedly improved the quality of both the Summary for Policymakers (SPM) and the chapters. On the technical side, we also thank our data visualization experts, Yuka Estrada and Maro Haas and team, for their work on the assessment figures and diagrams, and Mark Snethlage for his many contributions to data analysis and visualisation.

Our sincere appreciation is extended to the Swiss Government's Federal Office for the Environment for funding the TSU and to the University of Bern for hosting it. We also thank the Swiss Federal Office for the Environment for co-funding our lead author meetings in Engelberg (Switzerland), Zadar (Croatia) and Prague (Czech Republic). We would further like to thank the Indigenous and Local Knowledge TSU of IPBES for organizing the "Europe and Central Asia Dialogue Workshop on Indigenous and Local Knowledge" in Paris (France) in 2016 and the IPBES Capacity Building TSU who supported the "Capacity Development Writing Workshop for IPBES Experts from Central Europe, Eastern Europe and Central Asia" held in Antalya (Turkey), in 2016 and a regional dialogue meeting in Vácrátót (Hungary) in 2017. We are also very grateful to the many publishers, too numerous to list here, who freely provided figures.

We acknowledge and appreciate the members and observers of the IPBES Plenary, whose suggestions during the 6th session of the Plenary, held in Medellín, Colombia, in March 2018 led to significantly increased clarity and accessibility of the SPM. Finally, we express our sincere gratitude to Bureau members Ivar Baste and Senka Barudanovic for chairing the contact group at the 6th session of the IPBES Plenary with extraordinary professionalism, patience and sensitivity.

As we deliver the important results of this work, we are confident that the investment of so much time, passion, expertise and resources will pay exceptional dividends – informing better policies, decisions and action to protect the invaluable natural assets of our beautiful region for all the people of Europe and Central Asia.

Mark D.A. Rounsevell
Co-Chair

Markus Fischer
Co-Chair

PREFACE

WHAT IS AN ASSESSMENT?

An assessment is a critical evaluation of information, to inform decisions on a complex, public issue (MEA, 2005). An assessment does not generate new data, but seeks to create new understanding through summary, sorting and synthesis using different methods to manage complexity. It includes academic and grey literature, as well as insights from indigenous and local knowledge (ILK).

The IPBES Regional Assessment for Europe and Central Asia was conducted by a group of experts with a broad range of knowledge and skills, most of whom were nominated by Governments, and the remainder by organizations. The Assessment is supported by evidence, not based on advocacy, and relates to a particular time period (usually 1950-2050, but earlier or later where appropriate) and to the geographical domain of Europe and Central Asia.

THE IPBES CONTEXT FOR THE REGIONAL ASSESSMENT FOR EUROPE AND CENTRAL ASIA

Objective 2(b) of the IPBES work programme is to "strengthen the science-policy interface on biodiversity and ecosystem services at and across subregional, regional and global levels by producing "regional/subregional assessments on biodiversity and ecosystem services" for: Africa, the Americas, Asia-Pacific, and Europe and Central Asia (Decision IPBES-3/1: Work programme for the period 2014–2018: Annex IV-VII). The implementation of the Regional Assessment for Europe and Central Asia followed a scoping study that responded to requests by Governments, multilateral environmental agreements and other stakeholders in the formulation of key policy questions. These policy questions included: a) general questions relevant to all IPBES regional assessments and, b) questions specific to the Europe and Central Asia region. The scoping study resulted in a generic scoping report (Decision IPBES-3/1: Work programme for the period 2014–2018, Annex III: Generic scoping report for the regional and subregional assessments of biodiversity and ecosystem services (deliverable 2 (b))) and scoping reports for each of the four regions, which have guided the

implementation of the Regional Assessment for Europe and Central Asia according to the timetable outlined in **Figure 1**. The IPBES Plenary approved the summary for policymakers, and accepted the chapters of the Assessment Report, at its sixth session in March 2018.

Each of the four regional IPBES assessments share the same generic policy questions and follow the same chapter structure, which maps onto the IPBES conceptual framework. All regional assessments also integrate relevant aspects of the IPBES thematic and methodological assessments (outlined below) and consider trans-regional teleconnections in nature, nature's contributions to people[1] and good quality of life, and in the underlying drivers. While the Regional Assessment for Europe and Central Asia focuses on the regional scale, it also considers subregional or finer scales where necessary. Many examples of drivers, biodiversity, ecosystem services, and good quality of life concern national to local scales. Moreover, the local scale often offers the best opportunity for the integration of indigenous and local knowledge and other knowledge systems. Thus, the general coarse-scale focus of this assessment is rooted in a synthesis of information across a range of scales from local to the Europe and Central Asia region as a whole. The outcomes of the regional assessments are stand-alone products that also inform the IPBES Global Assessment on Biodiversity and Ecosystem Services (deliverable 2c).

THE ASSESSMENT PROCESS

The Regional Assessment for Europe and Central Asia was undertaken by an expert team of 118 individuals comprising two assessment co-chairs, a further 12 coordinating lead authors, 85 lead authors, six fellows and 13 review editors. The experts were selected in 2015 by the co-chairs; representatives of the IPBES Bureau and multidisciplinary expert panel (MEP) from the region; and the IPBES secretariat, from nominations by Governments and organizations, to cover a spectrum of disciplines including indigenous and local knowledge. The selected expert

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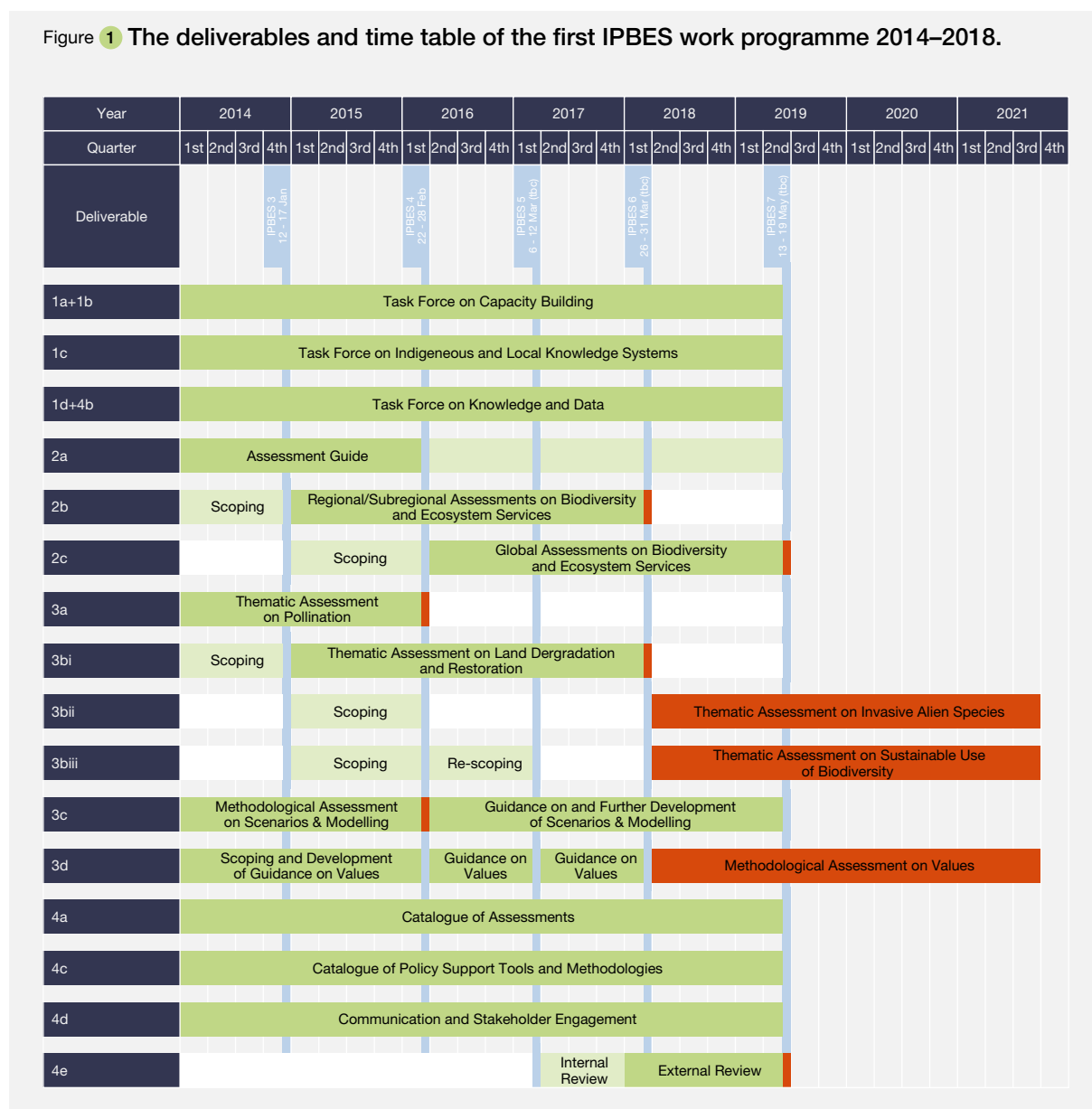1. Nature's contributions to people encompass the positive contributions, or benefits, and occasionally negative contributions, losses or detriments, that people obtain from nature. The term resonates with the original use of the term ecosystem services in the Millenium Ecosystem Assessment (MEA, 2005), and goes further by explicitly embracing concepts associated with other worldviews on human–nature relations and knowledge systems.

team was supported by numerous contributing authors. The 13 review editors assessed the adequacy of author responses to reviewer comments. The evidence presented in the assessment was derived from the peer-reviewed and publicly available literature or correctly cited and publicly available grey literature, as well as indigenous and local knowledge (Roué and Molnár, 2016).

Implementation of the assessment followed eight procedural steps. A first draft of the report chapters was prepared by the author team (1). This draft was peer reviewed in an open and transparent process by Governments, other stakeholders and all interested experts who responded to an invitation by the IPBES Executive Secretary by registering and submitting review comments (2). This facilitated

stakeholder engagement and provided a broad set of comments through which the assessment's legitimacy was enhanced. A second draft of the report chapters and first draft summary for policymakers (SPM) were prepared by the author team under the guidance of the review editors and the multidisciplinary expert panel, considering comments from the review (3). These two documents were reviewed a second time by Governments, and other stakeholders (4), leading to the preparation of the final draft of the report chapters and summary for policymakers by the author team under the guidance of the review editors and the multidisciplinary expert panel (5). The summary for policymakers was then translated into the six official languages of the United Nations, checked for accuracy by the author team, and prepared in formats suitable for

Figure 1 **The deliverables and time table of the first IPBES work programme 2014–2018.**

Year	2014				2015				2016				2017				2018				2019				2020				2021			
Quarter	1st	2nd	3rd	4th	1st	2nd	3rd	4th	1st	2nd	3rd	4th	1st	2nd	3rd	4th	1st	2nd	3rd	4th	1st	2nd	3rd	4th	1st	2nd	3rd	4th	1st	2nd	3rd	4th
Deliverable				IPBES 3 12 - 17 Jan				IPBES 4 22 - 28 Feb				IPBES 5 6 - 12 Mar (tbc)				IPBES 6 26 - 31 Mar (tbc)				IPBES 7 13 - 19 May (tbc)												
1a+1b	Task Force on Capacity Building																															
1c	Task Force on Indigeneous and Local Knowledge Systems																															
1d+4b	Task Force on Knowledge and Data																															
2a	Assessment Guide																															
2b	Scoping				Regional/Subregional Assessments on Biodiversity and Ecosystem Services																											
2c					Scoping				Global Assessments on Biodiversity and Ecosystem Services																							
3a	Thematic Assessment on Pollination																															
3bi	Scoping				Thematic Assessment on Land Dergradation and Restoration																											
3bii					Scoping												Thematic Assessment on Invasive Alien Species															
3biii					Scoping				Re-scoping								Thematic Assessment on Sustainable Use of Biodiversity															
3c	Methodological Assessment on Scenarios & Modelling				Guidance on and Further Development of Scenarios & Modelling																											
3d	Scoping and Development of Guidance on Values				Guidance on Values				Guidance on Values				Methodological Assessment on Values																		
4a	Catalogue of Assessments																															
4c	Catalogue of Policy Support Tools and Methodologies																															
4d	Communication and Stakeholder Engagement																															
4e													Internal Review				External Review															

indigenous and local knowledge holders (6). The final draft of the report chapters and summary for policymakers were made available to, and reviewed by, Governments who provided written comments (7), culminating in the review and approval of the summary for policymakers, and the acceptance of the report chapters at the 6th session of the IPBES Plenary in Medellín in March 2018 (8).

THE RELATIONSHIP BETWEEN THE REGIONAL ASSESSMENT FOR EUROPE AND CENTRAL ASIA AND THE OTHER IPBES ASSESSMENTS

Besides the four regional assessments, the IPBES work programme (see **Figure 1**) encompasses completed or ongoing assessments including the Thematic Assessment on Pollinators, Pollination and Food Production; the Methodological Assessment on Scenarios and Models of Biodiversity and Ecosystem Services; the Thematic Assessment on Land Degradation and Restoration; and the Global Assessment on Biodiversity and Ecosystem Services. These assessments report at the regional to global scales, and also provide important case studies at the landscape scale. The summary for policymakers of the Thematic Assessment on Pollinators, Pollination and Food Production, was approved and its chapters accepted at the 4th meeting of the Plenary of IPBES in 2016 (IPBES, 2016). It assessed the role of, and status and trends in, animal pollinators and pollination networks and changes in pollination that underpins food production. The assessment informs policy responses to declines and deficits in pollination and contributes to Aichi Biodiversity Target 14 on safeguarding and restoring ecosystems that provide essential contributions to people. The Thematic Assessment on Land Degradation and Restoration provides information to support decision-makers in reducing the negative environmental, social and economic consequences of land degradation, and in restoring degraded land to enhance nature's contributions to people. The assessment identifies areas of concern and the potential solutions to the challenges posed by land degradation as well as highlighting critical knowledge gaps and priority areas for research and investment (Decision IPBES-3/1, Annex VIII: Scoping for a thematic assessment of land degradation and restoration (deliverable 3 (b) (i))). The IPBES Global Assessment on Biodiversity and Ecosystem Services synthesizes evidence based on biodiversity and nature's contributions to people (IPBES/4/INF/9: Guide on the production and integration of assessments from and across all scales (deliverable 2 (a))) from across the Earth. It is based strongly on the outcomes of the four regional and subregional assessments, but also reports on literature that uses methods applied at the global scale. Where appropriate and necessary, information

elaborated during the progress toward these other IPBES assessments also contributed to the Regional Assessment for Europe and Central Asia.

THE POLICY CONTEXT

Almost all countries in Europe and Central Asia use (agreed-upon) relevant international frameworks to guide national strategy and action. The IPBES assessment of Europe and Central Asia was, hence, and as requested by the scoping document, undertaken in the context of the United Nations 2030 Agenda for Sustainable Development and Strategic Plan for Biodiversity 2011–2020. The assessment examines progress towards the Sustainable Development Goals of the 2030 Agenda and the Aichi Biodiversity Targets of the Strategic Plan. Its time frame covers current and projected trends corresponding with timelines for the 2030 Agenda (2030), and the Strategic Plan (2020) and its 2050 vision. The Strategic Plan for Biodiversity 2011-2020, in particular, exists in the broader context of the United Nations Decade on Biodiversity. In 2010, the Parties to the Convention on Biological Diversity invited the United Nations General Assembly to consider declaring 2011-2020 the United Nations Decade on Biodiversity (CBD, 2010), which the General Assembly did in a resolution in the same year, "with a view to contributing to the implementation of the Strategic Plan for Biodiversity for the period 2011-2020" (United Nations, 2011).

THE INVOLVEMENT OF DIFFERENT STAKEHOLDERS

Stakeholders can be considered in two groups, based on how they engaged with the Regional Assessment of Europe and Central Asia (Decision IPBES-3/4: Communications, stakeholder engagement and strategic partnership, Annex II: Stakeholder engagement strategy (deliverable 4 (d))):

1. *Contributors* - scientists, knowledge holders including indigenous and local knowledge holders, practitioners and others;

2. *End users* - regional governments, national Governments and multilateral environmental agreements, subnational and local governments, United Nations agencies, inter-governmental organizations, non-governmental organizations (NGOs), other practitioners within the private sector and the public.

As stakeholder engagement is an important element for the relevance, effectiveness, credibility and overall success of IPBES, stakeholder values, needs and concerns were embedded within the assessment process from the start. The Regional Assessment for Europe and Central Asia has engaged with the broader stakeholder community to better understand and communicate the causes of the loss of nature and nature's contributions to people, including the role of humans and the consequences for human well-being. Involving stakeholders is important in recognising their diverse conceptualisations of values, adding societal aspects when assessing drivers and scenarios and evaluating policy support tools. Although different stakeholders may have different priorities, they all aim to have their knowledge, views and values considered within the IPBES process, including its assessments. Stakeholders can bring complementary perspectives to those of Government, which also helps to identify and prioritize the most relevant knowledge gaps. Different stakeholder organizations can play an important role in the engagement of IPBES with different sectors of society. For these reasons, IPBES provides an opportunity for stakeholders to contribute to informing decision-making.

REFERENCES

CBD. (2010). *Decision X/8. United Nations decade on biodiversity 2011-2020.*

IPBES. (2016). *Summary for policymakers of the assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production*. S. G. Potts, V. L. Imperatriz-Fonseca, H. T. Ngo, J. C. Biesmeijer, T. D. Breeze, L. V. Dicks, L. A. Garibaldi, R. Hill, J. Settele, A. J. Vanbergen, M. A. Aizen, S. A. Cunningham, C. Eardley, B. M. Freitas, N. Gallai, P. G. Kevan, A. Kovács-Hostyánszki,

P. K. Kwapong, J. Li, X. Li, D. J. Martins, G. Nates-Parra, J. S. Pettis, R. Rader, & B. F. Viana (Eds.). Bonn, Germany: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

MEA. (2005). *Ecosystems and human well-being: Current state and trends, Volume 1.* Washington DC, USA: Island Press.

Roué, M., & Molnar, Z. (Eds.). (2016). *Knowing our lands and resources: Indigenous and local knowledge of*

biodiversity and ecosystem services in Europe and Central Asia. Paris, France: UNESCO.

United Nations. (2011). *Resolution 65/161. Convention on Biological Diversity*.

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SUMMARY FOR POLICYMAKERS

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KEY MESSAGES

A. A PRECIOUS ASSET: NATURE AND ITS CONTRIBUTIONS TO PEOPLE'S QUALITY OF LIFE IN EUROPE AND CENTRAL ASIA

Nature's contributions to people, which embody ecosystem services, are critically important for livelihoods, economies and a good quality of life, and are therefore vital to sustaining human life on earth. Nature has considerable economic and cultural values for societies. Nature also benefits, for example, human health through its role in medicines, the provision of food for varied diets and support to mental and physical health through green spaces. The knowledge and customary practices of indigenous peoples and local communities also enhance people's quality of life by fostering cultural heritage and identity. In Europe and Central Asia, which has an area of 31 million square kilometres, the regulation of freshwater quality has a median value of $1,965 per hectare per year. Other important regulating services include habitat maintenance ($765 per hectare per year); the regulation of climate ($464 per hectare per year); and the regulation of air quality ($289 per hectare per year).

Nature's contributions to people are under threat due to the continuing loss of biodiversity. Sustaining nature's contributions to people requires the maintenance of high levels of biodiversity. The continuing decline in biodiversity has had negative consequences for the delivery of many ecosystem services over the last decades. These include habitat maintenance, pollination, regulation of freshwater quantity and quality, soil formation and regulation of floods. These declines have occurred in part because of the intensive agriculture and forestry practices used to increase the provision of food and biomass-based fuels.

The region of Europe and Central Asia partially relies on net imports of renewable resources from outside the region. The population of Europe and Central Asia consumes more renewable natural resources than are produced within the region in spite of the increase since the 1960s in the production of food and biomass-based fuels. Central and Western Europe depends on food and feed imports equivalent to the annual harvest of 35 million hectares of cropland (2008 data), a land area the size of Germany.

Across Europe and Central Asia, nature's contributions are not evenly experienced by people and communities. In Europe and Central Asia, a combination of food provision and imports means that the region is currently food secure but, in some areas of Central Asia and Central and Eastern Europe, food security is threatened by exports arising from large-scale land acquisitions mainly by entities from both Western Europe and outside the region. Water security, which relies partially on nature's regulation of water quality and quantity, also varies across the region, with 15 per cent of people in Central Asia lacking access to safe drinking water. The decline of indigenous and local knowledge has negatively impacted on the heritage and identity of indigenous peoples and local communities.

B. THE BIODIVERSITY OF EUROPE AND CENTRAL ASIA IS UNIQUE BUT THREATENED

The biodiversity of Europe and Central Asia is in continuous strong decline. The extent of natural ecosystems has declined, e.g., wetland extent has declined by 50 per cent since 1970 and natural and semi-natural grasslands, peatlands and coastal marine habitats have been degraded. Ecosystems have considerably declined in terms of species diversity. Of the assessed species living exclusively in Europe and Central Asia, 28 per cent are threatened. Among all the assessed groups of species living in the region, particularly threatened are mosses and liverworts (50 per cent), freshwater fish (37 per cent), freshwater snails (45 per cent), vascular plants (33 per cent) and amphibians (23 per cent). Landscapes and seascapes have become more uniform in their species composition and thus their diversity has declined.

In recent years, national and international sustainability and conservation policies and actions have contributed to reversing some negative biodiversity trends. More sustainable management of fisheries and reduction of eutrophication has led to an increase in some fish stocks in areas such as the North Sea. Endangered habitats, such as Macaronesian woodlands, and species such as the Iberian lynx and European bison, have recovered substantially because of targeted conservation efforts.

Overall, progress towards healthy ecosystems is still insufficient. While some progress has been made in improving the status of biodiversity by safeguarding
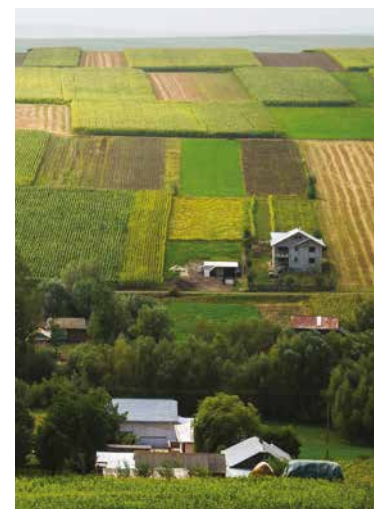
ecosystems, species and genetic diversity, biodiversity status and trends remain negative overall. Increasing conservation efforts and the sustainability of the use of biodiversity would enhance the chances of meeting national and international biodiversity targets.

C. DRIVERS OF CHANGE IN BIODIVERSITY AND NATURE'S CONTRIBUTIONS TO PEOPLE IN EUROPE AND CENTRAL ASIA

Land-use change is the major direct driver of the loss of both biodiversity and ecosystem services in Europe and Central Asia. Production-based subsidies have led to intensification in agriculture and forestry, and, together with urban development, have led to biodiversity decline. Increasing intensity often impinges on traditional land use. Ceasing traditional land use has reduced semi-natural habitats of high conservation value and associated indigenous and local knowledge, practices and culture across the region. Although protected areas have expanded in the region, protected areas alone cannot prevent biodiversity loss. Only where protected areas are managed effectively can they contribute to the prevention of biodiversity loss.

The impact of climate change on biodiversity and nature's contributions to people is increasing rapidly and is likely to be one of the most important drivers in the future. Trends in natural resource extraction, pollution and invasive alien species have led to considerable declines in biodiversity and ecosystem services, and are likely to continue to pose considerable threats, particularly in combination with climate change. Natural resource extraction is still a major pressure on biodiversity. Furthermore, despite effective regulations, pollution continues to pose a major threat to biodiversity and human health. Invasive alien species have increased in number – for all taxonomic groups across all the subregions of Europe and Central Asia – and this has severe effects on biodiversity and ecosystem services. The individual and combined effects of all the direct drivers have chronic, prolonged and delayed consequences for biodiversity and the provision of nature's contributions to people owing to considerable time-lags in the response of ecological systems.

Economic growth is generally not decoupled from environmental degradation. This decoupling would require a transformation in policies and tax reforms across the region. Economic growth, as measured through traditional gross domestic product (GDP), across Europe and Central Asia has indirectly reinforced drivers of biodiversity loss, which in turn has reduced nature's

contributions to people. Across the region, a range of policies, including environmental taxation, have been implemented to decouple economic growth from detrimental drivers. Furthermore, there still exist policy instruments, such as harmful agricultural and fishing subsidies, which continue to impede transitions towards a sustainable future. Decoupling would be assisted by new indicators that incorporate well-being, environmental quality, employment and equity, biodiversity conservation and nature's ability to contribute to people.

D. FUTURES FOR EUROPE AND CENTRAL ASIA

The continuation of past and present trends in drivers to, and beyond, 2030 (as represented in business-as-usual scenarios) will inhibit the widespread achievement of goals similar to and including the Sustainable Development Goals. Future scenarios that focus on achieving a balanced supply of nature's contributions to people and that incorporate a diversity of values are more likely to achieve the majority of such goals. Trade-offs are indicated between different ecosystem services under different future scenarios for Europe and Central Asia. Ways of resolving these trade-offs depend on political and societal value judgements. Scenarios that include proactive decision-making on environmental issues, environmental management approaches that support multifunctionality, and mainstreaming environmental issues across sectors, are generally more successful in mitigating trade-offs than isolated environmental policies. Scenarios that include cooperation between countries or regions are expected to be more effective in mitigating undesirable cross-scale impacts on biodiversity and ecosystem services.

Long-term societal transformation through continuous education, knowledge-sharing and participatory decision-making characterize the most effective pathways for moving towards sustainable futures. These pathways promote resource-sparing lifestyles and emphasize community actions and voluntary agreements supported by social and information-based instruments as well as rights-based approaches. They support regulating ecosystem services and highlight a diverse range of values in comprehensively considering biodiversity and nature's contributions to people across sectors, and across spatial and temporal scales. Other actions, such as technological innovation, ecosystem-based approaches, land sparing or land sharing, could support and pave the way for these more transformational solutions.

E. PROMISING GOVERNANCE OPTIONS FOR EUROPE AND CENTRAL ASIA

A mix of governance options, policies and management practices is available for public and private actors in Europe and Central Asia, but further commitment is needed to adopt and effectively implement them to address the drivers of change, to safeguard biodiversity and to ensure nature's contributions to people for a good quality of life. Well-designed, context-specific mixes of policy instruments building on, for example, ecosystem-based approaches, have been effective in the governance of biodiversity and nature's contributions to people. While legal and regulatory instruments are the backbone of policy mixes, economic, financial, social and information-based instruments provide additional incentives to trigger behaviour change. Developing rights-based instruments would fully integrate the fundamental principles of good governance, equalizing power relations and facilitating capacity-building for indigenous peoples and local communities. The mobilization of sufficient financial resources would strengthen institutional capacities to support research, training, capacity-building, education and monitoring activities. The removal of harmful subsidies in various sectoral policies, such as agriculture, fisheries and energy, in Europe and Central Asia, reduces negative impacts on biodiversity and allows for a more cost-effective use of public funds.

Mainstreaming the conservation and sustainable use of biodiversity and the sustained provision of nature's contributions to people into all sectoral policies, plans, programmes, strategies and practices could be achieved with more proactive, focused and goal-oriented approaches to environmental action. Partial progress has been made in tackling the underlying drivers of biodiversity loss, by mainstreaming across government and society. Mainstreaming could be harnessed in a three-step process by: first, raising awareness of the dependence of good quality of life on biodiversity; second, defining policy objectives concerning the ecological, economic and sociocultural needs for achieving sustainable development; and, third, designing instruments and policy mixes to support the implementation of effective, efficient and equitable policy and decision-making for nature and a good quality of life.

Better integration across sectors to coordinate biodiversity governance and the sustainable delivery of nature's contributions to people would avoid negative outcomes for nature and people. Improved coordination would enable better consideration of biodiversity and ecosystem services, taking trade-

offs between different policy and economic sectors into account. There is, for example, ample room for further exploiting this potential for the agriculture, forestry and fisheries sectors and urban planning. Regarding an economy-wide perspective, this includes measuring national welfare beyond current economic indicators that take account of the diverse values of nature. Ecological fiscal reforms would provide integrated incentives and provide leverage to redirect activities that support sustainable development.

Increasing participation and stakeholder involvement will help to integrate various forms of knowledge in policymaking and decision-making while promoting shared responsibility. The importance of the effective involvement of different actors is recognized in Western and Central Europe and increasingly also in Eastern Europe and Central Asia. This involvement can be strengthened by careful monitoring and evaluation, taking various values into consideration, including those of indigenous peoples and local communities.

Box SPM 1 **Region of Europe and Central Asia.**

Table SPM 1 **Subregions and countries of Europe and Central Asia according to decision IPBES-3/1, annex VII.**

SUBREGION	COUNTRIES
WESTERN EUROPE	Andorra, Austria, Belgium, Denmark, Finland, France, Germany, Greece, Iceland, Ireland, Israel, Italy, Liechtenstein, Luxembourg, Malta, Monaco, Netherlands, Norway, Portugal, San Marino, Spain, Sweden, Switzerland, United Kingdom of Great Britain and Northern Ireland
CENTRAL EUROPE	Albania, Bosnia and Herzegovina, Bulgaria, Croatia, Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Montenegro, Poland, Romania, Serbia, Slovakia, Slovenia, the former Yugoslav Republic of Macedonia, Turkey
EASTERN EUROPE	Armenia, Azerbaijan, Belarus, Georgia, Republic of Moldova, Russian Federation, Ukraine
CENTRAL ASIA	Kazakhstan, Kyrgyzstan, Tajikistan, Turkmenistan, Uzbekistan

Box SPM **1** **Region of Europe and Central Asia.**

The Europe and Central Asia region encompasses 54 countries **(Table SPM.1)** in four subregions **(Figure SPM.1)**. These countries vary greatly in size, including the largest and smallest on Earth, and have diverse governance structures, cultures, economies, ecoregions and sectors. The seas of the region are heterogeneous in terms of temperatures, currents, nutrient availability, depths and mixing regimes. There are great differences in data monitoring and availability across the region.

Figure SPM **1** **Region of Europe and Central Asia with the four IPBES subregions and regional oceans and seas.**

ECA SUBREGIONS AND SEAS

0   500 1,000   2,000   3,000 km

Projection: North Asia Lambert Conformal Conic
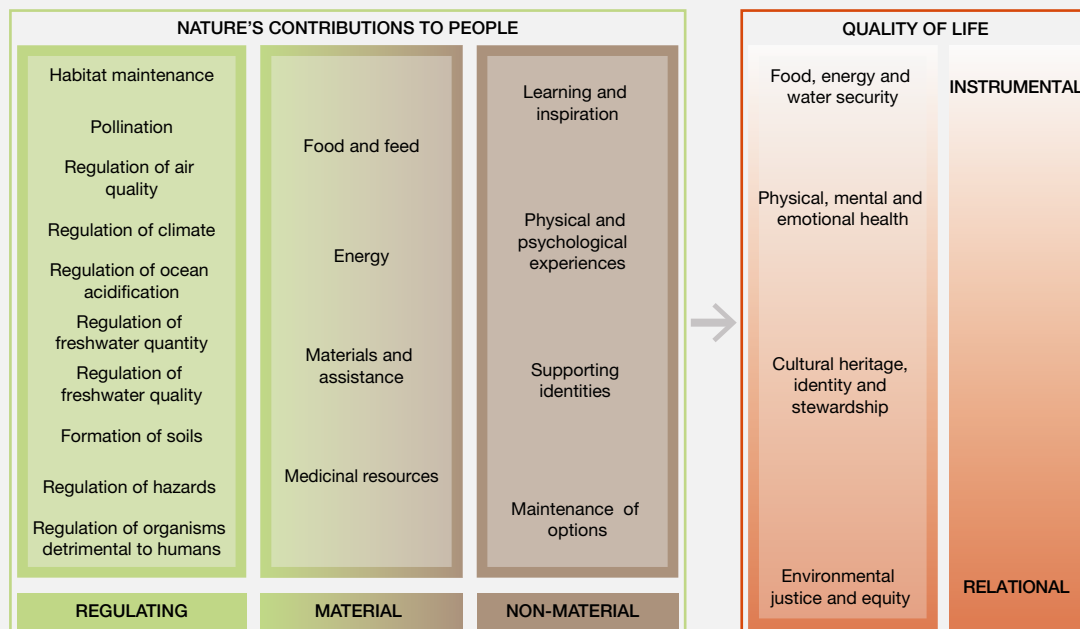Source: Natural Earth www.naturalearthdata.org

Box SPM ② **Nature's contributions to people.**

The regional assessment for Europe and Central Asia considers ecosystem services through the lens of nature's contributions to people (see appendix 2), which embodies both the scientific concept of ecosystem goods and services, and the notion of nature's gifts from indigenous and local knowledge systems. Nature's contributions can be beneficial or detrimental to people, depending on the cultural context, and are assessed from two complementary perspectives: one generalizing in nature and the other context-specific. The generalizing perspective includes 18 categories organized into three partially overlapping groups: regulating, material and non-material

contributions **(Figure SPM.2)** {2.1.1}. The context-specific perspective includes geographical and cultural aspects of indigenous and local knowledge systems. The grading of green and brown colours in **Figure SPM.2** indicates whether nature's contributions to people are associated more with natural or with cultural systems. Instrumental values refer to the value attributed to something as a means to achieve a particular end. Relational values are positive values assigned to "desirable relationships", such as those among people and between people and nature.

Figure SPM ② **Nature's contributions to people and their relation to quality of life in terms of instrumental and relational values.**

NATURE'S CONTRIBUTIONS TO PEOPLE			QUALITY OF LIFE	
Habitat maintenance	Food and feed	Learning and inspiration	Food, energy and water security	INSTRUMENTAL
Pollination				
Regulation of air quality			Physical, mental and emotional health	
Regulation of climate	Energy	Physical and psychological experiences		
Regulation of ocean acidification				
Regulation of freshwater quantity			Cultural heritage, identity and stewardship	
Regulation of freshwater quality	Materials and assistance	Supporting identities		
Formation of soils				
Regulation of hazards	Medicinal resources			
Regulation of organisms detrimental to humans		Maintenance of options	Environmental justice and equity	RELATIONAL
REGULATING	**MATERIAL**	**NON-MATERIAL**		

BACKGROUND

A. Nature and its contributions to people's quality of life in Europe and Central Asia

A1 **Nature provides valuable material (e.g., food), regulating (e.g., climate regulation and pollination) and non-material contributions to people (e.g., learning and inspiration) (Figure SPM.2). These contributions are essential for people's quality of life as they have substantial economic, social and cultural values** *(well established)*[3] **{2.3.5}.**

The highest valued regulating contributions to people in Europe and Central Asia include: the regulation of freshwater and coastal water quality (estimated to have a median value of $1965[4] per hectare per year) *(established but incomplete)*; habitat maintenance ($765 per hectare per year) *(unresolved)*; the regulation of climate ($464 per hectare per year); and the regulation of air quality ($289 per hectare per year) *(established but incomplete)* {2.3.5.2}. Monetary values for regulating contributions to people, however, are site-specific and vary significantly across the Europe and Central Asia region depending on location, habitat, extent of contribution and valuation method used.

Nature's material contributions to people have important values that are partly reflected in conventional market prices. Agricultural production across the 28 member States of the European Union generates profits ranging from $233 per hectare per year (cereals) to $916 per hectare per year (mixed crops), while wood supply from forests generates profits of $255 per hectare per year {2.3.5.1}.

Nature's non-material contributions to people, which include physical and psychological experiences linked to tourism and recreation, are estimated to have a median monetary value of $1,117 per hectare per year *(unresolved)* {2.3.5.2}. Other non-material contributions, such as cultural heritage and identity, may be valued using non-monetary approaches *(established but incomplete)* {2.3.5.2, 2.3.5.3}. Such values

are indicated through people's engagement with nature for leisure and tourism, spiritual and aesthetic experiences, learning, developing indigenous and local knowledge, and by their desire to conserve areas and iconic species *(well established)* {2.2.3}.

Nature and its contributions to people have value for human health *(well established)* {2.3.2}, including their role in contemporary and traditional medicine, dietary diversity *(well established)* {2.2.2.4, 2.3.2} and urban green spaces *(established but incomplete)* {2.3.2}. Unsustainable exploitation threatens the survival of, for instance some medicinal plants *(established but incomplete)* {2.2.2.4}.

Indigenous peoples and local communities hold distinct knowledge about nature and its contributions to people that have significant value for many local communities *(established but incomplete)* {2.3.3}. There has been, however, a loss of indigenous and local knowledge about ecosystems and species *(well established)* {2.2.3.1.2, 2.3.3} as well as declining trends of linguistic diversity (a proxy for indigenous and local knowledge) *(well established)* {2.2.3.1.2, 2.3.3}.

There is a range of monetary and non-monetary approaches to capture the multiple values of natures contributions to people. Novel approaches enable these values to be integrated into decision-making to maximize economic, social and quality-of-life benefits.

A2 **There are negative trends for the majority of nature's regulating, and some non-material, contributions to people in the Europe and Central Asia region between 1960 and 2016** *(well established)* **{2.2.1, 2.2.3, 2.2.5}. This has resulted partly from intensive agriculture and forestry practices used to increase the production of food and biomass-based fuels, which have had a negative impact on many regulating services, such as soil formation, pollination and the regulation of freshwater quality** *(well established)* **{2.2.1, 2.2.2, 2.2.5}. This continuing decline in regulating contributions can have**

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3. For explanation of confidence terms, see appendix 1.

4. These monetary values have been standardized to a common currency (the international dollar – $) and base year (2017). The standardization procedure adjusts values elicited in a particular currency and year to a standard currency and year using appropriate gross domestic product deflators and purchasing power parity (PPP) exchange rates.

Figure SPM 3 **Trends in nature's contributions to people (1960–2016) for Europe and Central Asia and the subregions.**

Trends are based on the evidence from publications and indicators reporting increasing, decreasing, constant or variable trends for each ecosystem service {2.2.5}. The higher level of confidence for the region of Europe and Central Asia compared with the subregions is the result of the extra publications that addressed the region as a whole. *Abbreviations:* WE = Western Europe, CE = Central Europe, EE = Eastern Europe, CA = Central Asia, ECA = Europe and Central Asia

		WE	CE	EE	CA	ECA
REGULATING NATURE'S CONTRIBUTIONS TO PEOPLE	Habitat maintenance	↘	↘	↘		↘
	Pollination	↘	↘	↘		↘
	Regulation of air quality	↕	↗	↗	↕	↗
	Regulation of climate	↗	↕	↗	↕	↕
	Regulation of ocean acidification					↕
	Regulation of freshwater quantity	↘	↕	↘	↘	↘
	Regulation of freshwater quality	↘	↘	↘		↘
	Formation and protection of soils	↘	↘	↘	↘	↘
	Regulation of coastal and fluvial floods	↕	↘	↘	↕	↘
	Regulation of organisms (removal of carcasses)	↗	↕	↗	↗	↗
MATERIAL NATURE'S CONTRIBUTIONS TO PEOPLE	Food	↗	↗	↗	↗	↗
	Biomass-based fuels	↗	→	→		↗
	Materials (wood and cotton)	→	→	→	→	→
NON-MATERIAL NATURE'S CONTRIBUTIONS TO PEOPLE	Learning derived from indigenous and local knowledge	↘	↘	↘	↘	↘
	Physical and psychological experiences	↕	↘	↘		↕
	Supporting identities					↕

↗ Increase → Stable ▢ Lack of evidence

↘ Decrease ↕ Variable

Confidence level:
➤ Well established
→ Established but incomplete/unresolved
→ Inconclusive

detrimental consequences for quality of life *(established but incomplete)* {2.3.1.1, 2.2.1.2, 2.2.1.5, 2.2.1.6, 2.2.1.7, 2.2.1.8, 2.2.2.1, 2.2.3.1}.

A total of 7 out of the 16 assessed nature's contributions to people are known to be declining in Europe and Central Asia, in particular regulating contributions and learning derived from indigenous and local knowledge *(well established)* {2.2.1, 2.2.3, 2.2.5}. These trends are consistent across the subregions of Europe and Central Asia **(Figure SPM.3)** *(well established)* {2.2.5}. Habitat maintenance, pollination *(established but incomplete)*, regulation of freshwater quantity and quality, formation and protection of soils and regulation of floods are declining because of land-use intensification designed to increase the production of crops, livestock, aquaculture, forest biomass and cotton, as well as urban development *(well established)*

{2.2.1, 2.2.2, 2.2.5}. Trade-offs between material and regulating contributions have compromised food and water security in some areas {2.2.1, 2.2.2, 2.2.5}.

The Europe and Central Asia region is currently food secure because of food production in the region and trade, despite the degradation of several of nature's regulating contributions and loss of food-related indigenous and local knowledge *(well established)* {2.3.1.1, 2.2.1.2, 2.2.1.5, 2.2.1.7, 2.2.1.8, 2.2.2.1, 2.2.3.1}. Soil erosion has affected 25 per cent of agricultural land in the European Union and 23 per cent in Central Asia. Combined with a decline in soil organic matter, this might compromise food production *(well established)* {2.2.1.8}. At the same time, between 2000 and 2010, erosion control increased by 20 per cent on arable land in Western and Central Europe {2.2.1.8}. Since 1961, Mediterranean and Central Asian countries have increased their dependence

on pollination by producing more pollinator-dependent fruits *(established but incomplete)* {2.2.1.2}. At the same time, however, the diversity and abundance of wild insect pollinators have declined since the 1950s and severe losses of the western honeybee have occurred in Europe since 1961 *(established but incomplete)* {2.2.1.2}. Continuing rural depopulation across the region and the loss of indigenous and local knowledge about traditional land use affects food availability, especially in remote areas *(established but incomplete)* {2.2.3.1.2, 2.2.3.2.1, 2.3.1.1, 4.5.5}. Wild fish catches have decreased since the 1990s, with more sustainable management practices being introduced only recently. Fish production from aquaculture increased by 2.7 per cent since 2000 *(established but incomplete)* {2.2.2.1.2}.

Water security depends partially on the regulation of water quality and quantity by ecosystems, which is impaired by pollution, decreasing floodplain and wetland area, overexploitation of freshwater bodies, and climate change *(established but incomplete)* {2.2.1.6, 2.2.1.7}. Nevertheless, 95 per cent of the people in Europe and Central Asia have access to safe drinking water, despite a 15 per cent decrease in water availability per capita since 1990 *(well established)* {2.3.1.3}.

A3 **Nature's contributions to people, and their influence on quality of life, are not always equally experienced across different locations and social groups in Europe and Central Asia *(established but incomplete)* {2.3.4}.**

Intra-regional equity in access to food and a balanced diet is largely achieved *(well established)* {2.3.1.1} as indicated by, for example, the average dietary energy supply, which ranges from 137 per cent in Western Europe to 121 per cent in Central Asia of the average dietary energy requirement for the population of the region {2.3.1.1}. However, large-scale land acquisitions in Central and Eastern Europe and Central Asia by entities from outside and within the region, mainly from Western Europe, may compromise the opportunities for certain groups of people to influence their own food systems *(established but incomplete)* {2.3.1.1}. Nature's contributions to people are factors in influencing the situation in which some 15 per cent of people in Central Asia, but only 1 per cent in Western Europe, lack access to safe drinking water *(well established)* {2.3.1.3, 2.3.4.2}. Within cities, inhabitants have unequal access to green spaces with consequences for public health and well-being *(established but incomplete)* {2.2.3.2, 2.3.4.2}. For example, residents in cities in the south of the European Union have less access to green space than residents of northern, western and central cities. Public access to forests for recreation is uneven across countries, with a high level of access (98–100 per cent) in Nordic and some Baltic countries and lower levels (under 50 per cent) in some other Western European countries *(well established)* {2.3.4.2}. There is also temporal

inequity as today's generations are benefiting from nature's contributions to people at the expense of future provision *(established but incomplete)* {2.2.3.4}.

A4 **The population of Europe and Central Asia uses more renewable natural resources than are produced within the region (Figure SPM.4) *(well established)* {2.2.4}. The region depends on net imports of both renewable natural resources and material contributions of nature to people *(well established)* {2.2.4}. Some of these imports to Europe and Central Asia negatively affect biodiversity, nature's contributions to people and food security in other parts of the world *(established but incomplete)* {2.2.4, 2.3.4}.**

Measures of ecological footprint[5] and "biocapacity"[6] show that Central and Western Europe import more of nature's contributions to people than Eastern Europe and Central Asia *(well established)* {2.2.4} **(Figure SPM.4)**. While most of Western and Central Europe and Central Asia have a "biocapacity" deficit, in Eastern Europe and northern parts of Western and Central Europe high footprints are offset by even higher biocapacities *(well established)* {2.2.4}. This negatively affects biodiversity, nature's contributions to people and food security both within Europe and Central Asia and other parts of the world *(established but incomplete)* {2.2.4, 2.3.4}. For instance, according to the technical report 2013-063 funded by the European Commission, 10 per cent of the world's annual deforestation was the result of consumption by the then 27 member States of the European Union *(established but incomplete)* {2.2.4.1}.

Western Europe's ecological footprint is 5.1 global hectares[7] per person and its "biocapacity" 2.2 hectares per person; Central Europe's footprint is 3.6 hectares per person and its "biocapacity" 2.1 hectares per person; Eastern Europe's footprint is 4.8 hectares per person and its "biocapacity" 5.3 hectares per person; and Central Asia's footprint is 3.4 hectares per person and its "biocapacity" 1.7 hectares per person *(well established)* {2.2.4} **(Figure SPM.4)**.

Food availability in Central and Western Europe relies significantly on imports from countries, both outside and
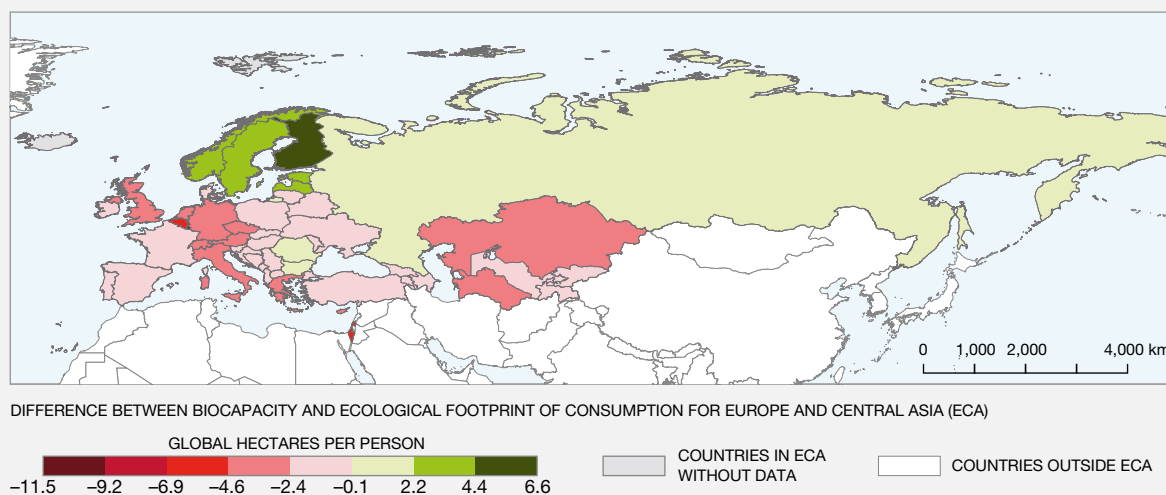
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5. Ecological footprint has a variety of definitions, but is defined by the Global Footprint Network as "a measure of how much area of biologically productive land and water an individual, population or activity requires to produce all the resources it consumes and to absorb the waste it generates, using prevailing technology and resource management practices." The ecological footprint indicator used in this report is based on the Global Footprint Network unless otherwise specified.

6. The definition that follows is for the purpose of this assessment only: "Biocapacity" has a variety of definitions, but is defined by the Global Footprint Network as "the ecosystems' capacity to produce biological materials used by people and to absorb waste material generated by humans, under current management schemes and extraction technologies." The "biocapacity" indicator used in this report is based on the Global Footprint Network unless otherwise specified.

7. A global hectare is a biologically productive hectare with world average biological productivity for a given year and depends on the land type.

Figure SPM ④ **Difference between "biocapacity" (on average 2.9 global hectares per person in the region) and the ecological footprint of consumption (4.6 global hectares per person; average deficit 1.7 global hectares per person).**

The ecological footprint quantifies the area needed to produce on a sustainable basis the renewable resources it consumes and thus can be used as a proxy for the use of certain of nature's material or regulating contributions to people and the area needed to assimilate CO_2 and other waste sustainably. "Biocapacity" refers to the capacity of a certain area to generate an ongoing supply of renewable resources and thus is a proxy for ecosystem productivity. A positive value (green) indicates a "biocapacity" reserve; a negative value (red) indicates a deficit. A deficit derives from the overuse of local renewable resources or the net import of renewable resources for consumption. Countries shaded in green have high "biocapacity", so they have a reserve despite having a higher ecological footprint than many other countries. Source: Based on Global Footprint Network (2017).

DIFFERENCE BETWEEN BIOCAPACITY AND ECOLOGICAL FOOTPRINT OF CONSUMPTION FOR EUROPE AND CENTRAL ASIA (ECA)

GLOBAL HECTARES PER PERSON

–11.5 –9.2 –6.9 –4.6 –2.4 –0.1 2.2 4.4 6.6

COUNTRIES IN ECA WITHOUT DATA   COUNTRIES OUTSIDE ECA

within the region, of the product of 35 million hectares of cropland harvested per year (2008 data), particularly from Argentina, Brazil, China and the United States *(well established)* {2.2.4}. Western Europe became less self-sufficient in crop production between 1987 and 2008, while the rest of Europe and Central Asia became more self-sufficient *(well established)* {2.2.4}. Seafood exports from Europe and Central Asia increased over the period 1976–2009, with Norway, Spain and the Russian Federation being the main exporters *(well established)* {2.2.4}. Over the period 1997–2012, there was a stable pattern of imports to Western Europe of roundwood and wood products from Central and Eastern Europe *(well established)* {2.2.4}.

A5 **Biodiversity loss impairs ecosystem functioning and, hence, nature's contributions to people *(well established)* {3.2.1, 3.2.2, 3.2.3}. The sustained delivery of these contributions requires the maintenance of different levels of biodiversity, i.e., genetic diversity, species diversity, and the diversity of ecosystems and of landscapes and seascapes *(well established)* {3.2.4}. At each of these levels, the sustained delivery of multiple contributions generally requires higher diversity than the delivery of single contributions *(well established)* {3.2.5}.**

Different organisms, species and communities differ in their contributions to ecosystem processes in Europe and Central Asia. Higher biodiversity therefore increases the capacity of terrestrial, freshwater and marine ecosystems to provide nature's contributions to people, such as soil formation, pollination, regulation of hazards, regulation of air and water quality, or the provision of materials, learning and inspiration *(well established)* {3.2.1, 3.2.2}. Higher biodiversity also stabilizes ecosystem functioning and improves capacity for evolutionary adaptation *(well established)* {3.2.3, 3.2.4}. The higher the number of nature's contributions to people to be provided, and the longer the time span and the larger the area of their provision, the more biodiversity is required *(well established)* {3.2.5}.

Ecosystem functioning is affected by genetic and phenotypic biodiversity within species, and by functional, taxonomic and phylogenetic diversity between species *(well established)* {3.2.4}. At the landscape and larger spatial scales, the increasing similarity of the sets of organisms found at different places, e.g., owing to the application of similar and intensive land use over large spatial scales, reduces nature's overall contributions to people *(established but incomplete)* because different sets of organisms contribute to different contributions of nature to people *(well established)* {3.2.5}. Thus, the supply of multiple contributions of nature to people requires the maintenance and promotion of high biodiversity at the landscape level *(established but incomplete)* {3.2.5}.

B. Trends in biodiversity and attribution to direct drivers

B1 Of the assessed marine habitats and species, a high percentage are threatened *(established but incomplete)*, varying between marine areas *(well established)* {3.3.4.1–7} **(Figure SPM.6).** The

abundance, range and habitat size of many marine species is shrinking under human pressures, including overfishing, climate change, pollution and invasive alien species *(well established)* {3.3.4.1–7, 3.4.6.1}.

Figure SPM 5 A **Extinction risk of species in Europe and Central Asia according to the International Union for Conservation of Nature (IUCN) Red List of Threatened Species in 2015.**

EX: extinct, CR: critically endangered, EN: endangered, VU: vulnerable, NT: near threatened, DD: data deficient, LC: least concern. Species in categories CR, EN, VU are considered threatened. The blue bar is the best estimate of the proportion of threatened and extinct species, assuming that the same proportion of DD species is threatened or extinct as of species with sufficient data (i.e., EX, CR, EN, VU, NT, LC). Only species in comprehensively assessed taxonomic groups are considered. Source: IUCN (2017).[8]
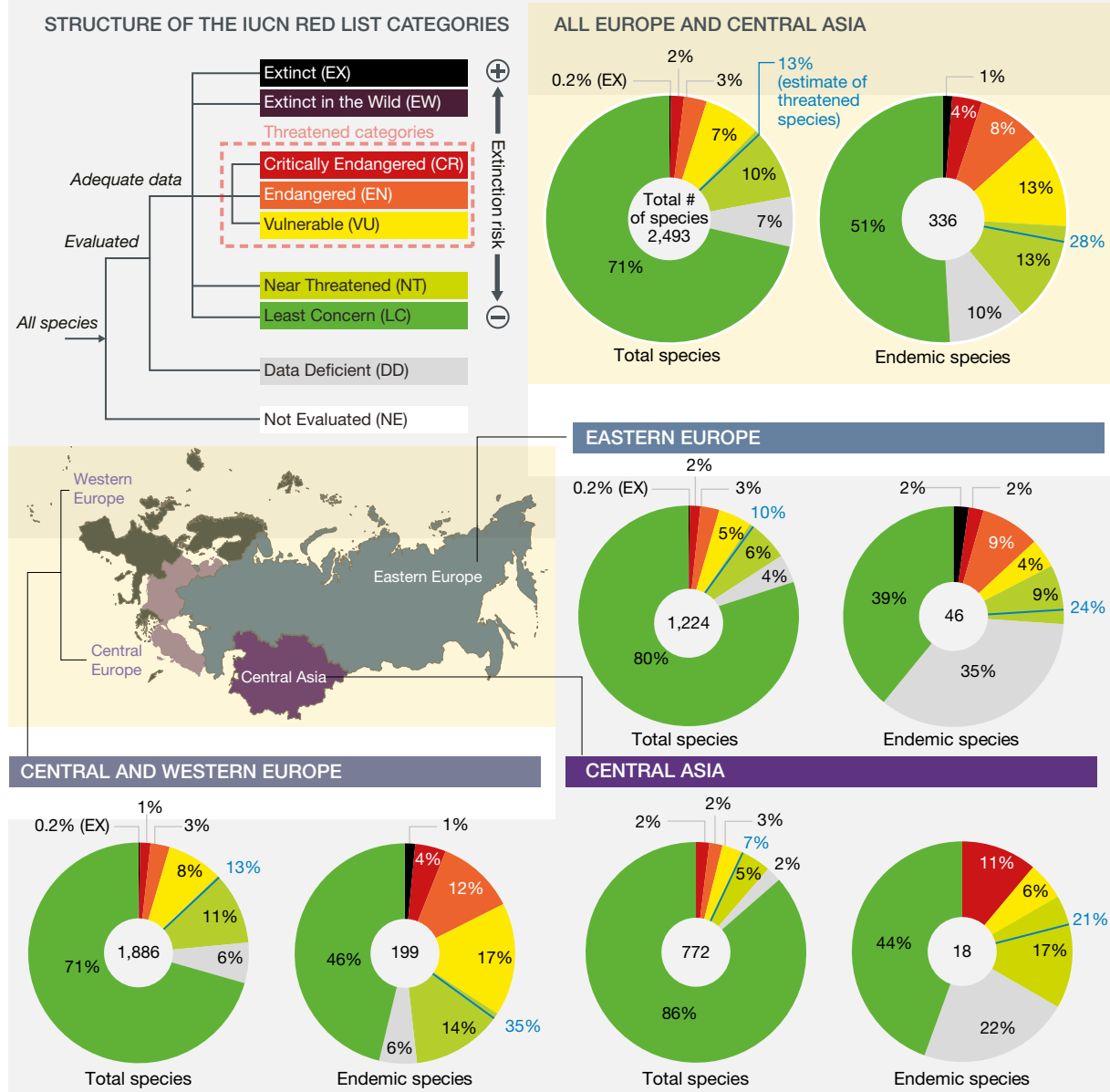
Figure SPM **5** **B** **Trend in Red List Indices of species survival weighted by the fraction of the distribution of each species within the region.**

The position on the vertical axis indicates the aggregate risk of extinction, the closer to one the lower the aggregate extinction risk. The slope indicates how rapidly this extinction risk is changing. For the region, the risk of extinction of species has increased over the last 20 years. Each line represents the most likely Red List Index value, considering uncertainty in the number of species threatened. The shading around each line represents the extremes, if all data deficient species were threatened with extinction (above the line), or if none of them were (below the line). Only birds, mammals and amphibians are considered here, as these are the only groups that have been comprehensively assessed at least twice. Source: IUCN, Red List of Threatened Species, version 2017-3.[8]
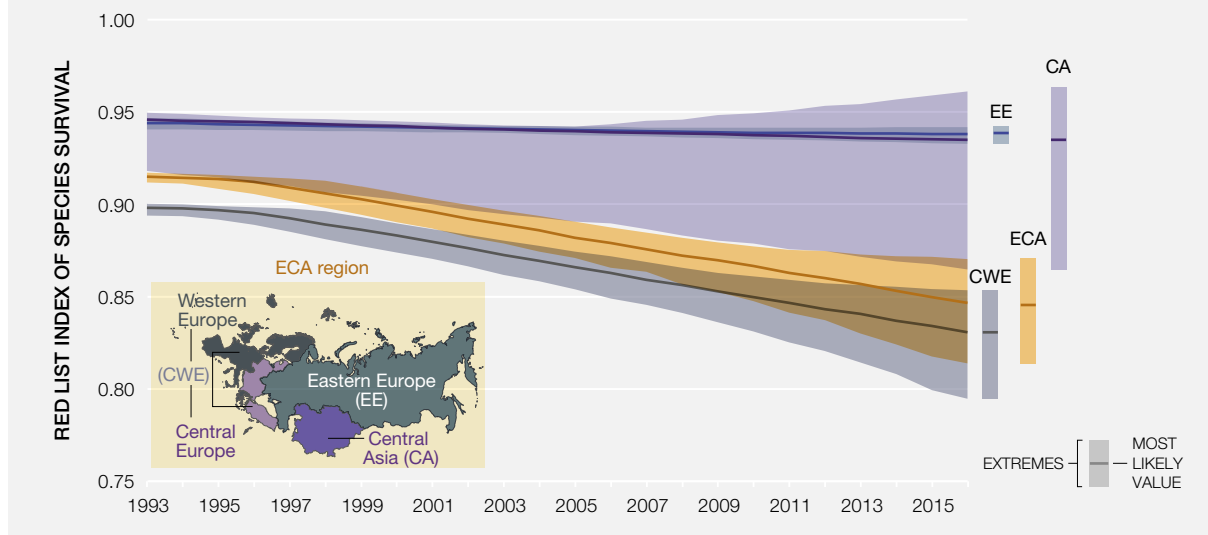
Present positive trends, mainly due to improved fishing practices, the establishment of marine protected areas and a reduction in eutrophication, include increases in some fish stocks in the North Sea and in plankton diversity in the Black Sea *(well established)* **{3.3.4.1, 3.3.4.4}. However, monitoring data are generally missing for most marine habitats and species** *(well established)* **{3.3.4}.**

In all, 53 per cent of the benthic shallow habitats in Western and Central Europe are data deficient. The corresponding figure is 87 per cent in the Black Sea, 60 per cent in the North East Atlantic, 59 per cent in the Mediterranean Sea and 5 per cent in the Baltic Sea *(well established)* {3.3.4.1–7}. Of the assessed benthic habitats, 38 per cent are classified as threatened (critically endangered, endangered or vulnerable), most of them in the Black Sea (67 per cent) and Mediterranean Sea (74 per cent), followed by the North East Atlantic (59 per cent) and the Baltic Sea (8 per cent) *(established but incomplete)* {3.3.4.1–7}. In the European Union, among assessments of the conservation status of species and habitat types of conservation interest covered by the European Union Habitats Directive, only 7 per cent of marine species and 9 per cent of marine habitat types show a "favourable conservation status". Moreover 27 per cent of species and 66 per cent of assessments of habitat types show an "unfavourable conservation status" and the

remainder are categorized as "unknown" *(established but incomplete)* {3.3.4}.

In Europe and Central Asia, 26 per cent of the marine fish species have known trend data. Of those, 72 per cent are stable, 26 per cent have declining populations and 2 per cent have been increasing over the last decade *(well established)* {3.4.6.1}. Seabirds, marine mammals and turtles, and habitat formers, such as seagrasses and kelps, also declined in abundance *(well established)* {3.4.2–4}. The distribution or phenology of marine phytoplankton, zooplankton, algae, benthic invertebrates, fishes, seabirds and mammals has shifted *(well established)* {3.3.4.1}. In all, 48 per cent of marine animal and plant species with known population trends (436 decreasing, 59 increasing, 410 stable) have been declining in the last decade, increasing the extinction risk of monitored species **(Figure SPM.5)** *(established but incomplete)* {3.4.1}. Most of these present trends are consistent with the individual and combined effects of mainly overfishing, climate change, pollution and invasive alien species *(established but incomplete)* {3.3.4.1–7}. The impact of pollution by microplastics on ecosystems was not known until recently, and evidence of those impacts is only now being assessed {3.3.4}.

B2 **Freshwater species and inland surface water habitats are particularly threatened in Europe and Central Asia** *(well established)***. A total of 53 per cent of the European Union's rivers and lakes achieved**

_____
8. Available from www.iucnredlist.org.

good ecological status in 2015 as defined by the European Union Water Framework Directive. Similarly 30 per cent of water samples in the Russian

Federation were above water quality standards *(well established)*. A total of 73 per cent of the assessments of the European Union's freshwater habitat types

Figure SPM ⑥ **Assessment of past (~1950–2000) and current (~2001–2017) trends in biodiversity status of marine, inland surface water and terrestrial ecosystems for the four subregions and the whole of Europe and Central Asia.**

The figure summarizes the trends in biodiversity status of the assessed units of analysis (habitat types). Biodiversity status represents the expert assessment of available indicators of habitat intactness, species richness and the status of endangered species. The trends are presented by unit of analysis and subregion for terrestrial and inland surface-water ecosystems, and by sea or ocean area for marine ecosystems {3.3; Box 3.3}. *Abbreviations:* WE = Western Europe, CE = Central Europe, EE = Eastern Europe, CA = Central Asia, ECA = Europe and Central Asia

show an unfavourable conservation status *(well established)* {3.3.3.1}. Across Europe and Central Asia, lakes, ponds and streams are altered and disappearing as a consequence of agricultural intensification, irrigation and urban development combined with climate change *(well established)* {3.3.3.1}. Notable is the case of the Aral Sea, once the fourth largest lake in the world, which has now almost disappeared owing to water abstraction for crop cultivation. The extent of wetlands in Western, Central and Eastern Europe has declined by 50 per cent from 1970, while 71 per cent of fish and 60 per cent of amphibians with known population trends have been declining over the last decade {3.3.3.1, 3.4.5, 3.4.6.2}.

Over 75 per cent of catchment areas in Europe and Central Asia are heavily modified and subject to multiple pressures. In 2015, good chemical status, as defined by the European Union Water Framework Directive, was not achieved for surface water bodies by 22 European Union member States and only 53 per cent of rivers and lakes had good ecological status as defined by the European Union Water Framework Directive despite some improvements {3.3.3.1}. In Western and Central Europe and the western parts of Eastern Europe[9] at least 37 per cent of freshwater fish and about 23 per cent of amphibians are currently threatened with extinction. In the same area, freshwater invertebrates are also threatened, with the most threatened group among those that are well monitored being gastropods (45–70 per cent of species threatened depending on whether or not data deficient species are considered threatened), followed by bivalves (20–26 per cent) and dragonflies (15–19 per cent) *(established but incomplete)* {3.4.5, 3.4.6.2, 3.4.8}.

Freshwater biodiversity trends are primarily driven by habitat destruction and modification caused by infrastructure for hydropower, navigation, flood protection, agriculture, urban development and water abstraction; pollution from agriculture and industry; the introduction of invasive alien species and their pathogens; and climate change *(established but incomplete)* {3.3.2.2, 3.3.3.4, 3.3.3.5.2}. Progress has been made in water protection in the European Union part of Western and Central Europe, in particular because of the European Union Water Framework Directive. The rate of natural habitat loss (e.g., wetlands) has slowed in Western, Central and Eastern Europe due to the implementation of binding nature conservation policies or the designation of conservation areas (e.g., Ramsar sites), *(established but incomplete)* {3.3.3.1}.

B3 **Terrestrial species and habitats have long-term declining trends in population size, range, habitat intactness and functioning. This decline is mainly due**

to land-use change, for example unsustainable agriculture and forest management, infrastructure, urban development or mining, causing habitat loss, modification and fragmentation, and due to climate change *(well established)* {3.3.2, 3.4}. The conservation status of some habitats and species that benefit from targeted conservation actions (e.g., large felids or some species listed in the European Union Birds Directive) has improved in recent years *(established but incomplete)* {3.4.13}.

Across Europe and Central Asia, 14 out of 15 habitat types have been declining in extent and biodiversity status since the 1950s **(Figure SPM.6)** {3.3.2.5}. These declines are continuing, albeit at a slower rate, with some exceptions in the Macaronesian and Atlantic Boreal regions of Western and Central Europe, where recoveries in habitat conservation status have been reported. Grasslands, tundra, mires and bogs have been the most affected habitats since the 1950s *(established but incomplete)* {3.3.2}.

Systematic assessments of habitat conservation status exist only for the European Union. There, 16 per cent of terrestrial habitat assessments in the period 2007–2012 had favourable conservation status; 3 per cent had unfavourable, but improving trends; 37 per cent had unfavourable, but stable trends; 29 per cent had unfavourable and declining trends; and 15 per cent had unknown or unreported trends relative to the period 2001–2006 *(well established)* {3.3.2}.

Since the 1950s, various biodiversity indicators have shown a decline in response to both abandonment of, and intensified use of, agricultural land *(well established* for Western Europe and Central Europe; *established but incomplete* for Eastern Europe and Central Asia) {3.3.2.9}. From 1980 to 2013, the abundance of farmland common bird species decreased by 57 per cent in Western and Central Europe *(well established)* {3.4.3}. The species diversity of arable crops has decreased by 20 per cent since 1950 in Western and Central Europe, and the abundance of rare arable plants has also decreased *(established but incomplete)*. The genetic diversity of plants cultivated in situ declined until the 1960s, owing to the replacement of landraces by modern cultivars, and no further reduction or increase of diversity was observed after the 1980s *(well established)*. Europe and Central Asia has over half of all known breeds of domesticated mammals and birds, but 75 per cent of local bird breeds and 58 per cent of local mammal breeds are threatened with extinction. The numbers of at-risk breeds have declined slightly since 1999, but exact quantification is hampered by the changing number of documented local breeds *(established but incomplete)* {3.4.13}.

Across Europe and Central Asia, 42 per cent of terrestrial animal and plant species with known trends

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9. The geographical scope here is continent-wide, extending from Iceland in the west to the Urals in the east, and from Franz Josef Land in the north to the Canary Islands in the south. The Caucasus region is not included.

have declined in population size over the last decade, increasing the extinction risk of monitored species *(established but incomplete)* **(Figure SPM.5)**. The main causes of this decline are habitat loss, degradation and pollution due primarily to unsustainable agriculture and forest management, natural resource extraction and invasive alien species *(established but incomplete)* {3.4, 3.3.2}. Monocultures, and all forms of homogenization of landscapes, such as the conversion of grasslands to crops, and agricultural intensification (especially the conversion of natural and semi-natural grassland to more intensively used pastures) have caused homogenization of ecological communities by supporting generalist species and impacting habitat specialists *(well established)*. Climate change is accelerating changes in species composition

and local extinctions in all habitat types *(well established)*, contracting glaciers, shifting the nival belt to higher altitudes *(well established)*, replacing polar deserts with tundra *(well established)*, expanding arid areas, and causing shifts in forest habitat types *(well established)* {3.3.2}. National and international conservation efforts have shown the potential to reverse these trends. The long-term population trends of 40 per cent of the breeding bird taxa in Annex I of the European Union Birds Directive are increasing, compared with 31 per cent for all breeding bird taxa {3.4.13}. Charismatic mammalian megafauna, such as the Amur tiger, Far-Eastern leopard, Iberian lynx, and European bison, are all recovering from the brink of extinction because of dedicated conservation efforts {3.4.3, 3.4.13}.

C. Drivers of change in biodiversity and nature's contributions to people in Europe and Central Asia

C1 **Land-use change, as one of the major direct drivers of change in biodiversity and nature's contributions to people in Europe and Central Asia, is often posing substantial risks for human well-being** *(well established)* **(4.2.1). There are examples of sustainable agricultural and forestry practices that are beneficial to biodiversity and nature's contributions to people in the region. However, the major trend is increasing intensity of conventional agriculture and forestry that lead to biodiversity decline** *(well established)*. **Ceasing traditional land use reduces semi-natural habitats of high conservation value** *(well established)* **and associated indigenous and local knowledge and practices** *(well established)* **{4.5.1, 4.5.5}. Protected areas have expanded, but this alone cannot prevent biodiversity loss** *(well established)* **{4.5.4}.**

Despite the development of more sustainable agricultural policies and practices in recent years in some countries, such as organic farming, conventional intensive agriculture, especially related to the excessive use of agrochemicals {4.5.1.1} reduces natural and semi-natural habitats, with severe negative impacts on biodiversity and ecosystem function *(well established)* {4.5.1, 4.5.2, 4.5.5}. This jeopardizes the sustainable management of land and food production *(established but incomplete)* **(Figure SPM.8)**

{4.5.1, 4.5.2}. Agri-environmental schemes, ecological restoration and sustainable approaches to agriculture, such as agroecology and agroforestry, mitigate some of the adverse effects of intensive agriculture *(established but incomplete)* {4.5.1, 4.5.2}. The efficiency of such measures depends also on the inclusion of traditional and local knowledge, and the consideration of biophysical and social-cultural contexts *(established but incomplete)* {4.5.1, 4.5.2, 4.5.3}.

Production-based subsidies have driven growth in agriculture, forestry and natural resource extraction, but this often impinges on traditional land users *(established but incomplete)* {4.5.1, 4.5.5}. The loss of traditionally managed semi-natural habitats has resulted in a decline and loss of associated biodiversity and ecosystem functions. Demographic trends, including urbanization, continue to diminish indigenous and local communities, with concomitant negative impacts on traditional land-use knowledge, culture and identities *(established but incomplete)* (4.5.5). The economic viability of indigenous and local communities can be supported by green tourism, demand for products derived from traditional practices and subsidies for traditional land uses *(well established)* {4.5.5}.

There are examples of sustainable forestry and agroforestry practices, however, the major trend across the region

Figure SPM 7 **Trends in the proportion of key biodiversity areas completely covered by protected areas in Europe and Central Asia.**

There are two types of key biodiversity areas, Important Bird and Biodiversity Areas (IBAs) and Alliance for Zero Extinctions sites (AZEs).
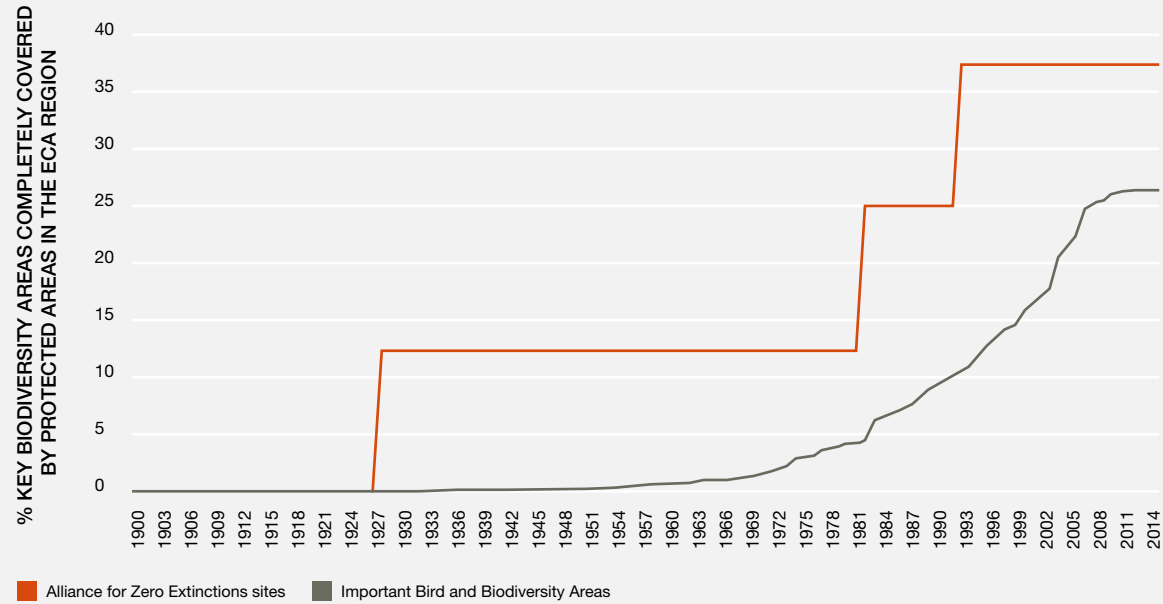

■ Alliance for Zero Extinctions sites   ■ Important Bird and Biodiversity Areas

Figure SPM 8 **Trends in direct drivers of biodiversity and nature's contributions to people in the last 20 years.**

The figure summarizes the trends in the five direct drivers for each of the assessed units of analysis (habitat types). The trends are presented by unit of analysis and subregion {see 4.2.1, 4.4, 4.5, 4.6, 4.7, 4.8, 4.9.2}. *Abbreviations:* WE = Western Europe, CE = Central Europe, EE = Eastern Europe, CA = Central Asia

is intensification of forest management that reduces biodiversity and many of nature's material and non-material contributions to people **(Figure SPM.8)**. Logging of intact forests continues across the region *(established but incomplete)* {4.5.3}. The trade-offs between the increasing intensity of forestry and delivery of multiple ecosystem services are recognized as a major challenge for forestry in Europe and Central Asia **(Table SPM.2)**.

Protected areas now cover 10.2 per cent of the region, 13.5 per cent of its terrestrial area and 5.2 per cent of its marine area *(well established)* {4.5.4} and their coverage of key biodiversity areas has been increasing **(Figure SPM.7)**. The prioritization and implementation of adequate legal frameworks for protected area development has largely been driven by the adoption of international agreements, as well as increasing public environmental awareness. The perceived trade-offs with economic development goals, however, have in many cases delayed the development of, or weakened, adequate nature conservation policies although this is variable across the region *(well established)*. The efficacy, connectivity and representativeness of protected areas are as important as their coverage, however, and conservation would also require fostering biodiversity outside protected areas *(well established)* {4.5.4, 3.3}. Eastern Europe and the Balkans have recently experienced armed conflicts, which negatively affect nature and its contributions to people {4.5.4.2}.

C2 **The impact of climate change on biodiversity and nature's contributions to people is increasing rapidly and is likely to be among the most important drivers in the future, in particular in combination with other drivers *(established but incomplete)* {4.7.1, 4.7.2, 4.9.2}.**

The region's climate is expected to be on average 1°C –3°C warmer in 2041–2060 than in 1986–2005, with larger increases in the north of the region *(well established)* {4.7.2.1}. Summers will be drier in the south of the region and winters wetter in the north, with increasing risks of extreme climatic events such as droughts and storms *(established but incomplete)* {4.7.1.2} **(Figure SPM.8)**. Indirect climate change effects, such as increased fire and flood risks and loss of permafrost, are already affecting biodiversity and nature's contributions to people *(well established)* {4.7.1.3, 4.7.2.5}. The extent of near-surface permafrost at high latitudes could decrease by between 37 and 81 per cent by 2100 *(established but incomplete)* {4.7.2.4}. In Arctic and alpine regions, permafrost melting will cause large greenhouse gas emissions, while short-term heat waves reduce biomass productivity and food availability for wildlife and livestock *(unresolved)* {4.7.1}.

Climate change shifts seasonal timing, growth and productivity, species ranges and habitat location, which affects biodiversity, agriculture, forestry, and fisheries *(well*

established) {4.7.1.1, 4.7.1.3}. Many species will not migrate or adapt fast enough to keep pace with projected rates of climate change *(established but incomplete)* {4.7.1}. Droughts decrease biomass productivity, increase biodiversity loss and net carbon flux to the atmosphere, and decrease water quality in aquatic systems *(established but incomplete)* {4.7.1.2, 5.2}. Climate change causes ocean acidification, rising sea levels and changes ocean stratification, reducing biodiversity, growth and productivity, impairing fisheries and increasing CO_2 release into the atmosphere *(established but incomplete)* {4.7.1.1, 4.7.1.3}.

Global economic growth is the main indirect driver of greenhouse gas emissions and hence climate change *(well established)* {4.7.3}. In contrast to global trends, primary energy consumption and fossil CO_2 emissions within the region have declined since 1990. Small increases in GDP growth with simultaneously decreasing energy production and CO_2 emissions from 2011 to 2014 suggest the decoupling of CO_2 emissions from GDP growth *(well established)* {4.7.3}. These apparent decreases may be explained, however, by increased transportation-related emissions in other regions and their inter-regional flows to Europe and Central Asia *(inconclusive)* {4.7.3} **(Table SPM.2)**.

C3 **Natural resource extraction, pollution and invasive alien species continue to reduce biodiversity and nature's contributions to people, and they increase with GDP and global trade. Recent policy intervention has reversed some negative impacts of these direct drivers.**

Extraction of biotic and abiotic natural resources has continued to reduce biodiversity and nature's contribution to people both within Europe and Central Asia and beyond. For biotic resources, the demand for fish in Western and Central Europe, coupled with the European Union Common Fisheries Policy that restricts extraction, contributes to unsustainable fishing practices and resource depletion outside Western and Central Europe. While awareness of local resource shortages, such as fish in Europe, would be expected to be prompted by price increases, displacement from interregional imports masks these feedbacks *(established but incomplete)* {4.2.5, 4.3.1, 4.4.1}.

As an example for abiotic resources, trade liberalization and increasing world market prices have increased extraction of mineral resources in Central Asia. Although this has resulted in the mining industry being one of the largest contributors to GDP in the subregion, this has led to the depletion of mineral resources and the loss of ecosystem services important to human health and well-being *(well established)* {4.4.4.2}.

These examples demonstrate that the depletion of natural resources may not be immediately apparent, due to factors

Table SPM 2 Impact of indirect drivers (rows) on direct drivers (columns) of biodiversity loss and nature's contributions to people in Europe and Central Asia.

The colour shows the impact of an indirect driver on a direct driver's effect on biodiversity and nature's contributions to people along a gradient from negative to positive effects. *Abbreviations:* WE = Western Europe, CE = Central Europe, EE = Eastern Europe, CA = Central Asia

	LAND USE CHANGE															
	Agricultural land use				Forestry				Traditional land use				Protected area development			
	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA
INSTITUTIONAL	√	√	×	×	√	√	√	√	√	√	~	~	√	√	~	~
ECONOMIC	~	~	~	~	×	~	×	~	√	√	×	×			×	×
DEMOGRAPHIC			~	~					×	×	×	×				
CULTURAL	√	~	×	×	√	√	√	×	~	~	~	~	√	√	×	×
TECHNOLOGICAL	~	~	~	~												

	Climate change				Pollution				Natural resource extraction				Invasive alien species			
	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA
INSTITUTIONAL	√	√	√	√	√	√	√	√	√	√		×	√	√	~	~
ECONOMIC	×	×	×	×	×	×	×	×	×	×		×	×	×	×	×
DEMOGRAPHIC					×	×	×	×	×	×	×	×				
CULTURAL					~	~	~	~	~	~	~	~	×	×	×	×
TECHNOLOGICAL	√	√	√	√	√	√	√	√	×	×	×	×				

× Negative	~ Both ways	√ Positive	Lack of evidence

such as global trade, which then masks or delays effective policy responses. In addition, harmful subsidies in the fishing and mineral industries reduce extraction prices and accelerate extraction levels despite declining stocks *(well established)* {4.4.1, 4.4.4}. The European Union and the Russian Federation continue to pay in total about $6 billion annually in such fishing subsidies *(well established)* {4.4.1.3}.

Recent regulations have reduced some pollution (for example, sulphur oxides, nitrogen oxides and heavy metals), but other pollution (ammonia, organic pollution and pesticides) and time-lag effects of pollution still threaten biodiversity. In Western and Central Europe terrestrial acidification has decreased since 1990 (from 30 per cent to 3 per cent of areas exceeding critical loads, while terrestrial eutrophication has decreased from 78 per cent to 55 per cent of areas exceeding critical loads *(well established)* {4.6.1, 4.6.3}. Marine and coastal eutrophication has decreased, but the proportion of marine dead zones due to oxygen depletion from nutrient and organic pollutants has increased markedly, reaching, for example, about 100 sites around Western European shores alone *(established but incomplete)* {4.6.1, 4.6.2}. Numbers of invasive alien species have increased for all taxonomic groups *(well established)* {4.8.2.1}. In Western and Central Europe, invasive alien species are increasing, although the recently

adopted European Union regulation on invasive alien species could curb the trend in the future {4.8.2, 4.8.3}. In Eastern Europe and Central Asia, rates of invasion are lower than in Western and Central Europe, but are expected to increase with increasing GDP and trade *(established but incomplete)* {4.8.1, 4.8.2} **(Table SPM.2)**. As direct drivers can have chronic, prolonged and delayed consequences for biodiversity and ecosystem services, owing to time lags in ecosystem response *(well established)* {4.5.1, 4.9.1}, phosphorous and nitrogen (except ammonia) pollution is decreasing but, owing to time lags, many lakes, rivers and coastal areas in Western and Central Europe still do not have a good ecological status {4.6.1, 4.6.2}. Time lags also occur between the initial introduction of invasive alien species and their impact *(well established)* {4.8.1}.

C4 Economic growth is generally not decoupled from environmental degradation. This decoupling would require a transformation in policies and tax reforms across the region *(established but incomplete)* {4.3.1, 4.3.2, 4.3.4}.

There is evidence of growth in GDP across Europe and Central Asia *(well established)*. For example, since 2000, gross domestic material consumption has increased across European Union member States, much of which has

been driven by growth-oriented policies *(well established)* {4.3.2}. However, this economic growth has indirectly reinforced drivers of biodiversity loss, which in turn has reduced nature's contributions to people. Such drivers have included land-use change, climate change, natural resource extraction, pollution and invasive alien species **(Table SPM.2)**.

Awareness of sustainability challenges has led to some institutional change in the region, including policies on climate agreements and a range of environmental policies. Furthermore, recent policy initiatives have suggested a focus on decoupling economic growth from environmental degradation {4.3.2, 4.3.4}. This decoupling would require

a transformation in policies and tax reforms at the global and national levels. Across the region, a range of policies for resource efficiency, including environmental taxation, have been implemented. The total revenue from environmental taxes in the European Union has declined from 6.8 per cent of the total revenues derived from all taxes and social contributions in 2002 down to 6.3 per cent in 2016 *(well established)* {4.3.1, 4.3.2}. Furthermore, there still exist policy instruments, such as harmful agricultural and fishing subsidies, which continue to impede transitions towards a sustainable future *(established but incomplete)*. Decoupling would be assisted by new indicators that incorporate well-being, environmental quality, employment and equity, biodiversity conservation and nature's ability to contribute to people.

D. Futures for Europe and Central Asia

D1 **Scenario studies for Europe and Central Asia, with time horizons up to 2100, show trade-offs between different ecosystem services with implications for biodiversity (Box SPM.3, Figure SPM.9) {2.2.6, 3.5, 5.3.3, 5.3.4}. Political and societal value judgements embedded within scenarios will determine how these trade-offs are resolved. Scenarios that assume proactive, environmental decision-making; promote environmental management approaches that support multifunctionality; and mainstream environmental issues across sectors, can mitigate undesirable trade-offs *(established but incomplete)* {5.3.3}.**

Moreover, scenarios that assume cooperation between countries or regions are more effective in mitigating negative impacts across geographic scales *(established but incomplete)* {5.3.3}. Such scenarios project more positive impacts across a broad range of indicators of biodiversity, nature's contributions to people and good quality of life than others *(established but incomplete)* {5.3.3, 5.6.1}.

Scenario studies (see **Box SPM.3** on scenario archetypes) suggest that reactive approaches to environmental issues will have mixed impacts. *Economic optimism* scenarios

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Box SPM **3** **Scenario archetypes.**

The scenario and modelling studies in the literature {5.2.3, 5.3.3.} were mapped to six existing scenario archetypes {5.2.2; Box 5.3}, which represent diverse plausible futures for Europe and Central Asia:

- *Business-as-usual* assumes the continuation of past and current trends in indirect and direct drivers.
- *Economic optimism* assumes global developments steered by economic growth, resulting in a strong dominance of international markets with a small degree of regulation.
- *Regional[10] competition* assumes an increasingly fragmented world with a growing gap between rich and poor; increasing problems with crime, violence and terrorism; and strong trade barriers.
- *Regional[10] sustainability* assumes a shift towards local and regional decision-making that is strongly influenced

by environmentally aware citizens. A proactive attitude to environmental management prevails, but poor international collaboration obstructs coordination to solve global environmental issues.
- *Global sustainable development* assumes an increasingly proactive attitude by policymakers and the public towards environmental issues, a high level of international cooperation and strong regulation.
- *Inequality* assumes increasing economic, political and social inequalities with power concentrated in a relatively small political and business elite who invest in green technology.

Each scenario archetype consists of different assumptions about future changes in direct and indirect drivers as shown in **Table SPM.3**.

Table SPM **3** **Trends in indirect and direct drivers assumed in six scenario archetypes covering time horizons up to 2100.**

Arrows in the table represent the expert interpretation of the magnitude of trends in drivers across all scenarios found within the archetypes. Colour coding represents the expert interpretation of the impact of the trend on biodiversity and nature's contributions to people {5.2.3}.

Scenario archetype	INDIRECT DRIVERS					DIRECT DRIVERS				
	INSTITUTIONAL (Environmental proactivity)	ECONOMIC (Gross domestic product)	DEMOGRAPHIC (Population)	CULTURAL (Sustainable consumption)	TECHNOLOGY	CLIMATE CHANGE (Temperature)	LAND USE CHANGE (Landscape homogeneity)	NATURAL RESOURCE EXTRACTION	POLLUTION	INVASIVE ALIEN SPECIES
Business-as-usual	↗ (positive) ↘ (negative)	↗	↗	↘ (negative)	↗ (positive) ↗ (negative)	↗ (negative)	↗	↗ (negative)	↗ (negative)	↗ (negative)
Economic optimism	↘ (negative)	↑	↑	↘ (negative)	↑ (positive) ↑ (negative)	↑ (negative)	↗	↑ (negative)	↗ (negative)	↗ (negative)
Regional competition	↘ (negative)	→	→	→ (neutral)	↓ (negative)	↗ (negative)	↗	↗ (negative)	↗ (negative)	↗ (negative)
Regional sustainability	↗ (positive)	↗	↗	↗ (positive)	→ (neutral)	↗ (negative)	↘	↘ (positive)	→ (neutral)	↘ (positive)
Global sustainable development	↗ (positive)	↗	→	↗ (positive)	↑ (positive)	↗ (negative)	↗	↘ (positive)	↘ (positive)	↘ (positive)
Inequality	↘ (negative)	↗	↘	→ (neutral)	→ (neutral)	↗ (negative)		↗ (negative)		

Legend:
↑ Strong increase   ↗ Increase   → Stable   ↘ Decrease   ↓ Strong decrease

→ Positive (green)   → Neutral (dark)   → Negative (orange)   → Not interpreted in terms of impacts   ☐ Lack of evidence

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generally lead to declines in biodiversity and regulating ecosystem services, but to increases in provisioning ecosystem services *(established but incomplete)* {5.3.3, 5.6.1}. *Regional competition* scenarios lead to the most negative impacts, particularly for non-material nature's contributions to people and indicators of good quality of life *(established but incomplete)* {5.3.3, 5.6.1}. In both types of scenarios, development is driven by economic growth, leading to strong positive effects for nature's contributions to people with market values and negative effects for contributions without market values *(established but incomplete)* {5.3.3, 5.6.1}. For example, scenarios for Western and Central Europe, which prioritize increases in food provision through agricultural expansion or intensification, lead to trade-offs with regulating contributions to people and biodiversity. Likewise, scenarios for Eastern Europe that focus on timber extraction lead to highly managed forests with decreased climate regulation and value for cultural or recreational purposes.

Sustainability-focused scenarios (e.g., *global sustainable development* or *regional sustainability*) assume a proactive approach to environmental issues that anticipates change and thereby minimizes adverse impacts and capitalizes on

opportunities {5.1.1}. Such scenarios cause increases in most of nature's contributions to people and good quality of life, but have mixed biodiversity trends *(established but incomplete)* {5.3.3, 5.6.1}. Trade-offs occur in these scenarios, especially involving land and water use (such as the effects of reduced agricultural intensity or of increases in bioenergy cropland, on other land uses and biodiversity) {5.3.3, 5.6.1}.

Impacts under *business-as-usual* scenarios are highly variable regionally. In general, the impacts on biodiversity, nature's contributions to people and good quality of life are more positive than for *economic optimism* and *regional competition*, but more negative than for *regional sustainability* and *global sustainable development* *(established but incomplete)* {5.3.3, 5.6.1}.

Scenarios considering climate change indicate increases in agricultural production for food, feed and bioenergy in the northern part of the European Union, but decreases in agricultural and timber production in the southern part **(Figure SPM.10)**. Major water shortages are projected in the

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10. Here the term "regional" is not meant to denote "IPBES regions", but reflects a more general meaning across the assessed literature, where it is used with reference to subnational, national or larger areas.

Figure SPM 9 Projected future impacts on biodiversity, nature's contributions to people and good quality of life according to six scenario archetypes for Europe and Central Asia up to 2100 (see Box SPM.3 for details of the scenario archetypes) {2.2.6, 3.5, 5.3.3}.

Green symbols with upward arrow indicate an increase, purple symbols with horizontal arrow a stable trend, and orange symbols with downward arrow a decrease. Thick arrows indicate evidence from the literature based on ten or more model indicators per scenario archetype, thin arrows indicate evidence based on fewer than ten.

long term for Central Asia, parts of Central Europe, and the Mediterranean, leading to key trade-offs for water use and management in different sectors, including the maintenance of environmental flows *(established but incomplete)* {5.3.3}.

Trade-offs depend on scenario assumptions about lifestyle and consumption, which affect the demand for nature's contributions to people, and policies affecting the

management and governance of resources. For example, *global sustainable development* scenarios assume changes in dietary preferences towards reducing meat consumption, behavioural changes to save water and energy, and the implementation of integrated and sustainable land and water management practices. These lead to positive outcomes for biodiversity, nature's contributions to people and good quality of life. Scenarios that assume strong international or

Figure SPM **10** **Trends in impacts on biodiversity, nature's contributions to people and good quality of life indicators that are consistent across most scenario archetypes (see Box SPM.3 for details of the scenario archetypes) {5.3.3}.**

The Western European region has been divided into four parts (northern, Atlantic, Alpine and southern), in view of the greater number of available studies.

WESTERN EUROPE

• Northern

Biodiversity	↘
Food and feed	↗
Energy	↗
Materials (forest products)	↗

• Atlantic

Regulation of climate	→
Food and feed (marine)	↗
Materials (forest products)	↘

• Alpine

Regulation of climate	↘
Physical and psychological experiences	→

• Southern /Mediterranean

Biodiversity	↘
Food and feed	↘
Materials (forest products)	↘
Water resources	↘
Physical and psychological experiences	↗

EASTERN EUROPE

Regulation of hazards (wildfires)	↘
Food and feed (fisheries)	↗

CENTRAL EUROPE

Regulation of climate	↘
Regulation of freshwater quality	↘
Food and feed	↗
Materials (forest products)	↗
Water resources	↘

CENTRAL ASIA

Regulation of freshwater quality	↗
Food and feed	↗
Water resources	↘
Security and livelihoods	↗

↗ Increase   → Stable to decrease   ↘ Decrease

transboundary coordination of adaptive measures between multiple stakeholders lead to more sustainable solutions across scales and regions. Scenario assumptions in *inequality* scenarios also affect how different social groups appropriate nature's contributions to people *(established but incomplete)* {5.2.3, 5.3.3}.

D2 **Future impacts on biodiversity and nature's contributions to people are underestimated because most scenarios consider only a few drivers, notably climate change** *(well established)* **{5.2.2, 5.3.2}. Single-driver scenarios also fail to capture driver interactions** *(well established)* **{5.2.2, 5.3.2}. Single-driver and single-sector approaches are likely to misrepresent the direction, magnitude or spatial pattern of impacts on biodiversity and nature's contributions to people, leading to poor management or policy decisions** *(established but incomplete)* **{5.3.1}.**

Many scenarios consider climate change as a single driver *(well established)*. The few multi-driver scenarios are largely based on the Special Report on Emissions Scenarios of the Intergovernmental Panel on Climate Change and, hence, focus on long-term climate change issues (to 2100). Pollution and invasive alien species are poorly represented in scenarios *(well established)* {5.2.2}. Land-use change is rarely considered as a direct driver of biodiversity and nature's contributions to people because land-use change scenarios focus more on the effects of indirect drivers (e.g., policy, social preferences and economics) on land use per se *(established but incomplete)* {5.2.1}. There are fewer scenarios of future land-use change impacts on biodiversity and nature's contributions to people than empirical studies of past trends *(established but incomplete)*. Single-driver scenarios fail to capture feedbacks and synergies between and amongst indirect and direct drivers operating across different scales *(established but incomplete)* {5.3.4}. Integrated scenarios and models are explicit about nature and cover multiple drivers, sectors and scales. This enhances the understanding of complex interdependencies between human and environmental systems to support coordinated decision-making {5.2.2, 5.3.1}.

Figure SPM **11** **Summary of the extent to which goals similar to the Sustainable Development Goals are expected to be achieved under the scenario archetypes up to 2100 and pathways to sustainability up to 2050 for Europe and Central Asia {5.3.4, 5.5.4}.**

Part **A** shows that the scenario archetypes *regional sustainability* and *global sustainable development* project a widespread achievement of goals (see **Box SPM.3** for description of the scenario archetypes). Part **B** introduces pathways that support the achievement of goals albeit to a different extent. This is exemplified in part **C**, where the wedges indicate the extent to which the pathways address each goal (see D3 for description of the pathways).

A: orange = widespread failure in the achievement of goals; green = widespread achievement of goals; grey = mixed achievement of goals. **B**: darker shades of green indicate a greater number of goals are addressed by the pathways. **C**: two examples of pathways with smaller and greater number of goals addressed.

A Achievement of goals similar to the Sustainable Development Goals

SCENARIO ARCHETYPES

Business-as-usual	✕
Economic optimism	∼
Regional competition	✕
Regional sustainability	✓
Global sustainable development	✓
Inequality	✕

✓ Widespread achievement of goals

∼ Mixed achievement of goals

✕ Widespread failure in the achievement of goals

B Number of goals similar to the Sustainable Development Goals addressed

PATHWAYS

Transition movements – resource sparing	Greater
Transition movements – collaboration	
Green economy – land sharing	
Low carbon – innovation	
Green economy – innovation	
Low carbon – regional multifunctionality	
Ecotopian – innovation	
Ecotopian – local multifunctionality	
Green economy – land sparing	Smaller

Number of Sustainable Development Goals addressed

C Examples of pathways

Green economy – land sparing

Transition movements – resource sparing

D3 **Pathways propose coherent sets of actions towards the sustainable futures envisioned for the region** *(established but incomplete)* {5.1.2, 5.4.3, 5.5.2}. **The most effective pathways stress longterm societal transformation (behavioural change) through education, knowledge sharing and participatory decision-making. These pathways emphasize nature's regulating contributions to people and the importance of considering diverse values** *(established but incomplete)* {5.5.2, 5.5.3, 5.5.4}.

Four types of pathways are specified. Two types of pathways do not challenge the economic growth paradigm *(green economy* and *low carbon transformation* pathways). They include actions related to technological innovation, land sparing or land sharing, and focus on combinations of top-down legal and regulatory instruments and economic and financial instruments. These pathways do not fully mitigate trade-offs and may not be able to achieve sustainable futures *(established but incomplete)* {5.5.2, 5.5.4, 5.6.1}. The third type of pathways focus on radical social innovation to achieve local food and energy selfsufficiency and local supply of nature's contributions to people *(ecotopian solutions).* They emphasize local multifunctionality, green infrastructure, urban design and

food production *(established but incomplete)* {5.5.2, 5.5.4, 5.6.1}. The fourth type of pathways emphasize a change towards diverse values, promoting resource-sparing lifestyles, continuous education and innovative forms of agriculture where different knowledge systems combine with technological innovation *(transition movements).* They achieve transformation using social and information-based policy instruments focusing on participatory processes, community actions and voluntary agreements. Rights-based instruments and customary norms, including indigenous and local knowledge, are used in combination with legal, regulatory and economic instruments *(established but incomplete)* {5.5.3, 5.6.1}. Actions proposed in all of the pathways can be combined. For example, short-term, incremental actions in *green economy* and *low carbon transformation* pathways may pave the way for more transformative *transition movements* pathways *(established but incomplete)* {5.5.4}. Despite distinct differences, all pathways stress some of the governance options highlighted in section E, including mainstreaming, integrated approaches that cut across sectoral boundaries, awareness-raising tools, education and participation to facilitate multi-actor governance *(established but incomplete)* {5.5.3}.

Box SPM **4** **Evidence from this regional assessment for Europe and Central Asia relevant in the context of the Aichi Biodiversity Targets and the Sustainable Development Goals.**

The Strategic Plan for Biodiversity 2011–2020, including its 20 Aichi Biodiversity Targets under five Strategic Goals, provides a framework for the United Nations system, including national Governments and others, for management and policy development on biodiversity. The 2030 Agenda for Sustainable Development, with its 17 Sustainable Development Goals, sets out the broader strategy towards global sustainability for the United Nations. This assessment summarizes the progress that the literature has reported towards these goals, as far as they pertain to the region and as far as there is sufficient evidence.

Evidence relevant in the context of the Aichi Biodiversity Targets
Evidence suggests progress in addressing the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society (Strategic Goal A) *(established but incomplete)***, although subsidies with negative impacts have not yet been reformed** *(well established).* Public awareness about the importance of biodiversity and ecosystem services (Aichi Biodiversity Target 1) appears to be increasing. Progress has also been reported in integrating biodiversity and ecosystem services into planning processes and national accounting in Western and Central Europe (Target 2) *(established but incomplete)* {6.6.2}. Substantial reforms could reduce the negative impacts of subsidies **(Table SPM.4)** {4.4.1}. Increasing positive incentives

for conservation could also improve progress towards Target 3 (harmful incentives eliminated, positive incentives developed and applied) **(Table SPM.4)** {6.2, 6.4.1}. Several countries have implemented ecological fiscal reforms, with mixed results *(established but incomplete)* {6.2, 6.4.1, 6.4.2}, but some policy instruments continue to have negative environmental impacts *(well established)* {4.3.1}. Without complementary strategies for reducing the impacts of consumption and production, more efficient resource use alone is unlikely to render current production and consumption patterns sustainable (Target 4 - sustainable consumption and production) **(Table SPM.4)** {6.5.4, 6.6.2, 6.6.3.2}.

Pressure from direct drivers on biodiversity is unlikely to be reduced *(established but incomplete)* **and the use of biodiversity is not yet sustainable** *(well established)* **(Strategic Goal B).** The evidence base in Europe and Central Asia related to the global Aichi Biodiversity Target 5 (habitat loss halved or brought close to zero) shows negative trends in biodiversity in agricultural areas {3.3.2.9}, important ecosystems such as seagrass beds {3.3.4}, and many fish stocks {4.4.1}*(established but incomplete).* Target 5 (habitat loss halved or brought close to zero) could, however, be achieved for terrestrial biodiversity in all subregions through, inter alia, effective and representative protected areas (see Target 11), mainstreaming biodiversity considerations

into and across all sectors and policies and integrated conservation management *(established but incomplete).* Contributions toward Targets 6 (sustainable management of marine living resources) and 10 (pressures on vulnerable ecosystems reduced) for the deep-sea are hampered by increased habitat degradation, and declines in biodiversity and ecosystem functioning. More effective fisheries management and increasing protected areas could improve this situation *(well established)* {3.3.4, 6.5.3}. Current trends in freshwater and terrestrial biodiversity suggest that it is highly unlikely that Europe and Central Asia will be able to fully contribute to Targets 7 (sustainable agriculture, aquaculture and forestry), 8 (pollution reduced) and 9 (invasive alien species prevented and controlled) *(well established)* {3.4.3}.

Progress has been made toward improving the status of biodiversity by safeguarding ecosystems, species and genetic diversity (Strategic Goal C) through protected areas *(well established)*. The extinction risk of domestic breeds is increasing and the genetic diversity of cultivated plants is decreasing, in spite of measures to counter this *(well established)*. Overall trends in biodiversity are still negative, however. Europe and Central Asia appears to achieve protected area coverage of 17 per cent of its terrestrial surface (Target 11) {3.2.9}, notwithstanding great variability in the level of protection. The European Union already protects about 25 per cent of its terrestrial surface. There has been a general increase in the number and extent of marine protected areas in the region. In 2017, 15 countries protected more than 10 per cent of their marine waters, and 12 per cent of the Baltic Sea area is protected *(well established)* {3.3.4.7}. Other marine systems, especially those further from the coast, are less protected *(well established).* The ecological representativeness, connectivity and management of protected areas have improved, but most still lack management measures to protect biodiversity, such as no-take zones *(well established)* {3.3.4}. In spite of some progress, current trends in biodiversity make it highly unlikely that the region will be able to contribute fully to achieving Targets 10, 11 and 12 (extinction prevented) {3.4, 3.5}. Downward trends in the Red List Index (increasing aggregate extinction risk) and Living Planet Index (decreasing population trends) also indicate that Europe and Central Asia will not be able to fully contribute to meeting Target 12. Europe and Central Asia are contributing to Target 13 (genetic diversity maintained) through the development of safeguards for rare domestic breeds and germplasms of cultivated plants. The extinction risk of domestic animal breeds is increasing, however, and there is evidence of the genetic erosion of cultivated plants under modern production systems *(established but incomplete).*

The Europe and Central Asia region has not advanced in enhancing the benefits to all people from biodiversity and ecosystem services (Strategic Goal D), as a consequence of the deterioration of nature's capacity to provide certain contributions to people *(well established)* {2.2.5} and the unequal distribution of nature's contributions *(established but incomplete)* {2.3.4}. Owing to biodiversity trends in freshwater, marine and terrestrial ecosystems,

it is highly unlikely that Europe and Central Asia will fully contribute to achieving Target 14 (ecosystems and essential services safeguarded) {3.3} **(Figure SPM.6)**. Progress is being made towards contributing to Target 16 (Nagoya Protocol in force and operational). By 2014, when the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization to the Convention on Biological Diversity entered into force, eight parties to the Protocol (15 per cent) in Europe and Central Asia had ratified the Protocol, while by 2017, the number had grown to 25 (46 per cent), including the European Union {6.4.1}.

Enhanced implementation through participatory planning, knowledge management and capacity-building (Strategic Goal E) has been positive where the Aichi Biodiversity Targets have informed the development of national-level targets. This has not been achieved, however, where indigenous and local knowledge and practices have declined or not been fully respected in relation to traditional land use *(well established)*. The Aichi Biodiversity Targets have been translated into national-level targets in all but 13 countries in the region. This suggests progress towards Target 17 (national biodiversity strategies and action plans adopted as policy instruments) {6.4.1}. The practices and knowledge of indigenous peoples and local communities in Western and Central Europe have continued to decline since the 1960s and have often not been fully respected or even marginalized, in contrast to Target 18 (traditional knowledge respected) *(well established)*. Evidence suggests that the further mobilization of financial resources (Target 20) is key for increasing the success of policy to achieve biodiversity conservation objectives *(well established)* {6.3.2, 6.3.3, 6.4.1, 6.5.4, 6.6.2, 6.6.4}.

Evidence relevant in the context of the Sustainable Development Goals

Progress in contributing towards achieving the Sustainable Development Goals has generally been positive in Europe and Central Asia in terms of environmental protection, human health, food security and water security (particularly in Europe) {2.3.1, 2.3.2} *(well established).* Nature offers various contributions to good quality of life, supporting the achievement of Goal 3 (good health and well-being) *(well established)* {2.3.2}. Conversely, the consumption of natural resources in Western Europe has increased large-scale land acquisition in other parts of the world, including Eastern Europe and Central Asia *(established but incomplete)* {2.2.4, 2.3.1.1}. This may contribute to not achieving Goal 2 (zero hunger), Goal 7 (affordable and clean energy) and Goal 12 (responsible consumption and production). The erosion of indigenous and local knowledge and the associated decline in sustainable traditional land use threatens the region's contribution to accomplishing Goal 2 and Goal 4 (quality education) *(established but incomplete)* {2.2.3.1.2}. Future climate and land-use change will decrease water security (Goal 6 - clean water and sanitation), with the number of water-stressed countries in Europe and Central Asia expected to increase by 2030 *(well established)* {2.3.1.2}. Some advances have been made towards accomplishing

environmental protection goals (Goals 14 – life below water and 15 – life on land), but the negative trend of biodiversity especially in agricultural areas currently restricts progress towards contributing to Goal 15 {3.3.2.9}. Despite some recent progress, the conservation of at least 10 per cent of coastal and marine areas by 2020, a target under Goal 14, has not been reached for all marine systems *(well established)*, although it has already been surpassed in some coastal areas of the North and Baltic Seas and by 15 countries *(well established).*

Beyond the Aichi Biodiversity Targets and Sustainable Development Goals
Looking beyond the 2030 timescale of the Sustainable Development Goals up to 2100, scenario analysis highlights that the continuation of past and current trends in drivers (as represented in *business-as-usual* scenarios) will inhibit the region from contributing to the widespread achievement of goals similar to and including the Sustainable Development Goals. In contrast,

scenarios which focus on achieving a balanced supply of nature's contributions to people and incorporate a diversity of values are more likely to contribute to achieving the majority of such goals *(established but incomplete).* A continuation of the business-as-usual approach in Europe and Central Asia is expected to result in failure to contribute to achieving most of the Sustainable Development Goals (contribution to achieving 4 out of 17), and Aichi Biodiversity Targets (contribution to achieving 8 out of 20) *(established but incomplete).* Scenarios of *economic optimism* are expected to enable the region to contribute to achieving 8 of the Goals, but only 4 of the 20 Targets. Scenarios of *regional competition* are expected to enable the region to contribute to achieving only two of the Goals and only one of the Targets *(established but incomplete).* By contrast, scenarios of *sustainability* are expected to enable the region to contribute to achieving the majority of the Goals (14) and Targets (14) *(established but incomplete)* {5.4, 5.6.}. A more comprehensive visual summary is provided in **Figure SPM.11**.

E. Promising governance options for Europe and Central Asia

E1 **Mainstreaming the conservation and sustainable use of biodiversity and the sustained provision of nature's contributions to people into policies, plans, programmes, strategies and practices of public and private actors could be achieved with more proactive, focused and goal-oriented environmental action, including quantitative goals *(well established)* {6.1, 6.3, 6.4, 6.5, 6.6; Figure 6.15}.**

The conservation and sustainable use of biodiversity in the more than 80 per cent of landscapes and seascapes outside protected areas would benefit from embedding biodiversity considerations into policies, strategies and practices of public and private actors that impact or rely on biodiversity {Table 6.1; Figure 6.2, Figure 6.15}. These considerations are equally important inside protected areas. Although progress has been made towards mainstreaming by setting up, reviewing and updating biodiversity strategies and action plans at multiple levels, existing legislation in all economic sectors could be implemented more effectively {6.3, 6.4.1} **(Table SPM.4)**. Mainstreaming the conservation and sustainable use of biodiversity would benefit environmental policies {6.4.2}, economic sectors and business actors depending on, or influencing, biodiversity {6.4.1, 6.5, 6.6; Table 6.10} **(Table SPM.4)**. Opportunities to successfully mainstream biodiversity and nature's contributions to people, in public and private policy and

decision-making **(Table SPM.4)** {6.6, 6.6.1; Figure 6.13}, could be harnessed by: first, raising awareness of the dependence of good quality of life on nature, enhancing capacity-building and strengthening participation of affected actors in decision processes; second, defining policy objectives concerning the ecological, economic and sociocultural needs for achieving sustainable living, taking account of the diverse values of nature for different stakeholder groups; and, third, designing instruments and policy mixes to support the implementation of effective, efficient and equitable policy and decision-making for nature and a good quality of life {6.6, 6.6.1}. Taking the European Union Common Agricultural Policy as an example, a number of factors would increase the effectiveness, efficiency and equity of related policy instruments. These factors include a better definition of clear and coherent objectives for the Common Agricultural Policy, simultaneously addressing multiple ecosystem services; a more defined focus on biodiversity conservation and the delivery of nature's contributions to people at the landscape level; a more explicit disclosure of trade-offs and synergies between different objectives; and more balanced and transparent funding between the production of agricultural commodities and the delivery of public goods {6.5.1.3}.

Table SPM ④ Policy options and opportunities for mainstreaming the conservation and sustainable use of biodiversity and the sustained provision of nature's contributions to people in Europe and Central Asia.

Building on three key steps of mainstreaming, options and opportunities for mainstreaming are provided for seven policy and economic sectors. The evidence shows that biodiversity and nature conservation will benefit from being mainstreamed in environmental policies and all economic sectors and their policies and that nature's contributions to people will benefit from being mainstreamed in all economic sectors, as well as the conservation sector. The table synthesizes those policy options and opportunities from the sectoral analyses in chapter 6 that are relevant to all sectors. It can be used by

	Sectors	CONSERVATION				ENVIRONMENT [1]			
STEPS	OPTIONS AND OPPORTUNITIES — Subregions	WE	CE	EE	CA	WE	CE	EE	CA
STEP 1: Raising awareness	Encourage education, joint learning and common understanding								
	Promote information sharing, transparency, knowledge management and training								
	Make trade-offs and tipping points visible at the relevant spatial scales								
	Encourage participation and dialogue among different actors								
	Make diverse values visible through national and business accounting								
	Mainstream recognition of need for profound societal transformation towards sustainability								
STEP 2: Defining policy objectives	Adopt and translate international and regional targets and standards into national and local strategies and action plans								
	Improve integration and coherence of legislation, sectoral policies and planning processes, to account for trade-offs and synergies								
	Develop context appropriate targets and objectives to stimulate positive change								
	Increase transparency and participation of a wide range of actors including indigenous peoples and local communities in decision making								
STEP 3: Designing instruments and policy mixes	*Legal and regulatory instruments*								
	Define and ensure property and access rights and responsibility								
	Set up, adjust and enforce legal and regulatory standards to sustain biodiversity and nature's contributions to people								
	Set up areas to protect biodiversity and nature's contributions to people								
	Economic and financial instruments								
	Phase out harmful subsidies	NA	NA	NA	NA				
	Tax and charge negative environmental impacts	NA	NA	NA	NA				
	Redistribute public revenues considering ecological objectives								
	Reward socio-economic activities delivering public goods								
	Secure conservation financing					NA	NA	NA	NA
	Foster sustainable technological and social innovation								
	Social and information-based instruments								
	Promote eco-labelling and certification schemes and improve their transparency and accountability								
	Promote voluntary agreements and partnerships for responsible management, which include self-enforcement mechanisms								
	Promote sense of agency and efficacy through the enhancement of public participation								
	Support social norms that promote sustainable lifestyles and practices								
	Rights-based approaches and customary norms								
	Strengthen the use of indigenous and local knowledge and practices								
	Strengthen the consideration of cultural properties and heritage in protecting sites and landscapes					NA	NA	NA	NA
	Strengthen the use of Social License to Operate or similar approaches to recognize the needs of indigenous peoples and local communities								

1. Include the following policy areas: Marine and freshwater quality and quantity, flood management, air and wider environmental pollution (including eutrophication and acidification), waste management, mitigation of and adaptation to climate change, soil management and land degradation. Options and opportunities in rows left blank have been covered by the other sectors, also in relation to their environmental outcomes.
2. Include the following policy areas: Energy, mining, manufacturing.
3. Include the following policy areas: Health, education and research, transport, tourism, finance.

policymakers of the subregions as a checklist to identify potential for improvement and for new policy instruments not yet initiated within the subregion. Although they have scope for improvement, legal and regulatory instruments are the most widely applied policy instrument category in all sectors and subregions, emphasizing their role as the backbone of policy mixes. Social and information-based instruments have been partly implemented in some subregions. There is also considerable scope for new or improved economic and financial instruments. Rights-based approaches and customary norms are the least developed and applied instrument category, indicating knowledge gaps (see **Box SPM.5**) or possibly a lack of attention or even acknowledgement to indigenous and local knowledge and practices.

AGRICULTURE				FORESTRY				FISHERIES				EXTRACTIVE & MANUFACTURING [2]				SERVICES [3]			
WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA

WE = WESTERN EUROPE   **CE** = CENTRAL EUROPE   **EE** = EASTERN EUROPE   **CA** = CENTRAL ASIA

■ EFFECTIVELY IMPLEMENTED   ■ UNDER DEVELOPMENT OR STARTED   □ NOT ASSESSED

■ IMPLEMENTED WITH SCOPE FOR IMPROVEMENT   ■ NOT YET INITIATED   **NA** = NOT APPLICABLE

E2 Developing integrated approaches across sectors would enable more systematic consideration of biodiversity and nature's contributions to people by public and private decision makers *(well established)* {6.1, 6.2, 6.4, 6.5, 6.6, 6.6.4.1; Figure 6.2}. This includes further options to measure national welfare beyond current economic indicators, taking account of the diverse values of nature {6.6.3.1}. Ecological fiscal reforms would provide an integrated set of incentives to support the shift to sustainable development *(established but incomplete)* {4.3–4.8, 6.4.1, 6.4.2, 6.6.2}.

Conventional sectoral approaches are insufficient to tackle interlinked environmental, economic and social challenges. Actions in one sector may affect other sectors, because policy design, instrument choice, or policy implementation rarely consider trade-offs {6.2, 6.4.1, 6.4.2, 6.6, 6.6.4.1, 6.6.4.2; Box 6.1, Box 6.9}. Without coordination between, and sustainable management practices within, sectors, there is evidence that agriculture, forestry, fisheries, mining, energy, manufacturing and the services sector may exert negative impacts on biodiversity, on nature's contributions to people and on the livelihoods of indigenous peoples and local communities {4.2.2, 6.4.2, 6.5.1–6.5.5, 6.6.4.1; Table 6.6}. Taking individual sectors as an example, a mismatch has been detected between the low degree of forest sector integration with other policy sectors on the one hand, and on the other its high potential to contribute to policy integration {6.5.2.3}. While some instruments of the European Union Common Agricultural Policy support extensive management practices, others are less well suited to, or implemented by, particularly, Central European countries of the European Union, to support indigenous and local knowledge and practices of small and semi-subsistence farms in high nature value farmland {6.5.1.2}. With regard to economy-wide policy integration, reflecting the real changes in the diverse values of nature's contributions to people in national income accounts is one option to provide better information and help to mitigate trade-offs {6.6.3.1}. Another option would be complementing national income accounts with satellite accounts containing information on the costs of ecosystem degradation. Ecological fiscal reform that creates an integrated set of incentives by redirecting taxation from labour to environment, including ecological indicators in intergovernmental fiscal relations and by greening public expenditure programmes, could support the shift to sustainable development {6.4.1, 6.4.2, 6.6.2}. Designing, implementing and assessing instruments in relation to their role in the overall policy mix would help to mitigate conflicting policy goals and trade-offs {6.2, 6.4.1, 6.5.5, 6.6.1, 6.6.2, 6.6.4.1, 6.6.5.5; Box 6.1}. The use of proactive strategies, tools and methodologies to account for diverse values and criteria, and of participatory processes can support trade-off analyses and facilitate policy integration {6.4.1, 6.4.2, 6.6.4, 6.6.5}.

E3 Effective governance of biodiversity and nature's contributions to people would benefit from well-designed mixes of policy instruments, suited to the context *(well established)*. Legal and regulatory instruments are the backbone of policy mixes, and economic, financial, social and information-based instruments provide additional incentives for Governments, businesses, non-governmental organizations and citizens. Further efforts would help to develop better rights-based approaches {6.2, 6.3, 6.4, 6.5, 6.6; Figure 6.2; Boxes 6.2, 6.4} **(Table SPM.4)**. A key factor constraining the effectiveness of existing policy mixes is limited enforcement owing, for example, to a lack of human resources, institutional capacity and financial means, or corruption *(well established)* {6.3.1, 6.4.1, 6.4.2}.

Where legal and regulatory instruments are concerned, the ratification and implementation of international treaties and transboundary agreements provide strong impetus for improving national and subnational policies in all sectors {6.3}. Marine protected areas, however, need more attention {4.5.4, 6.4.1}. For freshwater ecosystems, the European Union Water Framework Directive is of particular importance for achieving a good status for surface and groundwater {6.3.2.3, 6.4.2, 6.5.1, 6.5.2, 6.5.3, 6.5.4, 6.6.3, 6.6.5.5}, although integration and implementation of such novel governance approaches often remain incomplete, and ineffective when member States retain existing structures and procedures without transferring responsibilities and power to the river basin authorities {6.4.2}. Similar structures have been developed in non-European Union countries, such as Ukraine, which share river basins with European Union countries {6.4.2}. Targeted spatial and urban planning integrated across sectors and scales can support the conservation of biodiversity and nature's contributions to people, and enhance the quality of life of urban dwellers {6.6.4.2}.

Economic and financial instruments complement regulatory and other policy instruments by balancing conservation benefits and costs between actors and regions *(well established)* {5.5.3, 6.2, 6.3, 6.4, 6.5, 6.6}. Improving existing policies and developing and implementing new policies could help to avoid biodiversity loss and ecosystem degradation *(established but incomplete)* {6.2, 6.4.1, 6.4.2, 6.5, 6.6.2, 6.6.5.2; Tables 6.5, 6.6} **(Table SPM.4)**. Since markets undervalue nature's contributions to people, economic and financial instruments aim to change the behaviour of businesses, land users, citizens and public-sector actors, through incentives and disincentives to correct price signals. Environmental taxes, charges and fees make environmental pollution and habitat degradation more expensive, thereby making the polluter pay, whereas payments for ecosystem services or compensation payments reward conservation-friendly behaviour that is

otherwise not profitable or affordable {6.4.1, 6.4.2, 6.6.5.2}. Reforming environmentally harmful subsidies in sectors that negatively affect ecosystems (e.g., agriculture, fisheries, energy) would support more cost-effective use of public funds in reaching conservation objectives. Innovative economic and financial instruments include biodiversity offsets and habitat banking, tax reliefs, ecological fiscal transfers and integrated funding for biodiversity and climate-change adaptation {5.5.3, 6.4.1, 6.4.2, 6.5.1–6.5.5, 6.6.2, 6.6.3.2, 6.6.5.2}. Economic and financial instruments are more effective if customized to relevant scales, from global to national and local conditions in achieving conservation targets, while considering social impacts {6.2, 6.4, 6.6.2, 6.6.5}.

Social and information-based policy instruments have the capacity to integrate environmental concerns and to trigger behavioural change at the local, national and international levels, and to include consumers and producers in policy development *(established but incomplete)* {6.2, 6.3, 6.4, 6.5, 6.6.5.3; Table 6.5, Table 6.6} **(Table SPM.4)**. Enhanced consumer awareness, media coverage, business commitment and sustainable government procurement have increased the market shares of certified products {6.6.5.3}. Progress with certification is more advanced in countries with developed market economies and less so in countries in economic transition **(Table SPM.4)**. Owing to the lack of compliance mechanisms and clearly assigned responsibilities, there is a trade-off between the effectiveness of certification schemes and their accountability and impact. Efforts to change social norms through education and information-based campaigns promoting pro-environment behaviour have also been important {4.5.3, 5.5.3, 6.2, 6.4.1, 6.4.2.3, 6.5.1.2, 6.5.2–6.5.5, 6.6.5.3}.

Rights-based instruments and customary norms are increasingly supported and promoted by a wide range of multilateral environmental agreements, and by human rights *(established but incomplete)* {6.2, 6.3, 6.3.2.5, 6.3.2.6, 6.4, 6.5, 6.6, 6.6.5.4} (Table SPM.4). Those instruments integrate rights, norms, standards, and principles into policy, planning, implementation and evaluation, and offer ways to reconcile biodiversity conservation and human rights standards {6.2; Table 6.2}. While decisions by multilateral environmental agreements are implemented at the national level, the recognition of human rights, and in particular the rights of indigenous peoples, in relation to sustainable use of biodiversity varies considerably between countries in Europe and Central Asia **(Table SPM.4)**. Further efforts would be required for the full integration of the fundamental principles of good governance; equalizing power relations and facilitating capacity building.

For all these instruments and their combination in policy mixes, ecosystem-based approaches, such as successfully implemented in the Norwegian system of fisheries

management {Box 6.11}, the concept of nature-based solutions, as promoted by the European Union, or the idea of a circular economy adopt a more systemic perspective to environmental problems rather than addressing single issues {2.2.1.7, 6.4.2.1}.

E4 **A wide range of actors and stakeholders is increasingly integrated into governance processes. This can have a positive effect on biodiversity and nature's contributions to people if the effectiveness, efficiency and equity implications of such integration are carefully monitored, evaluated and improved *(well established)* {6.2, 6.4, 6.5, 6.6}. Lack of adequate financing is a major constraint on efforts to achieve biodiversity conservation and ecosystem restoration *(well established)* {6.4.1}.**

The role of multi-actor environmental governance is recognized in Western and Central Europe, and increasingly also in Eastern Europe and Central Asia. In parallel to top-down governance, decision-making concerning biodiversity and nature's contributions to people is increasingly devolved to public-private partnerships, co-management arrangements or even private governance, involving many stakeholders {6.2, 6.4, 6.5, 6.6; Tables 6.1, 6.8}. Promising developments include the establishment of new protected areas, and the protection of cultural landscapes through the United Nations Educational, Scientific and Cultural Organization (UNESCO) World Heritage Convention, the European Landscape Convention, and the International Union for Conservation of Nature (IUCN) protected landscape approach, where various forms of knowledge are integrated into management. Assessing the effectiveness, efficiency and equity of promising governance arrangements and taking power relationships and asymmetries into consideration require careful evaluation and monitoring {6.2, 6.4.2.2, 6.5.1.2, 6.5.1.5, 6.5.1.6, 6.2.2.2; Table 6.8; Boxes 6.7, 6.11}. This holds especially true for environmental governance in Central Europe, Eastern Europe and Central Asia with their rapid transformation processes since the early 1990s, moving from hierarchical, state-dominated processes to more collaborative governance processes {6.4.2; 6.5.1.4}. Another key challenge for policy success is posed by sufficient mobilization of financial resources. Increased funding from both public and private sources, together with innovative financing mechanisms, such as ecological fiscal transfers, would help to strengthen institutional capacities; to invest in research, training, capacity-building and education; to employ necessary staff; and to secure monitoring activities {6.3.2, 6.3.3, 6.4.1, 6.5.4, 6.6.2, 6.6.4}.

E5 **Dealing with change is a matter of societal choice (see D1). The way in which we choose to organize our societies and institutions, in both public and private spheres, is key to the realization of**

pathways towards the sustainable future envisioned by a diverse range of actors in Europe and Central Asia *(well established)* **{6.6.6}.**

The design of promising governance options and smart institutional arrangements supports the effective involvement of different actors in policy and decision-making with the aim of promoting shared responsibility for our common future. Developing pathways and corresponding experiments in a participatory manner, including all relevant stakeholder groups and indigenous peoples and local communities, enables the inclusion of a diversity of perspectives and promotes the necessary deliberation of strategic planning and agenda-setting {5.4.3, 5.5.1, 5.5.2, 5.5.6, 5.6.2}.

Governing direct and indirect drivers in complex adaptive systems, a process which often includes various forms of incomplete knowledge, would benefit from limiting institutional failures and promoting policy processes that stimulate adaptation and learning. Hence, policies, programmes and strategies may be seen as experiments that require governance and management for – rather than against – change, and systematic monitoring and evaluation. This can be achieved incrementally through adaptive governance and management and the systematic improvement of policy implementation, or via transition governance and management, and the organization of evolutionary processes of societal change {6.2, 6.4.2, 6.6, 6.6.6}.

Box SPM ⑤   ## Key knowledge gaps.

In the course of conducting this assessment, key information and data were not always available. Knowledge gaps are especially acute in the subregions of Central Asia and Eastern Europe, and in the Balkan countries in Central Europe {1.3, 1.6.1, 3.6, 5.6.2}. If future assessments are to provide a more comprehensive account of the status and trends in nature and its contributions to people, the following knowledge gaps would need to be addressed:

- **Gaps in our understanding of nature's contributions to people:** There is a need for better understanding, quantification and integrated monitoring of the diverse values of nature's contributions to people. Moreover, there is limited understanding of how these diverse values are endorsed by different social groups and genders. Indigenous and local knowledge systems and scientific knowledge could co-produce such understanding in the future {2.5}. There is also a lack of understanding about how biodiversity contributes to ecosystem services, especially in marine systems.

- **Gaps in our understanding of the contribution of indigenous and local knowledge:** Little research has been conducted on the integration of indigenous and local knowledge into national and international policy frameworks and initiatives to create synergies across knowledge systems. These knowledge gaps exist not only for biodiversity, but also in sectors of direct relevance to biodiversity, such as agriculture, forestry, fisheries, water and climate change {6.4.1.3, 6.4.2.4, 6.6.2}.

- **Gaps in our understanding of the status and trends of nature:** These gaps include habitat extent and intactness, and species conservation status and trends for the whole region, but critically for Eastern Europe and Central Asia. In addition, systematic and integrated biodiversity monitoring of fungi, non-vascular plants, invertebrates, marine and freshwater species and soil organisms are required to

better assess the status and trends for the whole region. Monitoring ecosystem functioning and species interactions is necessary to better understand the cascading effects of biodiversity changes and anticipate ecological tipping points.

- **Gaps in our understanding of drivers of biodiversity change:** A better understanding is needed of ways in which combinations of interacting indirect and direct drivers influence biodiversity and nature's contributions to people in various contexts. Furthermore, it is critical to understand time lags in the effect of drivers on biodiversity and nature's contributions to people to comprehend their real impact. In addition, there is a key gap in the identification, quantification and assessment of trends in drivers over time owing to their high spatial and temporal variability. There are also gaps in understanding the impact of climate change in combination with context-specific drivers on biodiversity and ecosystem services, especially with respect to tipping points and planetary boundaries. Moreover, there are gaps in understanding of the effects of interregional flows, especially the effects of global trade on ecological footprints and invasive alien species {4.7.1, 5.6.2}.

- **Lack of integrated scenario and modelling studies:** Scenarios rarely account for effects of multiple drivers and their interactions on impacts on the different components of biodiversity, nature's contributions to people and a good quality of life {5.6.2}. There is also a significant gap in terms of exploring the full range of synergies and trade-offs between the multiple aspects of biodiversity, ecosystem services and a good quality of life under different scenario archetypes and across different scales. It is also important to develop and couple process-based models of ecosystem functioning with the human dimensions of socioecological systems and to thoroughly evaluate these models, including the assessment of uncertainties {5.6.2}.

- **Gaps in the quantification and timing of pathways towards desired futures:** Pathways and envisioning studies are often not supported by modelling and, so, lack detailed quantification of goals and actions. Detailed description and sequencing of actions within pathways is rare, as is information on combinations of policy instruments to implement specific actions {5.6.2}. The incorporation of combinations of exemplary *transition movements* pathways into large-scale scenario exercises and into participatory scenario development is suggested as a way forward for better resolving trade-offs and for scaling-up local or sectoral solutions {5.6.2}.

- **Inadequate understanding of how to mainstream policy objectives within different sectors and integrate them across sectors and scales:** This requires a better understanding of the interaction between different policy instruments in existing policy mixes, not just the optimization of single instruments. More knowledge is needed about the effectiveness and efficiency of policy instruments that also consider institutional contexts, social impacts and how equity can be improved. There are further knowledge gaps on the effects of policy instruments on behaviour (e.g., of households and of companies) and on the economic and social systems within which these stakeholders operate {6.6.5}.

APPENDIX 1

Communication of the degree of confidence

Figure SPM Ⓐ ① **The four-box model for the qualitative communication of confidence.**

Confidence increases towards the top-right corner as suggested by the increasing strength of shading. Source: IPBES (2016).[11]
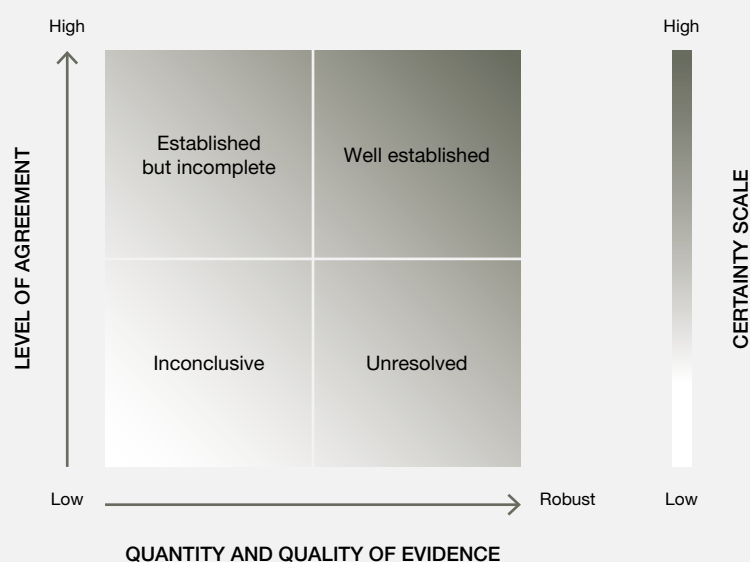
In this assessment, the degree of confidence in each main finding is based on the quantity and quality of evidence and the level of agreement regarding that evidence **(Figure SPM.A1)**. The evidence includes data, theory, models and expert judgement. Further details of the approach are documented in the note by the secretariat on the information on work related to the guide on the production of assessments (IPBES/6/INF/17).

The summary terms to describe the evidence are:

❯ **Well established:** comprehensive meta-analysis or other synthesis or multiple independent studies that agree.

❯ **Established but incomplete:** general agreement although only a limited number of studies exist; no comprehensive synthesis and/or the studies that exist address the question imprecisely.

❯ **Unresolved:** multiple independent studies exist but conclusions do not agree.

❯ **Inconclusive:** limited evidence, recognizing major knowledge gaps.

11. IPBES, Summary for policymakers of the assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production. S.G. Potts, V. L. Imperatriz-Fonseca, H. T. Ngo, J. C. Biesmeijer, T. D. Breeze, L. V. Dicks, L. A. Garibaldi, R. Hill, J. Settele, A. J. Vanbergen, M. A. Aizen, S. A. Cunningham, C. Eardley, B. M. Freitas, N. Gallai, P. G. Kevan, A. Kovács-Hostyánszki, P. K. Kwapong, J. Li, X. Li, D. J. Martins, G. Nates-Parra, J. S. Pettis, R. Rader, and B. F. Viana (eds.)., secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, 2016. Available from www.ipbes.net/sites/default/files/downloads/pdf/spm_deliverable_3a_pollination_20170222.pdf.

APPENDIX 2

Nature's contributions to people

This appendix describes the evolving concept of nature's contributions to people and its relevance to this IPBES regional assessment.[12]

Nature's contributions to people are all the contributions, both positive and negative, of living nature (i.e., diversity of organisms, ecosystems and their associated ecological and evolutionary processes) to the quality of life of people. Beneficial contributions from nature include such things as food provision, water purification, flood control and artistic inspiration, whereas detrimental contributions include disease transmission and predation that damages people or their assets. Many of nature's contributions to people may be perceived as benefits or detriments depending on the cultural, temporal or spatial context.

The concept of nature's contributions to people is intended to broaden the scope of the widely-used ecosystem services framework by more extensively considering views held by other knowledge systems on human-nature interactions. It is not intended to replace the concept of ecosystem services. The concept of nature's contributions to people is intended to engage a wide range of social sciences and humanities through a more integrated cultural perspective on ecosystem services.

Ecosystem services has always included a cultural component. For example, the Millennium Assessment[13] defined four broad groups of ecosystem services:

❯ Supporting services (now part of "nature" in the IPBES Conceptual Framework)

❯ Provisioning services

❯ Regulating services

❯ Cultural services

At the same time, there has been a long-standing debate in the ecosystem services science community, and in policy circles, about how to deal with culture. The social science community emphasizes that culture is the lens through which ecosystem services are perceived and valued. In addition, the groups of ecosystem services have tended to be discrete, while nature's contributions to people allow for a more fluid connection across the groups. For example, food production, traditionally considered to be a provisioning service, can now be categorized both as a material and a non-material contribution by nature to people. In many – but not all – societies, people's identities and social cohesion are strongly linked to growing, gathering, preparing and eating food together. It is thus the cultural context that determines whether food is a material contribution by nature to people, or one that is both material and non-material.

The concept of nature's contributions to people was developed to address the need to recognize the cultural and spiritual impacts of biodiversity, in ways that are not restricted to a discrete cultural ecosystem services category, but instead encompass diverse world views of human-nature relations. Nature's contributions to people also make it possible to consider negative impacts or contributions, such as disease.

There are 18 categories of nature's contributions to people, many of which closely map onto classifications of ecosystem services, especially for provisioning and regulating services. These 18 categories of nature's contributions to people are illustrated in **Figure SPM.2**. The 18 categories fall into one or more of three broad groups of nature's contributions to people regulating, material and non-material.

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12. Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R.T., Molnár, Z., Hill, R., Chan, K.M.A., Baste, I.A., Brauman, K.A., Polasky, S., Church, A., Lonsdale, M., Larigauderie, A., Leadley, P.W., van Oudenhoven, A.P.E., van der Plaat, F., Schröter, M., Lavorel, S., Aumeeruddy-Thomas, Y., Bukvareva, E., Davies, K., Demissew, S., Erpul, G., Failler, P., Guerra, C.A., Hewitt, C.L., Keune, H., Lindley, S., Shirayama, Y., 2018. Assessing nature's contributions to people. Science 359, 270–272. https://doi.org/10.1126/science.aap8826.

13. Millennium Ecosystem Assessment (2005). *Ecosystems and human well-being*. (Island Press, Washington, D.C.).

CHAPTER **1**

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CHAPTER **3**

STATUS, TRENDS AND FUTURE DYNAMICS OF BIODIVERSITY AND ECOSYSTEMS UNDERPINNING NATURE'S CONTRIBUTIONS TO PEOPLE

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STATUS, TRENDS AND FUTURE DYNAMICS OF BIODIVERSITY AND ECOSYSTEMS UNDERPINNING NATURE'S CONTRIBUTIONS TO PEOPLE

EXECUTIVE SUMMARY

Biodiversity loss impairs ecosystem functioning and, hence, nature's contributions to people (*well established*) (3.2.1, 3.2.2, 3.2.3). Sustainable delivery of these contributions requires the maintenance of genetic diversity, species diversity, and the diversity of ecosystems and land- and seascapes (*well established*) (3.2.4). The delivery of individual functions over time and at different places, and the delivery of multiple contributions, requires higher biodiversity than provisioning single services at one place and time alone (*well established*) (3.2.5). Higher biodiversity increases the capacity of terrestrial, freshwater or marine systems to capture resources, produce biomass, decompose and recycle nutrients, and to provide pollination (*well established*) (3.2.1, 3.2.2).

Higher biodiversity facilitates stable ecosystem functioning and improved capacity for evolutionary adaptation (*well established*) (3.2.3, 3.2.4). Higher biodiversity also increases ecosystem resilience and biological control of pathogens and invasive alien species (*established but incomplete*) (3.2.1, 3.2.3). To support ecosystem functioning, ecosystem stability over time, and adaptation to future environmental changes, biodiversity is required at different levels, from genetic and phenotypic diversity within populations, to diversity among populations and ecological or morphological types within species, species diversity and phylogenetic and functional diversity within communities, and diversity of communities, ecosystems and land and seascapes (*well established*) (3.2.3).

The higher the number of nature's contributions to people, the longer the time span, and the larger the area, the more biodiversity is required for their delivery (*well established*) (3.2.5). At the land and seascape and larger spatial scales, biotic homogenization, i.e. increasing similarity of the sets of organisms found at different places, reduces nature's overall contributions to people (*established but incomplete*), because of trade-offs between different facets of biodiversity and different contributions of nature to people (*well established*) (3.2.5). Thus, at the landscape and larger spatial scale the supply of multiple contributions of nature to people requires the maintenance and promotion of high biodiversity (*established but incomplete*). This implies high synergy at the land and seascape level between maintaining and promoting biodiversity and maintaining and promoting multiple contributions of nature to people (3.2.5).

Despite including some of the best-studied marine ecosystems, most of Europe and Central Asia's marine ecosystems, especially those deeper than 200 m, and most marine species are data deficient and their status and trends cannot be properly assessed (*well established*) (3.3.4). Of the assessed marine habitats and species, a high percentage are threatened (*established but incomplete*), varying between marine areas (*well established*) (3.3.4.1-7). The abundance, range and habitat size of many marine species is shrinking due to human pressures (*well established*) (3.3.4.1-7, 3.4.6.1). The distribution or phenology of many taxa has changed (*well established*) (3.3.4), including an "Atlantification" and "Pacification" of the Arctic Sea (*established but incomplete*) (3.3.4.5). Positive trends, mainly due to improved fishing practices or to a reduction in eutrophication, include increases in some fish stocks in the North Sea and in plankton diversity in the Black Sea (*well established*) (3.3.4.1, 3.3.4.4). Fifty-three per cent of the benthic shallow habitats in Western and Central Europe are data deficient. This figure is 87% in the Black Sea, 60% in the North East Atlantic, 59% in the Mediterranean Sea and 5% in the Baltic Sea (*well established*) (3.3.4.1-7). Of the assessed benthic habitats, 38% are classified as threatened (critically endangered, endangered or vulnerable), most of them in the Black (67%) and Mediterranean Seas (74%), followed by the North East Atlantic (59%) and Baltic Sea (8%) (*established but incomplete*) (3.3.4.1-7). In the European Union, among

assessments of the conservation status of species and habitat types of conservation interest, only 7% of marine species and 9% of marine habitat types show a "favourable conservation status". Moreover 27% of species and 66% of assessments of habitat types show an "unfavourable conservation status" and the reminder are categorized as "unknown" (*established but incomplete*) (3.3.4).

In Europe and Central Asia, 26% of the marine fish species have known trend data. Of those, 72% are stable, 26% have declining populations and 2% have been increasing over the last decade (*well established*) (3.4.6.1). Seabirds, marine mammals and turtles, and habitat formers, such as seagrasses and kelps, also declined in abundance (*well established*) (3.4.2-4). The distribution or phenology of marine phytoplankton, zooplankton, algae, benthic invertebrates, fishes, seabirds and mammals has changed (*well established*) (3.3.4). Such changes are particularly visible in the Arctic Ocean, where they were classified as "Atlantification" and "Pacification" with multiple ecosystem effects (*established but incomplete*) (3.3.4.5)**.** Many changes in species distribution or phenology lag behind the pace of climate change, however (*established but incomplete*) (3.3.4). Forty-eight per cent of marine animal and plant species with known population trends (436 decreasing, 59 increasing, 410 stable) have been declining in the last decade, increasing the extinction risk of monitored species (*established but incomplete*) (3.4).

Marine habitat and species trends are driven by individual and combined effects of overfishing, habitat degradation, climate change, pollution and invasive alien species (*established but incomplete*) (3.3.4.1-7). Invasion by alien species is observed in all marine areas of the region and is particularly fast in the Mediterranean Sea (*well established*). These invasions combined with species range shifts, are responsible for widespread biotic homogenization between subregions and systems (*well established*) (3.3.4.3). Invasive alien species, climate change and selective fishing reduce taxonomic and functional diversity by increasing generalist species and decreasing specialists (*well established*) (3.4). While fisheries are still the main driver of observed marine biodiversity loss across the region, e.g. in the Mediterranean Sea (*well established*), some fish stocks also improved due to decreased fishing pressure in some areas, e.g. the North Sea (*established but incomplete*) (3.3.4.1)**.** In some areas, eutrophication has decreased in recent years and e.g. plankton diversity of the Black Sea has recovered (*established but incomplete*) (3.3.4.4). Other forms of pollution, such as microplastics and noise, negatively affect marine animals, but a full assessment of their impact is still lacking (*established but incomplete*) (3.3.4).

Freshwater species and inland surface water habitats are threatened in Europe and Central Asia (*well established*). Only 53% of the European Union's rivers and lakes achieved good ecological status in 2015. 73% of the European Union's freshwater habitats have an unfavourable conservation status (*well established*) (3.3.3.1). Across Europe and Central Asia, lakes, ponds and streams are disappearing as a consequence of agricultural intensification, irrigation and urban development combined with climate change (*well established*) (3.3.3.1). The extent of wetlands in Western, Central and Eastern Europe has declined by 50% from 1970, while 71% of fish and 60% of amphibians with known population trends are declining (*well established*) (3.3.3.1, 3.4.5, 3.4.6.2). Over 75% of catchment areas in Europe and Central Asia are heavily modified and subject to multiple pressures, resulting in serious threats to biodiversity. In 2015, good chemical status, as defined by the European Union Water Framework Directive, was not achieved for surface water bodies by 22 European Union member States and only 53% of rivers and lakes had good ecological status, despite some improvements (*well established*) (3.3.3.1). Freshwater and saline lake species and habitats are the most threatened in the region. Most known population trends for freshwater and saline lake species have been declining, including fish, amphibians and invertebrates. In Western and Central Europe and the western parts of Eastern Europe at least 37% of freshwater fish and about 23% of amphibians are threatened with extinction. In the same area, freshwater invertebrates are also threatened, with the most threatened group being gastropods (45-70% of species threatened depending on whether or not data deficient species are considered threatened), bivalves (20 to 26%) and dragonflies (15 to 19%) (*established but incomplete*) (3.4.5, 3.4.6.2, 3.4.8).

The main drivers of trends in the biodiversity of inland surface waters are habitat destruction and modification caused by infrastructure for hydro-power, navigation, flood protection, agriculture, urban development and water abstraction; pollution from agriculture and industry; the introduction of invasive alien species and their pathogens; and climate change (*established but incomplete*) (3.3.3). Many lakes, ponds and streams are disappearing as a consequence of agricultural intensification, irrigation and urbanization combined with climate change (*well established*). Water bodies disappear particularly in the Mediterranean region and Central Asia. Lake Akşehir, for example, was among the largest freshwater lakes in Turkey, but has now completely disappeared due to loss of surface and ground water sources through intensive crop irrigation (3.3.3.1). The desiccation of the Aral Sea due to water abstraction for irrigation, followed by wind-borne pollution from former sediments, is globally considered as a major environmental disaster (*well established*) (3.3.3.2).

Water protection has progressed in Western and Central Europe, especially due to the European Union Water Framework Directive. The rate of wetland loss has slowed considerably in Central and Western Europe due to the implementation of binding nature conservation policies or the designation of conservation areas (e.g. Ramsar sites). Nevertheless, the deterioration of freshwater ecosystems is generally continuing in the region (*well established*) (3.3.3).

Most terrestrial species and natural habitats have long-term declining trends in abundance, range and habitat extent and intactness. This is mainly due to agriculture, forestry, transport infrastructure, urban development and climate change (*well established*) (3.3.2, 3.4). Most natural habitats have been declining in extent, especially subtropical and tropical forests with 20% left in Macaronesia and 10% in the Caucasus (3.3.2.5), with the highest loss occurred during the 20th century (*well established*) (3.3.2). These declines are generally continuing, albeit at a slower rate. Forests, grasslands and tundra have been the most impacted terrestrial habitats since the second half of the 20th century (3.3.2). Systematic assessments of habitat conservation status only exist for the European Union. There, 16% of terrestrial habitat assessments in the period 2007-2012 had favourable conservation status; 3% had unfavourable, but improving trends; 37% had unfavourable, but stable trends; 29% had unfavourable and declining trends; 11% had unfavourable status with unknown trend relative to the period 2001-2006 and 4% had unknown status (*well established*) (3.3.2.12).

Forty-two per cent of terrestrial European and Central Asian animal and plant species with known population trends declined in the last 10 years, 6% increased and 52% were stable (3.4.13) (*established but incomplete*).

The main causes of the decline of terrestrial species include habitat conversion and pollution due to agriculture and forestry practices, natural resource extraction, climate change and invasive alien species (*well established*) (3.4, 3.3.2). Loss of forest biodiversity continues due to loss of intact natural forest (*well established*), forest fires, loss of natural structures, such as dead trees (*well established*), fragmentation of populations (*well established*), loss of traditional forestry practices that created open forest (*well established*), increased number and strength of extreme weather events due to climate change (*well established*) and conversion of land use (*well established*). Since the 1950s, biodiversity has decreased in response to both abandonment of, and intensified use of, agricultural land (*well established* for Western Europe and Central Europe; *established but incomplete* for Eastern Europe and Central Asia) (3.3.2.9). The conversion of grasslands to crops and urban areas and conversion of semi-natural grassland to more intensively used pastures are among the main drivers of declining conservation status

of non-forested habitats and species (*well established*) (3.3.2.6). Climate change, including increased number and strength of extreme weather events, also accelerates turnover in species composition and species loss in all habitat types, shifts species distributions northwards and upwards on mountain slopes (*well established*), decreases the extent of glaciers (*well established*), decreases the extent of polar deserts with transformation to tundra (*well established*), expands deserts and shifts forest cover and types (3.3.2). Populations of invasive and alien species continuously increase in numbers, exacerbated in northern parts of Europe and Central Asia by climate change (*well established*) (3.3.2).

Drainage-based exploitation of boreal peatlands is gradually giving way to sustainable use, protection and restoration, while southern and mountain peatlands are still threatened by development (*well established*). Unique functions of peatlands such as carbon storage, water regulation and biodiversity maintenance are increasingly lost by drainage and over-utilization (*well established*) (3.3.2.8).

Europe and Central Asia has over half of all known breeds of domesticated mammals and birds, but 75% of local bird breeds and 58% of local mammal breeds are threatened with extinction (3.4.13). The species diversity of arable plants has decreased by 20% since 1950 in Western and Central Europe, and the abundance of rare arable plants has also decreased (*well established*) (3.3.2.9). The genetic diversity of plants cultivated *in situ* declined until the 1960s, due to the replacement of landraces by modern cultivars, and no further reduction or increase of diversity was observed after the 1980s (*well established*). The numbers of at-risk animal breeds have slightly declined since 1999, but exact quantification is hampered by the changing number of documented local breeds (*established but incomplete*) (3.4.13). From 1980 to 2013, the abundance of farmland common bird species decreased by 57% in Western and Central Europe, the abundance of grassland butterflies has declined since 1990 (*well established* for Western Europe) and there have been severe seasonal losses of honey bee colonies over the period 1961-2012 across Europe and Central Asia (*well established*) (3.3.2.9).

Between 44 and 68 recorded species endemic to Europe and Central Asia have become globally extinct since the 15th century (40-62 animals, four to six plants). In addition, between 20 and 88 recorded species have become regionally extinct in Europe and Central Asia (16-80 animals, one fungus and four to seven plants). 37 global extinctions involved marine and freshwater species and seven involved terrestrial species, while most recorded regional extinctions were of terrestrial species (*established but incomplete*). In addition to these extinctions recorded

at large scale, numerous extinction events were recorded at the country level (*well established*) (3.4.1). Around 13% of animal and plant groups living in Europe and Central Asia and comprehensively assessed by IUCN are endemic to the region (*well established*). Thirteen percent of species occurring in Europe and Central Asia with known conservation status are at high risk of extinction. Particularly threatened are mosses and liverworts (50%), freshwater fishes (37%), freshwater snails (45%), vascular plants (33%) and amphibians (23%). Of species endemic to Europe and Central Asia, 30% are threatened. The Central and Western European subregions have the highest percentages of threatened (13%) and endemic species (11%) and the highest percentage of threatened endemics (35%), with these percentages primarily driven by the many threatened endemic species in the Mediterranean hotspot and the Macaronesian Islands. Eastern Europe and Central Asia have lower percentages of species (<10%) and endemic species (<5%), and lower percentages of threatened endemics (<10%) (*established but incomplete*) (3.4.1).

The net change in extinction risk for mammals, birds and amphibians is 17 species moving one category closer to extinction every 10 years. Seven of these are in Western and Central Europe, six in Eastern Europe and four in Central Asia (*established but incomplete*) (3.4.13). From 2007 to 2012 the conservation status of 35 monitored plant and animal populations in EU-27 improved relative to the previous 6 years, versus 41 deteriorations (*well established*) (3.4.13). Overall, 118 monitored species of plants and animals in the European Union have unfavourable conservation status but improving trends, 572 have unfavourable conservation status and deteriorating trends and 905 have unfavourable status and stable or unknown trends (*well established*) (3.4.13).

In Western and Central Europe, the main drivers of recent past population declines across all realms are agriculture (use of biocides and chemicals affected 73% of assessed populations, intensification 42%, modification of cultivation practices 36%); reduction of habitat connectivity (55%); pollution of surface waters (56%); invasive alien species (46%); human induced changes in hydraulic conditions (43%); and forestry (removal of dead trees (39%), clearance (38%), logging of natural and plantation forests (38%)) (*well established*) (3.4.13). A separate assessment of threats to freshwater species found that at least 62% (n=13) of globally extinct species of European freshwater fishes were victims of water pollution and lake eutrophication. Destruction or modification of freshwater habitats, including water abstraction, affects 89% of amphibian threatened species and 26% of threatened freshwater invertebrates (*well established*) (3.4.5, 3.4.8). A quantitative assessment of drivers of biodiversity change in Central Asia and Eastern Europe was not possible due to a scarcity of data, but the same drivers with the addition of overexploitation (hunting,

trapping, fishing, harvesting) are reported as the main causes of known trends (*established but incomplete*).

Loss of taxonomic and functional diversity driven by increasing trends and expansion of generalist species and decline of specialists is documented across Europe and Central Asia and all taxa. On land, simplification of ecosystems through land-use intensification (agriculture, forestry, and urbanization) drives this phenomenon. In inland surface waters it is due to changes in water regime, eutrophication, salinity and introduction of invasive and alien species. In the seas, the main drivers are climate change, invasive alien species and fishing of selected species (*well established*) (3.3, 3.4). Loss of taxonomic, and even more so, of functional diversity driven by increasing trends and expansion of generalist species and decline of specialists is documented across Europe and Central Asia for all taxa (*well established*) (3.4). Biotic homogenization in agricultural areas has occurred for a range of biological groups, including birds, butterflies, cultivated plants, weeds, and domestic animals (*well established*). Intensification of forestry and urbanization also has resulted in biotic homogenization (*wel established*) (3.3, 3.4).

Bird communities have experienced extreme levels of biotic homogenization with near-extinction of habitat specialists, especially in grasslands of Western Europe and Central Europe due to landscape simplification. Other groups disproportionally affected are migratory species (hunting and trapping) and seabirds, due to bycatch from fisheries and predation by invasive species (*well established*) (3.4). Amongst forest plants, lichens, birds, mammals and arthropods show declines of specialists of old forests and of deciduous forests, and of cavity-nesters (3.3.2.3, 3.3.2.4, 3.3.2.5, 3.4). All these changes can be related to the intensification of forestry, which does not allow the development of structural elements benefitting specialist communities (*well established*) (3.4). Among freshwater fish communities, functional homogenization exceeds taxonomic homogenization sixfold. Species that are anadromous, slow-growing, large-body sized, diet or habitat specialists have been far more impacted than others. Body-size and specialization have also played a role in biotic homogenization of zooplankton communities (*established but incomplete*) (3.4). Large-bodied and other vulnerable marine fish species are the most threatened in large parts of Europe and Central Asia, and some have gone extinct (*well established*) (3.4.6.1).

Conservation efforts have shown the potential to reverse negative population trends (*well established*) (3.4.13). The long-term population trends of 40% of the breeding bird taxa in Annex I of the European Union Birds Directive are increasing compared with 31% for all breeding bird taxa (3.4.13). Charismatic mammalian mega-fauna,

such as the Amur tiger, Far-Eastern leopard, Iberian lynx, and European bison are all recovering from the brink of extinction because of dedicated conservation efforts (*well established*) (3.4.3, 3.4.13). The response of biodiversity to "ecologically-friendly" agricultural practices (stricter pesticide management, reduced tillage and organic farming) is generally positive, but depends on the landscape context, spatial scale of evaluation, and biological groups - with particularly beneficial effects on plants and pollinators (*well established*) (3.4).

Overall, impacts from direct drivers on biodiversity are maintained and the use of biodiversity is not sustainable in the region (3.3, 3.4). Progress has been made in the region in terms of the extent of protected areas (3.3). However, overall trends in biodiversity are still negative (3.3, 3.4). These trends suggest that the corresponding Aichi Biodiversity Targets and Sustainable Development Goals 14 and 15 are not likely to be met (*well established*) (3.3.2.12, 3.3.3.3, 3.3.4.8, 3.4.13). Aichi Biodiversity Target 5 (habitat loss halved or reduced) is unlikely to be achieved given the observed status and trends in extent and biodiversity of terrestrial, inland surface water, and marine habitat (3.3.,3.4). Based on current freshwater biodiversity trends, it is highly unlikely that Europe and Central Asia will achieve the respective Aichi Biodiversity Targets by 2020 (i.e. targets, 6-10) or Target 1 of the European Union Biodiversity Strategy (*well established*), in spite of some progress having been made (3.4, 3.3.3, 3.5.2). Although the rate of natural habitat loss (e.g. of wetlands) has slowed down in some Europe and Central Asia countries due to the implementation of binding nature conservation policies or the designation of sites (e.g. Ramsar), the decline in freshwater habitat continues (*well established*) (3.3.3). Achieving Targets 6 (sustainable management of marine living resources) and 10 (pressures on vulnerable ecosystems reduced) is hampered for the deep-sea by increased habitat degradation and declines in biodiversity (*established but incomplete*) (3.3.4). Achieving Aichi Biodiversity Target 11 for terrestrial ecosystems (at least 17% conserved through protected areas) appears to be on track, which is ensured for Western and Central Europe and likely to be met in Eastern Europe and Central Asia (Chapter 4). Despite some recent progress, Aichi Biodiversity Target 11 and target 14.5 of Sustainable Development Goal 14 have still not been reached for the marine systems of Europe and Central Asia (*well established*), although they have been surpassed in some coastal areas, e.g. of the Mediterranean and North Seas, and by 15 countries protecting more than 10% of their marine waters (3.3.4.8). Some marine systems, especially those further from the coast, are much less protected, however (*well established*). Downward trends in the conservation status of assessed taxa indicate that the Europe and Central Asia region is not on track to meet Target 12, in spite of some decreasing trends in extinction

risk (*well established*) (3.4). Despite some progress towards Target 13 (genetic diversity maintained) by developing safeguards for rare domestic breeds and germplasms of cultivated plants, the extinction risk of domestic animal breeds is increasing and genetic diversity of cultivated plants eroding under modern production systems (*established but incomplete*) (3.4.13). Despite advances in protected areas (relevant in the context of Sustainable Development Goals 14 – life below water and 15 – life on land), the negative trends observed for biodiversity currently restrict progress toward Goals 14 and 15 (*well established*) (3.3).

Under business-as-usual scenarios of future global change, the extent of coniferous forests is expected to be maintained or even increase. Meanwhile, tundra, other Alpine ecosystems, Mediterranean ecosystems, and broad-leaved and mixed forests are expected to substantially contract, because of climate and land-use change. Alpine, Scandinavian, and Icelandic glaciers are projected to retreat (3.5.1.3) (*well established*). The expected range of glacier losses depends on climate modelling scenario and varies from 20% to 90% of the 2006 ice volume. Climate change is also expected to further increase the stress on freshwater ecosystems, not only by changing species distribution but also by exacerbating the symptoms of eutrophication due to loss of planktivorous species through warming and salinization (*inconclusive*) (3.5.2). Mean species abundance, local functional and phylogenetic diversity and betwee-sites taxonomic diversity are expected to decrease throughout the 21st century, while local taxonomic diversity is expected to increase in some terrestrial and marine regions as a result of climate-driven range shifts (*established but incomplete*) (3.5.1, 3.5.3). Across species, range contractions are projected to be between 10% and 55% depending on climate scenario and taxonomic group considered (*established but incomplete*) (3.5.1.1). Biomass productivity may increase in some areas due to CO_2 fertilization and temperature increase, especially in the Arctice seas, lakes and boreal forests (*unresolved*) (3.5.1, 3.5.2, 3.5.3).

If key knowledge gaps would be addressed soon, future assessments could provide a more comprehensive account of the relationship between biodiversity and nature's contributions to people and of the status and trends of nature (*well established*) (3.6). Much more information is available on the relationship between biodiversity and ecosystem services from experiments than from the field. Among the experiments those manipulating plant diversity were overrepresented compared with those manipulating other taxa, and most concerned grasslands or aquatic mesocosms. For experiments and field studies addressing the relationship between biodiversity and ecosystem services, comprehensive information across all types of nature's contributions is not yet available (*well established*) (3.2, 3.6).

A broader knowledge basis on trends in habitat extent, intactness and species conservation status was available for Western and most of Central Europe than for Eastern Europe, Central Asia and Balkan countries in Central Europe (3.4, 3.6). For example, exact extent, biodiversity status and trends are hardly known for most terrestrial and freshwater ecosystems in Eastern Europe and Central Asia, and the chemical status of 40% of Western and Central Europe's surface waters remains unknown (*well established*). Biodiversity status and trends are also poorly known for most marine habitats. E.g. 30% of coastal marine habitat assessments in the Mediterranean reported unknown conservation status. Only a minor fraction of the deep-sea floor and of known seamounts have been subject to biological investigation (*well established*) (3.4, 3.6).

Major gaps on status and trends of taxonomic groups concerned invertebrates, most marine and freshwater species, bryophytes, lichens, fungi and microorganisms. Of the estimated 32,000 vascular plant species of Europe and Central Asia, IUCN evaluated 2,483 (approx. 8%) in the Red List of Threatened Species. Of the estimated more than 2,000 bryophyte and more than 7,000 lichen species in the region only 14 and 5 species, respectively, have been evaluated in the IUCN Red List. For invertebrates in general, and freshwater invertebrates in particular, even the current status is available only for a minority of species. Almost a quarter of all European freshwater molluscs are data deficient, many of them likely to be threatened. 76% of freshwater fishes and 83% of freshwater molluscs assessed have unknown population trends (*well established*) (3.4). One to two thirds of marine species are still to be described. Status and trends for marine biodiversity are mostly unknown, even for coastal habitats. Accordingly, 50% of the assessments under the European Union Habitats Directive reported unknown conservation status for cetaceans and turtles and coastal marine habitats in the Macaronesian biogeographic region. And 30% of coastal marine habitat assessments in the Mediterranean reported unknown conservation status. Only a minor fraction of the deep-sea floor and of known seamounts have been subject to biological investigation (*well established*) (3.3., 3.4, 3.6). Indigenous and local knowledge on biodiversity trends was only partially available (*well established*) (3.6).

Due to lack of quantitative knowledge the relative role of drivers of change in determining trends in extent and intactness of habitats and in species diversity and abundance could only be attributed in terms of a coarse classification. Moreover, information is lacking on the interacting effects of several drivers on biodiversity (*well established*) (3.3, 3.4, 3.6)

These knowledge gaps greatly reduce the ability to monitor progress towards international biodiversity targets and to inform policy to avert further biodiversity loss. For example, current instruments such as the European Union Habitats Directive and Natura 2000 programme do not consider algae, fungi or lichens, and only a small fraction of invertebrates (*well established*) (3.6).

3.1 INTRODUCTION

This chapter assesses, for Europe and Central Asia, evidence for the general role of biodiversity for nature's contributions to people (3.2). Then it assesses the past and current status and trends of terrestrial, inland surface water and marine biodiversity by ecosystems (units of analysis) (3.3) and by taxa (3.4). This is followed by an assessment of future trends of terrestrial, inland surface water and marine biodiversity (3.5). Finally this chapter assesses knowledge gaps (3.6) in these respects.

Whereas Chapter 2 of the IPBES Regional Assessment for Europe and Central Asia identifies strong evidence that nature's contributions to people are declining, this chapter provides an assessment of the general underpinning of nature's contributions to people by biodiversity. Moreover, while Chapter 4 establishes that natural resource extraction, land-use change, pollution, climate change, and invasive alien species are the main direct drivers driving biodiversity change in general, this chapter assesses the status and trends of marine, inland surface water and terrestrial biodiversity for different units of analysis and for different taxa, and it attributes these trends to the direct drivers.

3.2 THE RELATIONSHIP BETWEEN BIODIVERSITY AND ECOSYSTEM FUNCTIONS AND SERVICES

3.2.1 General importance of biodiversity for ecosystem functions and services

Theoretical, experimental and field studies have proven that biodiversity is one of the key factors in determining the mean level and stability of ecosystem properties and functioning, such as biomass production, decomposition and carbon sequestration (Cardinale *et al.*, 2012; Tilman *et al.*, 2014). Clear evidence of biodiversity effects on ecosystem functioning has been obtained from experiments, which overall showed that the impacts of diversity loss on ecological processes are of comparable magnitude to the effects of other global drivers of environmental changes such as climate change, ultraviolet radiation, increase in the concentration of CO_2, nitrogen addition, droughts and fires (Cardinale *et al.* 2012; Hooper *et al.*, 2012; Tilman *et al.*, 2012). Experiments can even underestimate biodiversity effects because they do not assess important properties of natural systems that enhance the positive diversity effects, such as

complex trophic structures, complementary and mutualistic interspecific relations, non-random biodiversity loss and spatial heterogeneity (Cardinale *et al.* 2012; Duffy *et al.*, 2009; supporting material Appendix 3.1[1]). In addition, biodiversity effects increase with time and at larger spatial scales (Cardinale *et al.* 2012; supporting material Appendix 3.1[1]), which means they may be stronger in real-world systems than in experiments. On the other hand, the range of species richness loss studied in typical biodiversity experiments is far greater than real world biodiversity loss (Vellend *et al.*, 2013).

Comparative field studies have the great potential to show the relevance of biodiversity in real world ecosystems, but they are often not suitable for demonstrating the causality of observed relationships and have difficulties in distinguishing the effects of biodiversity, versus environmental drivers, on ecosystem functioning. Thus, the analysis of field observations needs to separate effects of diversity from other confounding factors (supporting material Appendix 3.1[1]).

In addition to the general consensus about the key role of biodiversity in ecosystem functioning, there is increasing information on the relationships between biodiversity and ecosystem services and, hence, nature's contributions to people (Balvanera *et al.*, 2014). A comprehensive systematic literature review (Harrison *et al.*, 2014) showed that the majority of relationships between biodiversity attributes and the selected 11 ecosystem services were positive. The key role of biodiversity was demonstrated for certain provisioning services (such as wood production in plantations, production of fodder in grasslands, and stability of fisheries yields); and regulating services (such as pollination, resistance to exotic plant invasions and plant pathogens, aboveground carbon sequestration, soil nutrient mineralization, and bioremediation of contaminated water and sediments) (Cardinale *et al.*, 2012; Harrison *et al.*, 2014; Science for Environment Policy, 2015; Thompson *et al.*, 2012). For many other ecosystem services (e.g. long-term carbon storage, suppression of pests and animal disease), the evidence for biodiversity effects is mixed or there are still insufficient data (Balvanera *et al.*, 2014; Cardinale *et al.*, 2012). Overall, however, the evidence to date suggests that sustaining the long-term flow of many ecosystem services will require high levels of biodiversity (Science for Environment Policy, 2015).

In Europe and Central Asia, field studies revealed generally positive effects of biodiversity on ecosystem services. A European Union-wide assessment (Maes *et al.*, 2012) showed that biodiversity indicators (mean species abundance, tree species diversity and the relative area of Natura 2000 sites) and ecosystem service supply (aggregated index of four provisioning services, five regulating services and one cultural service) were positively correlated with each

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1. Available at https://www.ipbes.net/sites/default/files/eca_ch_3_ appendix_3.1_additional_references.pdf

Figure **3** **1** **Large-scale relationship between biodiversity and ecosystem service supply in the European Union.**

A Biodiversity and ecosystem services maps. Top left: Total ecosystem service supply calculated as the sum of standardized values for 10 ecosystem service indicators. Top right: Mean Species Abundance. Bottom left: The proportion of protected areas which are part of the Natura 2000 network. Bottom right: The forest tree species diversity measured using the average Shannon Wiener Diversity Index. **B** Relationship between biodiversity and total ecosystem service supply. Biodiversity is represented using three spatial indicators: Mean Species Abundance (MSA), forest tree species diversity and relative surface area of the Natura 2000 network. Ecosystem service supply is represented by total ecosystem service value. Dots represent the average value of total ecosystem service value for equally distributed classes of the biodiversity proxies. Error bars represent standard deviations. Source: Maes *et al.* (2012).

A

ECOSYSTEM SERVICES
TESV

0.12 — 1.76   1.77 — 2.15   2.16 — 2.25   2.26 — 2.64   2.65 — 4.29

MEAN SPECIES ABUNDANCE
MSA (%)

< 19   20 — 31   32 — 47   48 — 70   > 71

NATURA 2000 NETWORK
PROPORTION OF LAND (%)

< 1   2   3 — 9   10 — 30   > 30

TREE SPECIES DIVERSITY
MEAN SHANNON INDEX

0.00 — 0.02   0.03 — 0.10   0.11 — 0.23   0.24 — 0.45   0.46 — 1.90
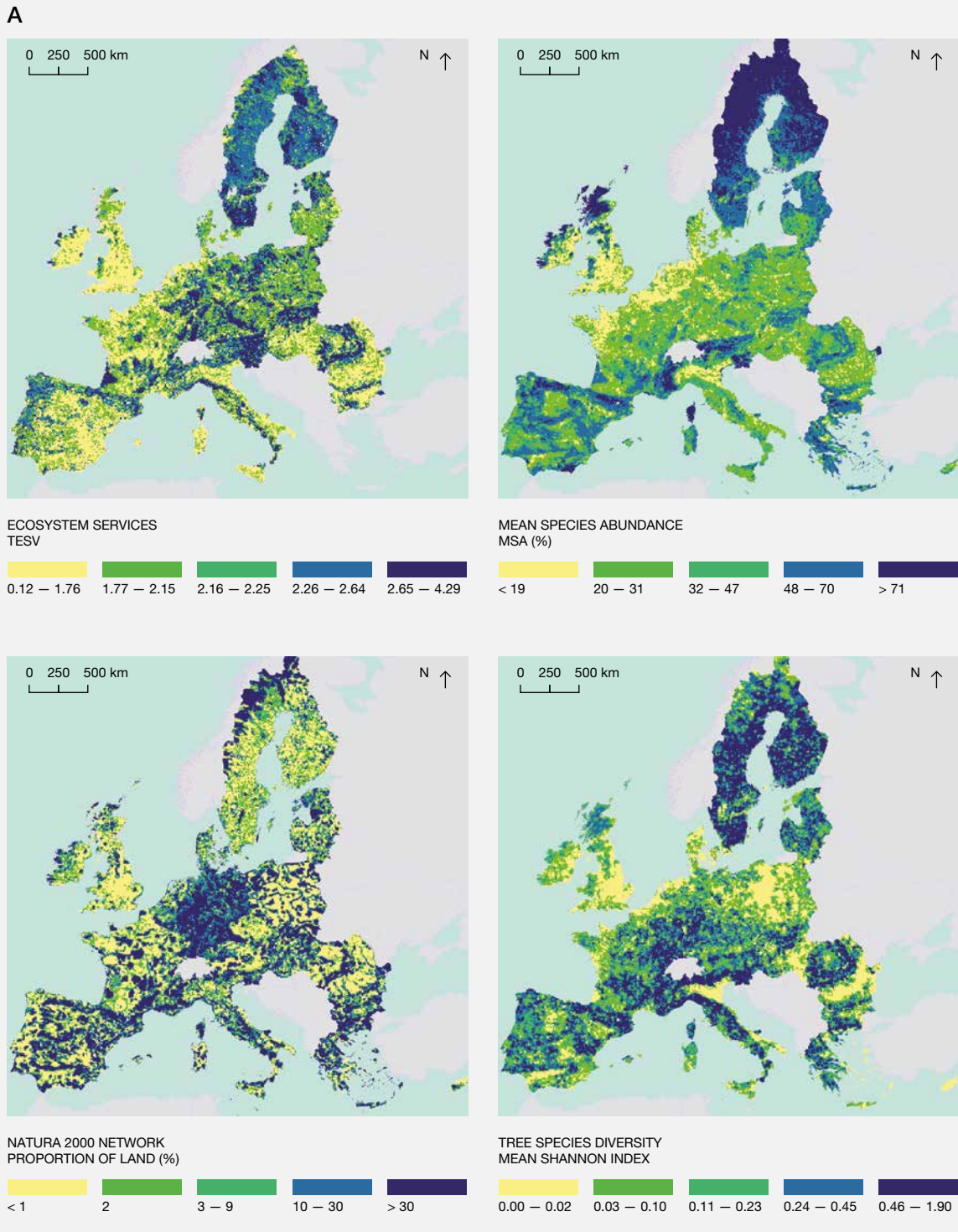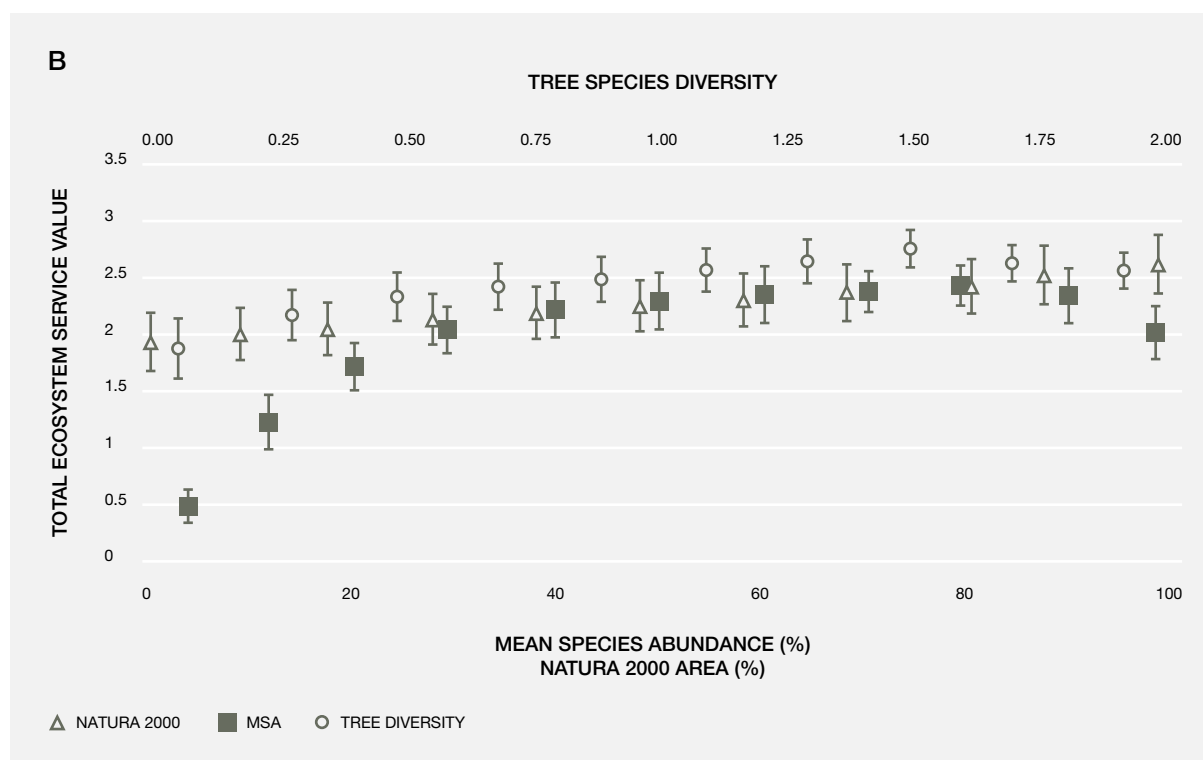
other **(Figure 3.1)**. Overall, habitats in a positive conservation status provided higher levels of biodiversity indicators and had a higher potential to supply ecosystem services, particularly regulating and cultural services, than unprotected areas.

An analysis of data of the UK National Ecosystem Assessment showed that biodiversity plays a key role in providing various types of ecosystem services: as a regulator of ecosystem processes, in providing final ecosystem services, and as a good with intrinsic value (Mace *et al.*, 2012).

An analysis of Swedish forest inventory data showed that relationships between tree species richness and several ecosystem services (production of tree biomass, soil carbon storage, berry production and game production) were positively linear to positively unimodal (Gamfeldt *et al.* 2013). Importantly, no single tree species was able to promote all services, emphasising the need for planting multiple tree species in forest stands to maintain multiple ecosystem services.

Regional studies in Finland (Hanski, 2014; Hanski *et al.*, 2012) confirmed that biodiversity increased immune regulation (von Hertzen *et al.*, 2011) and thus extended the view on ecosystem services to the field of maintaining human health. The findings suggest that loss of biodiversity reduces human exposure to beneficial environmental microbes, with essential immunoregulatory functions and, thus, leads to increasing prevalence of allergies and other chronic inflammatory diseases among urban populations worldwide.

3.2.2 Positive effect of biodiversity on the magnitude of ecosystem functioning

A scientific consensus has been reached that "there is now unequivocal evidence that biodiversity loss reduces the efficiency by which ecological communities capture biologically essential resources, produce biomass, decompose and recycle biologically essential nutrients" (Cardinale *et al.*, 2012). Both diversity within species (intraspecific diversity) and species diversity within communities are important for ecosystem functioning.

Numerous theoretical models describe how competition between individuals of both the same and different species predicts positive effects of species and functional diversity on biomass production and effectiveness of resource use (Tilman *et al.*, 2014). Several hypotheses predict that effects of diversity are more complex and variable in multi-trophic systems, i.e. of systems involving species of different trophic levels of the same food web (see supporting material Appendix 3.1[2]). Population genetics provides a theoretical foundation for the key importance of diversity within a population for population fitness (Lavergne *et al.*, 2010; Wennersten & Forsman, 2012) and thus, their capacity to provide ecosystem functions and services. Theory distinguishes two main classes of mechanisms by which diversity can positively affect ecosystem processes:

2. Available at https://www.ipbes.net/sites/default/files/eca_ch_3_ appendix_3.1_additional_references.pdf

Figure **3** **2** **The general form of the diversity-biomass production relationship. Effects of species richness on the standing stock abundance or biomass of the same trophic group.**

Each curve corresponds to data from a single study (grey circles and lines – terrestrial studies, black circles and lines – aquatic studies). Source: Cardinale *et al.* (2006). Reprinted by permission from Macmillan Publishers Ltd.
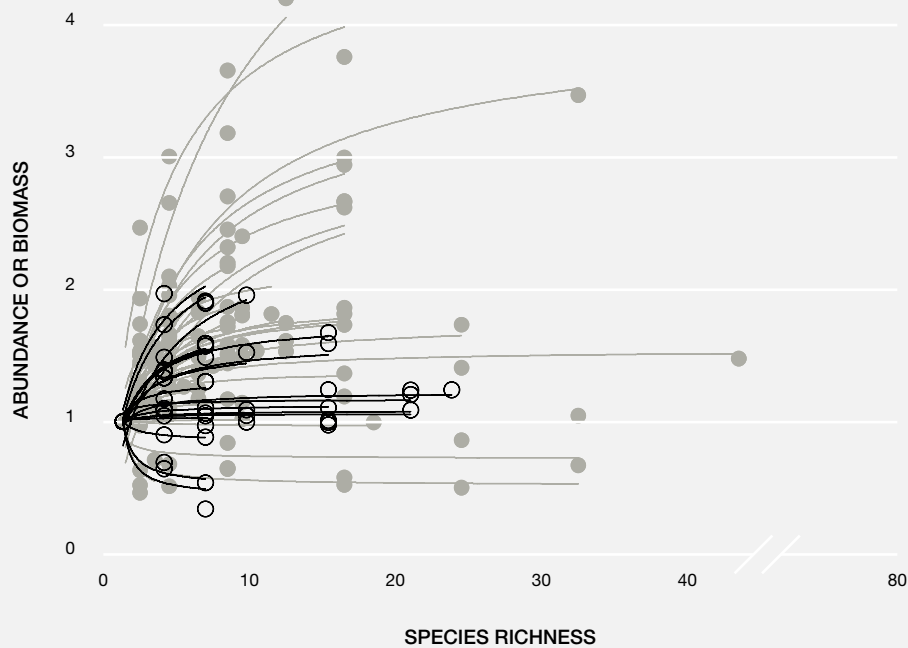
a) complementarity effects, i.e. functional complementarity in, for example, resource use of species or genotypes or phenotypes or due to positive (facilitative) species interactions; and b) selection effects, i.e. selection of particular functional traits of species or genotypes or phenotypes, with beneficial effects for ecosystem processes (for example the tendency of fast-growing plant species to become dominant in diverse communities) (Bolnick *et al.*, 2011; Forsman & Wennersten, 2016; Hughes *et al.*, 2008; Loreau, 2010).

Meta-analyses and reviews of hundreds of experiments revealed predominantly positive effects of species richness on community-level functioning (productivity, biomass, abundance, rate of nutrient cycling, invasion resistance, etc.). Negative effects were also found, but to a lesser extent **(Figure 3.2)** (Bardgett & van der Putten, 2014; Cardinale *et al.*, 2012; Gamfeldt *et al.*, 2015; Handa *et al.* 2014; supporting material Appendix 3.1[2]). Dozens of experiments with bacteria, plants, and invertebrate and vertebrate animals, showed positive effects of genetic diversity on ecosystem functioning (Forsman, 2014; Forsman & Wennersten, 2016; Hughes *et al.*, 2008; supporting material Appendix 3.1[2]). At the population level, high genetic diversity increases productivity, biomass, fitness, resistance and stability. At the community level, high genetic diversity (per species) decreases the probability of alien species invasions, disease levels, and increases the abundance and species

diversity of consumers. At the ecosystem level, high genetic diversity in dominant plant species increases decomposition rates and nutrient cycling (Forsman, 2014; Forsman, Wennersten, 2016; Hughes *et al.*, 2008).

Comparative field studies also demonstrated positive impacts of species and functional diversity on ecosystem functioning (productivity, biomass, aboveground carbon stocks, soil carbon content, nutrient cycling, resource use efficiency) in real-world terrestrial, freshwater and marine ecosystems across the world (Grace *et al*, 2016; Lewandowska *et al.*, 2016; Maestre *et al.*, 2016; Mora *et al.*, 2011; supporting material Appendix 3.1[2]). Field observations of plants, invertebrates, amphibians, reptiles, birds and mammals confirmed the importance of intraspecific diversity for population fitness and functioning (Forsman & Wennersten, 2016; Hughes *et al.*, 2008) that was also expressed in a decline of fitness and adaptability due to a loss of genetic diversity in small or anthropogenically disturbed populations (see supporting material Appendix 3.1[2]).

More specifically, comparative field observations also showed that positive biodiversity effects are widespread in Europe and Central Asia. Analysis of forests across Western and Central Europe revealed positive effect of tree species richness on biomass production (Jucker *et al.*,

Figure **3 3** **Shapes of relationships between biodiversity and ecosystem processes.**
From left to right: asymptotic saturated; positive linear; exponential; optimal.

A B C D

ASYMPTOTIC SATURATED POSITIVE LINEAR EXPONENTIAL OPTIMAL
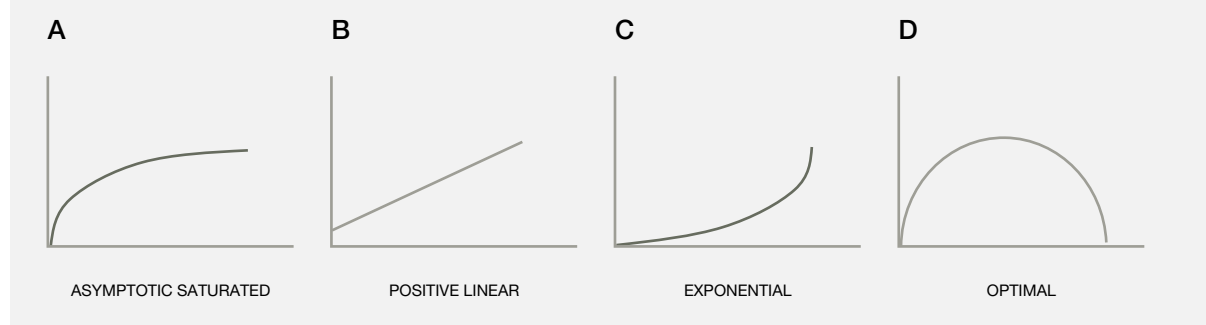
2014, 2016; Vilà *et al.*, 2013). Eight Western and Central European field studies of five animal groups (bees, carabid beetles, earthworms, soil nematodes and dung beetles), which deliver several key ecosystem functions (pollination, biocontrol of pests and weeds, bioturbation, nutrient cycling) revealed a positive relationship between functional diversity and ecosystem functioning provided by animals (Gagic *et al.*, 2015).

The shape of the relationship between biodiversity and ecosystem functioning is crucially important for ecological management. Most experiments that manipulated species richness revealed an asymptotic saturating relationship (see A in **Figure 3.3**) between diversity and ecosystem processes (Cardinale *et al.*, 2006, 2012). Most experiments that manipulated genetic diversity revealed a positive linear relationship (B in **Figure 3.3**) (Forsman & Wennersten, 2016). However, two recent large-scale field observational studies on sea communities detected exponential relationships (C in **Figure 3.3**) (Danovaro *et al.*, 2008; Mora *et al.*, 2011).

The asymptotically saturating pattern found in many experiments implies that the loss in ecosystem functioning accelerates as biodiversity loss increases. This suggests that the loss of a few species from a very species-rich community may have less deleterious consequences for ecosystem functioning than the loss of species from a species-poor community. In the case of a linear relationship, the loss of any species will equally decrease functioning. In the case of an exponential pattern, the loss of species will even cause an exponential decline in ecosystem functioning (Danovaro *et al.*, 2008; Loreau, 2008; Mora *et al.*, 2014). The unimodal shape suggests that there are optimal diversity values that correspond to maximum levels of ecosystem functioning, thus both a decrease and increase of diversity away from the optimal values leads to reduced ecosystem functioning. The optimal diversity values can often be regarded as typical for undisturbed populations and communities, which would suggest that the preservation of typical diversity may at the same time maintain ecosystem functioning (see 3.1.4).

Theoretically, the shape of the relationship between species richness and ecosystem processes depends on the degree of species niche overlapping and dominance - if species niches largely overlap (species are functionally redundant) the relationship is asymptotically saturating. If niches practically do not overlap (species carry out different functions) the relationship is close to linear (Loreau, 2000; Petchey, 2000; Tilman *et al.*, 2014). Mutualistic species interrelations can cause an exponential relationship (Loreau, 2008). The order of species extinctions also changes the shape of the relationship, particularly, saturating relationships are observed when species go extinct from the least efficient to the most efficient and exponential relationships when species are lost in the reverse order (see supporting material Appendix 3.1[3]). Unimodal relationships occur when ecosystem functioning peaks at intermediate biodiversity (D in **Figure 3.3**) and are predicted by some theoretical models (Bond & Chase, 2002; Bukvareva & Aleshchenko, 2013; Bukvareva, 2014). These were detected in some experiments manipulating genetic diversity (Caesar *et al.*, 2010; Burls *et al.*, 2014) and in wild populations of spruce and salmon (Altukhov, 2003). Experiments with communities of littoral psammophilous (i.e. sand-living) ciliates of the White Sea showed that the width of the group's trophic niche (i.e. the suite of used resources) was highest at intermediate species richness (Azovsky, 1989). Passy and Legendre (2006) found the highest biovolume (a surrogate for biomass) of algae at intermediate species richness in freshwater communities.

3.2.3 Effects of biodiversity on stability and resilience of ecosystem functioning

There is now a consensus that biodiversity increases the stability of ecosystem functions through time (Cardinale *et al.* 2012). Theoretical models predict that community stability is

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3. Available at https://www.ipbes.net/sites/default/files/eca_ch_3_appendix_3.1_additional_references.pdf

an increasing function of species richness, while population stability often decreases with species richness (Tilman *et al.*, 2014). Two main hypotheses known as the "portfolio effect" and the "insurance hypothesis" predict a stabilizing effect of species diversity. The "portfolio effect" posits higher likelihood of stabilization due to asynchrony in species responses to environmental fluctuations and stochastic

ecological mechanisms, where the decline of one species is compensated by an increase in another species (Loreau, 2010; supporting material Appendix 3.1[3]). The "insurance hypothesis", positing that more diverse communities have a higher likelihood that at least some species function well under various conditions, was supported by simulation models using data of Central European forests (Morin *et al.*,

Figure ③ ④ **A** Biomass stability in diverse grassland and forest communities is higher than within single species.

Communities had a lower inter-annual variability in total abundance than single species. The figure shows strong decreases in total abundance variability - and thus increased stability (arrows) - compared with the mean species variability, resulting from portfolio effects and species asynchrony. Four taxa with multiple species (arthropods, birds, bats and plants) in forests and grasslands were compared. Differences in stability between forests and grasslands in interaction with taxon were highly significant, whereas the relative stability gain between the two habitats was not. Source: Blüthgen *et al.* (2016).

B Community stability as a function of forest tree diversity.

The graph shows the fitted relationship between stability of aboveground wood production (AWP) and species richness across the entire plot network (black line) and for each site separately (coloured lines: Spain = red; Italy = orange; Germany = dark green; Romania = light green; Poland = light blue; Finland = dark blue). Source: Jucker *et al.* (2014).
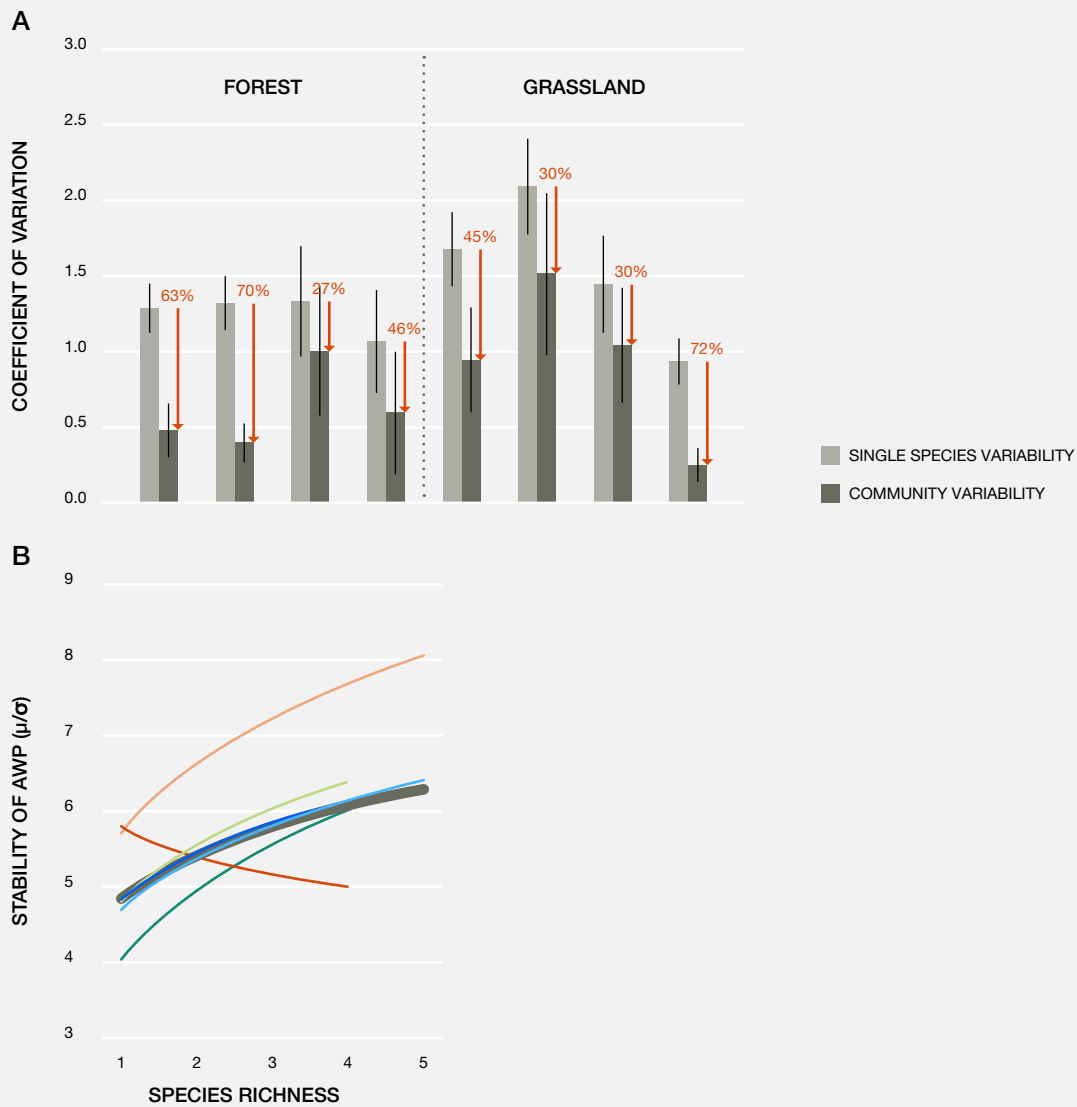
2014). Population models predict that similar mechanisms can provide stabilizing effects of intra-population diversity on populations and species (Bolnick *et al.*, 2011; Forsman & Wennersten, 2016; Hughes *et al.*, 2008; supporting material Appendix 3.1[4]). There is also evidence that resilience of ecosystem functioning (i.e. maintenance of ecosystem functioning under a range of environmental perturbations that could occur in the near future) is ensured by all levels of biodiversity - intraspecific genetic diversity, adaptive phenotypic plasticity, species diversity and spatial heterogeneity of habitats (Oliver *et al.*, 2015).

Grassland experiments fully confirmed theoretical assumptions showing that community stability increases with species richness due to averaging effects, while population stability decreases with species richness due to smaller population sizes (Griffin *et al.*, 2009; Gross *et al.*, 2014; Hector *et al.*, 2010; Tilman *et al.*, 2014; supporting material Appendix 3.1[4]). Furthermore, field observations in grasslands across five continents also showed positive relationships between species richness and stability in biomass production, but only in un-manipulated communities of non-fertilized grasslands (Hautier *et al.*, 2014). The importance of intraspecific diversity for population stability has been demonstrated for wild fish populations (see supporting material Appendix 3.1[4])*.

In Western and Central Europe, forest surveys showed that aboveground wood production is more stable in forests with higher tree species richness due to asynchronous responses of species to climate and due to greater temporal stability in the growth rates of individual tree species. Thus, the central role of diversity in stabilizing productivity was revealed for European forests (Jucker *et al.* 2014). Furthermore, studies of inter-annual fluctuations of 2,671 plant, arthropod, bird and bat species in German forests and grasslands demonstrated that species diversity provides community stability due to asynchronous changes in the abundance of different species (**Figure 3.4**, Blüthgen *et al.*, 2016).

3.2.4 Importance of all hierarchical levels of biodiversity

Measures of diversity other than species diversity have received less attention in literature on biodiversity and ecosystem functioning. However, intra-population diversity (i.e. genetic and phenotypic variation within populations) and intraspecific diversity (i.e. local populations, ecological and morphological forms composing species) are crucially important for fitness, adaptability and long-term viability of populations and species (Lavergne *et al.*, 2010). Maintaining the evolutionary perspective of species and ecosystems

is necessary to ensure ecosystem functioning and services into the future, while the loss of intra-population or intraspecific diversity undermines species' ability to adapt and evolve in a changing environment (Lavergne *et al.*, 2010; supporting material Appendix 3.1[4]). The loss of intra-population or intra-specific diversity also weakens and destabilizes ecosystem functioning (3.2.2 and 3.2.3). Diversity assessments that ignore intraspecific diversity may underestimate biodiversity changes and even lead to ineffective conservation practices. This is highly risky, since the loss of intra-specific diversity is already occurring and is projected to continue in the future in Europe and Central Asia (Balint *et al.*, 2011; Habel *et al.*, 2011; Neaves *et al.*, 2015; Pauls *et al.*, 2013; Taubmann *et al.*, 2011).

Experimental and field studies demonstrated that functional diversity (i.e. diversity of species functional traits or diversity of functional groups of species) is no less important than species diversity (Cardinale *et al.*, 2012; Cadotte *et al.*, 2011; Gagic *et al.*, 2015; Gamfeldt *et al.*, 2015; Gravel *et al.*, 2016; Lefcheck *et al.*, 2015; Mouchet *et al.*, 2010; supporting material Appendix 3.1[4]). Functional traits both of key species and rare species are important because the former have a large influence on community productivity (Cardinale *et al.*, 2012) and the latter can provide the most distinct trait combinations (Mouillot *et al.*, 2013). Phylogenetic diversity is the variation in the evolutionary origin of co-occurring species. It can be important for ecosystem functioning along with species and functional diversity (Cardinale *et al.* 2012; Mace *et al.*, 2003) (see supporting material Appendix 3.1[4]). Functional and phylogenetic homogenization across Europe were predicted for plants, birds and mammals, due to changes of climate and land use (Thuiller *et al.*, 2011, 2014b). Phylogenetic diversity over multiple taxonomic groups is considered as indicator of nature's contribution to people number 18, maintenance of options (Faith, 1992, Gascon *et al.*, 2015, Faith, 2017; Chapter 2, Section 2.2.3.4). Thus, accounting for these biodiversity facets appears important for the prediction of future ecosystem functions and services.

The structure of interspecific relations, including food webs, is also a key feature of biodiversity. Particularly, there is now consensus that the "loss of diversity across trophic levels has the potential to influence ecosystem functions even more strongly than diversity loss within trophic levels" (Cardinale *et al.*, 2012). Experiments and simulations demonstrated the importance of the structure of food webs for ecosystem functioning. For example, the loss of consumers at higher trophic levels can cascade through a food web to influence structure and functioning of the whole ecosystem (see supporting material Appendix 3.1[4]).

The diversity of ecosystems, communities and habitats is also of crucial importance for ecosystem functioning. Recent experiments demonstrated the importance of habitat

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4. Available at https://www.ipbes.net/sites/default/files/eca_ch_3_appendix_3.1_additional_references.pdf

diversity for ecosystem multifunctionality (Alsterberg *et al.*, 2017). Because biodiversity responds to environmental conditions and is itself driving ecosystem functioning (3.2.1), communities that are adapted to some conditions typically have high species diversity, while communities adapted to other, more stressful (e.g. Arctic), conditions have a low diversity. Global positive correlations between taxonomic diversity and temperature, evapotranspiration and other proxies of energy supply are well known (see supporting material Appendix 3.1[4]). Meta-analyses across similar communities, especially grasslands, at the global, or regional scales revealed positive, unimodal, and negative correlations between species richness and productivity (see supporting material Appendix 3.1[4]). However, this does not contradict the positive biodiversity effect on productivity within each local community (Loreau *et al.*, 2001; Schmid, 2002).

To maintain stable and effective ecosystem functioning in a landscape, maintaining undisturbed communities adapted to specific conditions (e.g. in peatlands, or rocky or sandy habitats) is required. Even though they typically may have lower species diversity than communities in other types of habitats, the diversity of undisturbed communities is still higher than the one of disturbed communities of the same habitat type. For example, Anderson *et al.* (2009) found that the distribution of carbon stocks in Britain was negatively correlated with species richness, as high carbon stocks were predominantly found in (inherently) species poor heathlands. In this case, communities typical of northern peat ecosystems, with low biodiversity, were likely most suitable for ecosystem functioning. Plant species from more diverse communities present in other habitat types are not adapted to the nutrient-poor conditions in peat ecosystems, and therefore do not function as well as the few species that are more typically found there. This case illustrates that the relevance of biodiversity for ecosystem functioning is revealed by comparisons between differently biodiverse ecosystems of the same type rather than by comparing between different types of ecosystems. Simply correlating biodiversity with ecosystem functioning across different ecosystem types ignores the fact that potential local biodiversity is not the same for all ecosystems, but depends on local environmental conditions (Schmid, 2002).

3.2.5 Long-term maintenance of multiple ecosystem functions and services

Maintaining multiple ecosystem processes at multiple places and times requires higher levels of biodiversity than does a single process at a single place and time, as shown by many studies (Byrnes *et al.* 2014; Cardinale *et al.* 2012; Hector & Bagchi, 2007; Isbell *et al.* 2011; Maestre *et al.* 2012; Zavaleta *et al.*, 2010). For example, Isbell *et al.* (2011)

demonstrated that 84% of the 147 grassland plant species in their study, including many rare species, promoted ecosystem functioning in at least one situation. Different species promoted different types of ecosystem functions, during different years, at different places, and under different environmental contexts. These results indicate that even more species will be needed to maintain ecosystem functioning than previously suggested by studies that have considered only the number of species needed to promote one function under one set of environmental conditions.

Inclusion of ecosystem multifunctionality (i.e. the provision of multiple ecosystem functions or services) in community models shows that multifunctional redundancy is generally lower than single-function redundancy. This means that a moderate loss of species can lead to a stronger loss of ecosystem multifunctionality than of individual ecosystem functions (Gamfeldt *et al.*, 2008). At the same time, the loss of multifunctionality due to biodiversity loss also depends on non-additive effects of biodiversity on individual functions (Gamfeldt *et al.*, 2017).

Field studies in Europe and Central Asia confirmed an important role of biodiversity for multiple ecosystem functions (ecosystem multifunctionality) in the real world. In a study across six Western and Central European countries, van der Plas *et al.* (2016a) showed that at local scales, relationships between local (so-called α) tree diversity and ecosystem multifunctionality can be either positive or negative, and strongly depend on how multifunctionality is quantified. However, larger scale (so-called β) tree diversity, quantified as the differences in tree species composition among sites, positively affected spatial turnover in the types of ecosystems that were provided at high levels (β-multifunctionality) and hence landscape-scale (so-called γ-) multifunctionality, across countries, emphasizing the scale-dependency of diversity-functioning relationships and the need for landscape-level forest diversity (van der Plas *et al.* 2016b) **(Figure 3.5)**. Hence, forest management leading to biotic homogenization can have negative consequences for large-scale ecosystem multifunctionality, whereas promoting forest stands varying in tree species composition will have positive influences on large-scale forest ecosystem multifunctionality.

A well replicated multisite study of 150 grasslands in Germany showed that plant biodiversity loss driven by land-use intensification also leads to loss of functions related to nutrient cycling, pest control, pollination and cultural services. While the effects on nutrient cycling, pest control and pollination varied among regions, effects of plant diversity loss consistently led to a loss in cultural services (Allan *et al.*, 2015). In the same grasslands, Soliveres *et al.* (2016a) revealed the importance of the diversity of locally rare species (plants, invertebrates, fungi, protists and bacteria) for ecosystem multifunctionality. Locally rare

Figure **3** **5** **Effects of local plot-scale richness of tree species (alpha-diversity) and the turnover of tree species among plots (beta-diversity) on local (α) multifunctionality, functional turnover (β multifunctionality) and landscape-scale (γ-) multifunctionality.**

Bars represent the standardized regression coefficients of α-diversity (light grey) and β-diversity (dark grey) in statistical models explaining α- **A**, β- **B**, or γ- **C** multifunctionality. Multifunctionality was quantified at different scales using a threshold approach, with thresholds of 40%, 50%, 70%, and 90%. In addition, sum-based γ-multifunctionality was calculated as the sum of scaled (between 0 and 1) individual function values. Diversity measures were calculated based on individuals of regionally common tree species. Source: van der Plas *et al.* (2016b).
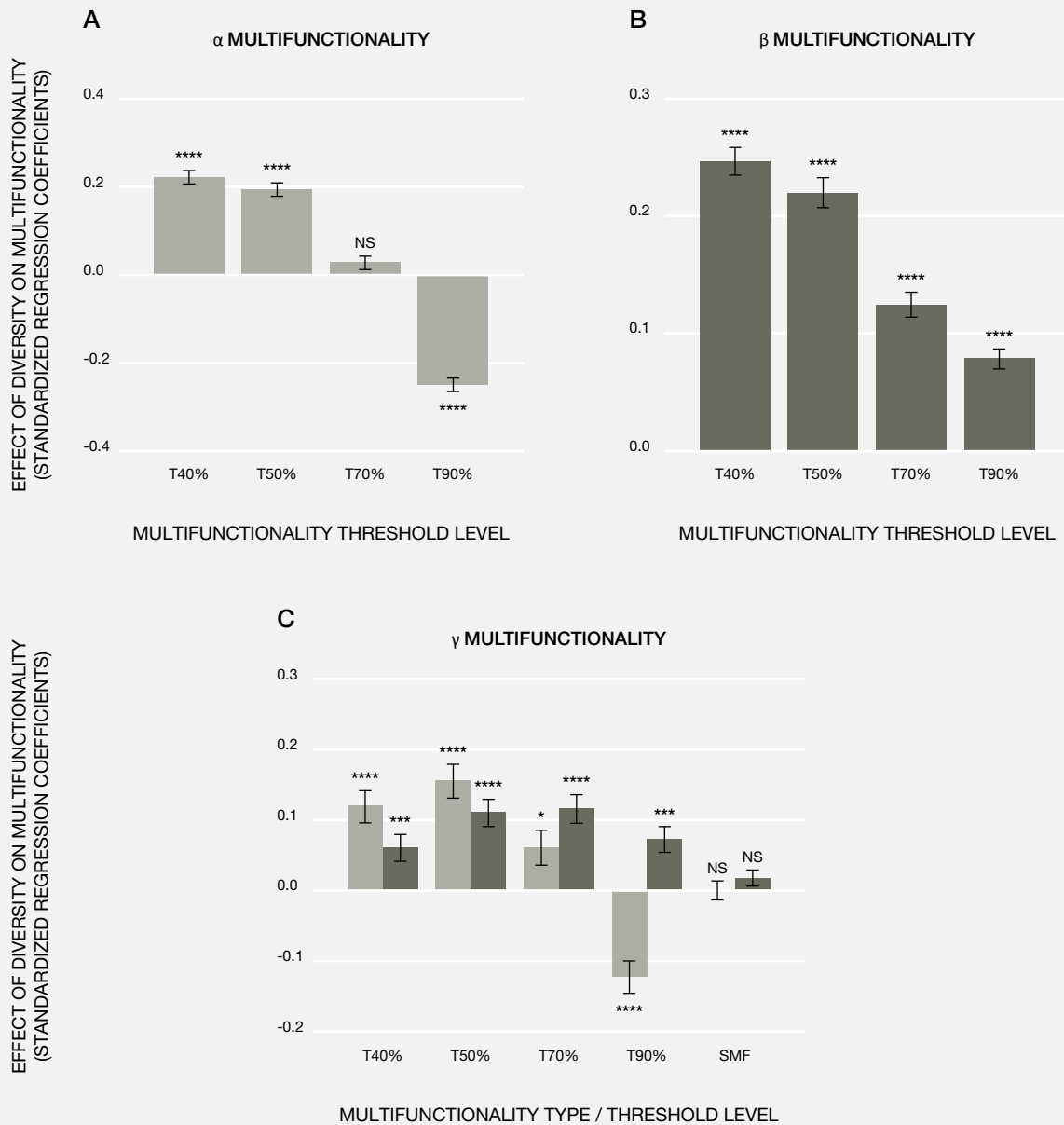
above-ground species were associated with high levels of multifunctionality, while common species were only related to average, not high, levels of multifunctionality. Furthermore, Soliveres *et al.* (2016b) showed that not only plants are important for multiple ecosystem functions and services, but that the diversity of other trophic groups, particularly aboveground herbivores and microorganisms,

is also extremely important for the maintenance of multiple ecosystem functions and services in grasslands.

Different ecosystem services profit from different types of management. Provisioning services often peak under intensive use of populations and ecosystems and at relatively low levels of biodiversity (Science for Environment Policy,

2015). Optimizing ecosystems for certain provisioning services, especially food, fibre and biofuel production has, however, greatly simplified their structure, composition and functioning across scales. While this simplification has enhanced certain provisioning services, it reduced others, particularly regulating services, and this simplification has led to major losses of biodiversity (Cardinale *et al.*, 2012). Mapping of four provisioning services, five regulating services and one cultural service across Western and Central Europe also revealed spatial trade-offs among ecosystem services, in particular between the provisioning service of crop production and regulating services (Maes *et al.*, 2011, 2012).

In summary, provisioning multiple ecosystem services requires maintaining and promoting high biodiversity within and between ecosystems. This implies high overall synergy between the goals of maintaining and promoting biodiversity and of maintaining and promoting multiple ecosystem services.

3.3 PAST AND CURRENT TRENDS IN BIODIVERSITY AND ECOSYSTEMS BY UNIT OF ANALYSIS

3.3.1 Introduction

Europe and Central Asia embrace a diversity of biogeographical regions from Arctic snow and ice-dominated systems in the north to Mediterranean forest and deserts in the south (Chapter 1, Section 1.3.4). The variety of the ecosystems also includes tundra, alpine and subalpine systems, temperate, boreal, tropical and subtropical dry and humid forests, peatlands, grasslands and deserts. The region also has important anthropogenic land cover types including agricultural and urban areas that are found across biogeographical regions. These categories are collectively referred to as terrestrial units of analysis, and in this section on past and current trends are addressed roughly sequentially from the north to the south of the region (Section 3.3.2), along with two examples of special ecosystems of relevance in the region, heathlands, and caves and other subterranean habitats. This is followed by a section on status and trends of biodiversity and ecosystems for inland surface waters (Section 3.3.3), which includes the categories of freshwater habitats and saline lakes. Finally, Section 3.3.4 addresses status and trends of biodiversity and ecosystems for marine systems, including the North Eastern Atlantic Ocean, Baltic Sea, Mediterranean Sea, Black and Azov Sea, Arctic Ocean, and North Western Pacific Ocean, and the

Deep Sea parts of the region and progress toward goals of Multilateral Environmental Agreements. The section is concluded by a box summarizing the trends for all terrestrial, inland surface water and marine systems in the overview **Table 3.5** and **Figure 3.43**.

3.3.2 Terrestrial Ecosystems

3.3.2.1 Snow- and ice-dominated systems

OVERVIEW OF THE SUB-SYSTEM

Glaciers and nival mountain belt

Currently glaciers are present in the high Arctic and in mountains in Europe and Central Asia. Glaciers extend for 55,800 km² in the Russian Arctic, 35,100 km² in Svalbard and 11,800 km² in Iceland. The average ice thickness varies from 280-300 m (Novaya Zemlya) to 200 m (Severnaya Zemlya) and 100 m (Franz-Joseph Land). Glaciers flowing into the sea break off forming icebergs in some coastal areas. In mountains, they extend for 25,400 km² in Scandinavia, the Alps, the Apennines and the Pyrenees, Siberia, the Caucasus, Altay, Tien Shan and Pamir (Milkov, 1977; UNEP-WGMS, 2008; Kotlyakov, 2010; AMAP, 2012; IPCC, 2013; Roshydromet, 2014; Zimnitskiy *et al.* 2015). The nival belt in mountains is characterized by extremely harsh conditions: low average annual temperature (<3.5°C) and a brief vegetation growing season (<10 days) (Körner *et al.*, 2011). In the higher mountains of the Europe and Central Asia region, "dry permafrost" in bedrock and moraines prevents the formation of continuous vegetation cover. In the northern Scandes, the lower limit for dry permafrost is currently at 1,300 m a.s.l. (Bockheim & Tarnocai, 1998).

Polar deserts

The Arctic deserts are spread over the far north of the Arctic Circle. The scant vegetation of the Arctic desert covers less than 50-60% of the soil surface, consisting of mosses, lichens, algae and a few species of higher plants (Milkov & Gwozdecky, 1969). These landscapes are common on Svalbard, Iceland, Arctic Ocean archipelagos and the Cheluskin Peninsula in Taimyr (Diakonov *et al.*, 2004; Matveeva, 2015).

The vegetation productivity here is negligible (Aleksandrova, 1983). Total biomass stock is less than 5 t/ha, dominated by above-ground biomass, thus distinguishing polar deserts from other habitats. Low vegetation productivity causes poor faunal diversity. At the extreme north of the zone only colonies of sea birds on rocky shores nest in summer and form so-called rookeries (especially on Novaya Zemlya and the Franz Joseph Land) (Milkov, 1977; Bliss *et al.*, 1981; CAFF, 2013).

PAST AND CURRENT TRENDS

The glaciation of the Russian Arctic has decreased by 725 km^2 in area and 250 km^3 in volume over the last 50 years, especially in western and central areas - 30% of it by icebergs and 70% by melting (Kotlyakov, 2010; Roshydromet, 2014). The mountain glaciers of southern Russia have decreased even more: by 40% in the Caucasus, 20% in the Altay and 30% in the Sayan Mountains relative to the mid-20th century (UNEP-WGMS, 2008). In the Alps, glaciers lost 35% of their total area from 1850 to 1970 and almost 50% by 2000 (Zemp *et al.* 2006). The lower limit for high alpine "dry" permafrost has been escalating rapidly over recent decades (IPCC, 2014a; Arctic Council, 2013).

Arctic deserts are extremely vulnerable to climate change because of greater than global average warming, decrease of ice and increase of permafrost melting in the Polar region. The warming and permafrost melting lead to more favourable conditions for plants, leading to an increase in species richness and productivity; and subsequently to the shift of vegetation type to tundra. While plant species richness increases, some vulnerable species are affected negatively and decline (Callaghan *et al.*, 2004; Callaghan *et al.*, 2005; Wolf *et al.*, 2008; IPCC, 2014a; Roshydromet, 2014). At the same time better climate conditions let people use natural resources more actively (Government of the Russian Federation, 2013).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Glaciers play an important regulating role for reindeer in the Altay Mountains in summer, as animals spend day time on ice to avoid blood-sucking insects. Shrinking of glaciers leads to concentration of reindeer in remaining places and limits their population size (Artemov *et al.*, 2013). The same function applies to snowbeds in the northern Scandes, where reindeer herds escape parasitic insects in warm summer days. This also allows them to see predators (e.g. wolverine) before they get close (Reimers *et al.*, 2006).

Reduction of the period during which the sea is covered in ice, means that polar bears are forced to stay on land for longer periods of time. Studies show that a one week shift in the ice melt in spring leads to 10 kg weight loss of a bear (Morgunov, 2011). Permafrost melting also leads to erosion of landscapes and destruction of the sea shore, as previously frozen surfaces become softer and more boggy. This can sometimes result in a fast decline in the area of islands – up to 10 m per year (Roshydromet, 2014).

Pollution and mining can have dramatic local effects, including complete destruction of vegetation. However, most Arctic deserts and mountain peaks are far from main industrial human activity, and are therefore not severely affected (CAFF, 2013; Shukurov *et al.*, 2015).

Tourism development, especially ski slopes, in high mountain ecosystems can cause their fragmentation, disturbance to animals, and land degradation in local plots (Sokratov *et al.*, 2014).

Poaching of rare charismatic animals for illegal trading leads to a decline of their populations in Arctic deserts, especially polar bears (classified as vulnerable A3c, Wiig *et al.*, 2015; CAFF, 2013). The snow leopard (vulnerable C1, McCarthy *et al.*, 2017) is another similar example of a species affected by illegal poaching and human-wildlife conflict in high mountains (Paltsyn *et al.*, 2012).

3.3.2.2 Tundra and mountain grasslands (only high elevation grasslands)

OVERVIEW OF THE SUB-SYSTEM

Tundra

Tundra refers to areas with permafrost, where the temperature is too low, precipitation too high and winds are too strong to allow for forest growth (Wielgolaski, 1972). Tundra is found on islands and on the mainland coast of the Arctic Ocean from the Kola Peninsula in the west to Chukotka in the east; and a vegetation belt in mountains from Scandinavia in the west to Kamchatka in the east and to Pamir and Tien Shan in the south (Milkov, 1977; Bliss & Matveyeva, 1992; Walker *et al.*, 2005).

Arctic tundra is a narrow strip along the ocean coast in Iceland, on many Islands in the Arctic Ocean and from the Barents Sea to Chukotka (Walker *et al.*, 2005). There are only two layers of vegetation, grasses and mosses, with some bushes and open soil (Diakonov *et al.*, 2004; Vasiliev *et al.*, 1941, Aleksandrova, 1970; Bliss *et al.*, 1981). Lichen and moss tundra is located in Iceland and in continental Eurasia, stretching in a band from the Kola Peninsula in the west to the Lena River in the east. Xerophilous and mesophilous mosses and some low shrubs are also abundant (Vasiliev *et al.*, 1941, Aleksandrova, 1970; Bliss *et al.*, 1981; Diakonov *et al.*, 2004). To the south on the continent the moss and lichen tundra is replaced by shrubs, commonly consisting of dwarf birches and bush willows. The lichen-moss layer contains more grasses and forest plants (Vasiliev *et al.*, 1941; Aleksandrova, 1970; Bliss *et al.*, 1981).

PAST AND CURRENT TRENDS

Remote and very slow naturaly recovering tundra areas were undisturbed by human impact for centuries. Currently

climate change affects the tundra through global warming, opening access into the Arctic. The overall trend is towards a greater human footprint (CAFF, 2013; Government of the Russian Federation, 2013).

The northward (and upward on mountains) range shift of species is also observed by both scientists and Arctic residents. Range shifts of plants averaging 6.1 km per decade toward the poles and 6.1 m per decade in altitude have been identified in response to a mean advancement of spring (initiation of greening) by two to three days per decade (Callaghan *et al.*, 2005; Morgunov, 2011, CAFF, 2013; IPCC, 2014a).

Lemming life cycles have changed in some Arctic regions probably due to changes in timing and quality of snow accumulation, with consequent impacts for lemming predators and alternative prey (Cornulier *et al.*, 2013, Henden *et al.*, 2010; Terraube *et al.*, 2011; Killengreen *et al.*, 2012; Terraube *et al.*, 2012; Schmidt *et al.*, 2012; Hamel *et al.*, 2013; Millon *et al.*, 2014).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Mechanical disturbance of the soil and vegetation cover leads to ecosystem fragmentation (Kumpula *et al.*, 2012), formation of artificial habitats and their colonization by weeds. Off-road driving with tracked vehicles poses a problem in parts of the Arctic, especially in Eastern Europe, where impacts on tundra vegetation can persist for decades following the disturbance (CAFF, 2013). Mechanical disturbances include thermokarst induced by the thawing of permafrost; freeze–thaw processes; wind, sand, and ice blasts; seasonal ice oscillations; slope processes; snow load; flooding during thaw; changes in river volume; coastal erosion and flooding. Biological disturbances include insect-pest outbreaks, peaks of grazing animals that have cyclic populations, and fire (Callaghan *et al.*, 2005). Overgrazing by domestic reindeer causes destruction of vegetation cover (Morgunov, 2011; Aleynikov *et al.*, 2014), - a widespread direct human-induced pressure on terrestrial Arctic in Europe and Central Asia (CAFF, 2013).

The Arctic stands out in terms of climate change effects on biodiversity (Callaghan *et al.*, 2005), including a prolongation of the growing season and an increase in productivity (for plants), nesting period (for birds), and warm season (for invertebrates). Climate change has led to a northward shift of the tundra-forest boundary; the extension of some species ranges, changing migration patterns; and to the introduction of alien species. An increase in the frequency of climatic anomalies such as winter thaw, summer frosts, increased precipitation, including snow, leads to the mass deaths of animals (e.g. reindeer and waterfowl) (Bhatt *et al.*,

2010; CAFF, 2013; Gauthier *et al.*, 2013; Hudson & Henry, 2009; IPCC, 2014a, 2014b; Morgunov, 2011; Raynolds *et al.*, 2006; Xu *et al.*, 2013).

Poaching and unregulated use of biological resources affect rare and vulnerable species. Polar bear poaching in Eastern Russian coastal tundra estimated at 100-200 animals per year (Kochnev, 2004; Morgunov, 2011; Kochnev & Zdor, 2014). While gathering goose down or hunting for birds and animals for food, local people may be unaware of the species national conservation status (Danilov-Danilyan *et al.*, 2001; Lavrinenko & Lavrinenko, 2006) and protection by law (Aleynikov *et al.*, 2014).

Intentional and unintentional introduction of alien species in Arctic ecosystems is ongoing. Fifteen alien invertebrate species, for example, have settled in Svalbard, many of them introduced via imported soils (Coulson, 2015).

Pollution by oil spills, mining or toxic waste dumps can transform or destroy vegetation cover and animal populations (Kumpula *et al.*, 2011; Virtanen *et al.*, 2002). Persistent organic pollutants and heavy metals accumulate in Arctic ecosystems, despite being produced and released in temperate and tropical regions, due to global atmospheric circulation (CAFF, 2013).

Subalpine and Alpine ecosystems

The alpine mountain belt is situated above subalpine and below snow and ice dominated ecosystems. The alpine vegetation comprises mainly perennial grasses, sedges, forbs, prostrate shrubs, cushions, tussocks, bryophytes and lichens (Körner, 2003; Körner *et al.*, 2011). It demonstrates high rates of local endemism (Grabherr *et al.*, 1995). Mountain tundra (as a variant of the alpine type) is most developed in Eastern Siberia, but can be found in all high mountains in Eurasia from the Urals to Kamchatka and from the Arctic to Tien Shan (Vasiliev *et al.*, 1941; Aleksandrova, 1970). Central Asian mountains contain a very specific variation of the alpine belt in extremely dry climate – alpine deserts in Pamir (Breckle & Wucherer, 2006).

The subalpine mountain belt is an ecotone zone between forest or steppe and alpine vegetation belts. It occurs at elevations from the sea level in the Kurily Islands in the Pacific Ocean up to 1,700-2,300 m in the Alps, Caucasus and Mediterranean mountains. The four main types of ecosystems in the subalpine belt are high-grass subalpine meadows; communities of dwarf bushes and shrubs; heathlands and grasslands consisting of short grasses; and subalpine thinned park type and crooked forests (Malyshev & Nimis, 1997).

Alpine and subalpine ecosystems stand out for their extremely high biodiversity. 20% percent (approx 2,500

species) of Europe's vascular plant flora were estimated to being predominantly alpine, i.e. occurring within only 3% of the continent's territory (Grabherr *et al.*, 1995; Väre *et al.*, 2003). Mountains around the Mediterranean basin, such as Sierra Nevada in Spain, are outstandingly rich in local endemic species (Pauli *et al.*, 2003) and there is a general south-to-north gradient of decreasing endemism in mountains across Europe (Favarger, 1972). The subalpine belt is especially diverse in mountains of Europe and Central Asia and includes a large part of endemic species. For example, in Central Asian mountains more than 600 species of vascular plants were found and 50 of them are endemics (Shukurov *et al.*, 2015, Kovalevskaya *et al.*, 1968-1993), in the Central Caucasus mountains the endemism level is higher: 197 from 595 species (Nakhutsrishvili, 2003).

PAST AND CURRENT TRENDS

The subalpine ecosystems in Europe and the Caucasus are strongly modified through a long history of human use. Humans converted large parts of subalpine woodlands into pastures and hay meadows, which resulted in a widespread increase in secondary grasslands below the tree line. The actual tree lines have shifted downwards especially in densely populated mountains such as the Alps and the Caucasus (Körner, 2003; 2012). Unlike in the Alps, in southern Siberia the altitudinal range of the subalpine belt is mostly conditioned by natural factors (Malyshev & Nimis, 1997).

In the Carpathians the subalpine scrub communities almost completely disappeared, being transformed into so-called polonina with matgrass (*Nardus stricta*) swards or communities dominated by blueberry (*Vaccinium myrtillus*) with very low plant diversity (Kricsfalusy *et al.*, 2008). Overgrazing in Tien-Shan in the second half of the 20th century was five to ten times over the tolerance limit (Shukurov, 2007). Pamir alpine deserts are 20% moderately degraded, 25% strongly degraded and 55% extremely degraded (Breckle & Wucherer, 2006). As a result, wild species were crowded out by livestock and their number has dramatically declined (Korotenko & Domashov, 2014). This subsequently led to a decline in the number of predators and scavengers (Shukurov, 2007).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Mountain meadows in Europe and Caucasus suffer from overgrazing, which leads to simplifying of ecosystem structure and decline of population abundance and species richness (European Commission, 2016).

Alpine grasslands today undergo rapid transition driven by changes in land use and climate. Thermophilous species

increased while cold-adapted high-elevation species declined in European mountains (Gottfried *et al.*, 2012; Grabherr *et al.*, 2010; Malanson *et al.*, 2011). Upward altitudinal shifts of alpine plant species ranges have repeatedly been observed in mountains (Klanderud & Birks, 2003; Kokorin *et al.*, 2011; Pauli *et al.*, 2012; Wipf *et al.*, 2013), which has led to increased species numbers on mountain tops in northern parts of Europe, but to declines in Mediterranean mountains (Jiménez-Alfaro *et al.*, 2014; Pauli *et al.*, 2012). The rate of tree line change varies across the region, usually several meters per decade and more than 100 m in Sweden and Norway (IPCC, 2014a).

Species population dynamics may lag behind climatic changes due to the persistence of some alpine plant species (Dullinger *et al.*, 2012). Abandonment of traditional farming and rural depopulation has become an evident trend in European and Caucasus mountains (Keenleyside *et al.*, 2010). The consequence is natural reforestation (Gehrig-Fasel *et al.*, 2008; Sitzia *et al.*, 2010), which reduces landscape heterogeneity, increases fire risks and exacerbates human-wildlife conflicts (Körner, 2003; Navarro & Pereira, 2015; Wilson *et al.*, 2012).

Landscape fragmentation and degradation as a result of the development of ski and tourism centres in high mountains have local negative impacts on biodiversity, species decline (especially rare and endemic species) and homogenization. Disturbance of vegetation on steep slopes may result in mudslides and water erosion (Belonovskaya, 1995; Akatov *et al.*, 2003). In Kyrgyzstan mining in high mountain ecosystems has led to degradation, fragmentation and pollution of vulnerable subalpine and alpine grasslands (Korotenko & Domashov, 2014; Shukurov, 2007).

Often changes are due to combinations of drivers. For example, species richness in Scottish alpine areas over a 20–40 year period increased in most habitats, while β-diversity declined, resulting in increased homogeneity of vegetation. Key northern and alpine species declined, while lowland generalist species increased. This change was consistent with impacts of climate change, but other elements of spatial pattern (decline in lichen richness in high deposition areas) were consistent with effects of nitrogen pollution (Britton *et al.*, 2009), which transforms species composition significantly (Bassin *et al.*, 2007).

3.3.2.3 Temperate and boreal forests and woodlands

OVERVIEW OF THE SUB-SYSTEM

Broad-leaved, mixed and coniferous forests constitute most of the potential natural vegetation in about 80% of

Europe (Bohn *et al.*, 2000) and Central Asia. Other patchily distributed forest types include water-influenced forests like black alder carrs and ravine forests on steep slopes. The vast area of boreal forest includes much of Fennoscandia, the middle and northern part of European Russia, Southern Siberia and far eastern part of Russia, covering ca. 809 million ha (Federal Forestry Agency, 2013).

PAST AND CURRENT TRENDS

The main past and current trends in biodiversity have been deforestation and fragmentation. In Western and Central Europe, conversion of deciduous forest to agriculture caused large breaks in connectivity and loss of the typical plant communities. Recovery of ecosystems lags far behind the efforts made in afforestation in these predominantly agricultural landscapes (Hermy *et al.*, 1999).

High biodiversity of various taxa of forest ecosystems is associated with natural disturbances like fires, wind and insect outbreaks, creating patches of dead trees and heterogeneity at different spatial scales. Up to 4,000 species are dependent on coarse woody debris as habitat (Stokland *et al.*, 2012). Protected areas focus on limiting human intervention in forests, with the aim of conserving species dependent on forest cover continuity, deadwood and large trees. Many bryophytes, lichens, fungi, vascular plants, saproxylic beetles and birds and cavity-nesters are associated with old forests (Bilz *et al.*, 2011; Moning & Müller, 2009, Paillet *et al.*, 2010; Roberge *et al.*, 2015; Gregory *et al.*, 2007; Scheidegger *et al.*, 2012; Virkkala *et al.*, 2008).

In the 18th century modern forestry began to reduce traditional forest pasturage, litter raking, charcoal making, pollarding and coppicing (Szabó, 2013). This replacement to high forests led to loss of species like threatened butterfly species associated with these open habitats (Konvička *et al.*, 2006). These changes have shifted the composition of understorey towards more shade-tolerant and nutrient-demanding species (de Frenne *et al.* 2013, Kopecký *et al.*, 2013) and forest vegetation has undergone significant loss of plant specialists (see e.g. De Frenne *et al.*, 2013; Keith *et al.*, 2009), loss of lichens (Reinecke *et al.*, 2014) and decreased multi-functionality at landscape-scale (van der Plas *et al.*, 2016a). This process is visible also in some coniferous forests (Hedwall & Brunet, 2016). Traditionally managed open forest habitats included in European Union protected habitats currently have unfavourable status (EEA *et al.*, 2016).

For the 2007–2012 period, the 27 European Union member States reported that only 26% of forest species and 15% of (non-Mediterranean) forest habitats (29 habitats) of European interest, as listed in the European Union's Habitats Directive, are in favourable conservation status (EEA, 2015a). An additional 7 have unfavourable but improving status with respect to the 2001-2006 period, 54 are deteriorating, 102 are stable or have unknown trends. In Central Europe, 248 assessments were performed (combinations of habitats and countries) and of these 56 were favourable, 16 unfavourable but improving, 46 unfavourable and declining, 123 unfavourable but stable, and 7 had unknown or unreported status and trends. In Western Europe, 380 status and trend assessments were performed, of which 66 were favourable, 25 unfavourable but improving, 83 unfavourable and declining, 125 unfavourable stable and 81 unknown or unreported. The most endangered habitats are forests along rivers and on bogs and water-influenced habitats such as ravine and boreal coniferous forests, riparian alluvial forests, lichen Scots pine forests, old acidophilous oak woods on sandy plain, Fennoscandian wooded pastures and swampy forests (Janssen *et al.*, 2016).

For people in remote forested areas, old-growth mountain or boreal forests are the only source of wood, and a source of food. Planted forests exclusively used for timber cover about 10% of the European Union (EEA, 2016). Throughout the forests of Europe and Central Asia, biodiversity is an important source of non-wood products (berries, mushrooms, game animals and recreation). Mustonen & Helander (2004) reported a decline of certain berry plants such as marsh whortleberries, traditionally collected by the Sami people in Finland.

A significant upward shift in the optimal elevation of forest herb and woody species occurred during the 20th century in various Western, Central and Eastern European forests, including primary forests (Engler *et al.*, 2011; Lenoir *et al.*, 2008; Šebesta *et al.*, 2011). Across several regions, the upper elevational limits of both tree seedlings and saplings were significantly higher than of adults (e.g. Vitasse *et al.*, 2012). However, despite the observed climate change, tree distribution of life-stage has not changed directionally (Máliš *et al.*, 2016). Drought is also known to be increasing fire risk in boreal forest (Drobyshev *et al.*, 2012) which, coupled with inadvertently human-caused ignition, can cause extensive wildfires.

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Direct drivers such as the expansion of infrastructure (urban and transport), unsustainable silviculture (including alteration and embankment of streams and spring drainages, excessive use of chemicals and clear-cutting and afforestation by monocultures of invasive species), conversion to agricultural land use and the lack of natural processes (e.g. floods in floodplain forests or fires in taiga forests) have affected forest ecosystems (Forest Europe, 2015) **(Figure 3.6)**.

Figure **3** **6** State and loss of intact forests in Russia in 2002–2015. Source: WWF (2015).

- **Intact forest areas in 2013**
- **Loss due to fires**
- **Loss due to logging**
- **Losses due to exploration**
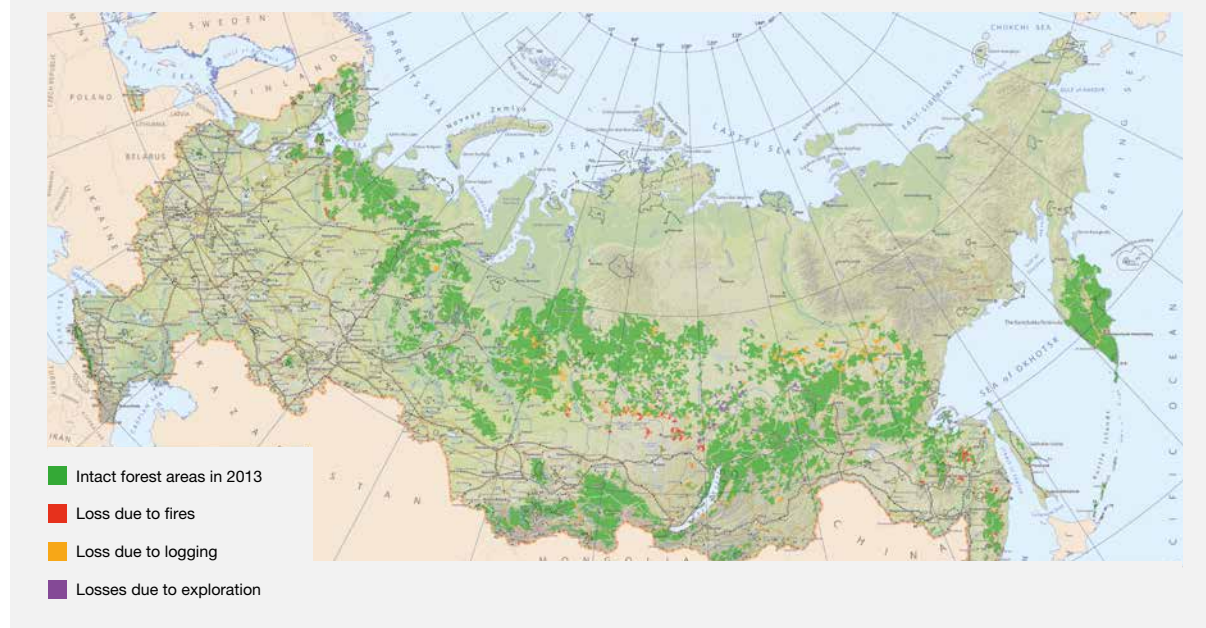
It is difficult to disentangle the influence of various drivers on forest indicators; however, repeated surveys (see e.g. initiative www.forestreplot.ugent.be) have revealed significant changes in species composition and distribution ranges.

In Central and Western Europe there have been trends of increasing integrated forest management for conservation of biological diversity by close-to-nature forest management without clear cuts to increase continuity of forest structures, and emulation of natural disturbances (creation of dead wood and natural rejuvenation (Kraus & Krumm, 2013). Large populations of game animals can decimate natural rejuvenation by browsing (Kuijper *et al.*, 2010) or rooting (Brunet *et al.*, 2016).

In addition, the current large tree plantations are prone to invasions by species in the forest understorey (Essl *et al.*, 2010; Pyšek *et al.*, 2009). Among the problematic invasive alien species, 33 (invertebrates, vascular plants and fungi) are regularly found in European Union forest ecosystems or are dependent on trees (EEA, 2016).

Past deposition of SO_2 caused acidification of soil in some areas (Krám *et al.* 2012), resulting in the widespread dieback of Norway spruce plantations and mountainous coniferous forests and associated decline of ectomycorrhizal fungi in the 1980s (Arnolds, 1991). Current atmospheric nitrogen deposition in areas of Central and Western Europe has caused soil eutrophication (Hédl *et al.*, 2011, Lomský *et al.*, 2012, Šebesta *et al.*, 2011), in general and and locally close to urban and industrial areas (Kotlyakov, 2000). This has caused changes in forest plant communities (Ewald *et al.*, 2013; Verheyen *et al.*, 2012).

3.3.2.4 Mediterranean forests, woodland and scrub

OVERVIEW OF THE SUB-SYSTEM

This unit stretches west to east from Portugal to Jordan, includes ecosystems on Madeira, the Azores and Canary Islands in the Atlantic Ocean (Conservation International, 2011; FAO, 2013b) and is characterized by cool wet winters and dry hot summers causing water stress (Allen, 2014; Gauquelin *et al.*, 2016). Similar conditions can be found in the Crimean Peninsula, Turkey, and in lower parts of the Caucasus and Central Asian Mountains, which are sometimes also considered as a part of the Mediterranean area (Takhtazhyan, 1978; Şekercioglu *et al.*, 2011). Further to the east, juniper and pistachio woodlands and scrub become the most common ecosystems (Fayvush & Aleksanyan, 2016; Şekercioĝlu *et al.* 2011; Government of Tajikistan, 2016; Government of Turkmenistan, 2015). In Central Asia, they are composed of pistachio-almond dry woodlands (Venglovsky, 2006).

The Mediterranean area is among the world's richest places in terms of plant diversity, with 25,000 plant species, 50% of them endemic (Blondel *et al.*, 2010). There are 290 tree species (Noce *et al.*, 2016), of which 200 are endemic (Quézel & Médail, 2003; Gauquelin *et al.*, 2016). Two thirds of Mediterranian amphibian species, 48% of reptiles, a quarter of mammals, 14% of dragonflies, and 3% of birds are endemic (Mittermeier *et al.* 2004; Paine & Lieutier, 2016; Lefèvre & Fady, 2016; FAO, 2013b). With 52 plant refuges during ice ages (Médail & Diadema, 2009), the Mediterranean is recognized as a Global Biodiversity Hotspot (Mittermeier *et al.*, 2004).

In scrublands, the dominant maquis has many local names reflecting indigenous and local knowledge, such as matorral in Spain, phryganae in Greece or bartha in Israel. It is characterized by hard-leaved shrubby evergreen species of genera *Cistus, Erica, Genista, Juniperus, Myrtus, Phillyrea,* and *Pistacia*. The term "garrigue" is restricted to the limestone, semi-arid, lowland and coastal regions of the basin and is maintained by grazing and fires.

PAST AND CURRENT TRENDS

The State of Nature in the European Union reports that 139 habitats where assessed for the Mediterranean Ecoregion in the European Union. In the period 2007-2012, 27 of these were of favourable conservation status, 1 is unfavourable but improving with respect to the 2001-2006 period, 49 are deteriorating, 62 are stable or have unknown trends. In Central Europe, 38 assessments were performed (combinations of habitats and countries) and of these 37 were favourable and one unfavourable but stable. In Western Europe, 467 assessments were performed, of which 139 were favourable, 6 were unfavourable but improving, 104 unfavourable and declining, 107 unfavourable stable and 109 unknown or unreported status and trends (EEA, 2015a).

Originally, the Mediterranean region was largely covered by evergreen oak forests, deciduous, semi-deciduous, and conifer (pine, juniper) forests (De Beaulieu *et al.*, 2005). However, as a result of centuries of deforestation, no intact forest is left in the region (Blondel *et al.*, 2010; CEPF, 2010a). Both human intervention and climatic conditions are favouring the development of shrublands and then of sclerophyllous and secondary coniferous forests, replacing the primary semi-deciduous and deciduous forests (Abdurakhmanov *et al.*, 2003; Blondel *et al.*, 2010; CEPF, 2010a; Allen, 2014). UNEP *et al.* (2009) reported that 67% of the sub-system of the Mediterranean forest, woodland and scrub had been converted before 1950, whereas recent changes only represent 3% in terms of area. During the 1990-2005 period the area covered by forest generally increased except in Croatia and Bosnia-Herzegovina (UNEP *et al.*, 2009; FAO, 2013b). Plantations cover about 11% of the area, mostly formed by pines and eucalyptus (Wingfield *et al.*, 2015; de Rigo *et al.*, 2016).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Mediterranean forests are the product of a long history of agro-sylvo-pastoral management by rural populations and of interactions between local societies and the state (mainly through the public forest administration) (Blondel *et al.*, 2010; Gauquelin *et al.*, 2016; Kouba *et al.*, 2015; Lefèvre & Fady, 2016; Paine & Lieutier, 2016; Médail & Diadema, 2009). At low altitudes, the present Mediterranean forests have been managed by coppicing, wood cutting, prescribed

fires and grazing, while at higher altitudes they have been conserved (Blondel *et al.*, 2010). Fires and herding are particularly important drivers of vegetation dynamics and selection of plant traits (Arianoutsou, 2001; CEPF, 2010a; de Rigo *et al.*, 2016). After the Second World War, land abandonment resulted in land cover changes from pastures to scrub and later to closed forests, and was accompanied by significant changes in biodiversity (Gauquelin *et al.*, 2016; Lavergne *et al.*, 2005; Mazzoleni *et al.*, 2004; Sirami *et al.*, 2010). The most important threats for Mediterranean woodland species are habitat loss, fragmentation, land degradation and anthropogenic fires causing primary forest cover to decrease and an increase of secondary forest and shrubland (Abdurakhmanov *et al.*, 2003; Peñuelas *et al.*, 2002; FAO, 2013b; Government of Armenia, 2015; EEA, 2002). Nitrogen pollution from agriculture (Sutton *et al.*, 2014; Feest *et al.*, 2014), and unsustainable harvesting and hunting (Peñuelas *et al.* 2002; IUCN, 2008; FAO, 2013a) also contribute to biodiversity loss.

Climate change, withan increase in temperature and frequency of natural disasters, especially droughts, leads to aridification, desertification and a decline of species richness (Allen, 2014; FAO, 2013b; IPCC, 2014a).

With the exception of some pyrophytic species like *Cistus* sp. (EEA, 2004), pine woodlands are more sensitive than scrubland and oak forest to more frequent wildfires (Pausas *et al.*, 2008; Moreira *et al.*, 2011; Dias *et al.*, 2016).

Invasive alien species, including forest pests and diseases (potentially favoured by fires) and plantations of exotic tree species also contribute to biodiversity loss (de Rigo *et al.*, 2016; EEA, 2004; IUCN, 2008). Eucalyptus monocultures can be infected with up to 150 pathogens, spreading more easily in uniform conditions (de Rigo *et al.*, 2016; Wingfield *et al.*, 2015), while pine invasion promotes soil acidification, causing a decrease of taxonomic and phylogenetic diversity (Selvi *et al.*, 2016).

3.3.2.5 Tropical and subtropical dry and humid forests

OVERVIEW OF THE SUB-SYSTEM

There are several types of laurel subtropical forests ("laurifolia" or "laurisilva") on islands in the North Atlantic Ocean, belonging to the Macaronesian biogeographical province. They occupy territories with medium to high precipitation of Azores, Canary Islands and Madeira islands between altitudes of 600 and 1,500 m (Dias *et al.*, 2005; Fernández-Palacios & Arevalo, 1998) and demonstrate a high species diversity: 12,660 species of fungi, plants and animals with extremely high level of endemism – 30% (3,570 species) (Moya *et al.*, 2004).

Two types of mixed and deciduous humid subtropical forests with evergreen elements grow in the Caucasus Mountains: Colchic in the west (Georgia, Russia and Turkey) and Hyrcanic in the east in Azerbaijan (Akhani *et al.*, 2010; Chitanava, 2007; Grossheim, 1926; Gerasimov *et al.*, 1964; Prilipko, 1970; Safarov, 1979; Safarov, 2009; Solomon *et al.* 2014). Other researchers consider Colchic and Hyrcanic forests as specific temperate rainforests, rather than subtropical, because of the mild climate conditions with cooler winters than in many subtropical regions and the presence of the sclerophyllous species only in undergrowth and absent from the tree layer (Borsch *et al.*, 2014; Maharramova *et al.*, 2015; Nakhutsrishvili *et al.*, 2015; Zazanashvili & Mallon, 2009). Colchic forests include about 3,600 vascular plant species, and Hyrcanic forests more than above 1,200 species (Abdurakhmanov *et al.* 2003; Akhani *et al.* 2010; Chitanava, 2007; Filibeck *et al.* 2004; Grossheim, 1926; Gerasimov *et al.*, 1964; Prilipko, 1970; Safarov, 1979, 2009; Solomon *et al.* 2014; Tutayuk, 1975; **Figure 3.7**). Twenty to 30% of Caucasian flowering plants, fish, and terrestrial vertebrates and invertebrates are endemic. Endemism in terrestrial molluscs can reach 75% (CEPF, 2004; Mumladze *et al.*, 2008; Nakhutsrishvili *et al.*, 2015; Zazanashvili & Mallon, 2009). Due to the high diversity of relict Arcto-Tertiary species (Gegechkori, 2011) and the high level of

endemism these forests are included in the Caucasus Global Biodiversity Hotspot (CEPF, 2010b; Mittermeier *et al.*, 2004), the Western Caucasus UNESCO World Heritage Site (Succow & Uppenbrink, 2009) and Global 200 WWF ecoregions (WWF, 2006).

Tugai is a type of gallery forest and shrubland interspersed with grasslands along the Caucasus and Central Asian rivers, similar to natural riparian forests in the northern part of Europe and Central Asia (Glazovsky, 1990; Sadygov, 2012; Shukurov, 2009). Primary wild walnut-fruit forests and woodlands are a specific feature of Central Asian mountains and relict ecosystems, remaining as refuges during ice ages. They occupy mountain slopes at 800 - 2,100 m a.s.l. (Janick, 2003; Shukurov *et al.*, 2005; Venglovsky, 2006).

PAST AND CURRENT TRENDS

Subtropical forests have been transformed by human activities. Currently, native subtropical forests in Europe and Central Asia occupy only 20% of initial laurel forest area (Fernández-Palacios & Arevalo, 1998) and about 10% of Colchic, Hyrcanian, Amu Darya and Azerbaijan Tugai forest area. Mostly they have been transformed into agricultural lands. Remaining subtropical forests are fragmented and

Figure ③ ❼ **Location of Colchic and Hyrcanic forest areas. Source: Nakhutsrishvili *et al.* (2015).**

degraded because of logging and overgrazing, or replaced by Mediterranean type vegetation. This has been the case with laurel forests in Macaronesia and Tugais in Armenia (Government of Azerbaijan, 2014; Bikirov, 2012; Burkhanov, 2013; Fayvush & Aleksanyan, 2016; Fernández-Palacios & Arevalo, 1998; Ionov & Lebedeva, 2004; Janick, 2003; Jungius, 2012; Mumladze *et al.*, 2008; Nakhutsrishvili *et al.* 2011; Shukurov *et al.*, 2015; Treshkin, 2001; Turdieva *et al.*, 2007; Yusifov & Hajiyev, 2004; Zazanashvili & Mallon*,* 2009). 97% of Macaronesian Laurisilva is in Madeira, and is in unfavourable but stable conservation status, the remaining 3% is in the Canary Islands and is considered in favourable conservation status (EEA, 2015a).

The total number of regional extinctions from subtropical forest is unknown. However, noteworthy is the global extinction of the Caspian tiger (*Panthera tigris* ssp. *virgata*). About 50% of natural Azorean species are in danger of extinction (Dias *et al.*, 2005). Twenty-one species of mammals, birds, reptiles and amphibians in Caucasus forests are globally threatened and included in IUCN Red Lists as vulnerable, endangered or critically endangered. Of these, 8 are endemic (West Caucasian tur (*Capra caucasica*), Clarks' lizard (*Darevskia clarkorum*)*,* Charnali lizard (*Darevskia dryada*)*,* large-headed water snake (*Natrix megalocephala*)*,* Caucasian viper (*Vipera kaznakovi*)*,* Black Sea viper (*Vipera pontica*)*,* Caucasian salamander (*Mertensiella caucasica*) and Persian mountain salamander (*Iranodon persicus*) (Nakhutsrishvili *et al.*, 2011). The population of Bukhara deer declined to 100 animals in Tajikistan (Bannikov & Zhirnov, 1971; Jungius, 2012).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Overgrazing affects all types of remaining subtropical forests, but especially damaged wild walnut-fruit forests in Central Asia because of the lack of good pastures. So far, numbers of livestock (mainly goats and sheep) have been growing. Only in Kyrgyzstan in walnut-fruit forests it increased by 5% from 2012 to 2013 (Asykulov & Chodonova, 2015). This leads to structural homogenization, extinction of rare and endemic species, and introduction of weeds and invasive species like Japanese spiraea (*Spiraea japonica*) (Nakhutsrishvili *et al.*, 2011; Treshkin *et al.*, 1998; Shukurov *et al.*, 2005; Fernández-Palacios & Arevalo, 1998; Prada *et al.*, 2009; Asykulov & Chodonova, 2015).

In Macaronesia tourism exerts pressure on ecosystems through recreational activities, disturbance of species, and risk of fires and wastes. From 1960 to 1998 the number of tourists increased from 940,000 to 12.5 million people each year, 6 times as large as the local population. The resulting anthropogenic pressure impairs natural ecosystem functioning (European Comission, 2014; Fernández-Palacios & Arevalo, 1998).

In Azerbaijan water storage facilities have transformed the water regime downstream the Kura River, making it drier and without regular floods. These conditions are unsuitable for Tugai forest (Sadygov, 2012).

Aridification caused by global warming negatively affects Hyrcanian and walnut-fruit forests - dryer forest types (IPCC, 2014a) and they lose mesophytes (Program and Action Plan, 2015). Increasingly, the frequency of catastrophic floods and forest fires caused by climate change, cause the decline of species richness in all subtropical forests within Europe and Central Asia (Prada *et al.,* 2009; Succow & Uppenbrink, 2009; Zazanashvili & Mallon, 2009).

Invasive species affect all types of subtropical forests (Fernández-Palacios & Arevalo, 1998; Shukurov, 2016). The Azorean flora consists of 1,007 plants species, 707 of which have been introduced mostly in the last fifty years (Dias *et al.*, 2005). An invasion of box tree moth (*Cydalima perspectalis*) in 2012 in the Caucasus exemplifies the magnitude of pest damage in the region. It has developed in an active pest outbreak in 2013, and expanded from common box (*Buxus sempervirens*) into an endemic relict box (*Buxus colchica*) in the wild (Gninenko *et al.*, 2014). In 2015 it reached Abkhazia and Crimea, and has destroyed most Buxus colchica in the Caucasus Colchic forests (Abasov *et al.*, 2016). Pest outbreak effects have been exacerbated by destabilisation of ecosystems due to pesticide application (Shukurov, 2016). Pollution from agricultural sources has caused a strong decrease in the area covered by subtropical forest ecosystems (Kuz'mina & Treshkin, 1997; Shukurov *et al.*, 2005; Zazanashvili & Mallon, 2009).

In Central Asia, mining projects have been developed in walnut-fruit forests, polluting air and water and leading to the degradation of the forest vegetation (Janick, 2003).

After the collapse of the Soviet Union many fields and plantations were abandoned and a process of natural reforestation started (Nakhutsrishvili *et al.* 2011). Today these are gradually being returned to agriculture, thereby preventing the expansion of subtropical ecosystems in this way (Shukurov *et al.*, 2015).

Countries have recognized the necessity to conserve the remaining subtropical forests and species and to establish protected areas (Turdieva *et al.*, 2007; Government of Kyrgyzstan, 2014; Government of Tajikistan, 2014; Government of Turkmenistan, 2015; Government of Kazakhstan, 2015; Government of Uzbekistan, 2015). For example, all remaining laurel forests in the Canary Islands are protected (Fernández-Palacios & Arevalo, 1998), and 37% of Hyrcanian forests in Europe and Central Asia are covered by protected areas (Nakhutsrishvili *et al.*, 2015). Programmes on forest restoration have started in some countries (ENPI-FLEG, 2015), to promote the recovery

of species diversity and habitat. Due to implemented measures, populations of some threatened species have become stable or even slowly growing, such as Bukhara deer in Kazakhstan (Greifswald, 2010; Government of Kazakhstan, 2015; Government of Turkey, 2014; Government of Uzbekistan, 2015; Government of Tajikistan, 2014; Government of Kyrgyzstan, 2014; Government of Tajikistan, 2016; Government of Turkmenistan, 2015).

3.3.2.6 Temperate grasslands

OVERVIEW OF THE SUB-SYSTEM

The ecosystem comprises dry or seasonally wet (not overwetting) non-coastal land, more than 30% covered by natural vegetation. Vegetation is dominated by herbaceous, shrub plants and trees. Actively managed grasslands and cultivated lands, high-mountainous (alpine) grasslands, and arid dwarf-shrublands (semi-deserts) are covered in 3.3.2.9, 3.3.2.2. and 3.3.2.7, respectively. Three main grassland types are distinguished in Europe and Central Asia (namely: Steppes, azonal/extrazonal natural dry grasslands, and secondary (semi-natural) grasslands (Bohn *et al.*, 2004; Dengler *et al.*, 2013; Dengler *et al.*, 2014; Ellenberg & Leuschner, 2010; Veen *et al.*, 2009; Vrahnakis *et al.*, 2013; Wesche *et al.*, 2016)). The natural grasslands of the two first types are essentially self-sustaining if the wild herbivore assemblage is present or replaced with a domestic one. The man-made grasslands of the last type require continuous management to preserve their current status (or restore them). The area of original extent of steppes in Europe and Central Asia was assessed as 1,700,000 km^2. The actually remaining steppe area was assessed as 670,000 km^2 (Henwood, 2010).

Europe and Central Asia's grasslands are global hotspots of small-scale (at scales below 100 m^2) vascular plant diversity. Some prominent examples are Transylvanian and Carpathian dry meadows (or meadow steppes) where up to 98 vascular plant species can co-exist on 10 m^2 and 133 species on 100 m^2 (Dengler *et al.*, 2014(b; Török *et al.*, 2016; Wilson *et al.,* 2012). This richness may result from the traditional management practices of local people (Babai & Molnár, 2014). More than 18% of Europe's endemic vascular plants are bound to grassland habitats (Habel *et al.*, 2013; Hobohm & Bruchmann, 2009).

Europe and Central Asia's grasslands provide important habitats for many species of global conservation concern, such as the saiga antelope (*Saiga tatarica*), great bustard (*Otis tarda tarda*), sociable sapwing (*Vanellus gregarious*) (IUCN, 2017b). In Europe, the birds associated with grasslands (and low intensity agricultural) habitats have the highest proportion of threatened species (23%) compared with other habitats (BirdLife International, 2015).

The steppe habitats of Russia harbour 11 mammal species of global conservation concern. The Federal Red Data Book of Russia listed 14 mammal and 14 bird species strongly linked to steppe habitats (two are extinct in the wild in Russia) (Antonchikov, 2005; Smelansky & Tishkov, 2012), and 30 insect species inhabiting only grasslands (presumably steppes) comprising 31% of the whole list of insects (94 taxa) (Red Data Book of Russian Federation, 2001). In Ukraine (Parnikoza & Vasiluk, 2011; Vasiluk *et al.*, 2010) steppe animals comprise 29% of the list of the national Red Data Book (159 from 553). Among 826 species of plants listed in the Red Data Book of Ukraine 33.4% (276) can be found in steppe habitats only (Korotchenko & Peregrym, 2012; Parnikoza & Vasiluk, 2011).

European grasslands have been recognized as threatened hotspots of biodiversity which emphasizes their high conservation priority (Dengler *et al.*, 2014; Habel *et al.*, 2013; Török *et al.*, 2016). Fifty three grassland habitats, distinguished in Europe, are assessed as threatened to some degree, including 12 critically endangered or endangered habitats (Janssen *et al.*, 2016). Nearly half of the bird species associated with grasslands have a threatened population status in the European Union (EEA, 2015d).

In the steppes of Russia only 10% of the protected areas are covered by grasslands (Tishkov, 2005); only 11 of 151 Russian federal strict nature reserves and national parks conserve significant steppe tracts and they comprise only ca. 1% of the total area of federal protected areas (Smelansky & Tishkov, 2012).

PAST AND CURRENT TRENDS

Each main type of grasslands has had a distinct trend. Here we treat Steppe and Azonal natural grasslands together due to similar trends.

Historically, dry grasslands in Europe and Central Asia were ploughed up and turned into croplands on a massive scale. This was the fate of the wet grasslands as well, which were directly drained, or were drained due to drainage of the neighboring arable fields in the landscape (Stoate *et al.*, 2009). The process had accelerated in the Central Europe in 17th century and came to an end in the mid-20th century in Siberia and Kazakhstan (Hejcman *et al.*, 2013; Moon, 2013; Smelansky *et al.*, 2006). In England and Wales 97% of semi-natural grassland disappeared by the mid-20th century (Bullock, 2011). Thus, extensive decline in area, increasing fragmentation, and loss of diversity were dominant trends in grasslands for centuries. As a result, only 3-5% of natural steppe grasslands (and azonal grasslands to a significantly lesser extent) remained relatively intact in Europe (Henwood, 2010) and ca. 20% in Russia (Smelansky & Tishkov, 2012). In the only country where grassland are in relatively good status, Kazakhstan, at least 70-80% of the original extent

of grassland remains (from 10% to 90% for different steppe and semi-desert types) (Henwood, 2010; Rachkovskaya & Bragina, 2012). An example of a more detailed assessment can be found in Hungary, where approximately 251 thousand ha (6.8%) of the original total of 3.7 million ha of forest-steppe vegetation survived, of which only 5.5% of the stands may be considered natural, 38% semi-natural, 46% moderately degraded, and 10% strongly degraded (Molnár *et al.*, 2012).

The second most important trend in the last millennium was loss or significant decline of two keystone herbivore guilds naturally grazing over grasslands in Europe and Central Asia: wild nomadic ungulates (Pärtel *et al.*, 2005); and colonial burrowing rodents and lagomorphs (Davidson *et al.*, 2012). Both guilds are the main ecosystem engineers in their grassland ecosystems through grazing, trampling, defecating, and digging activities.

A general trend common for steppes and semi-natural grasslands is a strong dependence on traditional agricultural systems, evolved over centuries of land use by local people (Schneider-Binder, 2007). Many grassland variants in Europe and Central Asia developed under or were supported by traditional low-intensity agricultural land use including livestock grazing, hay making, manuring, tillage and burning regimes (Smelansky, 2003). Many grassland species, for example some birds and insects, are dependent on specific agricultural practices in both Europe (Benton *et al.*, 2002; Cardador *et al.*, 2014; Donald *et al.*; Stoate *et al.*, 2009) and Central Asia (Kamp *et al.*, 2011; Wright *et al.*, 2012). For example, critically endangered sociable lapwing (*Vanellus gregarius*), endemic black lark (*Melanocorypha yeltoniensis*), and some other typical steppe birds strongly prefer heavily-grazed habitats for nesting, but moderately-grazed habitat is optimal for nesting success (Fijen *et al.*, 2015; Sheldon *et al.*, 2013; Watson *et al.*, 2006). Historically, grazing patterns in the steppes of Central Asia were created and maintained by the traditional mobile pastoralists acting for centuries (Krader, 1955; Leeuwen *et al.*, 1994).

In contrast to traditional farming systems, the more recent intensification of farming has resulted in a dramatic decline of grassland biodiversity. Data from Western Europe show a strong decline of grasslands birds and a 45% decline in the butterfly population in recent decades (Donald *et al.*, 2006; EEA *et al.*, 2013).

In general, habitat and species trends for grasslands in Europe and Central Asia are negative **(Table 3.5)**. Habitat degradation is still increasing and habitat area decreasing principally as a result of massive land-use changes and pollution, but significant subregional variation is observed. The conservation status of many endangered species remains unchanged or even becomes worse due to land-use change, overexploitation and pollution. Only species richness is relatively stable, except for semi-natural grasslands,for which it has a negative trend. Climate change accelerates these trends.

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

The most important direct drivers that strongly affect temperate grassland area are ploughing, afforestation, mining and excavation, settlements and industrial area encroachment, land abandonment and climate change (Cerqueira *et al.*, 2015; Kamp *et al*., 2016; Korotchenko & Peregrym, 2012; Prishchepov *et al.*, 2013; Rachkovskaya & Bragina, 2012; Smelansky *et al.,* 2006; Smaliychuk *et al.*, 2016; Smelansky & Simonov, 2008; Smelansky & Tishkov, 2012) **(Table 3.5)**.

Biodiversity and functioning of ecosystems are shaped by differences in subbiome (grassland) type, and latitudinal and evolutionary gradients, site factors (slope, aspect, nutrient status, levels of alkalinity/acidity and moisture), livestock breeding (grazing and mowing), fire, fertilization (manuring, nitrogen deposition), species invasion, and successional dynamics (specifically as a result of abandonment) (Smelansky *et al.*, 2006; Faber-Langendoen & Josse, 2010; Kamp *et al.*, 2016; Korotchenko & Peregrym, 2012; Merunková *et al.*, 2014; Smelansky & Tishkov, 2012).

Except for latitudinal and subbiome differences, the abovementioned drivers are caused or influenced by society. Grazing is a major direct factor influencing biodiversity and ecosystem functioning (Augustine & Mcnaughton, 1998; Díaz *et al.*, 2007). Fire is another major factor, both through wildfires and prescribed burning. Wildfires (including uncontrolled burning) are practiced in extensive areas in Ukraine, Russia, some Central European countries (Romania, Hungary, Bulgaria) and in Central Asia (Valkó *et al.*, 2016; Smelansky *et al.*, 2015) as well as in the Mediterranean (Keeley *et al.*, 2012; Valkó *et al.*, 2016). Fertilization leading to eutrophication is especially important for semi-natural grasslands in Western and Central Europe (Duprè *et al.*, 2010). Many drivers lead to fragmentation of grasslands producing a loss of grassland-specific species and degradation of ecosystems.

3.3.2.7 Deserts

OVERVIEW OF THE SUB-SYSTEM

Deserts comprise low and high altitude plains with precipitation of no more than 100 mm/year (FAO, 1989) or no more than 250 mm/year (as per Koeppen-Geiger Classification, Kottek *et al.*, 2006), with rare or absent

vegetation on desert soils (Kharin, 2002). While the largest extent of deserts is found across Central Asia, the most arid desert in the region is located in Israel (Western Europe in this assessment). The Central Asian deserts extend from the Kopetdag and Paropamiz mountains in the south, to a latitude of 48° north and from the Caspian Sea in the west to the foothills of Jungar Alatau, Tien Shan and Pamir-Altai mountains in the east. This spans about 1,400 km from north to south and 2,700 km from west to east (Akzhygitova *et al.*, 2003). The Negev Desert in Israel is expanding from the south-eastern section of the Mediterranean Sea eastwards and south-eastwards and with extension northwards along the Dead Sea Rift Valley (Evenari *et al.*, 1982).

Central Asian deserts include: northern or steppified deserts (or semi-deserts) with wormwood gramineous and salt grass plant associations; middle deserts or the true deserts with perennial saltworts and wormwoods and saxaul (*Haloxylon ammodendron*) on sands; and southern deserts with a different composition of wormwoods and salt grass species. Deserts in foothills and in intermontane areas are specifically different in terms of species composition but occupy comparatively small areas (Akzhygitova *et al.*, 2003, **Figure 3.8**).

While the Central Asian deserts form part of the Irano-Turanian floristic region (Takhtadzhyan, 1978; Shmida, 1985), the Negev Desert also has Irano-Turanian vegetation. It becomes increasingly arid towards the south, with features of Saharo-Arabian vegetation. The region has been continuous with the African continent since the Permian (Trewick, 2017; Ziv *et al.*, 2014). Additional more recent geological processes making it a major biodiversity corridor between Africa and Eurasia include the rifting of the Dead Sea Rift Valley (Anker *et al.* 2009).

Aralkum is a new desert formed as a result of the drying up of the Aral Sea following extensive water consumption for irrigation. The current flora consists of 34 families of plants with 134 genera and 300 species. Aralkum covers an area of over 38,000 km^2 and is a source of windblown dust. Dust storms carry away about 100 million tons of toxic dust and salts annually, including fertilizers and pesticides that have been washed away from irrigated fields (Breckle *et al.*, 2012).

Figure 3 8 **Vegetation of Central Asian Deserts. Source: Akzhygitova *et al.* (2003).**

1. Основные подразделения растительного покрова.
Пустыни: *1* — северные, *2* — средние, *3* — южные, *4* — предгорные, *5* — растительность солончаков.
6 — Растительность долин рек и оазисов. *7* — Растительность гор.

1. Main subdivisions of vegetation cover.
Deserts: *1* — northern, *2* — middle, *3* — southern, *4* — piedmont, *5* — vegetation of solonchaks.
6 — Vegetation of river valleys and oases. *7* — Vegetation of mountains.
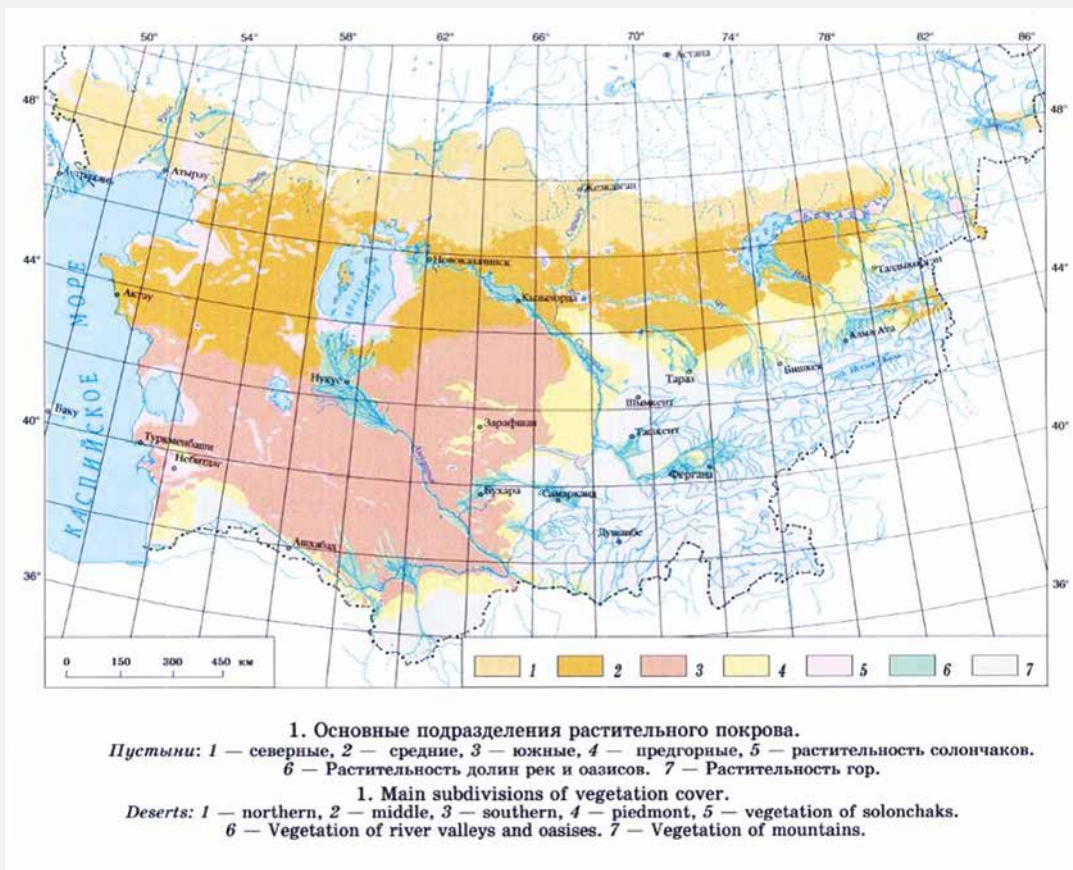
PAST AND CURRENT TRENDS AND THEIR ATTRIBUTION TO DIRECT DRIVERS

The most evident changes of population abundance, functional diversity, and habitat extent in Central Asian deserts were caused by land transformation, fragmentation and degradation (Zoï, 2011). Desert habitats in Central Asia have been fragmented by agriculture for cotton and food production (Kharin, 2002). The irrigated area of Central Asian deserts more than doubled during the 20th century (from 25,000-35,000 km^2 to 70,000-80,000 km^2 and reached 100,000 km^2 in 2013) (Kurtov, 2013). Land degradation caused a species richness decline due to high salinity of abandoned fields. Overall, 40 to 60% of irrigated soils in Central Asia are salt-affected or waterlogged (Gupta *et al.*, 2009; Zoï, 2011). In the Negev wind and water erosion plays an additional significant role (Verheye, 2009). Removal of sand by winds stimulates sand desert expansion by 3-4 m per year on average and up to 9-12 m in Turkmenistan (Veisov *et al.*, 2008).

Central Asian deserts traditionally have been used by local people as pastures – up to 1,700,000 km^2 during the Soviet Union period (until 1991) (Vinogradov, 1977). Poor pasture management and overgrazing deteriorate the natural vegetation (Gupta *et al.*, 2009; Turdiboeva, 2015). They were partly abandoned at the end of the last century, but most of the area still suffers from overgrazing, which causes land degradation and species richness decline (Kharin, 2002; Shukurov, 2016). Different natural conditions in the Negev Desert supported different land-use patterns: crop husbandry at the north and grazing in the south, which were based on water-harvesting practices. The history of ecosystem transformation in the Negev is as long as in the Mediterranean (Verheye, 2009). Until recently, the process of desertification did not affect the Negev profoundly. This was mainly due to large-scale afforestation programmes, restrictions imposed on grazing, and large water subsidies from the less arid parts of Israel to its more arid areas (Portnov & Safriel, 2004). Presently, overgrazing and aridification contribute to biodiversity decline (Verheye, 2009; IPCC, 2014a).

Aridification due to climate change is leading to the increase in desert area and consequent a decline in biodiversity in the centre of deserts (Berseneva, 2006; IPCC, 2014a). It also leads to the spread of deserts to the north and high into the mountains in response to warming. This results in loss of biodiversity in former semi-deserts and dry steppes (Glazovsky & Orlovsky, 1996; IPCC, 2014a).

Fragmentation of habitats by linear infrastructure interrupts migration routes, for example for globally threatened ungulates leading to decline of their populations: saiga antelope (*Saiga tatarica*), khulan (*Equus hemionus*), Goitered

gazelle (*Gazella subgutturosa*) (Olson, 2013; Rosen Michel & Röttger, 2014).

Pollution by fertilizers, pesticides, defoliants used in agriculture (Zoï, 2011), and from mining extraction has a large impact, locally up to the total loss of the vegetation cover (Luryeva, 2014). A particular feature of Central Asian deserts is the impact of the Aralkum that is causing overall species richness decline due to the windborne transfer of hazardous substances from remaining sediments of the former Aral Sea bottom to the surrounding areas (Alikhonov, 2011; Zoï, 2011; Breckle *et al.*, 2012).

In spite of large number of invasive species (57 in Turkmenistan alone) and their competition with native ones, they mainly occur in agricultural and urban territories, so their impact is generally not considered a significant driver of the decline of the number or abundance of populations of native species (Kamakhina, 2008).

3.3.2.8 Peatlands

OVERVIEW OF THE SUB-SYSTEM

Peatlands are areas where water-saturated soil causes the accumulation of incompletely decomposed plant material ("peat")**.** A peatland which is actively accumulating peat is called mire. Several English terms (e.g., marsh, swamp, fen, bog) are used for naming different mire types (Joosten *et al.*, 2017). Henceforth, this assessment report will use the term peatland. Most national definitions require "peatland" to have a minimum peat depth of 30 cm with peat of >30% by dry mass (Joosten & Clarke 2002, Parish *et al.*, 2008, Rydin & Jeglum, 2013). Peatlands have organic soils (histosols), which include soils with shallower organic layers and less organic matter (FAO, 2015b). Areas with shallow peat (< 30 cm) may cover large areas, as in tundra and boreal zones (e.g. Vompersky *et al.*, 1996, 2011), and in the field are difficult to distinguish from real peatlands, but are usually overlooked and not considered as they usually count as tundra or boreal area (Minayeva & Sirin 2012). Most peatlands of Europe and Central Asia were formed after the last Ice Age (~10,000 years ago), and only very few are much older (Joosten *et al.*, 2017).

Peatlands often demonstrate a unique structural and functional integrity which has developed over centuries. Saturated peatland conditions select the plant species that may grow and form peat. The accumulated peat (which may consist to more than 90% of water) regulates the moisture balance and further determines the habitat for plant growth. Changes in water regime or vegetation may lead to peat and peatland degradation, causing enormous emissions of greenhouse gases (Parish *et al.*, 2008, Hiraishi *et al.*, 2014). Under favourable conditions, however, peatlands may recover (Bonn *et al.*, 2016, Minayeva *et al.*, 2017a).

PAST AND CURRENT TRENDS

Peatlands are found in every country in Europe and Central Asia. In Western Europe peatlands cover 276,323 km², of which 48% are degraded by drainage for agriculture, forestry and peat extraction, or destroyed by infrastructure development, construction, or flooding by dams **(Figure 3.9)**. In Central Europe peatlands cover 47,829 km², of which 74% are drained and degraded. In both subregions some 10% of the former peatland area does no longer have enough peat to be considered as peatlands. In the European Union part of Western and Central Europe 51% of mires and bogs assessments were classified as "unfavourable bad" and another 34% as "unfavourable inadequate" (EEA, 2015a). In Eastern Europe (including only the European part of the Russian Federation) peatlands cover 267,130 km² of which 38% are drained and degraded (Joosten *et al.*, 2017 and **Figure 3.9**). In the entire Russian Federation peatlands

occupy 1,390,000 km² or 8.1% of the country and together with shallow peat lands (<30 cm) as much as 3,690,000 km² or 21.6%. Most peatlands (85%) and shallow-peat lands (84%) are found in the Asian part of the Russian Federation. Almost 20% of the peatlands are underlain by permafrost, of which 5.3% are polygon mires and 14.5% palsa mires (Vompersky *et al.*, 1996; 2005; 2011). Trees are present on 38% of the peatland area, while about 62% is open. Also, 53% of the shallow-peatlands are open (Vompersky *et al.*, 2011). Most peatlands in Russia are still in a natural state. Degraded peatlands are concentrated in the western and central part of European Russia (Minayeva & Sirin, 2005; Minayeva *et al.*, 2009). In Central Asian countries, peatlands cover only a few thousand square kilometres and are mainly situated in the highlands of Pamir, Tyanshan and Altay (Aljes *et al.*, 2016; Kats, 1971). Highland peatlands play a crucial role for maintaining ecosystem productivity, conserving biodiversity, preserving permafrost, and regulating

Figure **3** **9** **Proportion of current peatland area (% total country area) and proportion of degraded peatland area (% total peatland area) in Western and Central Europe and the Western part of Eastern Europe.**

Source: Based on data from Joosten *et al.* (2017) and Global Peatland Database/Greifswald Mire Centre. Map prepared by C. Tegetmeyer. Note: in many countries, the original peatland area was substantially larger than the current peatland area.

PEATLAND AREA (% total country area)

| ≤ 2 | ≤ 5 | ≤ 10 | ≤ 20 | > 20 |

PEATLAND DEGRADATION (% total peatland area)

| ≤ 25 | ≤ 50 | ≤ 75 | ≤ 95 | ≤ 100 |
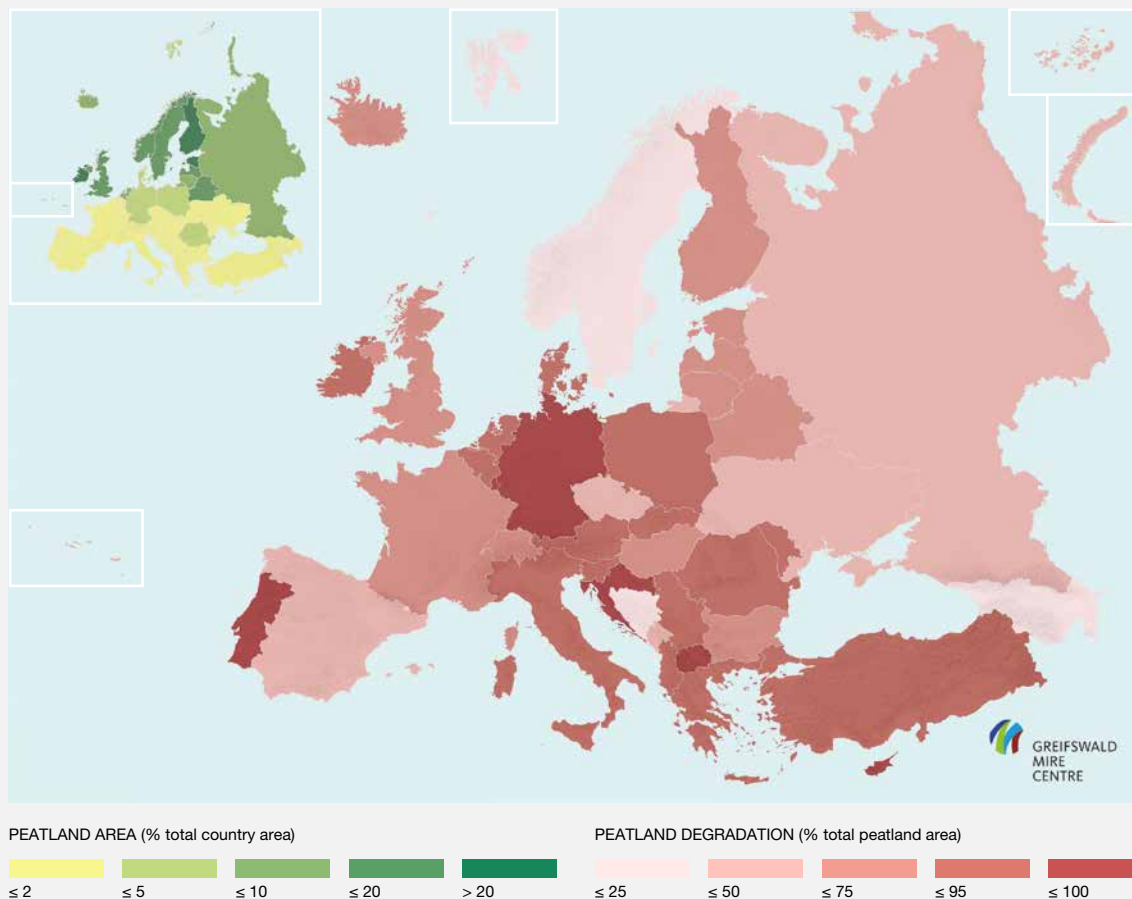
water supply (Müller *et al.*, 2016). However, they are often overlooked, not considered as peatlands, treated as dry meadows, and therefore rapidly disappearing.

Peatlands in Europe and Central Asia in the past demonstrably suffered from long-term climate warming (Klimanov & Sirin, 1997), but their diversity and the variety of geographical conditions prohibit drawing unequivocal general conclusions on their reaction to climate change, especially on the scale of decades (Parish *et al.*, 2008).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Peatlands as ecosystems are rather well adapted to climate change (Minayeva & Sirin, 2012). However, especially in the 19th and 20th centuries, anthropogenic transformation (drainage for agriculture, forestry, peat extraction, infrastructure) has substantially reduced this resilience. Northern permafrost peatlands, which are most sensitive to climate change, are progressively affected by industrial development and intensification of traditional land use (Minayeva & Sirin, 2009; 2010). In the temperate and boreal zones peatlands have been widely drained and used for forestry, agriculture and peat extraction. Many of the earlier drained areas are currently abandoned and subject to – sometimes catastrophic – fires (Minayeva *et al.*, 2013; Sirin *et al.*, 2011).

Boreal peatlands currently show a gradual reverse from drainage-based exploitation towards protection and restoration. In the temperate zone a growing appreciation for ecosystem services has initiated peatland rewetting projects to reverse the impacts of drainage. At the same time, however, the demand for biomass has caused massive expansion of biomass cultivation on peatlands with deeper drainage and more fertilization, which dramatically changes peat soil properties. In semi-arid and desert regions peatlands are being destroyed by overgrazing and drainage, while highland peatlands are often affected by mining. Overgrazing on peatlands leads to peat degradation, massive CO_2 emissions, and a loss of storage and retention capacity for carbon and water (Sirin *et al.*, 2016). All these hazards are aggravated by climate change, especially by decreasing precipitation, rising temperatures, and increased probability of catastrophic events such as droughts, rain storms or fires.

The resilience of natural peatlands to climate change is based on their self-regulation, but this capacity is not unlimited (Minayeva & Sirin, 2012). Substantial changes in peatland hydrology (by drainage), soil hydraulic properties (by long-term drainage), and peatland relief (by oxidation, subsidence and peat extraction) make spontaneous and supported recovery more and more complicated (Parish *et al.*, 2008). In damaged peatlands, climate change is

Figure **3** **10** **Spatial distribution of the main changes in the extent of agroecosystem types (classes 01 to 10, 13 and 14) between 1990 and 2006 in the EU-27. Source: Levers *et al.* (2015). With permission of Springer.**

01 Increasing yields in permanent crops
02 Increasing cropland yields
03 Intensification to high-intensity cropland
04 Intensification to med-intensity cropland
05 Declining cropland yields
06 De-intensification high-intensity cropland
07 De-intensification med-intensity cropland
08 Declining grassland yields
09 De-intensification high-intensity livestock farming
10 De-intensification med-intensity livestock farming
11 Intensification of wood production
12 Forest expansion over grassland
13 Deforestation for agric. expansion
14 Cropland-grassland conversion
15 Permanent cropland loss
16 Urban expansion
17 Stability

expected to increase the probability of catastrophic events, such as peat fires (Minayeva *et al.*, 2013; Sirin *et al.*, 2011), erosion, and inundation, and will impair the further provision of important ecosystem services, such as carbon storage and water regulation (Parish *et al.*, 2008, Bonn *et al.*,

2016). As peatland degradation enhances climate change (because of the enormous emissions involved) (Hiraishi *et al.*, 2014), the impact on biodiversity reaches far beyond the boundaries of the peatland itself.

Figure ③ ⑪ Ⓐ **Major gross area changes between the different types of agroecosystems observed for EU-27 and Switzerland per decade since 1910. Source: Fuchs *et al.* (2015).** Ⓑ **Extent of agroecosystems under management; and** Ⓒ **experiencing cessation of management assessed in 2005 for areas in eight countries of Baltic, Central Europe and Eastern Europe. Source: Alcantara *et al.* (2012).**

A

Legend:
- Grass to crop (e.g. cropland expansion)
- Crop to grass (e.g. land abandonment or pasture expansion)
- Afforestation/Reforestation
- Deforestation

B — AGROECOSYSTEMS UNDER MANAGEMENT

C — AGROECOSYSTEMS UNDER CESSATION OF MANAGEMENT
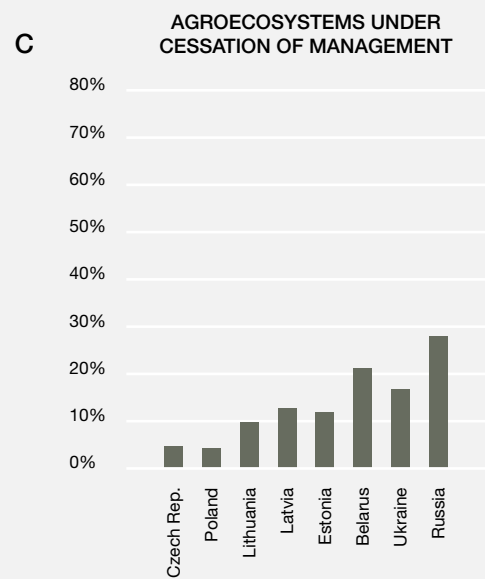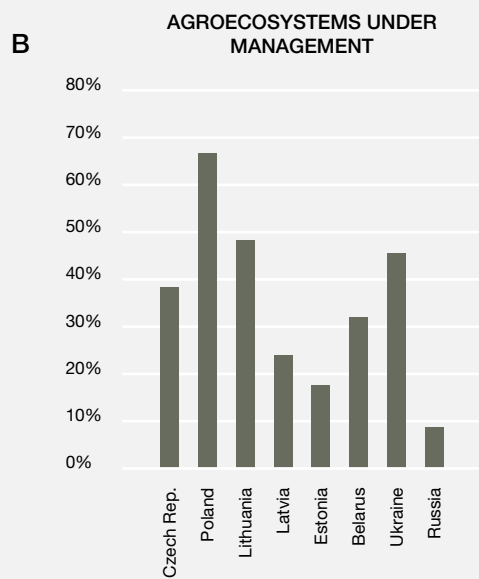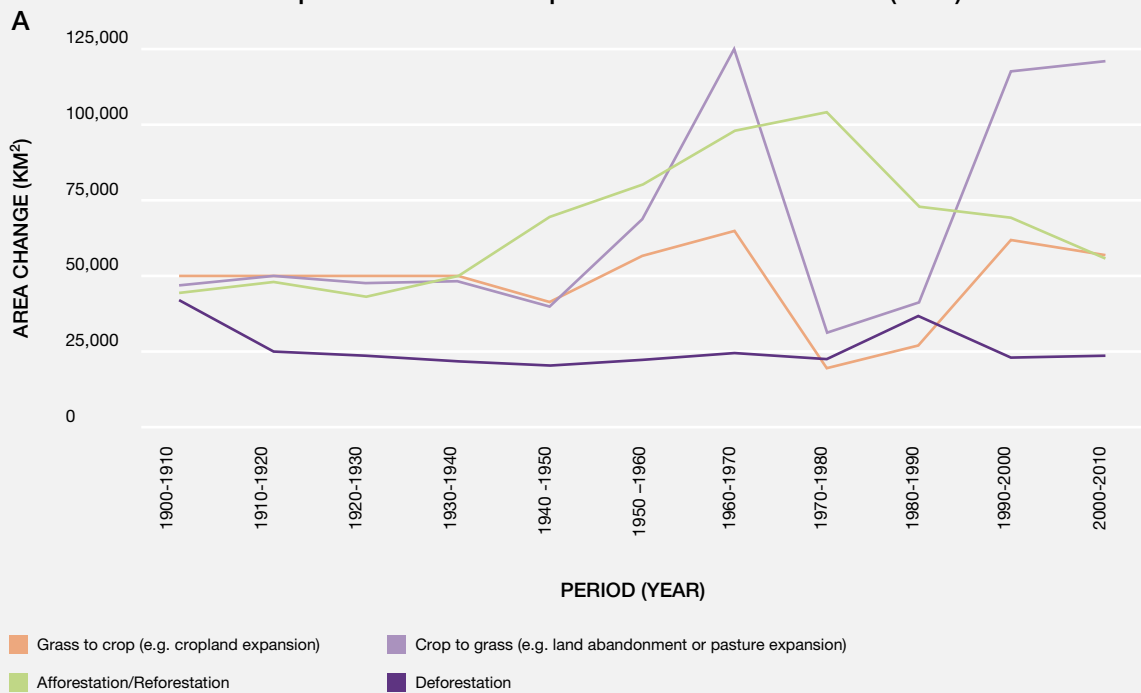
3.3.2.9 Agricultural areas

OVERVIEW OF THE SUB-SYSTEM

Agroecosystems include croplands, orchards, horticultural systems and managed grasslands (note that alpine grasslands and natural or semi-natural grasslands are addressed in other sections). Agricultural areas cover around half of the land area and thus represent the largest terrestrial unit of analysis over Europe and Central Asia (EEA, 2015a; FAO, 2013a; Levers *et al.*, 2015).

PAST AND CURRENT CHANGES IN THE EXTENT AND DIVERSITY IN AGROECOSYSTEMS

The legacy of traditional, low-intensity and diverse agricultural systems in Europe and Central Asia is a rich diversity of habitats and landscapes, generally supporting high levels of biodiversity (STOA, 2013 and references therein). However, agroecosystems and their diversity have changed dramatically since the early 1950s, and there has been an increase of highly modified and simplified agroecosystems and agricultural landscapes, in particular in Europe (Poláková *et al.*, 2011). From 1990 to 2006, land-use conversion, de-intensification and intensification took place on 26%, 18% and 15% of land areas, respectively, which corresponds with huge changes in the extent of different agroecosystem types **(Figure 3.10)**.

De-intensified agroecosystems dominated in Eastern Europe and Central Europe (3.5; see also Kuemmerle *et al.*, 2016) and in Central Asia (Kraemer *et al.*, 2015), along with abandoned farmland after the collapse of the Soviet Union (e.g. 26 million ha in Russia, Belarus, Ukraine and Kazakhstan (Lambin and Meyfroidt, 2011). A vast area experienced spontaneous recovery of forest and steppe ecosystems (Kamp *et al.*, 2015). Remote, economically unproductive agroecosystems are increasingly abandoned, reforested, or included in rewilding schemes (MacDonald *et al.*, 2000; Navarro and Pereira, 2012). For the EU-27 plus Switzerland, gross changes in the extent of the different types of agroecosystems resulted in changes to 56% of the area (ca. 0.5% /yr) between 1900 and 2010. This covers twice the area of net changes, with main changes being cropland or grassland dynamics and afforestation **(Figure 3.11)**. Within agricultural landscapes, decreased crop diversity, decreased coverage of natural and semi-natural areas (hedgerows, isolated trees, ponds, permanent grasslands) and lower connectivity between the remaining natural and semi-natural habitats are generally observed in response to intensification of agricultural systems (Robinson and Sutherland, 2002; Stoate *et al.*, 2001, 2009). For instance, hedgerow length and connectivity have strongly decreased in Western Europe (Deckers *et al.*, 2005; Robinson and Sutherland, 2002).

Ample information is available on the status and temporal trends of biodiversity for some broad taxonomic or functional groups in Europe and Central Asia, or at least for Western Europe and Central Europe. A vast number of scientific papers report temporal trends of biodiversity components in agricultural areas in locations or (sub)regions of Europe and Central Asia. Well established information exists for farmland birds (e.g. work of the European Bird Census Council covering at least 28 countries), arable flora (meta-analyses covering croplands from many countries), grassland butterflies (covering 19 countries), and the diversity of avian and mammalian breeds (syntheses performed by FAO over Europe and Central Asia). For the diversity of cultivated crop plants, comprehensive information exists for the number of varieties conserved ex situ, but not for the trends in the (genetic and functional) diversity of major cultivated varieties actually cultivated, i.e. grown *in situ*. In contrast to the Western Europe and Central Europe subregions, agricultural lands in Eastern Europe and Central Asia are often not recognized as having high conservation value, and research on trends of biodiversity in agricultural areas is rare. We summarize the major trends for different components of biodiversity in agricultural areas of Europe and Central Asia in **Table 3.1**.

Farmland birds - From 1980 to 2013, the abundance of common farmland bird species has continuously been decreasing (by 57% in total) in Europe, although the slope of decrease is lower since the 1990s **(Figure 3.12)**. Since 1990, the decline is more pronounced for northern Europe, intermediate for western Europe and new European Union member States, and less important for southern Europe **(Figure 3.12)**. In addition, the functional diversity of farmland bird communities is changing. The abundance of 17 granivorous species and 14 insectivorous species decreased strongly (56% and 46%, respectively), while the abundance of other species (one herbivore, two omnivores, one carnivore and one aerial insectivore) remained constant over 28 European countries (Inger *et al.*, 2015)[5]. Overall, farmland bird communities become more homogeneized (Doxa *et al.*, 2012).

Over the past 25 years in the Eastern Europe and Central Asia subregions the dynamics of farmland bird populations have been mainly driven by the crucial land-use changes related to transition from the Soviet-era planned economy to a market economy. The 1991-2001 period was characterized by massive land abandonment, decreasing crop yields and livestock numbers, and decline of fertilizer and pesticide use, which led to increases of the abundance and species richness of farmland birds in the steppe and forest-steppe geographical zones (Bolnykh & Vengerov 2011; Kamp *et al.*, 2011, 2015; Korovin, 2015), whereas

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5. Inger *et al.* kindly reanalysed their published data and computed trends for farmland birds for the present assessment

Table 3 1 **Summary of the major trends reported for several components of biodiversity in agricultural areas in Europe and Central Asia, based on the analysis of over 150 temporal trends reported in the literature.**

For trends in pollinators, see IPBES (2016a). ↑/↓ denote strong and consistent increase/decrease in the indicator; ↗/↘ denote moderate and consistent increase/decrease in the indicator; ↔ stable indicator; ↕ variable trend in the indicator. The numbers reflect the impact of the driver on the trend: 0 no or marginal impact; 1 moderate impat; 2 high impact. P=Past, C=Current. ECA=Europe and Central Asia, WE=Western Europe, CE=Central Europe, EE=Eastern Europe, CA=Central Asia.

Biodiversity Indicators		General trends – Past (ECA WE CE EE CA)	General trends – Current (ECA WE CE EE CA)	Climate P	Climate C	Land use* P	Land use* C	Pollution* P	Pollution* C	Exploitation P	Exploitation C	Invasives P	Invasives C
Farmland birds	Species richness	↘ ↘ ↘ ↘ ↘	↘ ↘ ↘ ↔ ↘	1	1	2	2	1	1	0	0	0	0
Farmland birds	Species abundance	↓ ↓ ↘ ↘ ↘	↘ ↘ ↘ ↗ ↔	1	1	2	2	1	1	0	0	0	0
Farmland birds	Functional diversity	↘	↘	1	1	2	2	1	1	0	0	0	0
Avian breeds	Breed richness	↘ ↓ ↘ ↘ ↘	↘ ↘ ↘ ↘ ↘	0	0	2	2	0	0				
Avian breeds	Local breed abundance	↘ ↓ ↘ ↘ ↘	↘ ↘ ↘ ↘ ↘	0	0	2	2	0	0				
Avian breeds	Genetic diversity	↘ ↓ ↘ ↘ ↘	↘ ↘ ↘ ↘ ↘	0	0	2	2	0	0				
Mammal breeds	Breed richness	↘ ↓ ↘ ↘ ↘	↘ ↘ ↘ ↘ ↘	0	0	2	2	0	0				
Mammal breeds	Local breed abundance	↘ ↓ ↘ ↘ ↘	↘ ↘ ↘ ↘ ↘	0	0	2	2	0	0				
Mammal breeds	Genetic diversity	↘ ↓ ↘ ↘ ↘	↘ ↘ ↘ ↘ ↘	0	0	2	2	0	0				
Butterflies	Species richness	↘	↘			2	2	2	2	0	0		
Butterflies	Species abundance	↘	↘			2	2	2	2	0	0		
Butterflies	Functional diversity	↘	↘	1	1	2	2	2	2	0	0		
Arable plants / weeds	Species richness	↘	↘	0	0	2	2	2	2	0	0	1	1
Arable plants / weeds	Species abundance	↘	↘	0	0	2	2	2	2	0	0	1	1
Arable plants / weeds	Functional diversity	Shift	Shift	0	0	2	2	2	2	0	0	1	1
Cultivated plants	Ex situ variety richness	↑ ↑ ↑ ↑ ↑	↑ ↑ ↑ ↑ ↑	0	0	2	2	2	2				
Cultivated plants	Ex situ genetic diversity	↕ ↕ ↕ ↕ ↕	↕ ↕ ↕ ↕ ↕	0	0	2	2	2	2				
Cultivated plants	Ex situ functional diversity	↘	↘	0	0	2	2	2	2				
Cultivated plants	In situ genetic diversity	↘	↘	0	0	2	2	2	2				

*Agriculture intensification encompasses both land-use change and pollution (fertilization, pesticide application)

in the forest zone this promoted an opposite trend (i.e. decreasing abundances and diversity) due to spontaneous reforestation, decreased open habitat areas and reduced habitat diversity (Borisov *et al.*, 2014). At least in part of the Central Asia and Eastern Europe subregions, farmland bird populations have decreased again since the early 2000s (Kamp *et al.*, 2015).

The abundance of grassland butterflies has declined by 30% in 22 European countries from 1990 to 2015 **(Figure 3.13)**. Butterfly communities also became more homogeneized (Eskildsen *et al.*, 2015). However, this negative trend has been locally reversed in some cases **(Box 3.1)**.

Agriculture-detrimental and beneficial insects – Temporal trends in the abundance or distribution of insects, which can cause major changes for agriculture have been reported.

For instance, important changes in the distribution of crop pests, in particular due to climate change in northern areas of Europe and Central Asia, have been reported (Roshydromet, 2014; **Figure 3.14 A**). Evidence also accumulates of significant declines for both managed and wild bees (including bumblebees) over the past 60 years in Europe, which has been recently synthesized by a thematic IPBES assessment (IPBES, 2016a). In particular, there have been severe losses of honey bee colonies reported for the 1961-2012 period in many countries of Europe and Central Asia **(Figure 3.14 B)**. However, in the countries of Central Europe, Eastern Europe, and Central Asia subregions, the hive numbers show marked trends of recovery during the past decade (Kazstat, 2005, 2016; Rosstat, 2015).

Animal genetic resources for food and agriculture – Geogaphical Europe and the Caucasus have by far

Box **3** **1** Reversing the decline of biodiversity in agricultural areas: a success story for a butterfly species.

Ecological knowledge was successfully used to guide innovative conservation practices allowing the reversal of the decline of *Maculinea arion*, a charismatic specialist whose larvae parasitize *Myrmica* ant societies (Thomas *et al.*, 2008). *M. arion* larvae were found to be adapted to a single host-ant species inhabiting a narrow niche in grassland. Inconspicuous changes in grazing and vegetation structure caused host ants to be replaced by other ant species unsuitable for the butterfly larvae, explaining the extinction of European *Maculinea* populations. Once this problem was identified, ecosystems were perturbed by appropriate practices, and the predicted subsequent recovery of the butterfly and ants was validated for 78 sites. Such successful identification and reversal of the problem provides a paradigm for other science-based actions to reverse the decline of biodiversity in agricultural areas.

Figure **3** **12** **A** Temporal variations in the abundance of common farmland birds between four European regions between 1990 and 2011. Source: Pe'er *et al.* (2014). **B** Temporal variation in the abundance of common farmland birds for 28 European countries and for 39 species from 1980 to 2015. Source: http://www.clo.nl/en/indicators/en1479-farmland-birds.
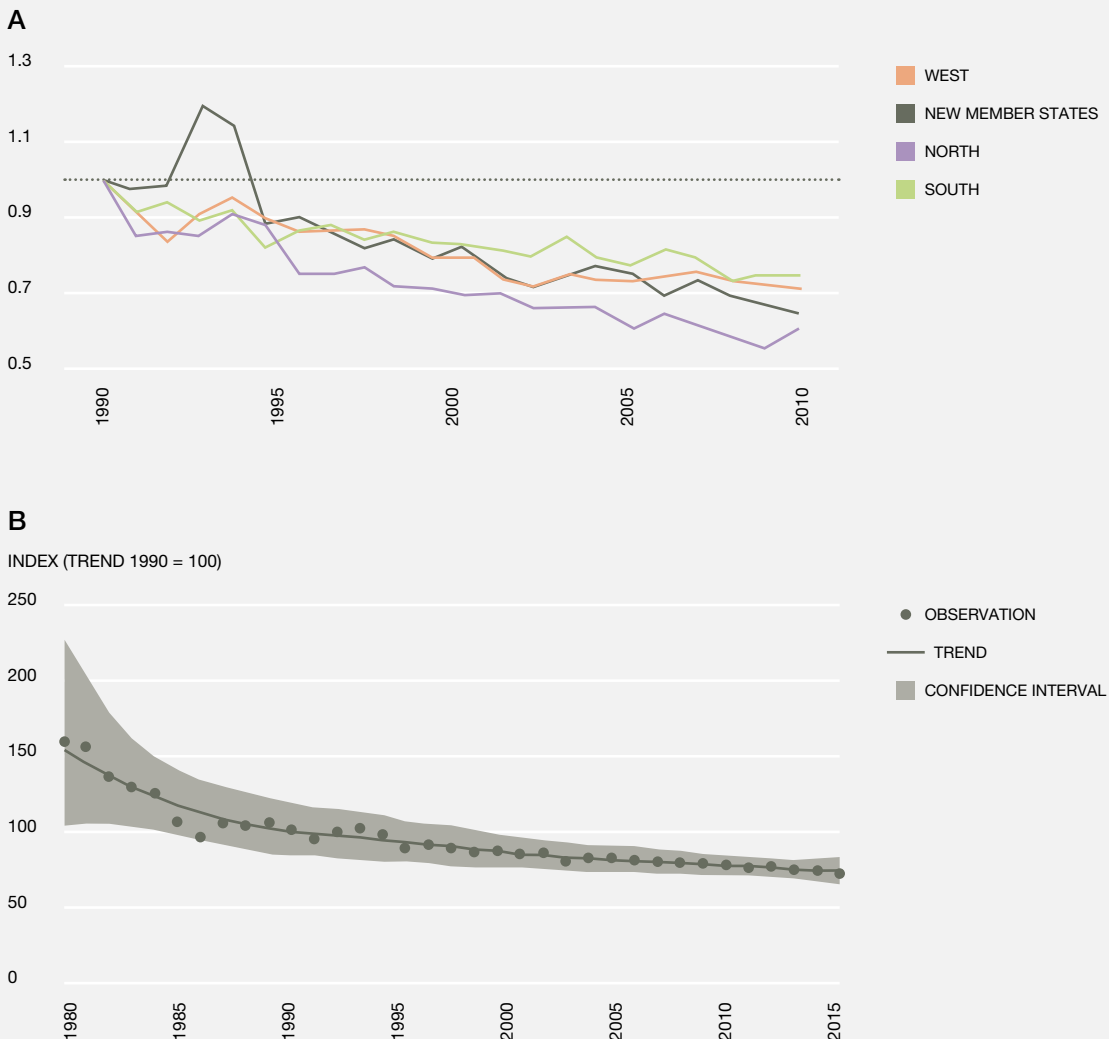
the highest proportion of animal breeds at risk in the world (31 and 35 per cent of mammalian and avian breeds, respectively) and the highest absolute number of at-risk breeds (446 mammalian and 75 avian breeds corresponding to 79% and 91% of total breed extinction at global scale, respectively) **(Figure 3.15)**. In several countries, populations of native breeds, although generally well adapted to local circumstances and resources and forming an important part of our cultural heritage and regional identity in Europe and Central Asia, remain at critically low numbers, being replaced by a few and widespread highly productive breeds. Native breeds make up only a small part of the total population, and nearly 40% of native breeds are at risk in Europe and Central Asia, i.e. the highest value for all global regions (FAO, 2015a). Overall, the situation of animal genetic resources is stable but negative in Europe and Central Asia.

Arable plants and weeds - The species diversity of arable plants has decreased since 1950 (by around 20%) (Richner *et al.*, 2015). The abundance of arable plants has also decreased (Meyer *et al.*, 2013; Richner *et al.*, 2015). In particular, the abundance of rare arable plant species characteristic of traditional management has decreased since the 1950s. These trends probably hold true all over Europe and Central Asia. The functional diversity of arable plants has changed from the 1950s to 2011, with an

increase of arable weeds linked to high nutrient demand and resistance to extreme pH, and herbicides (Richner *et al.*, 2015). 25% of weed taxa are threatened in Tajikistan, including 18 endemic and four subendemic plants (Nowak & Nowak, 2015; Nowak *et al.*, 2014).

Plant genetic resources and crop wild relatives for food and agriculture - The number of plant varieties conserved *ex situ* has increased in Europe, as a result of selection and efficient storage approaches. However, much of the diversity of crop wild relatives and underused species relevant for food and agriculture still needs to be secured for present and future use (FAO, 2015a). Regarding the genetic diversity of crop plants actually cultivated *in situ*, a reduction in diversity occurred up to the 1960s due to the replacement of landraces by modern cultivars and to the low number of cultivars actually cultivated over large areas, while no further reduction or increase of diversity was observed after 1980 (Bonnin *et al.*, 2014); but the trend is likely species-specific. However, the actual genetic diversity of crop species found in fields is often not documented.

Among 572 species of European wild relatives of economically important crop species, 11% are threatened, and a further 4.5% of the species are near threatened (Bilz *et al.*, 2011; Kell *et al.*, 2012). More species are threatened at national level.

Figure **3 13** Temporal variation in the abundance of 17 grassland butterfly species averaged across 22 European countries during the 1990–2015 period. Source: van Swaay *et al.* (2017).

BUTTERFLY CONSERVATION EUROPE / STATISTICS NETHERLANDS
PAN-EUROPEAN GRASSLAND BUTTERFLY INDICATOR

CONFIDENCE LIMITS   •••••• INDEX   ——— TREND

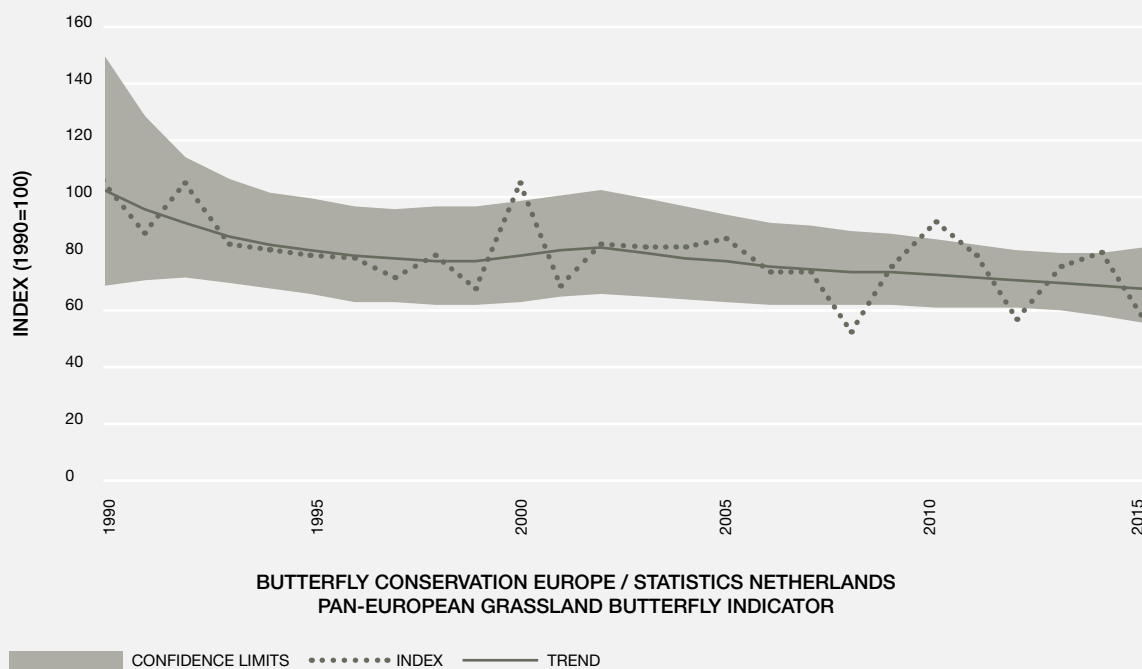Figure **3** **14** **A** **Change in the climatic range of the Colorado beetle from 1991 to 2010 compared with 1951 to 1970. 1: unsuitable areas for the beetle; 2: range increment; 3: suitable in both periods. Source: Popova & Semenov (2013).** **B** **Annual growth rate (%/yr) in the number of honey bee colonies for countries reporting those data to FAO between 1961 and 2012. Source: FAO (2013a).**
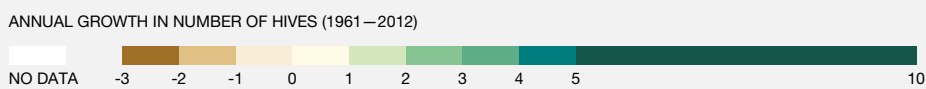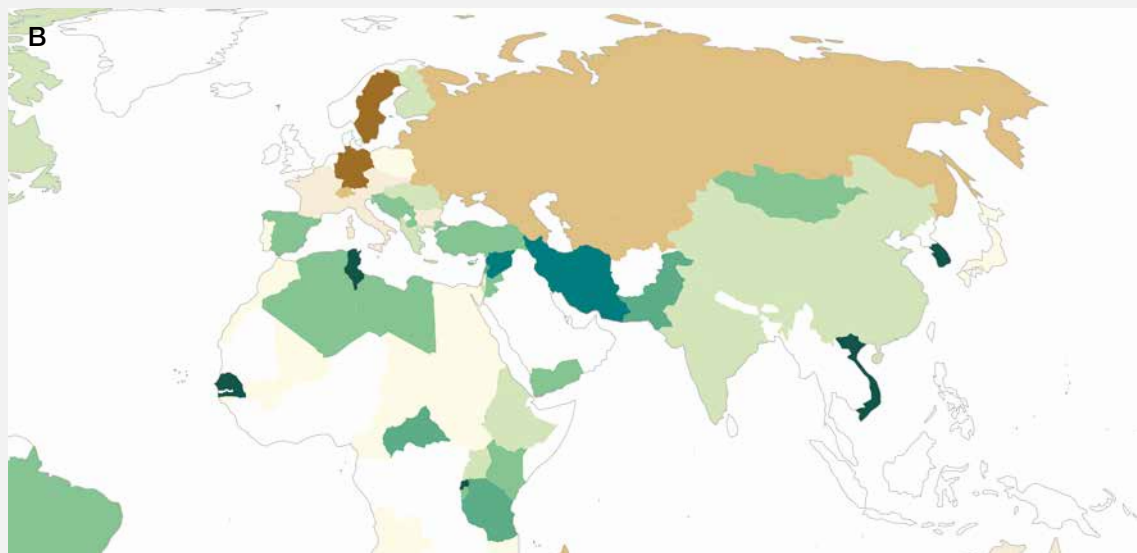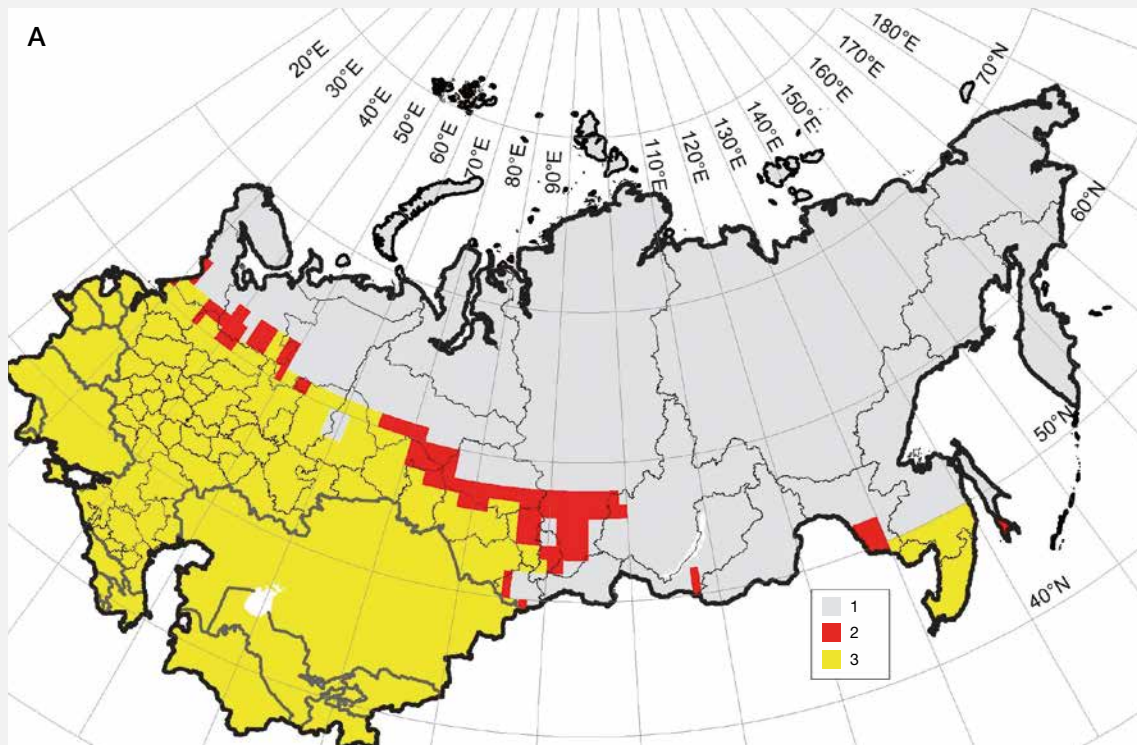
Figure **3 15** **Risk status of mammalian breeds in geographical Europe and the Caucasus up to December 2005. Relative (figures) and absolute (tables) numbers are presented. Source: FAO (2007).**
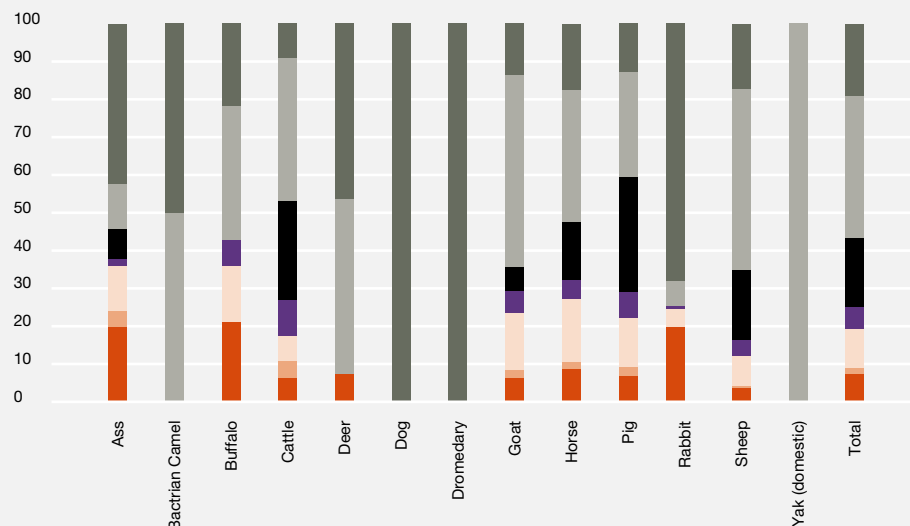
RISK STATUS	Ass	Bactrian Camel	Buffalo	Cattle	Deer	Dog	Dromedary	Goat	Horse	Pig	Rabbit	Sheep	Yak (domestic)	Total
Critical	10		3	35	1			15	41	23	38	26		192
Critical-maintained	2			24				5	8	7		5		51
Endangered	6		2	37				34	78	42	8	65		272
Endangered-maintained	1		1	51				13	22	21	1	32		142
Not at risk	6	2	5	205	6			117	163	90	13	375	1	983
Extinct	4			142				16	72	102		148		484
Unknown	21	2	3	48	6	13	1	30	79	41	127	133		504
TOTAL	50	4	14	542	13	13	1	230	463	326	187	784	1	2628

While scientific publications on biodiversity trends in agricultural areas in Central Asia and some parts of Europe are not numerous, precious information can be derived from indigenous and local knowlegde. For instance, Hungarian herders have a deep understanding of biodiversity and its trends in managed grasslands, and they also report a biodiversity decline, in particular for bird and plant species richness and abundances (Molnár, 2014; Varga & Molnár, 2014).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Main drivers influencing biodiversity in agricultural areas: The moderate utilization of lands in historical times was associated with high species richness in the rural landscape of Europe (Kull and Zobel, 1991; Pykälä, 2003), leading to the concept of high nature value farmland (Halada *et al.*, 2011). The traditional, non-intensive agriculture and the management of marginal lands generally have a positive role in maintaining high biodiversity levels.

As reported by a large majority of the many studies on this subject, lower biodiversity levels are very generally observed with increasing intensification of agricultural systems (Le Roux *et al.*, 2008). These are mostly related (1) at the landscape scale to decreased percentage of natural and semi-natural elements, decreased habitat diversity or crop diversity, and to a lesser extent reduced coverage of extensively managed crops; (2) at the field scale to increased addition of pesticides and fertilizers, and other practices like drainage; and to a lesser extent (3) to decreased connectivity between habitats (Le Roux *et al.*, 2008; STOA, 2013) **(Figure 3.16)**.

Overall, the effects of the level of agricultural intensification on the diversity of several taxonomic groups are now well documented, but are complex and depend on both the considered group, aspect of intensification and spatial scale (e.g. Jørgensen *et al.* 2016b for farmland birds). Intensive farming also has impacts on biodiversity outside agricultural areas and outside Europe and Central Asia (STOA 2013). In

Figure **3** **16** The species richness of aboveground invertebrate groups (here bugs, carabids and hoverflies) increases with the percentage of semi-natural elements and crop diversity in twenty-five 16-km^2 agricultural landscapes from seven European countries. Source: Billeter *et al.* (2008).
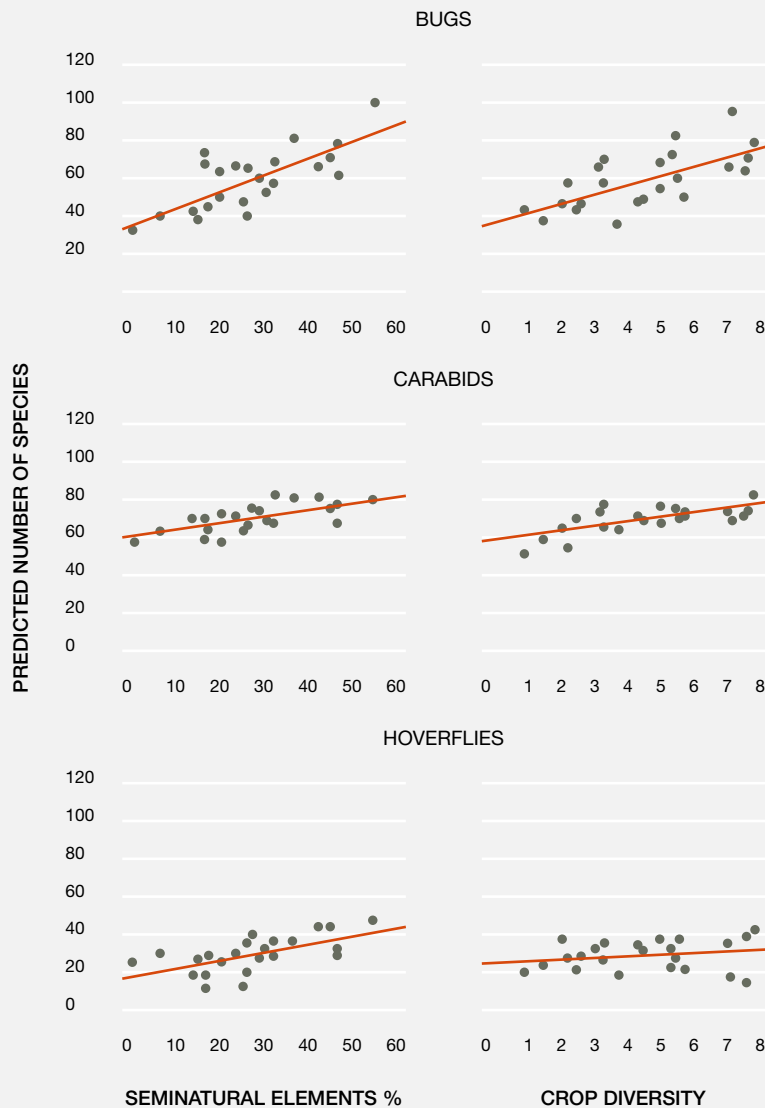
parallel, partial or complete abandonment of agricultural management on non-intensively managed systems is a major threat to biodiversity in Europe and Central Asia (Billeter *et al.*, 2008; STOA, 2013), and many studies have reported that biodiversity declines following abandonment for several biological groups (Le Roux *et al.* (2009) and references therein). In parts of Central Asia, the decline in cooperative farms and intensive agriculture based on relatively few economically important crops has led to a return to a more diverse crop production, offering opportunities to biodiversity.

Main drivers influencing particular taxonomic groups:
The steep decline in farmland bird populations during the 1980s and 1990s was associated with increasing agricultural specialisation and intensity in some areas, and large-scale marginalisation and land abandonment in others (Pe'er *et al.*, 2014). As these changes have expanded eastwards, a steeper decline has been reported in Central Europe in recent years (EBCC, 2013). Agri-environment schemes implemented after revision of the European Union's agri-environmental programmes in 2007 were not more effective for farmland bird diversity than schemes implemented before revision (Batáry *et al.*, 2015). In post-

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Box ③ ② **Does biodiversity increase in response to agri-environmental schemes?**

A meta-analysis (Batáry *et al.*, 2015) showed that agri-environment schemes benefit species richness and abundance, but several reviews reported that current schemes are not sufficient to reverse the decline of farmland biodiversity in

Europe (Berendse *et al.*, 2004; Kleijn *et al.*, 2006). This is likely due to the fact that many agri-environment schemes do not sufficiently target biodiversity conservation or are not applied over a sufficient land cover (STOA, 2013; see Chapter 6).

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soviet countries in Eastern Europe and Central Asia, the dynamics of farmland bird populations were mainly driven by land-use changes linked to the transition to the market economy (Kamp *et al.*, 2015; Kessler & Smith, 2014).

Intensifying agriculture on the one hand, and abandoned land (mainly in Eastern Europe and Southern parts of Western and Central Europe) on the other, are the two main driving forces affecting the populations of grassland butterflies (van Swaay *et al.*, 2015).

Evidence has accumulated of a significant decline in populations of bees (including bumblebees) over the past 60 years in geographical Europe, resulting mainly from agriculture intensification (IPBES, 2016b). Many of the environmental threats to wild bee diversity in Europe are associated with modern agriculture and, in particular, shifting agricultural practice and increasing intensification of farming (Nieto *et al.*, 2014). In addition, while agriculture has become increasingly pollinator-dependent, the number of honeybees required to provide crop pollination across 41 countries from the region has risen 4.9 times faster than honeybee stocks between 2005 and 2010 (Breeze *et al.*, 2014; Schatz & Dounias, 2016).

Bats, rodents, and herbivorous and carnivorous mammals, are all in decline due to agriculture intensification in geographical Europe since mid-20th century (e.g. Flowerdew, 1997; Pocock and Jennings, 2008). Among different drivers linked to intensive agriculture (Stoate *et al.*, 2009), molluscicides and rodenticides are considered the greatest risk to mammals, both through primary and secondary exposure (Shore *et al.*, 2003), while poisoning by pesticides persists or tends to decrease locally (Barnett *et al.*, 2006). It is noteworthy that several large mammals such as the wolf (*Canis lupus*), brown bear (*Ursus arctos*), lynx (*Lynx lynx*) wild boar (*Sus scrofa*), and moose (*Alces alces*), are probably gaining from land abandonment, expansion of forest cover or subsequent increase in ungulate mammal prey in Europe and Central Asia (Moreira and Russo, 2007; Falcucci *et al.*, 2007; Russo, 2007; Sieber *et al.*, 2015).

The role of ecologically-friendly agricultural practices:
During recent decades, agricultural practices and systems alternative to intensive ones have been developed (including new practices or previously widespread ones),

such as leaving field margins unsprayed, stricter pesticide management, reduced tillage, and organic farming (EBCC, 2017; see Chapter 4 for details and temporal trends). The effects of these "ecologically-friendly" agricultural practices on biodiversity are generally positive, but can vary, e.g. according to the landscape context and spatial scale of evaluation **(Box 3.2)**.

In particular, organic farming has been shown to increase local species richness of wild organisms, although with large variation between studies (Tuck *et al.*, 2014). The effect differs between taxonomic groups (Dicks *et al.*, 2016; Fuller *et al.*, 2005), with particularly beneficial effects on plants and pollinators (Batáry *et al.*, 2011, Tuck *et al.*, 2014). Other differences between studies can be attributed to the effect of landscape context (Tuck *et al.*, 2014), the local extent of organic farming (Gabriel *et al.*, 2010) and time since conversion to organic farming (Jonason *et al.*, 2011). However, beneficial effects of organic farming may be mainly local (Bengtsson *et al.*, 2005), and it is not clear whether effects on local biodiversity scale up to effects on biodiversity at regional scales (Gabriel *et al.*, 2006; Schneider *et al.*, 2014).

Given the low uptake of organic farming in areas with high agricultural intensification, where the effects on biodiversity would be greatest (Rundlöf & Smith, 2006), the actual effect of organic farming on general biodiversity trends may be smaller than expected. Organic farming may contribute to the maintenance of agriculture in marginal areas of high value for biodiversity (Gabriel *et al.*, 2009), but the extent of this effect remains unknown.

The question of how farmland conservation initiatives actually contribute to the policy objectives of halting the decline of agrobiodiversity largely remains to be addressed in a quantitative manner (see Kleijn *et al.* (2011) and references therein) and using adequate indicators.

3.3.2.10 Urban areas

OVERVIEW OF THE SUB-SYSTEM

Urban green infrastructures comprise systems of indigenous habitats, formal (e.g. parks, cemeteries) and informal

(e.g. ruderal, transportation areas) green space, artificial habitats (e.g. green roofs and walls, ponds), semi-natural and rural habitats. Taxa that occupy these habitats vary in their sensitivity to urbanization, with some assemblages comprising generalist species and others retaining specialist species and contributing more to biodiversity (Niemelä & Kotze, 2009). During the expansion phases of cities, both through outward expansion into the peri-urban region and densification, changes occur in the provision of green space and the composition of species assemblages (Kotze *et al.*, 2014). The European Union "Plan of Action on Subnational Governments, Cities and Other Local Authorities for Biodiversity (2011-2020)" emphasizes the essential role of cities in achieving the Aichi Biodiversity Targets. Also the 7th Environment Action Programme supports the development of initiatives for the conservation of biodiversity.

Urbanization has changed habitats, both spatially and through the release of heat, waste, nutrients and contaminants. Cities generate novel habitats and assemblages, as many species adapt to urban conditions, and urban habitats acquire characteristic communities. Disturbance is typical of urban habitats and they tend to remain at early to mid-successional stages, which can have high levels of species diversity. A number of the species that have become most adapted to cities originate in rocky habitats, such as the rock pigeon (*Columba livia*), the common swift (*Apus apus*) and the alpine swift (*Tachymarptis melba*) (Kelcey & Rheinwald, 2005). In Central Asia, the core urban avian fauna comprises 7 to 17 species (Fundukchiev, 1987) with distinctive adaptive traits to urban conditions.

Such novel features as green roofs and green walls have been introduced into many cities as potential means of enhancing the provision of supplementary habitats. Studies show that these can develop diverse assemblages of arthropods and vascular plants (Madre *et al.*, 2013), and they probably have the potential to support the biodiversity of some taxa.

PAST AND CURRENT TRENDS

As a result of intensive urbanization in the 20th and 21st centuries, patches of indigenous habitats have become fragmented, and many species have declined or disappeared. The overall result is generally a loss of species across most taxa, particularly specialized species, and a subsequent assemblage of mostly generalist species that are adapted to urban conditions.

Many species have adapted to urban conditions and are recognized as typical urban species. These include the European red fox (*Vulpes vulpes*) and feral pigeon (*Columba livia domestica*), and in Central Asia the common myna (*Acridotheres tristis*). In addition, many species have been

periodically recorded as expanding into urban areas, such as the flying squirrel (*Pteromys volans*) in Helsinki (Mäkeläinen *et al.*, 2016), the Eurasian eagle owl (*Bubo bubo*) in numerous cities (König & Weick, 2008) and the Eurasian lynx in Tallinn and Espoo. These probably result from declines in resources in peri-urban regions and availability of resources within urban regions. Vulnerable taxa, such as ground-nesting birds, do not persist in cities due to many threats.

Fish species have declined in urban areas, with the loss of migratory species, such as salmon (*Salmo salar*), sturgeon (*Acipenser sturio*) and river lamprey (*Lampetra fluviatilis*), through fragmentation due to obstacles to free movement along rivers. Modification of rivers by straightening channels, dredging and canalizing, has resulted in the loss of species that inhabit or breed in gravel beds and river margins. Recently there have been initiatives in many cities to restore natural features of rivers, improve water quality and enhance connectivity. Some fish species that are present outside urban areas, such as three fish species endemic to the River Danube, *Gymnocephalus schraetzer*, *G. baloni* and *Zingel zingel*, which are all occasionally recorded in Budapest (Tóth-Ronkay *et al.*, 2015), have potential to benefit from restoration of urban river systems.

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

There is a high level of impact of land-use change on both the extent of habitat and the biodiversity status. The loss, degradation and isolation of both terrestrial and aquatic habitats, is a major cause of declines in biodiversity. Habitat loss is mainly due to replacement of green space with urban infrastructure, and the conversion of indigenous habitats to managed habitats, such as parks and gardens (Kabisch & Haase, 2013). There is a high level of variation within the region **(Figure 3.17)** (Siedentop & Fina, 2012).

Habitat degradation includes qualitative changes in habitats that are not destroyed, but converted, such as woodlands converted to parks, species-rich grasslands - to lawns, or water bodies that are dredged, drained, canalized or diverted into pipes. Homogenization due to management practices leads to loss of specialized species and domination of communities by a small number of generalist species.

Relict natural habitats such as steppe grasslands and limestone caves in Budapest (Tóth-Ronkay *et al.*, 2015) and calcareous sand dunes in Rotterdam (Van de Poel *et al.*, 2015), support communities of specialized species, though fragmentation often leads to species losses and reduces the potential for re-colonization. Large old mature trees in parks, often more common even than in mature forests, can provide nesting cavities for birds and support communities

of saproxylic insects (Venn *et al.*, 2015) and fungi, such as polypores, though they have been reduced in some cities for safety reasons. Such habitats may be lost outside cities and become increasingly valuable for biodiversity (Gilbert, 1989), depending on their size and capacity to retain characteristic species communities.

Fragmentation and loss of connectivity is one major cause of biodiversity decline. Migratory species such as the common frog (*Rana temporaria*), which migrates between running water, still water and terrestrial habitats during its annual life cycle, are particularly vulnerable (Šťastný *et al.*, 2015). Fragmentation and isolation of habitats results from the development of urban infrastructure, particularly communication networks, such as roads, but can also include noise, light and chemical barriers both within and between habitats (Vershinin *et al.*, 2015). Some cities retain large green space elements and extensive corridor networks, often following the courses of rivers, such as riparian forests (Herrera *et al.*, 2015).

Climate change has less negative impacts in urban areas than in many other systems, as urban areas are warmer, lighter and drier, and thus their assemblages tend to contain mainly thermophilic species. However, cities in northern parts of the region, such as Helsinki and Rotterdam, are experiencing an ongoing influx of species of many insect taxa, including Lepidoptera, Carabidae, Odonata and Apidae, as a consequence of range

expansions due to climate change (Moerland *et al.*, 2015; Venn *et al.*, 2015).

Pollution affects habitats and communities most intensively and extensively in urban regions. Pollutants include heavy metals, pesticides, nutrients, salt, other chemicals, heat and light. In many cities, legislation has been introduced to control pollution, with consequent decreases in their levels in green infrastructure. Heavy metals are common in most urban soils, and lead levels can be high adjacent to major roads, due to the use of lead in petrol fuels prior to the 1990s. Pesticide residues (DDT, DDD, DDE, phosphorus organic-trichloroform) are present in high concentrations in suburban regions of some eastern European cities (Peskova, 2000). Some rare plants, with tolerance to metals, including a number of orchid species, occur at sites containing calcareous metalliferous spoils in the UK (Johnson *et al.*, 1978). The urban heat island phenomenon can increase temperatures by approximately 2–3°C in the urban core (Vershinin *et al.*, 2015). In northern Europe, many cities contain thermophilic species due to suitable microclimatic conditions. These include fish and amphibians in aquatic habitats and also fig trees (*Ficus carica*), for instance, in some UK cities (Gilbert, 1989). Thermal pollution can also result in phenological changes (Belimov & Sedalishchev, 1980; Fominykh & Lyapkov, 2011; Piano *et al.*, 2017). High levels of light pollution, particularly in Central and Western Europe **(Figure 3.18)** cause a disorientating effect on some nocturnally flying insect

Figure **3** **17** **Patterns of annual change of urban land cover across 26 European countries for the periods A 1990–2000 and B 2000–2006. Source: Siedentop & Fina (2012) based on CORINE land-cover data.**

| Now new urban area | less than 0,5% | 0,51% — 1% | 1,01% — 2,5% | more than 2,51% |
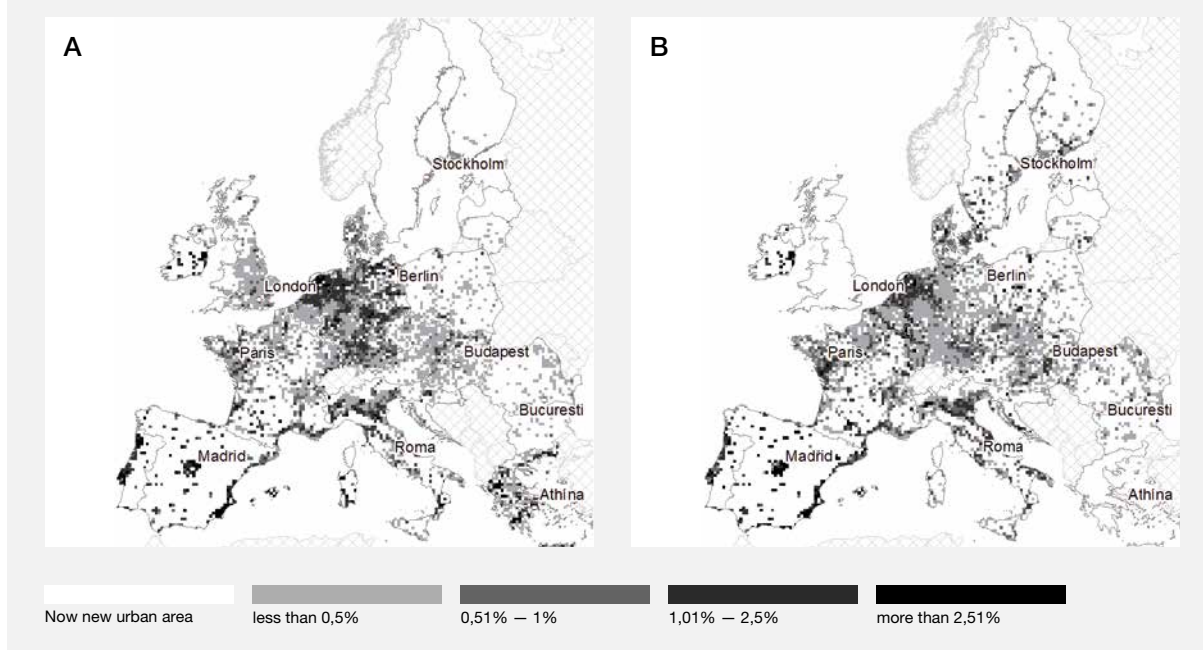
taxa and can compromise pollination (Knop *et al.*, 2017). Bats, amphibians and entomophagous mammals use this niche, i.e. streetlights, illuminated buildings, for foraging. Recently there have been initiatives to reduce the amount of energy used for lighting and the amount of light lost into the atmosphere.

Overexploitation in the urban systems is attributed to excessive utilization of recreational areas, which can lead to erosion. Tourism pressure has also had an impact on vulnerable biotopes in the Mediterranean region (Mansuroglu *et al.*, 2006). Land-use change, recreational activities and the intensification of fish farming have also affected populations of amphibians, as has the spread of the chytrid fungus *Batrachochytrium dendrobatidis*, which has devastated amphibian populations in many parts of Europe and Central Asia (Šťastný *et al.*, 2015; Tóth-Ronkay *et al.*, 2015).

Alien and invasive species seriously affect ecological equilibria, and displace indigenous species or hybridize with them (Rhymer & Simberloff, 1996). Urban sites are among the most invasible biomes (Richardson & Pysek,

2006). Exotic species are a problem in most cities. Both escapes of garden plants and the release of pets maintain alien species populations (Herrera *et al.*, 2015). It has been estimated that 2,000 exotic species of arthropods were introduced to Europe during the 20th century (Kobelt & Nentwig, 2008), mostly via cities. In the case of taxa introduced incidentally via anthropogenic activity, such as spiders and other arthropods, the majority of these arrive via international trade (Kobelt & Nentwig, 2008). Many cities have programmes for the control of alien invasive species, though a major problem is the delay between recognition of invasiveness and initiation of control measures. Some invasive plant species, such as *Elodea canadensis, Solidago canadensis, or Heracleum* species have colonized virtually the whole of Europe. Invasive plant and tree species, such as *Robinia pseudoacacia* and *Acer negundo*, also lead to homogenization of woodlands and loss of microhabitats and associated communities.

In aquatic communities, introductions of alien fish species including carp (*Carassius* spp), rainbow trout (*Oncorynchus mykiss*), silver carp (*Hypophthalmichthys molitrix*) and

Figure **3** **18** **NASA satellite image of global city lights (2008). Source: Craig Mayhew and Robert Simmon, NASA GSFC. Based on data from the Defense Meteorological Satellite Program.**

eel (*Anguilla anguilla*), reduce the potential for restoring indigenous communities (Herrera *et al.*, 2015; Šťastný, 2015; Tóth-Ronkay *et al.*, 2015).

CONSERVATION INITIATIVES

There are many cases of habitat and population restoration and species reintroductions in cities of Europe and Central Asia (McNeill, 2010). Many of these have been accomplished through EU LIFE actions. Many cities have biodiversity plans, or biodiversity incorporated into other strategic policy. There is ongoing encroachment of large areas of green space for development, due to the dwindling availability of suitable land for construction. Wetlands, rocky hills and other habitats have been conserved and many, such as the riparian forests of Dresden and Leipzig, have been protected (Haase & Gläser, 2009).

Parks and woodlands can also be valuable, and in many cities they are now managed less intensively, with retention of decaying wood for saproxylic species. Spider assemblages of cities are diverse and include a considerable number of species benefitting from humans and urban spaces (Fedoriak *et al.*, 2012). Many of these are also present in green infrastructure and some species have adapted to inhabiting buildings since the 1930s.

Lepidopteran species of meadows and open habitats, are particularly sensitive to urbanization, with poor levels of diversity in urban areas and higher diversity restricted to more natural areas at the periphery (Šťastný, Červený, Řezáč, *et al.*, 2015). The decline of Lepidoptera has resulted from intensive urban development, widespread use of pesticides during the post-war period and light pollution, which attracts and disorientates males of nocturnal species (Manu *et al.*, 2015). Many cities have had more diverse assemblages of Lepidoptera during the early 20th century (e.g. Manu *et al.*, 2015). Replacement of vegetation with solid surfaces is probably a major reason for this decline. River banks and remnant forest habitats still retain some noteworthy species, such as the ash hawkmoth (*Dolbina elegans*) in Bucharest (Manu *et al.*, 2015).

Cities also provide opportunities through the allocation of municipal resources to conservation for the maintenance of urban biodiversity. This can include mowing and grazing of meadows for the benefit of plants and insects (Venn *et al.*, 2015), management of wetland vegetation for amphibians (Šťastný, Červený, Rom, *et al.*, 2015) and control of invasive species. This is particularly important for species that decline due to the cessation of suitable management regimes of semi-natural habitats. However, many of these are affected by landscape change on such a large regional scale that local initiatives alone do not have the capacity to improve the situation dramatically.

3.3.2.11 Special systems

3.3.2.11.1 Heathlands

OVERVIEW OF THE SUB-SYSTEM

Dwarf-shrub dominated heaths are among the principal cultural landscapes of the Atlantic regions of Western Europe (Janssen *et al.*, 2016). These heathlands developed about 4,000 years ago as a result of forest clearances, and have since been maintained by a land-use regime that may include year-round free-range grazing by domestic ungulates, prescribed burning, cutting of vegetation and turf for fuel, and harvesting of vegetation for fodder (Gimingham, 1972; Kaland, 1986; Odgaard, 1994; Jansen *et al.*, 1997). Heathlands have since been an intrinsic part of the agricultural system, with the pattern and intensity of their use closely linked with the local agricultural economy (Diemont & Jansen, 1998; Kaland, 1986, Diemont *et al.*, 2013).

Heathlands harbour unique landscape and habitat qualities and specialized biodiversity, and are thus of nature conservation interest (Janssen *et al.*, 2016, Halada *et al.*, 2011, Rosa Garcia *et al.*, 2013, Halvorsen *et al.*, 2015, Nybø & Evju, 2017, Webb *et al.*, 2010). They support characteristic plant and animal assemblages (Webb, 1986), which respond to, and in part are dependent on, the interplay between traditional management practices and underlying environmental variability (vascular plants and bryophytes (Vandvik *et al.*, 2005; Velle *et al.*, 2014), carabid beetles (Bargmann *et al.*, 2015), other insects (WallisDeVries *et al.*, 2016), and soil invertebrates (Ponge *et al.*, 2015)). The long-term land-use history of heathlands has also had evolutionary consequences, for example, *Calluna vulgaris* seed germination is stimulated by smoke in heathlands, a trait that is lacking in populations from other habitats not regularly subject to burning, such as alpine areas (Vandvik *et al.*, 2014).

PAST AND CURRENT TRENDS

Traditional management practices maintained open heathlands until the beginning of the 20th century. During their maximum extent **(Figure 3.19)** heathlands occurred over several million hectares, but today less than 350,000 ha remain (Diemont *et al.*, 1996; Webb 1998).

The heathlands of Western Europe are now threatened throughout their range (https://bd.eionet.europa.eu/; Lindgaard and Henriksen 2011). In the Mediterranean parts of Western Europe, major heathland habitat types (European Union habitat number 4010 - Northern Atlantic wet heaths with *Erica tetralix*, 4020 - Temperate Atlantic wet heaths with *Erica ciliaris* & *E. tetralix*, and 4030 - European dry heaths), are reported by European Union member States to be in

Figure **3** **19** **Distribution of lowland heathlands in Western and Central Europe during their maximum extent, ca. 1850. Source: Haaland (2002).**

Figure **3** **20** **Distribution and conservation status of Habitat 4030 Dry Heaths across Europe.**

Green: Favourable status. Grey: Unknown. Yellow: Unfavourable-Inadequate status. Red: Unfavourable-Bad status. Note that heaths of countries not party to the Habitats Directive are not reported and mapped in Eionet. Source: https://bd.eionet.europa.eu.

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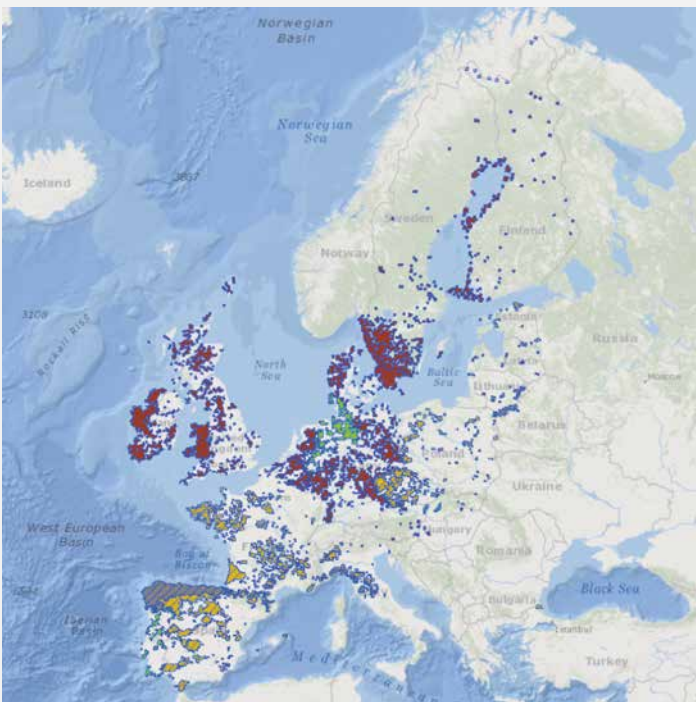
Figure **3** **21** The traditional management cycle with prescribed burning, grazing, and mowing, and the successional dynamics occurring after abandonment of management in heathlands. Source: Developed from Gimingham (1972).
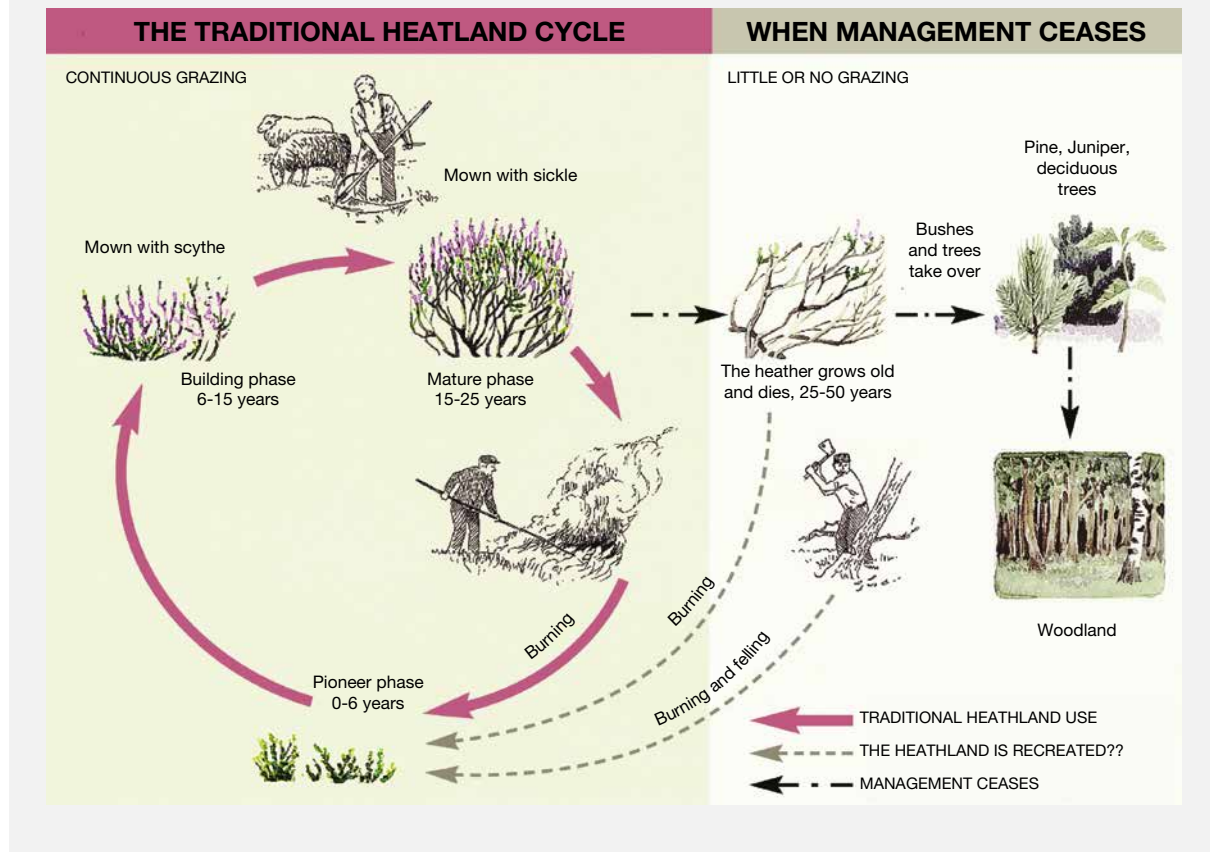
"inadequate" conservation status. In the Continental, Boreal and Atlantic parts[6] these same habitats are reported to have "bad" conservation status. Dry Atlantic coastal heath with *Erica vagans* (Habitat 4040) is somewhat less threatened, its status being classified as "inadequate". The European Red List of Habitats (Janssen *et al.*, 2016) classifies some heathland types (F4.1 Wet heath and F4.2 Dry heath) as "vulnerable". Approximately one third of the latitudinal distribution of heathlands is found in Norway, which is not party to the Habitats Directive. The corresponding Norwegian Red List for ecosystems and habitat types classifies northern coastal heathlands as "endangered" (Lindgaard & Henriksen 2011, **Figure 3.20**).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Across their range in Western Europe, heathlands thus suffer from poor conservation status and loss of biodiversity and ecosystem functions. The underlying

drivers of these negative trends vary. Pollution (especially atmospheric nitrogen deposition and associated acidification), land-use intensification, and habitat loss or fragmentation are the main drivers in the central parts of the heathlands' range (Härdtle *et al.*, 2009; Aerts & Heil, 2013). Land abandonment and habitat conversion, including afforestation, dominate in the southern, westernmost, and northern parts (Britton *et al.*, 2017; Fagúndez, 2013; Halvorsen *et al.*, 2015; Nybø & Evju, 2017). Contrasting processes may drive changes within the same landscape or region. For example, in the UK declines in the quality of lowland heaths have occurred due to increasing stocking in privately owned sites and succession towards woodland in areas managed for forestry or conservation (Diaz *et al.*, 2013). In the uplands of the UK over-exploitation for sheep grazing is a critical concern (Pakeman & Nolan, 2009). In some important parts of heathland range future prospects are undermined by controversies over their ecological importance and the sustainability of management regimes **(Figure 3.21)** (Davies *et al.*, 2016).

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6. Continental, Boreal and Atlantic parts of Western Europe as per EU Habitats Directive

3.3.2.11.2 Caves and other subterranean habitats

OVERVIEW OF THE SUB-SYSTEM

Subterranean habitats represent an extreme environment with unique particularities including trophic dependence on surface ecosystems. The relative constancy of abiotic factors makes these habitats and their associated fauna one of the most vulnerable on Earth to any disturbance (Juberthie, 2000). The absence of photosynthetic activity, limited supply of organic material, as well as stable temperature, high relative humidity and low rates of evaporation create an environment that determines the distribution and population density of cave fauna (Holsinger, 1988). Subterranean ecosystems encompass terrestrial and aquatic systems - the latter constituting freshwater, anchialine (with an underground connection to the ocean) and marine systems.

We distinguish to two types of subterranean systems, subterranean terrestrial systems (dry caves, epikarst, MSS (*milieu souterrain superficiel*)) and subterranean aquatic systems (flooded caves, groundwater, interstitial).

Typically, two main zones are recognized in the karst: epikarst (cutaneus zone, the surface; and soil and subcutaneus zone, the regolith and enlarged fissures) and endokarst (vadose zone, water unsaturated, and phreatic

zone, water saturated) (Ford & Williams, 2007; Palmer, 1991). Karst systems provide heterogeneous habitats of interconnected cracks, fissures and draines, filled with air or water. The karst process is polyphasic through geological time and related to the change of level of sea and landmass, so inactive (fossil) caves may be present at different elevations. Moreover, shallow subterranean habitats, as areas of habitable space that are less than 10 metres in depth beneath the surface (ranging from large areas such as lava tubes, to tiny areas such as cracks in cave ceilings or pore spaces in soil) have little in common with caves except for the absence of light and a specialized fauna with typical "cave" morphology (Culver & Pipan, 2014).

Subterranean habitats and there fauna are extremely vulnerable and endangered mostly by anthropogenic influences (pollution, overexploitation of caves, changing of water regime, building of hydropower plants and dams) as well as climate changes. Ecological categories are defined as stygoxene and trogloxene (stygo- relates to aquatic and troglo- to terrestrial) species, which spend their complete life cycle in surface environments and are only accidentally found in subterranean habitats; stygobite and troglobite species, which spend their complete life cycle in subterranean environments; stygophiles and troglophiles may have several kinds of life cycles— some populations live in surficial habitats and others in

Figure **3** **22** **Map of species richness patterns of Western and Central European obligately subterranean terrestrial species (troglobionts).**

The blue triangles are areas with few if any troglobionts, the orange triangles are areas with fewer than 50 species, usually much fewer than 50, and the orange circle is Ardeche, with fewer than 50 species in 5000 km² of area or less. The red circles are the diversity hotspots in Slovenia and Ariege. Red triangles are other possible diversity hotspots. The boundary of the Pleistocene ice sheet is shown as a scored solid line. A pair of dashed lines indicates the hypothesized position of the high-diversity ridge. Source: Adapted from Culver *et al.* (2006); Culver & Pipan (2013).
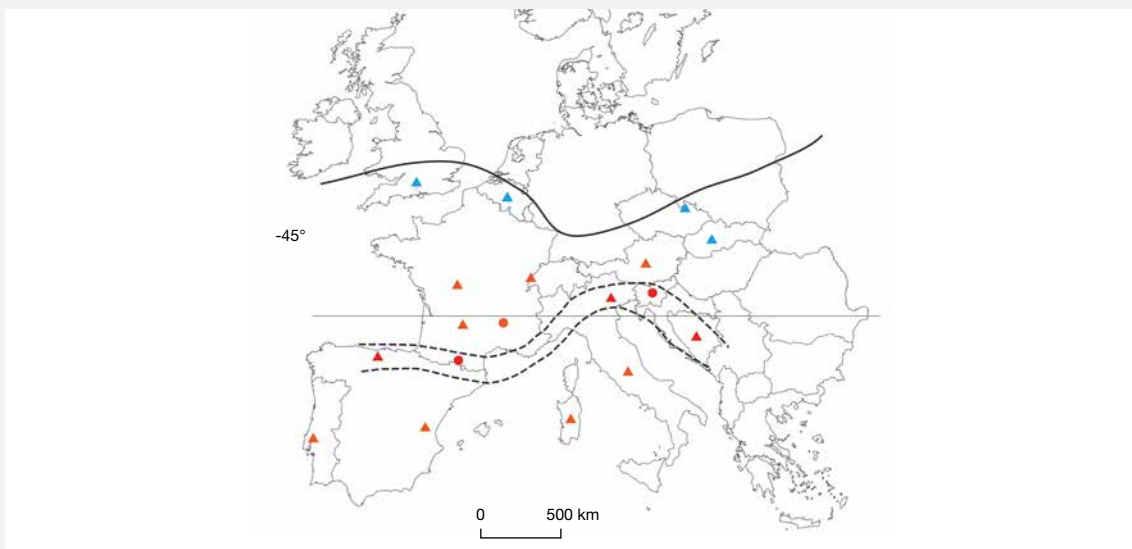
subterranean habitats, or individual life cycles necessitate use of both surface and subterranean environments (Gibert & Deharveng, 2002).

By 2000, approximately 5,000 obligate subterranean aquatic (stygobionts) and terrestrial (troglobionts) species from Central Europe had been described. 1,200 had been described from Asia, 500 from Africa, and 1,000 from North America (Gibert & Culver, 2005). Central Europe is both a hotspot of subterranean biodiversity and a hotspot of research into subterranean biology, both historically

and at present (Deharveng *et al.*, 2009). The Dinaric karst in the western Balkan Peninsula is a global hotspot of subterranean biodiversity, with more than 900 aquatic and terrestrial obligate subterranean species recorded (Sket, 2012a). Troglobiotic beetles are considered the most important contributors to terrestrial subterranean biodiversity in most temperate karst regions, including the Dinaric karst, where they present about 42% of the terrestrial troglobionts (Sket *et al.*, 2004). Subterranean biodiversity in Europe is actually higher than on other continents as indicated by (Culver & Sket, 2000).

Figure **3 23** **Distribution of the species richness of obligately subterranean terrestrial (troglobiotic) beetles in Dinaric karst at different grid cell sizes: Ⓐ 80 x 80, Ⓑ 40 x 40, (C) 20 x 20, Ⓓ 10 x 10, Ⓔ 5 x 5 km.**

Included are all records with localities of positional accuracy of 3 km or less, including 254 species (Lambert Conformal Conical Projection). Source: Zagmajster *et al.* (2008).

A

	1
	2 — 13
	14 — 27
	28 — 38
	45

B

	1
	2 — 7
	8 — 15
	16 — 22
	23 — 26

C

	1
	2 — 5
	6 — 11
	12 — 16
	17 — 19

D

	1
	2 — 3
	4 — 7
	8 — 11
	12 — 13

E

	1
	2 — 3
	4 — 7
	8 — 11
	12 — 13
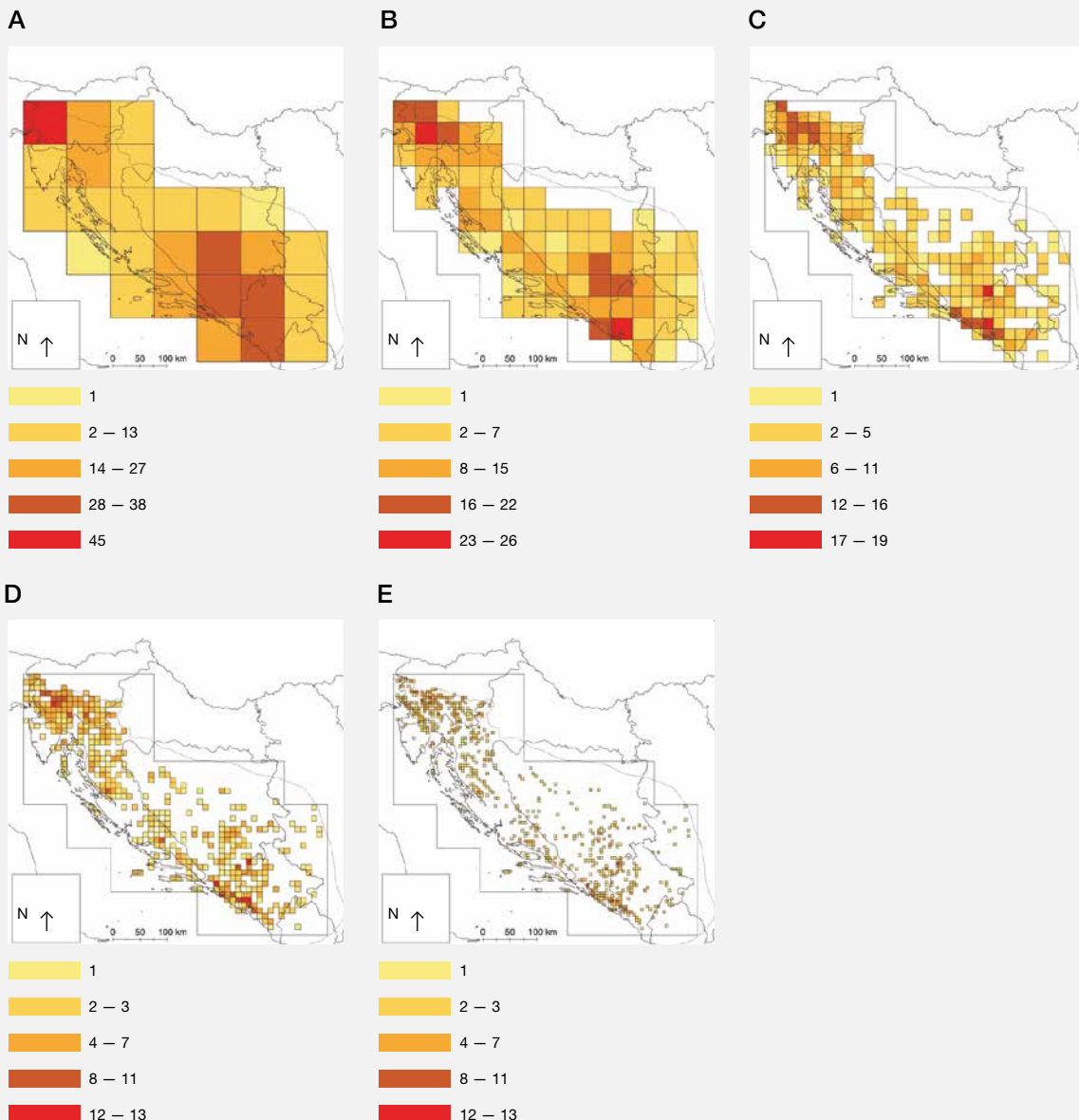
Figure **3** **24** **Map of obligately subterranean aquatic (stygobiotic) species numbers in 0.2 x 0.2 ° grid cells distributed across six Western and Central European countries. Source: Deharveng *et al.* (2009).**

NUMBER OF SPECIES
(thresholds at 25% and 50% of the maximum number of species)

41 — 79 (4 cells)

21 — 40 (29 cells)

1 — 20 (1,195 cells)
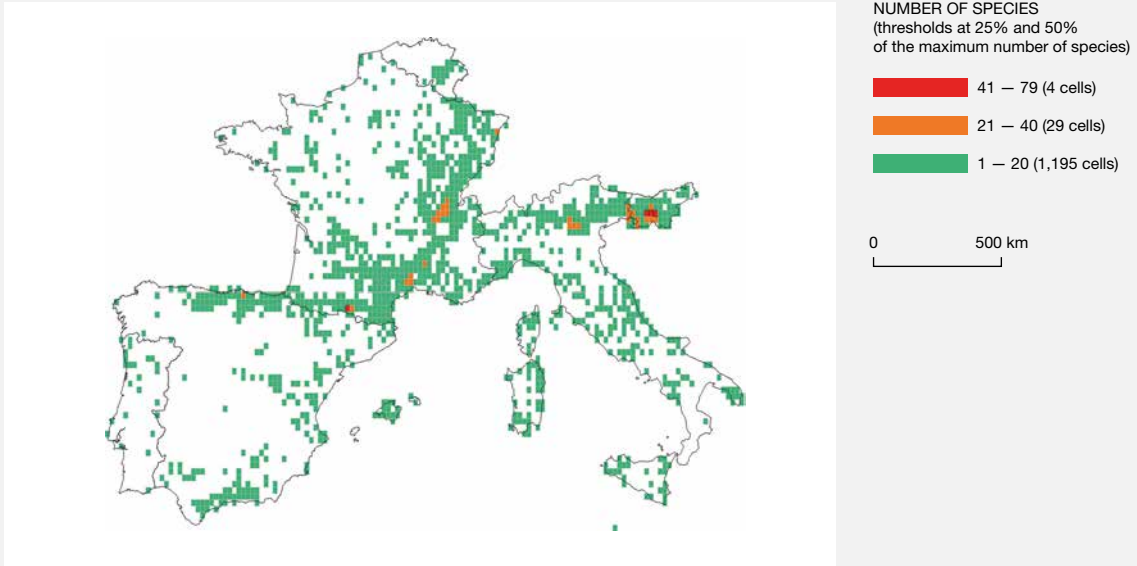
Figure **3** **25** **Olm *Proteus anguinus*, an endemic species of Dinaric karst (Rupećica Cave in Ogulinsko Zagorje, Ogulin, Croatia, 2014). Photo: Dušan Jelić.**

There are also visible geographic patterns within Western and Central Europe. The first one is a gradient in species richness with diversity decreasing from south to north and highest biodiversity within the mid-European high subterranean diversity ridge **(Figure 3.22)**. For details see Culver & Pipan (2013).

Some of the biodiversity hotspots are in the western Balkans (northeast Italy, Slovenia, Croatia, Bosnia and Herzegovina, and Serbia) and the Pyrenees (France and Spain). Increased diversity of stygobionts in the western Balkans could be explained by the complex biological and geological history of the Dinaric mountains (Sket, 1999) and complex history of the Mediterranean Sea (including its almost complete

drying about 6 million years ago during the Messinian crisis) **(Figure 3.23, Figure 3.24)**.

Population data are deficient compared with Western Europe, but information has recently improved for a few species, including olm *Proteus anguinus* (Trontelj & Zakšek, 2016; Trontelj *et al.*, 2009), chiropteran species (data collected by EUROBATS) and the bivalves *Congeria kusceri* and *C. jalzici* (Bilandžija *et al.*, 2014; Jovanović Glavaš *et al.*, 2017).

The Dinaric Arc is a habitat to one of the best-known representatives of stygofauna, the cave dwelling, blind salamander (olm; **Figure 3.25**). It is only found in the

Figure ③ ㉖ **Planned hydropower plants in the Balkan Peninsula. Source: Schwarz (2012).**
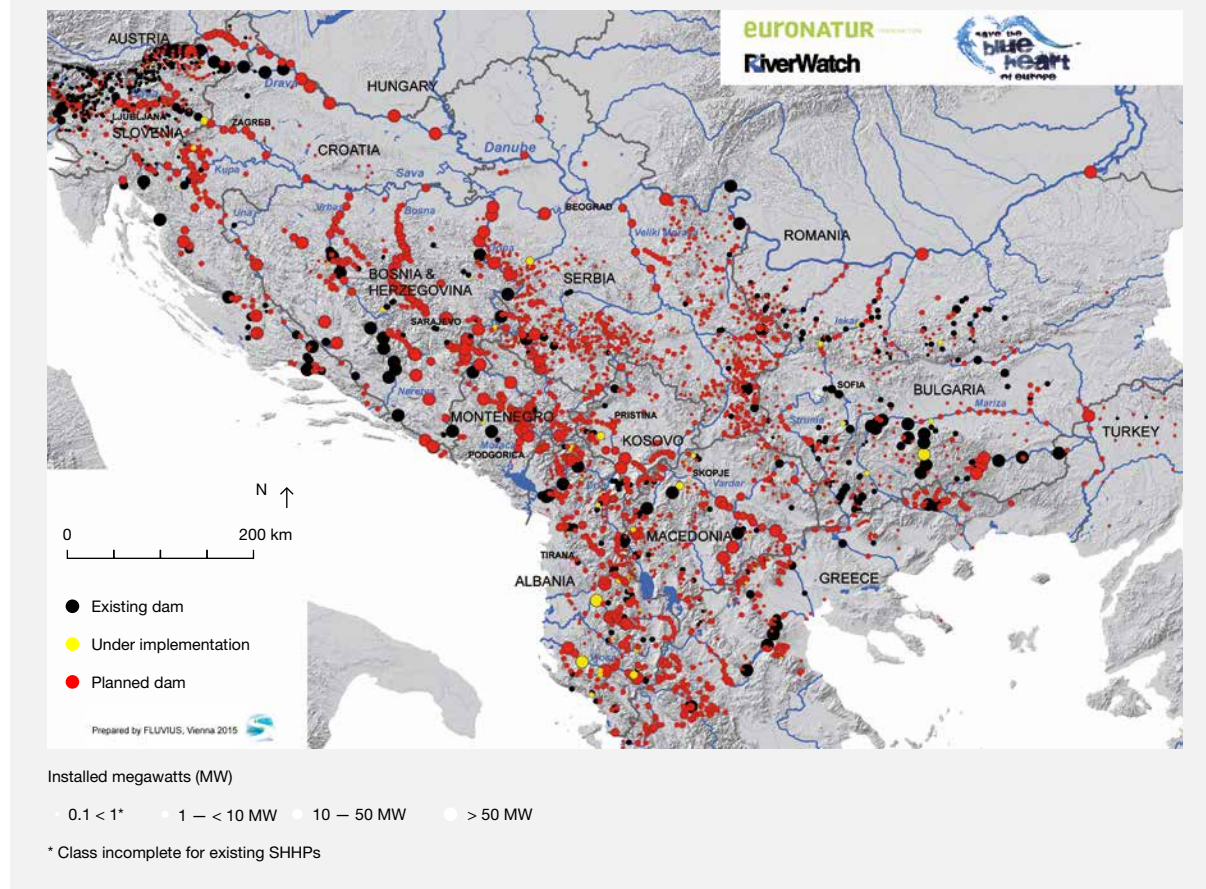
Dinaric karst region of the Balkan Peninsula (Italy, Slovenia, Croatia and Bosnia and Herzegovina; endemics of Dinaric karst) and is a globally vulnerable species (VU) (Arntzen *et al.*, 2009). Its distribution is severely fragmented, and there is a continuing decline in the extent and quality of its habitat (underground aquifers) (Jelić *et al.*, 2012; Sket, 2012b). The olm is the largest strictly cave adapted (stygobiont) species in the World (23-25 cm) and, until recently, it was the only exclusively cave-dwelling vertebrate species found in Europe. Then, in 2012 the first cave loach (*Cobitis damlae*), was discovered in the Dalaman river drainage which flows into the karstic plain of western Turkey (Erkakan & Ozdemir, 2012).

IMPACT OF DIRECT DRIVERS ON SUBTERRANEAN HABITATS AND FAUNA

In the nineteenth and early twentieth centuries, some animals were caught in large numbers for illegal trading purposes (Sket, 2012b). The trend of collecting and trading in rare and endangered fauna or even paleontological samples (for example *Ursus speleous*) still persists (Lukić-Bilela *et al.*, 2013).

The main threats are habitat loss, water regulation and flooding, dam projects, overextraction, quarries, and pollution. Moreover, due to a lack of research species are likely being lost before they are even scientifically described.

Shifts in water level regimes and seasonal cave flooding due mainly to hydropower development pose extreme threats to underground ecosystems. More than 2,700 new hydropower plants are being implemented or planned in the south of Central Europe (area of the Balkan Peninsula) **(Figure 3.26)**.

Above-ground pollution was reported to seeps directly into the subterranean habitats and destroys unique biodiversity (Danielopol *et al.*, 2003; Slingenberg *et al.*, 2009).

Climate change impacts these fragile ecosystems through reduction of water in aquifers and lack of seasonal flooding (Hunkeler, 2007). Cave temperature are generally strictly connected with the external climate (Badino, 2004) and thus increase.

Subterranean ecosystems are generally extremely oligotrophic habitats, receiving very little degradable organic

matter from the surface. Conversely, anthropogenic impacts on underground ecosystems (for example from intensive tourism and recreational caving) cause important alterations to the whole subterranean environment. In particular, artificial lighting systems in show caves support the growth of autotrophic organisms (the so-called *lampenflora*), mainly composed of cyanobacteria, diatoms, chlorophytes and mosses (Mulec & Kosi, 2009; Falasco *et al.*, 2014).

3.3.2.12 Progress towards Multilateral Environmental Agreements for terrestrial ecosystems

EUROPEAN UNION BIODIVERSITY STRATEGY

The European Union Biodiversity Strategy Target 1 "Fully implement the Birds and Habitats Directives" and Target 2 "Maintain and restore ecosystems and their services" define actions to ensure habitats and ecosystems protection. According to the 2015 mid-term review of the implementation of the Strategy by the European Environment Agency progress toward these targets is insufficient: 15.6% of terrestrial habitat assessments in the period 2007-2012 had favourable conservation status; 3.3% had unfavourable, but improving trends; 36.7% had unfavourable, but stable trends; 28,8% had unfavourable and declining trends; 11.2% had unfavourable status with unknown trend relative to the period 2001-2006 and 4.3% have unknown status (EEA, 2015d).

At the same time the network of Natura 2000 sites has progressed and is largely completed for terrestrial habitats, since 2010 it has grown by 1.4% and in 2015 covered 18.1% of land in the European Union. Overall, the European Union biodiversity targets 1 and 2 will not to be fully met by 2020 should the rate of progress not improve.

AICHI BIODIVERSITY TARGETS

Aichi Biodiversity Target 5 requires at least to halve the rate of loss of all natural habitats, including forests, and where feasible to bring it close to zero, and significantly reduce degradation and fragmentation. This is to be achieved through improvements in production efficiency and land-use planning, and enhanced mechanisms for natural resource governance combined with recognition of the economic and social value of ecosystem services provided by natural habitats (Nelson *et al.*, 2009, see Chapter 4). The emphasis for this target is specially made on preventing the loss of high-biodiversity value habitats, such as primary forests and wetlands. Recent evidence suggests that the rate of deforestation in Europe and Central Asia is decreasing (see 3.3), with some variations by country in Central Europe and Central Asia. Concerning terrestrial habitats, achievement of Target 5 is unlikely without increased implementation of

integrated forest management targeted at conservation of biodiversity and without halting negative trends of biodiversity in agricultural and other areas in Europe and Central Asia.

The network of Natura 2000 sites has progressed and is largely completed for terrestrial habitats, covering about 18% of the land in Western Europe and Central Europe. Countries in Central Asia and Eastern Europe traditionally report on the coverage of strictly protected areas and do not account for other effective area-based conservation measures. In their national biodiversity strategies and action plans (NBSAPs) reports to the Convention on Biological Diversity Eastern European and Central Asian countries committed to achieve protected areas coverage by 2020 at the level of 12% in Eastern Europe and 15% in Central Asia, and at the level of 22% and 19% for all types of sustainably managed and protected terrestrial areas. Thus, Western and Central Europe has largely progressed toward achieving Aichi Biodiversity Target 11. Further, the implementation of the NBSAPs commitments of 2017 would allow for meeting Aichi Biodiversity Target 11 for terrestrial ecosystems in Eastern and Central Europe.

3.3.3 Inland surface waters

3.3.3.1 Freshwater systems

OVERVIEW OF THE SYSTEM

Freshwater habitat includes streams, rivers, lakes, ponds (temporary or not) and also their sources (glaciers, aquifers or rainfall). Freshwater biodiversity includes organisms that either live permanently in water, or spend part of their life cycle in water. The freshwater ecosystems of Europe and Central Asia are very diverse. Based on the distribution and composition of freshwater fish species and major ecological and evolutionary patterns, almost 60 different freshwater "ecoregions" were depicted for this area (Abell *et al.*, 2008). They include large rivers in the Atlantic, Arctic and Pacific Ocean basins and the Mediterranean, Black, Caspian and Aral Sea basins. Lakes of different sizes are numerous in all subregions with Lake Baikal in eastern Russia dominating in size and volume, containing almost 20% of the world's freshwater. Overall, almost 60% of world water volume stored in lakes is located in Europe and Central Asia (Messager *et al.*, 2016). Out of four global biodiversity hotspots identified for the region, the Mediterranean basin is considered a hotspot for freshwater systems.

Freshwater systems are consistently at higher risk than their terrestrial or marine counterparts (Dudgeon *et al.*, 2006) and the quantity and quality of habitats and abundance of many species is declining in Europe and Central Asia. Agriculture

is the biggest user of fresh water, constituting 70–90% of the annual water demand for many countries (Rabalais *et al.*, 2010), and this is expected to further increase due to a growing population. In many regions, the lack of regulation of groundwater extraction has led to a decline in water tables. If all of the water in a river is used by agriculture and industry, leaving nothing for the aquatic environment, freshwater biodiversity will inevitably decline and freshwater ecosystems will disappear. Of course, this crisis point is unlikely to happen if technological solutions (e.g., change in farming practices, recycling waste water) are put in place to close the gap between supply and demand. Climate change is expected to intensify the hydrological cycle and alter evapotranspiration, with implications for ecosystem services but also feedback to regional and global climates. As a result, increased stress on freshwater ecosystems is expected in the coming decades.

The overall diversity of freshwater species in Europe and Central Asia has routinely been reported to increase towards lower latitudes, along with the proportion of threatened species. However, according to Dehling *et al.* (2010), in Europe this pattern differs for lentic (standing water) and lotic (running water) animal species. In Europe and Central Asia there is a high proportion of freshwater species with unknown population trends, for example in the case of 76% of European freshwater fishes and 83% of freshwater molluscs (Cuttelod *et al.*, 2011). This highlights the urgent need for monitoring and data collection across the region. However, according to Vörösmarty *et al.* (2010), the highest incidence of freshwater biodiversity threats worldwide is for Europe and Central Asia and correlates with the incidence of human water security threats.

PAST AND CURRENT TRENDS

Unfortunately, historical information and long-term data are rare for freshwater biodiversity and thus the patterns of species richness, for example, are known with much less confidence than for terrestrial systems (Carpenter *et al.*, 2009; Strayer & Dudgeon, 2010; Tockner *et al.*, 2008; Tockner *et al.*, 2011). This lack of quantitative freshwater biodiversity data is severe (e.g. 32% of IUCN evaluated freshwater invertebrate species in Europe are data deficient) especially for Central Asian freshwater ecosystems, as they have not yet benefited from IUCN Red List assessments.

The extent of wetlands in Western, Central and Eastern Europe has declined by 50% from 1970 to 2008 (Dixon *et al.*, 2016). According to the State of the Environment Report review of the state of freshwater systems, only 53% of geographical Europe's rivers and lakes have a good ecological status in 2015 (EEA, 2015a) **(Figure 3.27)**, despite several major European water initiatives in the past 15 years. Ecological status is a criterion for the quality of the structure and functioning of surface water ecosystems.

Based on current freshwater biodiversity trends, it is highly unlikely that Europe and Central Asia will achieve the relevant Aichi Biodiversity Targets by 2020 (i.e. Targets 2, 3, 4, 6, 7, 8, 9, 10, 11, 12, 13, 14) or Target 1 of the European Union Biodiversity Strategy. Furthermore, several water bodies in the region are drastically declining in size, and many ponds and streams are even disappearing from the landscape as a consequence of agricultural intensification, draining, dam construction and urbanization in combination to climate change (UNDP, 2015; Jeppesen *et al.*, 2015; Bagella *et al.*, 2016; Bogatov & Fedorovskiy, 2016; Boix *et al.*, 2016). Examples of water bodies disappearing are particularly found in the Mediterranean region and Central Asia (Jeppesen *et al.*, 2015). An example is Lake Akşehir, which was previously one of the largest freshwater lakes in Turkey, but completely disappeared due to loss of surface and ground water sources through intensive crop irrigation (Doğan, n.d.; Jeppesen *et al.*, 2009).

In the Mediterranean region, there is sometimes no legal requirement for a permanent minimum water outflow from dams and this often has dramatic consequences in summer when rivers dry out downstream (Benejam *et al.*, 2016; Freyhof, 2011).

A further issue of concern is the conservation of ponds in Europe and Central Asia at landscape scale, which harbour a significant proportion of aquatic biodiversity but are under increasing pressure. They have been historically neglected particularly in the Mediterranean region (Boix *et al.*, 2016; Céréghino *et al.*, 2008) and remain excluded from the provisions of the European Union Water Framework Directive. Natural wetlands (marshes and bogs) decreased by 5% between 1990 and 2006, one of the largest proportional land cover change of all habitats (EEA, 2010). In the Mediterranean region, temporary ponds contain rare, endemic or Red Data List species and as such form an irreplaceable type of habitat for a variety of freshwater biota (Céréghino *et al.*, 2008). However, the shallowness and small size of many temporary ponds have made them very vulnerable to human impacts as they can easily be drained for agriculture, urbanization, tourism, or industrial purposes (Boix *et al.*, 2016; Zacharias *et al.*, 2007). Moreover, annual rainfall has been declining substantially since 1900 in several parts of the Mediterranean region owing to climate change, and already dry periods in rivers and wetlands have been markedly prolonged.

European Union member States reporting under the Habitats Directive indicate that 17% of Europe's freshwater habitats have an "unfavourable to bad" conservation status, while 56% were classified as "unfavourable to inadequate" (EEA, 2015a) **(Figure 3.27)**. Yet relatively unaffected parts of the European Union include parts of the Balkans which, although not devoid of pressures, are freshwater biodiversity hotspots of continental and global value (Griffiths

Figure **3** **27** State of Western and Central European rivers and lakes. Good ecological status is defined "slightly" differently than high ecological status (with no or minimal human impact) and represents the target value that all surface water bodies have to achieve in the near future. Source: EEA (2015a).

Percentage of classifed water bodies
in less than good ecological status or potential
in rivers and lakes

- < 10%
- 10 — 30%
- 30 — 50%
- 50 — 70%
- 70 — 90%
- ≥ 90%
- EEA member countries not reporting under Water Framework Directive
- No data
- Outside coverage
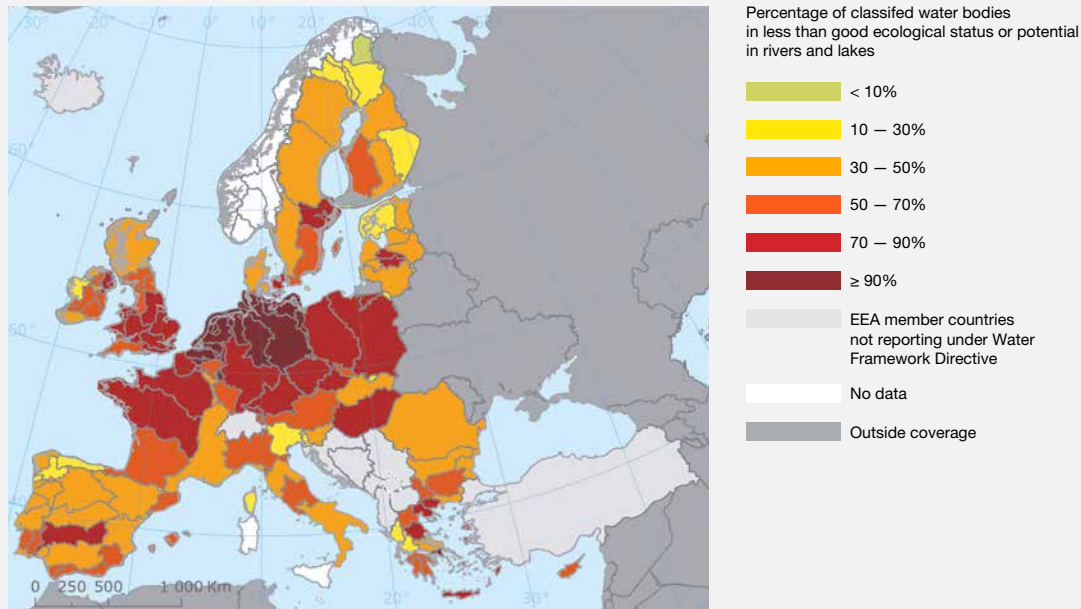
et al., 2004). Concerning species, 30% assessments have an "unfavourable to bad" conservation status and 45% assessments were classified as "unfavourable to inadequate" (EEA, 2015a). For Eastern Europe, fresh water quality remains poor, with variation from contaminated to extremely polluted for the majority of large rivers in Russia (Government of the of Russian Federation, 2016). In Central Asia in mountainous regions water bodies were assessed as clean and even very clean, when in lowlands they were assessed as moderately polluted and sometimes as extremely polluted (UNECE, CAREC, 2011).

Increased air temperatures result in melting of the glaciers which feed rivers and streams of Central Asia (e.g. Amu Darya, Syr Darya), causing changes in their hydrological regime (Zoï, 2009). Many formerly perennial wetlands are now seasonal, while several formerly seasonal wetlands are now rarely flooded. In other parts of Europe and Central Asia, recent climate change has produced contrasting trends. For example, floods in the Artic Ocean basin are becoming more prevalent due to an increase in winter runoff over the past 30 years, underpinned by the melting of Central Asian glaciers (Georgievsky, 2016; Gurevich, 2009). The Central Asian subregion also suffers from a drastic water loss that constitutes over 70% of global net permanent water loss. This water loss is due a combination of drought and human activities including river diversion, damming and unregulated water intake (Pekel *et al.*, 2016).

In addition, in the southern Caucasus and in Central Asia, there is a decline in surface water quality due to poor water treatment facilities. This leads to an increase in organic pollution, with about 20% of untreated sewage directly discharged into rivers (Barenboim *et al.*, 2013; Georgiadi *et al.*, 2014). Freshwater salinization is also a threat across Europe and Central Asia (Cañedo-Argüelles *et al.*, 2016; Jeppesen *et al.*, 2015), however, it is most relevant for the arid parts of Central Asia and the Mediterranean region due to irrigation and land washing salt pollution (Crosa *et al.*, 2006; Jeppesen *et al.*, 2015; Karimov *et al.*, 2014a). The lack of international and inter-sectoral coordination (e.g. between the irrigation and energy sectors) of water resource management in Central Asia and the Caucasus in the construction of irrigation systems, canals and water storage reservoirs in the lower reaches and deltas of the Central Asian Amu Darya, Kura, Syr Darya, Hrazdan and Ural Rivers has resulted in a severe environmental crisis (Petr *et al.*, 2004). Overall, despite contrasting trends in the availability of water resources in part of Europe and Central Asia (i.e. drying of ponds, flooding of rivers), the resulting environmental trend is a rapid decline in freshwater habitat quality and the decline in the most fragile species.

According to a recent study that identified the most important catchments for the conservation of freshwater biodiversity in geographic Europe (see Carrizo *et al.*, 2017), protected areas do not currently provide sufficient

Figure **3** **28** **Critical catchments (i.e. catchments that contain sites likely to qualify as freshwater "key biodiversity areas") for fishes, molluscs, odonates and aquatic plants, with 706 catchments shaded by the number of distinct trigger species. Source: Carrizo et al. (2017).**

NO. TRIGGER SPECIES

- 0
- 1 — 2
- 3 — 5
- 6 — 12
- 13 — 21
- 22 — 69
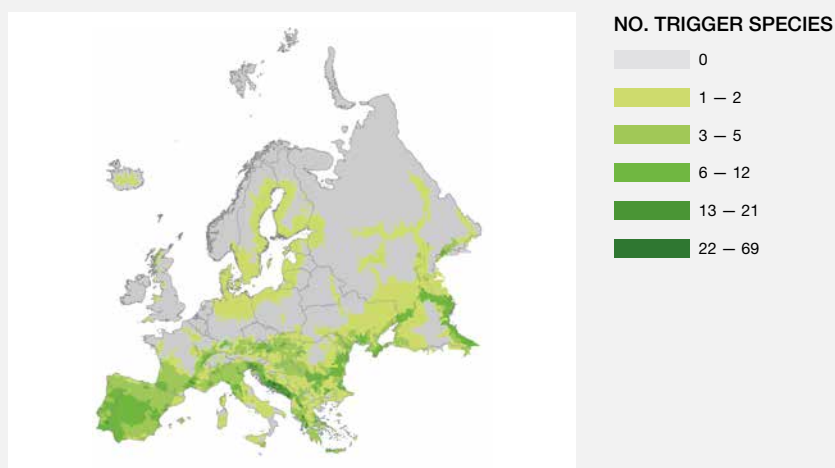
coverage to the most important "critical catchments" (i.e. catchments that contain sites likely to qualify as freshwater "key biodiversity areas") **(Figure 3.28)**. Without improvement to the current configuration and perhaps management, European countries are unlikely to meet international obligations to reverse the loss of freshwater biodiversity.

Alien species trends

The rate at which alien freshwater species have been introduced in Europe and Central Asia has doubled in the space of 40 years, with the principal motives being aquaculture (39%) and improvement of wild stocks (17%) (EC, 2014; Gozlan, 2008, 2015). Most sought-after freshwater species have already been introduced in Europe and Central Asia rivers and lakes and have contributed to biotic homogenization (Gozlan, 2016; Vilà & Hulme, 2017). In Central and Western Europe, 16% of lakes contain alien fish species (Jeppesen, Winfield, et al., 2017). The role of alien species in the emergence of novel diseases in the region has clearly been demonstrated in the last three decades through the increased geographic distribution of pathogens and parasites and also as facilitators of host-switching (Peeler et al., 2011). In the European Union, the historical trends of alien species introduction have been slowed down due to legislation (European Union, 2007) concerning use of alien and locally absent species in aquaculture. This regulation establishes a "framework governing aquaculture practices to assess and minimize the possible impact of non-native species on aquatic habitats and in this manner contributes to the sustainable development of the sector".

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Analyses of data on freshwater biodiversity show that more than 75% of Europe and Central Asia catchment areas are subject to multiple pressures and have been heavily modified, resulting in serious threats to their biodiversity (EEA, 2010; Tockner et al., 2008). General threats to inland water ecosystems such as overexploitation, water pollution, flow modification, habitat degradation, invasive alien species and salinization (Dudgeon et al., 2006), are also the most relevant for Europe and Central Asia. Vörösmarty et al. (2010) have classified the importance of these drivers for biodiversity status and have shown that the main drivers threatening biodiversity in areas where incident threat is greater than the 75th percentile (i.e. most of the region), is water resource development (e.g. dams, river fragmentation), followed by pollution (e.g. organic pollution and sediment loading). In comparison, the effects of fishing and aquaculture pressure remain relatively limited, while the impact of alien species is projected to increase in the future (EEA, 2015d). This is further illustrated by another recent study at continental scale based on 4,000 monitoring sites across Europe (Malaj et al., 2014) which showed that the health of almost half of all European freshwater ecosystems are at risk from toxic organic chemical pollution. The chemical risk to freshwater ecosystems is strongly influenced by human land use, with areas of natural vegetation at significantly lower risk. Pollution pressures particularly affect central and north-western parts of Western European areas with intensive agricultural practices and high population density. Notably, the chemical status of 40% of Europe's surface waters remains unknown (EEA, 2015b) and

a good chemical status (as defined by the European Union Water Framework Directive in terms of compliance with all quality standards established for chemical substances at European Union level) was not achieved in surface water bodies in 22 member States in 2015. Furthermore, although in most parts of Europe the potential for hydropower is almost fully exploited, the Balkans, which are a freshwater biodiversity hotspot of continental and global value, rank under the top world regions concerning planned dams and impoundments (Griffiths *et al.*, 2004; Zarfl *et al.*, 2015). The boom in hydropower development threatens the remaining free-flowing rivers and near-natural freshwaters including in Siberian rivers (Saltankin, 2012). Similarly, according to current plans, Turkey's rivers and streams will see the construction of almost 4,000 dams, diversions, and hydroelectric power plants for power, irrigation, and drinking water by 2023 (Şekercioĝlu *et al.*, 2011).

According to the State of the Environment Report 2015's (EEA, 2015a) review of the health of freshwater systems in Western and Central Europe, the pressures reported to affect most surface water bodies are pollution from diffuse sources, in particular from agriculture, causing nutrient enrichment. More than 40% of rivers and coastal water bodies and more than 30% of lakes and transitional waters in European Union subregions are affected by diffuse pollution from agriculture (EEA, 2012). Between 20% and 25% are subject to point source pollution, for example, from industrial facilities, sewage systems and wastewater treatment plants. Across Europe and Central Asia, industrial and agricultural developments also influence water quality and threaten biodiversity in some highly diverse ecosystems (e.g. Selenga River and Lake Baikal in eastern Russia) (Sorokovikova *et al.*, 2013). Nevertheless, pollution and nutrient enrichment are the only pressures that are reported to be decreasing in part of Western and Central Europe (EEA, 2015a; Jeppesen *et al.*, 2005). Agriculture is the main reason for groundwater over-abstraction, an activity that is frequent in areas with low rainfall and high population density, and in areas with intensive agricultural or industrial activity, such as Italy, Spain, Greece and Turkey, among others. The result is sinking water tables, empty wells, draining of wetlands, higher pumping costs and, in coastal areas, the intrusion of saltwater from the sea which degrades the groundwater (Rabalais *et al.*, 2010). Climate change and other components of global change, such as a growing population demanding higher food production, are expected to intensify these problems. Global warming can also exacerbate the symptoms of eutrophication in lakes and thus lower nutrient loading will be needed in a future warmer world to achieve the same ecological status as today (Jeppesen *et al.*, 2017).

Invasive alien species

Although increasing with the number of introductions, the risk of ecological impact after the introduction of an

alien freshwater fish species is less than 10% for the great majority of alien freshwater species introduced (Gozlan, 2008). However, alien species are very numerous in many freshwater bodies (Altermatt *et al.*, 2014) there are specific threats associated with the introduction of freshwater species which clearly need to be mitigated, such as the risk of alien pathogen introductions (Peeler *et al.*, 2011) and alien species that have been clearly identified as ecosystem engineers. The heightened risk associated with these species is that they are especially difficult to eradicate (Cacho *et al.*, 2006) and capable of significantly altering the functioning of ecosystems.

3.3.3.2 Enclosed seas and saline lakes

The Aral Sea

OVERVIEW OF THE SYSTEM

In the mid-twentieth century, the Aral Sea was the fourth largest lake in the world with an area of 67,499 km^2 (Aladin & Plotnikov, 2008) and water volume 1,064 km^3 (Glazovsky 1990). The biodiversity of this moderately saline (around 10 g/l salt) lake (Dobrovolskii and Zalogin, 1982) included about 200 species of invertebrates (Plotnikov, 2016), 34 fish species (Aladin and Plotnikov, 2008; Ermakhanov *et al.*, 2012; Zonn *et al.*, 2009), and 30 species of macrophytes (Zhakova, 2013).

The Aral Sea is, however, now a much smaller and more saline body of water **Figure 3.29**. Salt-dust and sandstorms originating from the desiccated seafloor are affecting agricultural systems and the livelihood and health of the people in the region (Breckle *et al.*, 2012). Full restoration of the Aral Sea in the foreseeable future appears impossible (Micklin, 2007).

PAST AND CURRENT TRENDS

From the 1960s, the Aral began to shrink because of large-scale water extraction from the two main in-flowing rivers, the Amu Darya and the Syr Darya (Boomer *et al.*, 2000). The sea split into two isolated lakes, the Small and Large Aral Lakes. By 1989 the Large Aral Sea divided further into Western and Eastern parts (Aladin & Plotnikov, 2008). By 2014, the eastern part of the Large Aral Sea had dried completely, but later some water appeared again **(Figure 3.29)** (Lindsey, 2016; NASA, 2014). Climate changes have also contributed to transformation of the Aral Sea (IPCC, 2014b).

The desiccation of the Aral is considered the world's worst aquatic ecology crisis in recent history (Pekel *et al.*, 2016). Negative effects of the Aral's retreat on the ecology, economy, and quality of human life in the region are manifold and dramatic (Micklin, 2007; Zavialov, 2005).

Figure 3 29 **NASA's image: shrinking of the Aral Sea. 1 – Small Aral; 2 – Large Aral; 3 – Western Aral; 4 – Eastern Aral. The fine line shows the approximate shore line in 1960. Source: Lindsey (2016).**

Figure 3 30 **The Caspian Sea. Source: NASA (2004).**

A dam separating the Small Aral basin from the Large Aral basin has resulted in an increase in the water level and decrease in salinity of the Small Aral. As a result the biodiversity of invertebrates has increased (Plotnikov, 2016). The Small Aral was stocked with fish and now even provides some commercial fish yields. The Large Aral Sea has split to several hypersaline lakes with biodiversity limited to species which are tolerant to high salinity, with a few species of invertebrates (Plotnikov, 2016) and macrophytes (Zhakova, 2013), but no vertebrates (Aladin *et al.*, 2017).

The Caspian Sea

OVERVIEW OF THE SYSTEM

The Caspian Sea is the largest saline inland sea or lake in the world, it contains about 40% of all inland lake waters (Messager *et al.*, 2016) **(Figure 3.30)**. This brackish water body, with salinity up to 14 g/l (Mamaev, 2002), is a home to 1,814 species and subspecies (Dumont *et al.*, 1999; Kasymov, 1987; Kazantcheev, 1981). Endemism at the species level is very high, especially among molluscs and fish. There are five sturgeon species that are endemic or shared only with the Black Sea and constitute 85% of the standing stock of the world's sturgeon population (Dumont *et al.*, 1999; Mamaev, 2002). The only aquatic mammal is the endemic Caspian seal (*Pusa caspica*) (Mamaev, 2002), assessed as endangered by the IUCN (Goodman & Dmitrieva, 2016). The Caspian Sea lies on migration routes of many birds and offers refuge for a number of rare and endangered bird species (Mamaev, 2002).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Changes in the level of the Caspian Sea play a significant role for ecosystems, but their causes are uncertain. They may be caused partly by climate change and decrease of inflow after the construction of dams on the Volga River (Barannik *et al.*, 2004; IPCC, 2014a; Dobrovolskii and Zalogin, 1982; Mamaev, 2002). Since 1995 the level of the Caspian Sea has not changed significantly, but it is impossible to predict the scale and direction of future fluctuations (Pekel *et al.*, 2016).

The Caspian Sea is threatened by pollution from untreated wastewater from industry and agriculture along the Volga River (an estimated 80% of the total load) (Glantz & Zonn, 1997) offshore oil and gas production, processing, extraction and transportation, and shipping. Industrial pollution impacts biological processes including the growth of commercially important fish (Dumont *et al.*, 1999; Mamaev, 2002).

The Lenin Canal between the Don and Volga Rivers, which opened the Caspian to maritime navigation in 1954, led to invasions by Mediterranean biota such as small crustaceans, marine molluscs (e.g. *Mytilaster Zineatus*) and comb-jelly (*Mnemiopsis lediyi*), which drove some endemic species (e.g. the bivalve *Dreissena caspia* or one of the main fish resources *Clupeonella*) to almost total extinction (Dumont *et al.*, 1999; Rintelen & Van Damme, 2011; Zoï, 2012).

Fishing has significantly dropped during the 1990s, and slowly grew thereafter (Makoedov *et al.*, 2007; **Figure 3.31**). During the 1990s, illegal fishing vastly increased and negatively impacted mostly sturgeon and salmon. A special moratorium on sturgeon fishing was signed by five Caspian countries in 2013. All Caspian sturgeon species are protected under CITES (the Convention on International Trade in Endangered Species of Wild Fauna and Flora), but the Convention is not in force in Turkmenistan. A quota system, introduced together with a temporary ban on pelagic fishing, does not appear to have been effective in reviving the dwindling sturgeon population (Mamaev, 2002).

The population of the Caspian seal (*Pusa caspica*, a globally endangered species) has declined by 70% in the last

Figure **3** **31** **Total fishing catch in the Caspian Basin. Source: Makoedov *et al.* (2007).**

FISH CATCH, THOUSAND TONS
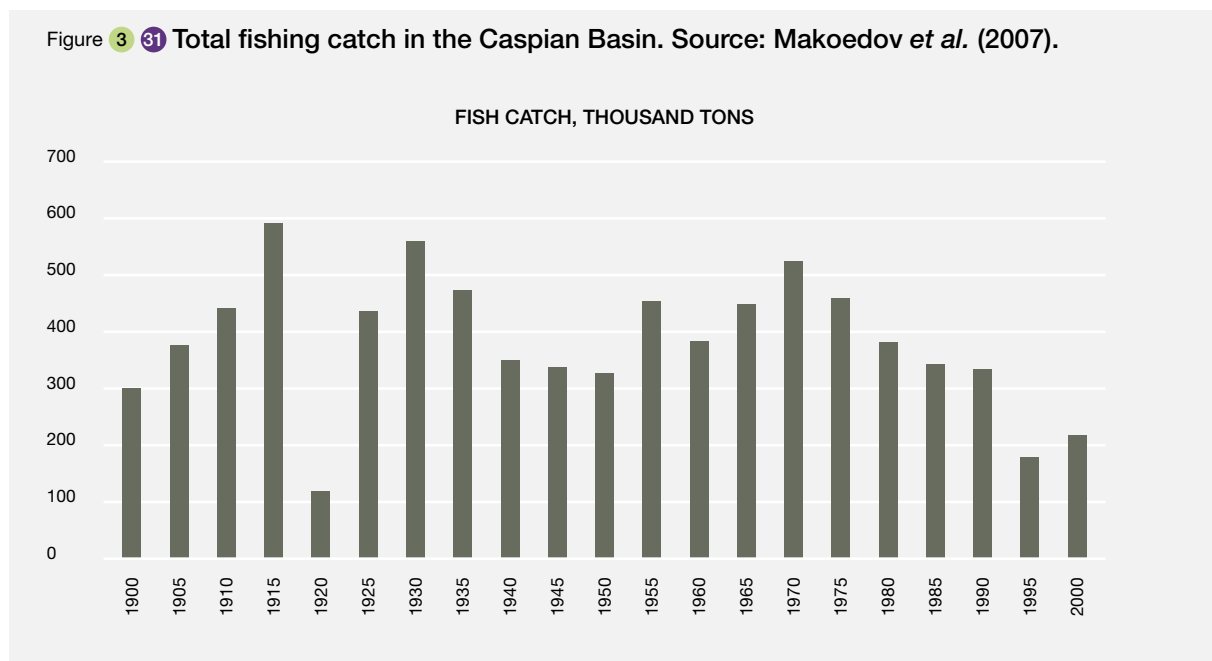
twenty years. This is primarily due to unsustainable hunting, trapping as by-catch of the illegal sturgeon fishery, and loss of prey-base due to fishing and invasive species (Goodman & Dmitrieva, 2016; Harkonen *et al.*, 2012). A canine distemper epidemic starting in April 2000 also contributed to the seal decline (Mamaev, 2002). Limitations on hunting were introduced in the 1940s but illegal killing of seals is still common (CEP, 2007; Mamaev, 2002).

Saline lakes

OVERVIEW OF THE SYSTEM

In Western and Central Europe saline and brackish lakes can be found predominantly in the Mediterranean region (Čížková *et al.*, 2013). To the east, saline water bodies are found in many terminal basins in a wide territorial belt with semiarid or arid climate including Turkey, the Caucasus, Central Asia and southern Siberia (Comin and Alonso, 1988; EEA, 2002; Kazanci *et al.*, 2004; Kotova *et al.*, 2016; Kulagin *et al.*, 1990; Montes & Martino, 1987; Orlov *et al.*, 2011; Örmeci & Ekercin, 2005; Government of Turkey, 2014; Stenger-Kovács *et al.*, 2014; Williams, 1981; Zektser, 2000).

The biodiversity of saline and brackish lakes is variable and depends strongly, among other factors, on salinity (Balushkina *et al.*, 2008; Boros *et al.*, 2013; Brucet *et al.*, 2012; Ventosa & Arahal, 2009). It can be quite high in large and moderately saline lakes, for example Lake Issyk-Kul (Kulagin *et al.* 1990; Savvaitova & Petr, 1999). Generally, however, increased salinity leads to a decrease in biodiversity (Kipriyanova *et al.*, 2007). In hypersaline lakes like the Dead Sea in Israel or Lake Elton in Russia, only some algae (*Dunaliella salina*), halophilic bacteria and fungi can be found (Nissenbaum, 1975). At the same time, many hypersaline lakes harbour high and unique bacterial diversity that has high scientific, ecological and biotechnological values (Oren, 2006).

Saline and brackish lakes in Europe and Central Asia are crucially important for birds during seasonal migrations and wintering. Many of them are located along transcontinental migration routes, as for example, the Torey lakes in the Daurian steppe in Russia. Some are crucial stops along the Australian-Asian migration route, providing temporary habitats for rare species such as 70% of the world population of the threatened white-headed duck (*Oxyura leucocephala*), which overwinters at Lake Burdur, Turkey, which is a designated Ramsar site (Ramsar, n.d.).

Figure **3** **32** **Lake Chany. An example of a large saline lake with fluctuating water level, salinity and biodiversity. Source: Landsat-8 (2016).**

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

There are no comprehensive assessments of changes in biodiversity in saline and brackish lakes in Europe and Central Asia. Many saline lakes in the region experience large fluctuations in water level and salinity, with corresponding biodiversity and ecosystems shifts (Namsaraev *et al.*, 2008) **(Figure 3.32)**.

Fishery volume exceeds sustainable use and fish resources dwindle in the largest saline and brackish lakes in Central Asia (Karimov *et al.*, 2009; Thorpe *et al.*, 2011; Zoï, 2012), however rehabilitation measures for fish resources (stocking of lakes with fish larvae, protection of spawning areas, etc.) usually are not conducted (Karimov, 2011).

Water withdrawal for irrigation from tributaries led to the decline of many saline lakes' area and volume, rise in salinity and destruction of fish spawning areas and species' migration routes (Bai *et al.*, 2004; Karimov *et al.*, 2009; Government of Turkmenistan, 2015). Another factor that contributes to decline of water level in saline lakes is climate change. This process is especially strong in the arid zones of Europe and Central Asia (IPCC, 2014b). It affects salinity level and, as a result, leads to decline in biodiversity and threatens the total extinction of the majority of species (Bai *et al.*, 2004).

It is projected that many lakes in the Mediterranean climate zone will be markedly affected by aridification and water abstraction, with related changes in water level, salinity, biodiversity and the ecology of lakes and reservoirs (Jeppesen *et al.*, 2015). Artificial saline lakes are also created in natural depressions of Central Asia by storing collector-drainage water after irrigation (Stone, 2008; Thorpe *et al.*, 2011; Yakubov, 2011). They are extremely polluted by agricultural chemicals, initially with low biodiversity limited

to some algae and bacteria (Glazovsky, 1990; Orlov *et al.*, 2011). However, there are projections that these man-made ecosystems can be important for biodiversity conservation, fisheries, migration birds and recreation (Karimov *et al.*, 2014b; Government of Uzbekistan, 2015; Thorpe *et al.*, 2011).

As large saline and brackish lakes have a long history of isolation from each other, they have been refugees for rare and endemic species. These species are more strongly affected than others by non-native invasive species, which reach saline lakes sometimes accidentally, sometimes through introduction by humans to improve fisheries, like in Issyk-Kul lake (Kulagin *et al.*, 1990; Thorpe *et al.*, 2011).

3.3.3.3 Implementation of the Ramsar Convention by the countries of Europe and Central Asia

All countries in Europe and Central Asia are Contracting Parties to the Ramsar Convention, except San Marino.

According to a national reports review undertaken by the Secretariat of the Convention (Ramsar, 2015a, 2015b), Ramsar wetlands in the region face increasing pressures from rapid urbanization and land-use changes for tourism, infrastructure development (transport and energy) and non-sustainable exploitation of natural resources (e.g. water, gravel, peat, oil, gas). Ongoing climate change increases environmental risk and the frequency of natural hazards such as floods, droughts, storms and landslides, especially in Central Asian countries. The regulating services that wetlands can provide are only rarely taken into account. Wetlands in Eastern Europe and Central Asia are under increasing pressure especially from conversion due to population increase (Central Asia) and development

Table ③ ② Implementation of the Ramsar Convention in Europe and Central Asia: reporting statistics.

Subregion	Number of countries reporting to the Convention	Total sites number	New sites last reporting period	Sites under threat or with changed ecological character	Official reporting on Ramsar site ecological character change
Western Europe	18 of 24	805	46	62 (8% of all sites)	17 (27% of sites changed or under threat)
Central Europe	13 of 18	174	11	27 (15%)	15 (55%)
Eastern Europe	5 of 7	110	6	17 (15%)	1 (6%)
Central Asia	1 of 5	21	1	1 (n/a)	1 (n/a)

Table ③ ③ Implementation of the Ramsar Convention in Europe and Central Asia: progress toward goals. Yes = goal achieved; In part = goal partially achieved; No = Goal not achieved.

Subregion	Goal 1: Wise use of wetlands	Goal 2: Network of Wetlands of International Importance (Ramsar Sites)	Goal 3: International cooperation	Goal 4: Institutional capacity and effectiveness
Western Europe	Yes	No	Yes	No
Central Europe	Yes	In part	Yes	No
Eastern Europe	In part	Yes	Yes	In part
Central Asia	In part	In part	In part	In part

projects (Eastern Europe), overuse of wetland resources, expansion of human habitats and infrastructure, agricultural, recreational and development activities, and pollution. In Central Asia there are difficulties with water availability for wetlands, and there is competition for water within and between countries. There are cases in Central Asia of wetland loss due to the natural disasters – such as droughts and landslides.

An assessment of Ramsar Convention implementation was undertaken **(Table 3.2)** considering progress towards the four main goals of the Convention: 1 - wise use of wetlands, 2 - creating a network of wetlands of international importance (Ramsar Sites), 3 - international cooperation, and 4 - institutional capacity and effectiveness **(Table 3.3)**.

The number of Ramsar sites is highest in Western Europe, while these sites cover smaller areas than in other subregions. Western Europe is also more active in designating new sites. Fewer sites in Western Europe are under threat than elsewhere. Nevertheless, those that are under threat are reported by NGOs or local communities, and seldom via official channels to the Ramsar Secretariat. Eastern and Central Europe has a higher portion of endangered sites, but more often reported via official channels. In Central Europe 55% of sites with changing ecological character were reported via official channels, while in Eastern Europe it was only in 5% of cases. Central Asia cannot be assessed due to a lack of information except for Kazakhstan, which also reports its endangered Ramsar site officially and was visited by a Ramsar mission.

As part of the wise use of wetlands, countries are reporting on successful wetland restoration projects and work related to water policies and river basin management including the European Union Water Framework Directive **(Table 3.3)**. Within goal 2, countries report on the development of management plans for Ramsar Sites and

the implementation of their provisions; wetland monitoring and inventory activities; and the preparation and designation of new Ramsar Sites and synergies with the European Union Natura 2000 network of protected areas. Goal 3 is on international cooperation. The steps to meet goal 4 mostly are communication, education and outreach activities, including World Wetlands Day; and the development of national policies for conservation, biodiversity and wetlands including national biodiversity strategies and action plans.

The greatest difficulties reported are limited administrative capacity resulting from limited human and financial resources; slow administrative processes to put effective policies in place; and insufficient coordination between wetland, water, and river basin management authorities. Progressing with wetland ecosystem conservation on the ground is difficult, because it needs to be based on time-consuming inter-sectoral stakeholder consultations. Agricultural, urban and land-owner interests hinder the implementation of Ramsar objectives. The lack of political interest, economic incentives in the absence of wetland valuations, and sufficient wetland inventories are reported by Europe and Central Asian countries.

3.3.4 Marine systems

The marine environment of Europe and Central Asia, which includes open ocean areas and semi-enclosed seas encompassing several marine ecoregions (Spalding *et al.*, 2007) is very diverse at genetic, community, ecosystem and seascape levels. This environment has been significantly impacted by human activities for millennia but marine research in some parts of the region is well established, resulting in some of the best studied marine ecosystems in the world. Still about 53% of the benthic shallow habitats in Western and Central Europe were found to be data deficient in recent habitat assessments (Gubbay *et al.*, 2016). Of

the assessed benthic habitats, about 38% were classified as threatened in the categories critically endangered, endangered and vulnerable. In the European Union, among assessments of the conservation status of species and habitat types of conservation interest, only 7% of marine species and 9% of marine habitat types show a "favourable conservation status". Moreover, 27% of species and 66% of assessments of habitat types show an "unfavourable conservation status" and the remainder are categorized as "unknown".

For the purpose of the current assessment the marine environment was divided into the different ocean basins and semi-enclosed seas of the region including the North East Atlantic Ocean, with different sections for the Baltic, Mediterranean and Black Seas, the Eurasian Arctic Ocean and the North West Pacific Ocean, focusing on the exclusive economic zones of countries of Europe and Central Asia, and of the relevant regional agreements.

3.3.4.1 North East Atlantic Ocean

OVERVIEW OF THE SYSTEM

The European part of the Atlantic Ocean (*sensu lato*, i.e. North Sea, Irish Sea, English Channel, Iberian coast, and the Macaronesean Island coasts except for Cape Verde) encompasses large latitudinal gradients, several biogeographic provinces from Artic to warm temperate systems realms (Spalding *et al.*, 2007), and a diversity of ecosystems and habitats, including complex structural habitats like seagrass meadows, kelp forests and biogenic reefs, providing a diverse set of nature's contributions to people (Prather *et al.*, 2013; Smale *et al.*, 2013; Worm *et al.*, 2006). Despite knowledge gaps, several trends are well established thanks to the sustained observation of marine biota particularly in the Celtic Sea, English Channel, North Sea and Bay of Biscay (e.g. Barceló *et al.*, 2016; Beaugrand *et al.*, 2009; Daan *et al.*, 2005; EEA, 2015c; Frederiksen *et al.*, 2013; Mieszkowska *et al.*, 2014; OSPAR, 2010, 2017).

PAST AND PRESENT TRENDS

Changes in distribution and species abundance are the most well documented trends, across diverse taxonomic groups, as illustrated in over 670 observational data points extracted from Poloczanska *et al.* (2013)2 and summarized in **Figure 3.33**.

Shifts in range, in particular northward expansion of more than 140 km per decade on average across taxa (Poloczanska *et al.* 2013), have been shown **(Figure 3.34)**. This is exemplified by the subtropicalization of European pelagic fish communities (Montero-Serra *et al.*, 2015), by movements of calanoid copepods towards the north at

rates of up to 23 km per year between 1958 and 2009 (Beaugrand *et al.*, 2009) and by shifts of the centre of the distribution for about 60% of 65 marine invertebrates studied in the North Sea (Hiddink *et al.*, 2015). Range shifts occur not only in latitude, but also along depth gradients (e.g. Dulvy *et al.* (2008) for fishes; Hiddink *et al.* (2015) for marine invertebrates). Range shift is, however, not fast enough to keep pace with climate change for many species (Hiddink *et al.*, 2015), so other effects of climate change, such as phenological changes, are also observed. Also, as shown **Figure 3.34**, the rate of change varies across taxa: northward expansion of benthic algae display an average range shift of 42 km per decade which is an order of magnitude slower than that documented for fishes (Perry *et al.*, 2005; Poloczanska *et al.*, 2013). Importantly, although documented in a few taxa only, such range shifts can provoke the loss of particular genetic clades (e.g. in the macroalga *Fucus vesiculosus*; Nicastro *et al.*, 2013) and impoverished genetic diversity at species level, with putative ecological and economic impacts (Parmesan, 2006).

In the 20th century almost all fish stocks of the North Atlantic have been depleted in abundance, with consequential impacts on stock biomass, size distribution, and diversity (reviewed in Rice *et al.*, 2016). Many fish stocks are currently overfished. However, in the 21st century, fishing has been reduced in most parts of the North East Atlantic shelves, and there is evidence of recovery in most of these areas, albeit at different rates for different species (Rice *et al.*, 2016). A combination of range shifts and fishing is responsible for genetic changes, such as declines in genetic diversity in fishes, as observed in the North Sea cod (Hutchinson *et al.*, 2003). Populations of most marine bird species have been declining since 2002 (Frederiksen, 2010), with the exceptions only of the northern gannet (*Morus bassanus*) and great skua (*Stercorarius skua*), both likely benefiting from increasing availability of fishery discards, and, for the gannet, from recovery from past persecution. These changes in abundance lead to local population and species decline, which affect a variety of fish and bird taxa, as detailed above, but also primary producers such as phytoplankton, with important consequences for trophic networks (McQuatters-Gollop *et al.*, 2007), and marine invertebrates including crustaceans, annelids, and molluscs (OSPAR, 2008; Wiens, 2016).

Another clearly documented change is biotic homogenization, due to species range shifts (e.g. for fishes assemblages, Magurran *et al.*, 2015) combined with the introduction of alien species. An estimated 237 species have been introduced into the North East Atlantic (Galil *et al.*, 2014), having steadily increased by about 173 species from 1970 to 2013. Many of these alien species were introduced deliberately (e.g. the Asian oyster (*Magallana gigas*), with which many other "hitch-hiking" species have been accidentally introduced). This is a consistent past

Figure 3 33 **Number of species for which changes in distribution, abundance or functioning (including demography, phenology, assemblages) have been documented since the 1950s (time series over more than 23 years), per functional groups.**

Data are shown separately for each taxonomic (functional) group. The dotted line provides the percentage of species with no change observed over all the trends considered. Source: Data for the North East Atlantic extracted from the Table S1 in Poloczanska *et al.* (2013).

DOCUMENTED CHANGES (SPECIES NUMBER)
PER FUNCTIONAL GROUPS

DISTRIBUTION · ABUNDANCE · COMMUNITY CHANGE · DEMOGRAPHY · PHENOLOGY ······ % NO CHANGE
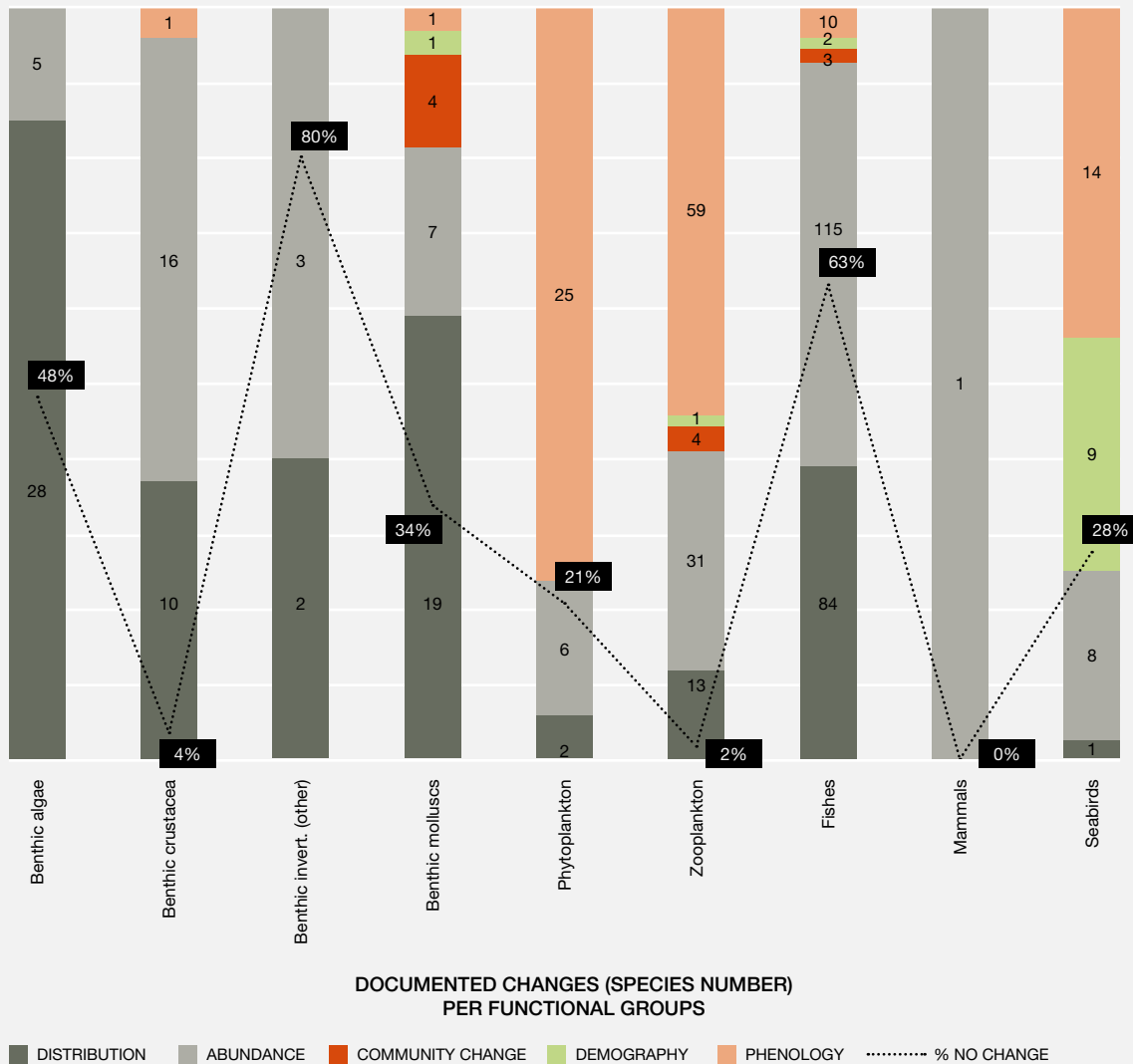
and current trend over a large range of taxa (Seebens *et al.*, 2017).

Changes in distribution and abundance also impact habitat-structuring species, such as seagrass and kelp forests, which are both natural carbon sinks and thus may contribute to carbon sequestration, or biogenic reefs, for example *Sabellaria spinulosa* or flat oyster reefs, both of which are included on the OSPAR list of threatened or declining habitats (OSPAR, 2008). Disease outbreaks have also been reported in cold-water corals, like the seafan *Eunicella verrucosa* (Hall-Spencer *et al.*, 2007), a structuring perennial species listed on the IUCN Red List of threatened

species. The decline in extent and abundance of these diverse structuring species modifies ecosystem functioning as well as the contributions that they provide to people. For instance, a shift from kelp canopies to turf-forming seaweeds has a global impact on community structure and function (Smale *et al.*, 2013) as well as on fisheries (Bertocci *et al.*, 2015). These habitat-forming species are insufficiently monitored (e.g. for kelps see Araújo *et al.*, 2016), but current trends have already documented declines, as exemplified by *Cymodocea* meadows, with estimated declines of between 15% and 80% in extent along the Iberian Peninsula coasts. Changes in ecosystem functioning (e.g. food web and trophic network) have also been well-

Figure **3** **34** **Range shift (northward expansion – leading edge; southern contraction – trailing edge) of taxonomic/functional groups in the North East Atlantic.**

Numbers above bars indicate the number of taxa for which data are available. Source: Data extracted from Table S6 by Poloczanska *et al.* (2013).

RANGE SHIFT (KM PER DECADE)

■ LEADING EDGE   ■ TRAILING EDGE
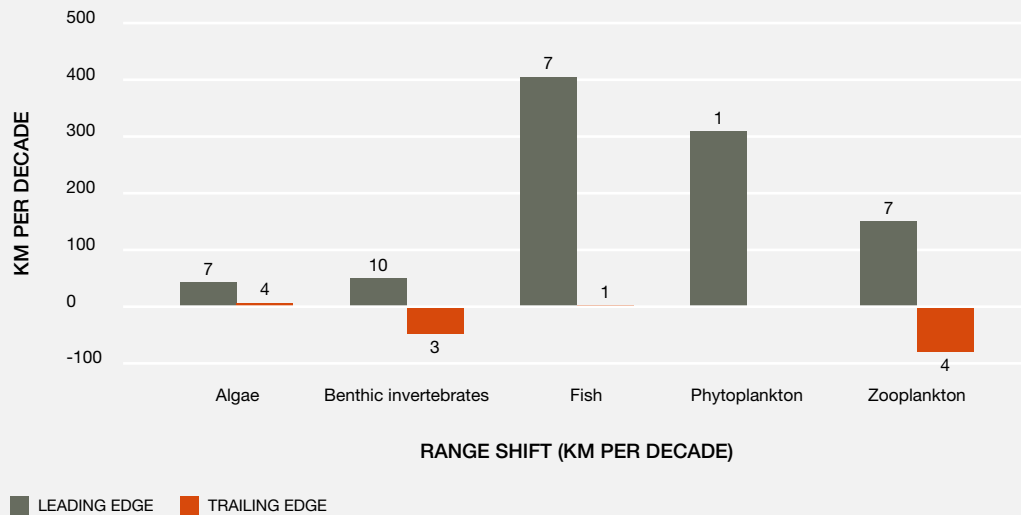
documented in some areas, e.g. in pelagic systems of the North Sea (e.g. copepods-fishes; Beaugrand, 2004; Kirby & Beaugrand, 2009).

Phenological changes (e.g. earlier timing of recruitment) are an important component of these changes in ecosystem functioning. They can affect populations through diverse mechanisms and with large impacts such as mismatches with food resource availability and increased mortality because of non-favourable environmental conditions (Thackeray *et al.*, 2010 and references therein). They have been established with confidence for several taxonomic groups (Edwards & Richardson, 2004; Kirby & Beaugrand, 2009; Poloczanska *et al.*, 2013; Thackeray *et al.*, 2010). For some taxa such as marine invertebrates (Thackeray *et al.* 2010), rates of advance in seasonal timing was shown to increase over recent decades.

Changes in patterns and processes, as detailed above, are indicative of a decline in biodiversity status, now and in the past, at species, community and ecosystem levels.

Although biodiversity decline and changes in ecosystem functioning are widespread, a few trends are indicative of partial recovery when compared with past-trends. With the exception of Atlantic cod, there are signs of improvement in fish stocks and biomass, especially compared with other Western European waters such as the Mediterranean Sea (Fernandes *et al.*, 2017). The number of assessed stocks that are above their maximum sustainable yield has dropped from 94% in 2007 to 41% in 2014 in European Union

Atlantic and Baltic waters, which has been explained by an overall decrease in the level of fishing pressure (Daan *et al.*, 2005; EEA, 2015b). Moreover, with 3,203 marine protected areas extending over 171,174 km², 5.9% of the surface of the North East Atlantic benefits from protection. There are, nevertheless, discrepancies between sea areas (e.g. 14.7% Greater North Sea vs. 5.9% Bay of Biscay and Iberian coasts) and distance from the shore (52.1% of 0-1 nautical miles zone vs. 2.3% beyond 12 Nautical miles). The increase in network coverage is a positive current trend, but still below the Aichi Biodiversity Target 11 of 10% of marine habitats under protection (EEA, 2015a; OSPAR 2017) over the whole North East Atlantic area. In addition, only 10% of marine habitats that have been assessed have a favourable conservation status (EEA, 2015b), with contrasted features across areas. For instance, while the Macaronesian region reported 33% of favourable habitat conservation status, the other areas of the North East Atlantic reported 71% of unfavourable-bad assessments (EEA, 2015a). Finally, no fauna extinction has been documented so far, maybe due to major knowledge gaps for important taxonomic groups like marine invertebrates (McCauley *et al.*, 2015).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

The primary pressures responsible for past regime shifts in shelf ecosystems are overfishing, pollution and climate driven changes including Arctic ice melting and ocean warming. In terms of the importance of these direct drivers for past trends, they are graded as high impact **(Table 3.5)**.

Overall, several studies point with a high confidence to climate change, including ocean acidification as the main emerging driver in the North East Atlantic (Barceló *et al.*, 2016; Beaugrand *et al.*, 2013; Birchenough *et al.*, 2015; Fossheim *et al.*, 2015a; Hiddink & ter Hofstede, 2008; Montero-Serra *et al.*, 2015; Poloczanska *et al.*, 2016). Eighty-six percent of the changes documented by Poloczanska *et al.* (2013) are consistent with expectations based on climate change effects, although most often (82% of the cases examined), other drivers are acting simultaneously. These include natural resource exploitation with direct (e.g. overfishing) or indirect (e.g. trawling and demersal fishing activities on benthos) effects, land and water use (eutrophication, pollution, including plastics and microplastics), habitat changes (marine urbanization) and invasive species. There are also substantial cumulative impacts of this diverse set of drivers (Halpern *et al.*, 2015).

Between the past and current periods, the importance of the effect of climate change has not decreased. Conversely, the importance of changes due to natural resource exploitation has likely been decreasing (i.e. graded as moderate for current trends in **Table 3.5.** For example, in benthic communities bottom trawling is one of the main pressures (Rice *et al.*, 2016), but recoveries have been observed following cessation of this activity (Kaiser *et al.*, 2006). Similarly, overfishing remains high (50% of fish stocks in the North East Atlantic) but positive trends are now observed. For example, fishing effort decreased by 25% from 2000 to 2006 in the Greater North Sea (EEA, 2015c; OSPAR, 2010). The same can be said for pollution: coastal benthic communities have been strongly affected by nutrients and pollutants runoff and climate change (Rice *et al.*, 2016) but nutrient inputs are now reduced, even if still cause for concern (OSPAR 2010, 2017). However other categories of pollutants (e.g. xenochemicals, microplastics) might have substantial effect, but have not yet been assessed (see Chapter 4). Conversely, besides climate change, the impact of man-made structures on seabed and coastal habitats has been increasing. These include structures associated with urbanization of coastal areas, coastal land defences and a growing number of offshore structures (EEA, 2015c), and associated ecosystems and species. The importance of invasive alien species has been increasing in a recent past, with 44 high-impact species (de Castro *et al.*, 2017) **(Box 3.3; Table 3.5)**.

3.3.4.2 Baltic Sea

OVERVIEW OF THE SYSTEM

The Baltic Sea is a shallow brackish waterbody characterized by strong seasonal variability and decreasing gradients of salinity and temperature from south-west to north-east. It is an almost non-tidal sea that spans from the temperate, highly populated and industrialized south with intensive agriculture, to the boreal and rural north. It is a young, low diversity ecosystem inhabited by species of both marine and freshwater origin, migratory species and glacial relicts (Segerstråle, 1957). Despite being well-studied compared with other aquatic systems (Costello *et al.*, 2010), several ecosystem parts are still under-investigated (Ojaveer *et al.*, 2010).

PAST AND CURRENT TRENDS

International coordination of research in the Baltic Sea has been ongoing since the beginning of the 20th century, but long-term datasets are only available from the 1950s for benthos, plankton and fishes (Ojaveer *et al.* 2010). Due to the absence of long-term monitoring for many other taxa, several parts of the ecosystem are under-investigated and thus under-evaluated. Several biodiversity assessment tools have been created for the assessment of biodiversity, but most of them have only been applied in marginal areas of the Baltic Sea (Andersen *et al.*, 2014; Aunins & Martin, 2014). The overall health of the Baltic Sea is currently in a bad state, with significant decline in the status of biodiversity in large areas (BalticSTERN, 2013; HELCOM, 2009, 2010), as can be seen in the indicators in **Figure 3.35**. Only the Bothnian Sea and some coastal areas in the Bothnian Bay have an acceptable status in terms of different elements of biodiversity. The grey seal population is in good status in the whole Baltic Sea **(Figure 3.35)**.

Regime shifts and fish trends

In general, fish communities of the Baltic Sea are very unstable due to substantial decline or lack of large predatory fish in the system. Several species are of concern in achieving the Baltic Sea Action Plan community level targets (HELCOM, 2009). Several currently threatened or declining fish species are negatively influenced by eutrophication and pollution (Fernandes *et al.*, 2017; HELCOM, 2009). Coastal fish species are declining in shallower areas, mainly due to increasing temperature (Snickars *et al.*, 2015). Latest assessments indicate a good biodiversity status for about half of the assessed coastal area (HELCOM, 2017f). In addition, reduced salinity reduces the food base for benthic feeding fish in deeper areas (Snickars *et al.*, 2015).

The open areas of the Baltic Sea have undergone several regime shifts in the 20th century (Österblom *et al.*, 2007). Such changes are primarily caused by the combination of weakened top-down pressure and increased primary production (Möllmann *et al.*, 2007). These ecosystem shifts are well observed in cod populations. The current decline of cod populations can be attributed to the large scale fishing industry and results in a significant increase in sprat populations (HELCOM, 2010), changes in zooplankton

Figure **3** **35** **Status of biodiversity core indicators by sub-basin of the Baltic Sea.**

Green circles indicate good status, red circles indicate not good status, and empty circles indicate that the core indicator is applicable for the sub-basin, but has not been assessed. Absent circles indicated that the indicator is not applicable. For coastal indicators, pie charts show proportion of coastal assessment units per sub-basin in good status (green), not good status (red) and not assessed (empty). Source: HELCOM (2017e).
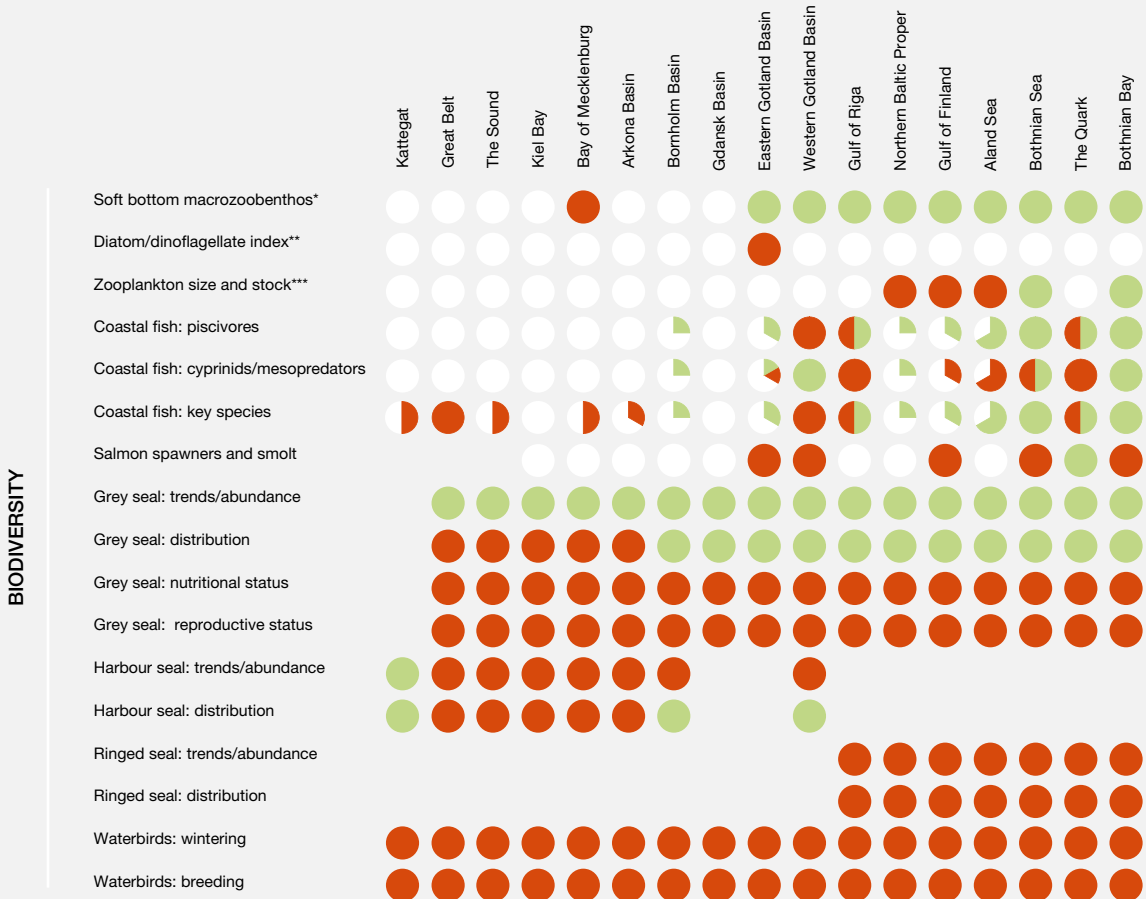
* Core indicator agreed to be tested in the HELCOM assessment
** Pre-core indicator agreed to be tested in the HELCOM assessment`
*** The indicator 'Zooplankton size and stock' is under testing for the Gdansk Basin

communities (Rönkkönen *et al.*, 2004) and thus reduced growth of the Baltic herring. In addition, changing climate conditions and lack of saline water inflows have created environmental conditions unsuitable for marine fishes (e.g. cod). Although, in some areas, signs of recovery have been observed for cod populations (Cardinale & Svedäng, 2011), recovery to safe biological limits has not yet been reached (HELCOM, 2010). Sturgeon, a very important commercial species for centuries, is now a red-listed species. A reintroduction programme has being developed with eggs from the St. John river in Canada (Kolman *et al.*, 2011). In the open sea, a good status in terms of fish biodiversity has not been achieved in any assessment area (HELCOM, 2017f).

Marine mammal trends

In the early 1900s strong hunting pressure followed by toxic pollution substantially decreased all populations of marine mammals in the Baltic Sea resulting in a "critically endangered status" for the Baltic Sea harbour porpoise (Hammond *et al.*, 2008; HELCOM, 2009, 2013) and an order of magnitude decrease in the number of seals (Harding & Härkönen, 1999). Although, the conservation status of marine mammals in the Baltic Sea was considered as unfavourable for most of the species assessed (EEA, 2015d), there are some signs of an increase of top predators, mostly seals and predatory birds, during recent decades (HELCOM, 2013, 2017d). Population size of grey

seals is considered as favourable in several Baltic Sea areas **(Figure 3.35)** and this recovery is interfering with fishing activity and an unknown number of seals are drowning in fishing gear every year (Vanhatalo *et al.*, 2014). But the assessment of their nutritional and reproductive status is still not good **(Figure 3.35)**. In addition several migratory bat species populations are negatively impacted by wind turbine development (Voigt *et al.*, 2012). An expert evaluation of endangered species in the Baltic shows that a number of species are still at risk of extinction (HELCOM, 2013).

Marine bird trends

No clear trends are evident for marine bird populations, but populations are not considered stable in the Baltic Sea. Substantial long-term declines can be attributed to anthropogenic factors, through lower reproductive success. However, some bird species (e.g., cormorants) may benefit from certain anthropogenic activities (HELCOM, 2009). A cascading effect from overfishing, that targets predator fish, has also improved the food base for some birds, as more prey becomes available to them (e.g. auks) (HELCOM, 2009). In addition, climate change has impacted the range and population size of migrating species through changes in breeding areas (HELCOM, 2009, 2017b). In recent decades, over half of wintering water bird species have declined significantly and the reasons for their decline are not currently understood (BalticSTERN, 2013; HELCOM, 2017c).

Plankton trends

The species dominance and biodiversity of phytoplankton have significantly changed over the past 100 years (Feistel *et al.*, 2008; Hällfors *et al.*, 2013; HELCOM, 2009; Wasmund *et al.*, 2008). In recent decades, however, there have been few clear trends. Long-term increases in cyanobacteria blooms present a challenge to achieving good Baltic Sea Action Plan environmental status (HELCOM, 2009). During the past few decades, the dominant zooplankton taxa have undergone considerable changes, driven by natural shifts and human impacts. These changes are causing a cascading effect in the food web, affecting upper trophic levels (HELCOM, 2009).

Benthos and habitat forming species trends

Currently, macrobenthic communities are severely disturbed and degraded in several Baltic Sea areas (HELCOM, 2009; Norkko *et al.*, 2007) and long-term patterns indicate a "shifting baseline" (HELCOM, 2009). From 1994 to 2005 marine invertebrates in the Kattegat area decreased from 230 to 180 species and this decline continued until 2011, when some taxonomic groups were found to have only one third of the species recorded in 1994 (EEA, 2015a). In general, the dominance of perennial habitat-forming macrophytes, such as bladder wrack, eelgrass and charophytes, is gradually decreasing and currently being replaced by phytoplankton and fast growing annual phytobenthic species (Dahlgren & Kautsky, 2004; HELCOM, 2009, 2010; Korpinen & Jormalainen, 2008). However, some range expansion in several important macroalgal species has been observed in the area of the Northern Baltic Proper (HELCOM, 2009, 2013). For example, bladder wrack has increased its range in depth (HELCOM, 2009) and its status is considered of least concern in the most recent assessment (HELCOM, 2013). Eelgrass populations have undergone several restoration attempts after being almost destroyed by diseases in the 1930s. Long term trend indicates significant fluctuations in eelgrass distribution in the Baltic Sea, with higher instability in sheltered areas (Frederiksen *et al.*, 2004). In addition, mussel beds have undergone significant transformation and further decline is expected due to the range expansion of invasive species preying on mussels (Westerbom *et al.*, 2002; HELCOM, 2009; Ojaveer *et al.*, 2016). In open sea areas soft bottom invertebrate communities are in good condition in a large part of the Baltic Sea (HELCOM, 2017e, 2017f).

Invasive species trends

The number of non-indigenous species in the Baltic Sea is growing (HELCOM, 2009, 2017e, 2017g). Over half of those recorded have become established in at least one of the Baltic Sea countries (Ojaveer *et al.*, 2016).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Eutrophication, overfishing, and a significant decline in the abundances of marine mammal populations were the most important drivers of change in the Baltic Sea in the 20th century (Ojaveer *et al.*, 2010). Currently major environmental problems include eutrophication caused by increasing river runoff, overfishing, hazardous substances, risk of chemical or oil spills, invasive species, habitat loss due to anthropogenic factors, and climate change induced changes, i.e. in temperature and salinity (BalticSTERN, 2013; Costello *et al.*, 2010) **(Box 3.3, Table 3.5)**. Assessments of the status of widespread pressures like marine litter, including microplastics and underwater sound are currently unavailable, but need to be assessed (HELCOM, 2017e). Most areas are subject to multiple stressors (Andersen *et al.*, 2015).

Eutrophication

All open waters and coastal areas of the Baltic Sea, with the exception of some areas in the Bothnian Bay, are changing due to eutrophication (HELCOM, 2010). Altogether 97% of the surface area in the Baltic Sea is eutrophic (HELCOM,

2017e). The sea floor area where hypoxia occurs has increased 10-fold over the last 115 years (Carstensen *et al.*, 2014). In open waters, the increase of oxygen-deficient zone areas is the main driver of change in biodiversity and benthic community functioning (Carstensen *et al.*, 2014; HELCOM, 2009). Areas with eutrophication-induced coastal hypoxia are becoming more common both in deep and shallow water habitats (Conley *et al.*, 2011). In the northern Baltic Sea, hypoxic disturbance degrades the structure and function of seafloor communities and sediment nutrient cycling (BalticSTERN, 2013; Villnäs *et al.*, 2012). There are improvements in eutrophication status that are direct consequences of long-term efforts to reduce nutrient inputs (Andersen *et al.*, 2015; HELCOM, 2017e), but the overall target of a Baltic Sea unaffected by eutrophication has not yet been met (Svendsen *et al.*, 2015).

Overfishing

Overfishing is one of the main drivers of change in the Baltic Sea ecosystem, because low diversity systems are more prone to cascading effects caused by the decline of top predators (BalticSTERN, 2013). Technical improvements in fishing methods have increased landings since the second half of the 20th century in the overpopulated Baltic Sea area. In addition, construction and regulations in main watercourses have disturbed the natural reproduction of migratory fish species (BalticSTERN, 2013). Since the collapse of the cod stock in the 1980s, landings have been reduced, but due to a shifting regime the cod stocks have not recovered (HELCOM, 2010).

Invasive species

Fewer non-indigenous species are recorded in the Baltic Sea than in other European Seas (Galil *et al.*, 2014). Nevertheless, due to low native species diversity, underrepresentation of several ecosystem traits, and overall large disturbances in habitats, alien species are having severe impacts on the Baltic Sea ecosystem (BalticSTERN, 2013; Leppäkoski *et al.*, 2002). Ecological impacts caused by the invaders vary depending on how they differ from natives in their life form and resource usage (HELCOM, 2009).

Climate change

Climate change amplifies the effect of all other drivers of change (Snickars *et al.*, 2015). In the Baltic Sea eutrophication rates are increasing through increased nutrient fluxes from increased river runoff. Warmer temperature and an increase in extreme temperatures are making the areas better suited for the establishment of alien species. Moreover, increased riverine flows result in lower salinities with detrimental impacts on all species of marine origin.

In summary, the Baltic Sea is well studied and its ecosystems and biodiversity have been very degraded in the past. Management plans for recovery have been in place for some years, and although in general the status of biodiversity is still considered poor, some signs of recovery have been observed.

3.3.4.3 Mediterranean Sea

OVERVIEW OF THE SYSTEM

The Mediterranean Sea, covering approximately 2,500,000 km², is a remnant of the Tethys ocean, an ancient ocean from the Mesozoic era. The sea's main hydrologic features are: i) a microtidal regime; ii) scarce freshwater inputs compensated by inflow of Atlantic surface water; iii) highly saline (38 to 39.5‰) concentration basin with higher evaporation eastwards; iv) oligotrophy, with organic carbon inputs 15-80 times lower in the eastern than in the western basin and extremely low concentrations of chlorophyll-a in surface offshore waters (ca 0.05 µg l^{-1}); and v) with almost constant temperature from about 300-500 m downwards, with bottom temperatures about 12.8 - 13.5°C in the western basin and 13.5 - 15.5°C in the Eastern basin.

STATUS AND TRENDS

Despite covering only 0.82% of global oceanic surface, the Mediterranean sea is host to more than 17,000 described marine species, representing an estimated 7% of the world's marine biodiversity, including about 25 to 30% of endemic species (Coll *et al.*, 2010b; Mouillot *et al.*, 2011). Longitudinal and latitudinal patterns distinguish a dozen biogeographic regions, from the Alboran Sea to the Levantine Basin (Bianchi *et al.*, 2012), and a great number of unique ecosystems (Coll *et al.*, 2010b; Danovaro *et al.*, 2010). The apparent eastwards decrease in biodiversity follows a gradient of production, but its true extent is still not clear. Biodiversity is generally higher in coastal areas and on continental shelves. Biodiversity, excepting bacteria and archaea, decreases with increasing water depth, but to a different extent in different taxa. Danovaro *et al.* (2010) estimate the deep-sea biodiversity of the Mediterranean (excluding prokaryotes) at 2,800 species, of which two thirds remains undiscovered.

In recent habitat Red List assessments carried out for 47 benthic shallow (<200 m depth) habitats off the northern shores of the Mediterranean, 60% were considered data deficient. Of the remaining habitats 74% (14 habitats) were threatened (Gubbay *et al.*, 2016).

Some fish and invertebrate populations have been decimated in recent years. Of the 519 native marine fish species and subspecies in the Mediterranean Sea,

more than 8% (43 species) were classified in threatened categories (critically endangered, endangered or vulnerable). Of the 15 critically endangered species, 14 are sharks and rays. Thirteen species are listed as endangered, nine of them sharks and rays (Abdul Malak *et al.*, 2011). Cartilaginous fishes in general are declining in abundance, diversity and range (Cavanagh & Gibson, 2007). In the red list assessment of Mediterranean Anthozoans 69 species (51%) were listed as data deficient, and from the remaning about 25% were found to be threatened with extinction (critically endangered, endangered or vulnerable), including two of the endemic species (Otero *et al.*, 2017).

Mediterranean phyto- and zooplankton blooms, including jellyfish and comb jellies, are regional, seasonal and species-specific phenomena. These blooms have likely benefited from overfishing, eutrophication, habitat modification, aquaculture, global warming and human-mediated dispersal (Boero, 2013). Documented increases in bloom frequency, duration, and spatial extent have negatively impacted food web structure, as well as economy and human health (Ferrante *et al.*, 2013) although in some cases, jellyfish can contribute to maintain water quality and prevent phytoplankton blooms exerting a top-down control of the trophic web (Pérez-Ruzafa *et al.*, 2002).

Concurrent expansion of the range of warm-water species (native, recent Atlantic thermophilic entries, tropical Erythraean aliens - that entered the Mediterranean through the Suez Canal) and contraction of that of cold-water species, disrupt the present biogeographic patterns within the basin and place cold-water species under threat (Bianchi *et al.*, 2012; Galil *et al.*, 2017). In the past decade Erythraean aliens have increasingly been recorded on the deeper shelf (> 80 m) and even on the upper slope (Innocenti *et al.*, 2017).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Recent increase of littoral residents, from 44 million in 2000 to 590 million expected in 2050 (Tosun, 2011), and tourists: 270 million in 2010 to 346 million expected in 2020, coupled with intensification of anthropogenic activities, is driving unprecedented changes in the Mediterranean Sea (Micheli *et al.*, 2013; EEA, 2015c). Habitat loss, environmental degradation and pollution are chronic and ubiquitous. Symptoms of complex and fundamental alterations to native populations, habitats and ecosystems proliferate, including increases in exotic species. The biota across wide swaths of the Mediterranean Sea, including marine protected areas, seagrass beds (Boudouresque *et al.*, 2009), algal mats, and biogenic reefs have already been severely altered (Airoldi and Beck, 2007) with direct ecological, economical and human health impacts (Galil *et al.*, 2015, 2017). Coastal lagoons are increasingly endangered by anthropogenic

impacts (climate change, sea-level rise, massive introduction of invasive alien species, industrial scale aquaculture operations and fisheries) to the detriment of their role as a reservoir of genetic diversity (Pérez-Ruzafa *et al.*, 2011; Pérez-Ruzafa & Marcos, 2012).

Over half of all fish species are affected either directly or indirectly by fishing activities (Abdul Malak *et al.*, 2011; Vasilakopoulos *et al.*, 2014). Fishing, either through targeted or multi-species fisheries, is by far the most common threat to fish biodiversity, affecting 33% of native marine fish species, with an additional 18% threatened by by-catch. In the Mediterranean, 85% of the stocks are currently overfished and populations of many commercial species are characterized by truncated size- and age-structures (Colloca *et al.*, 2013). Overfishing has also led to a reduction of genetic diversity outside marine protected areas (Pérez-Ruzafa *et al.*, 2006). Larger coastal species and species that occur in areas subjected to prolonged or intensive fishing pressure are of particular concern (Abdul Malak *et al*., 2011). An analysis of the status of the Mediterranean fisheries (1970-2010), using various indicators (total landings, mean trophic level and fishing-in-balance index) confirmed that the fisheries resources of the Mediterranean are at risk from overexploitation. The pattern of exploitation and the state of stocks differed among the regions, with the eastern Mediterranean fisheries being in the worst shape, and declining (Tsikliras *et al.*, 2015).

The effectiveness of management initiatives implemented in the context of the European Common Fisheries Policy has been questioned with regard to the Mediterranean (Vasilakopoulos *et al.*, 2014; Cardinale & Scarcella, 2017). However, some of the analyses that compare the fishing activity in the North East Atlantic and in the Mediterranean do not take into account some of the differentiating characteristics of each region, and fail to discuss the role of marine protected areas as a complementary management tool (Pérez-Ruzafa *et al.,* 2017).

Marine protected areas provide benefits not only for recovering target fish stocks, but also to biodiversity (Pérez-Ruzafa *et al.*, 2017), maintaining assemblage structure and ecosystem equilibrium (Claudet *et al.*, 2008; García-Charton *et al.*, 2008; Lester *et al.*, 2009; Guidetti *et al.*, 2014; Sciberras *et al.*, 2015) preserving ecological interactions (Guidetti, 2006a, 2006b) and maintaining genetic diversity (Pérez-Ruzafa *et al.*, 2006). These effects can take place in a relatively short time (Pérez-Ruzafa *et al.*, 2017) and so the number of marine protected areas has been increasing significantly (see MAPAMED for trends in the Mediterranean, http://www.medpan.org/en/mapamed). There are 1,231 (7.14% of sea surface area) marine protected areas under legal designation in the Mediterranean, even if only 76 of those have no-go, no-take or no-fishing zones, that are the widest measures of protection for biodiversity (0.04% of sea

surface area). A recent report (MedPAN & RAC/SPA, 2016) admitted that "… for the majority of sites little is known on whether management measures are implemented and if they are, whether these measures are effective to reach the sites' conservation targets." Surveys conducted in marine protected areas situated along the Levant coastline recorded large populations of mostly Erythraean exotic species (Sala *et al.*, 2011; Yokes & Baki, 2012; Guidetti *et al.*, 2014; Vergés *et al.*, 2014). These marine protected areas are "hot spots" of exotic biodiversity and serve as "seed banks" for secondary spread. A study by IUCN, WWF and MedPAN found "Uncertainty and lack of information regarding marine introduced species was high in the marine protected areas we surveyed, as in average half marine protected area (54.8%) managers did not know the status of the introduced species reported (there)." (Abdulla *et al.*, 2008).

The number of alien species, currently 740 multicellular species (**Figure 3.36**, with their distribution), is substantially greater for the Eastern than the Western Mediterranean Sea with new introductions registered on monthly basis (Galil *et al.*, 2015; Galil *et al.*, 2017). The most common vectors in the Mediterranean are the Suez Canal (60%) **(Figure 3.36)** and vessels (21%). The invasion of the "killer alga" *Caulerpa taxifolia* raised concern over its impact on *Posidonia* meadows (Bulleri & Piazzi, 2014), on the trophic chain (Alomar *et al.*, 2016; Deudero *et al.*, 2011; Felline *et*

al., 2014; Terlizzi *et al.*, 2011), nutrient cycles (Gennaro *et al.*, 2015), sediments (Balata *et al.*, 2015), and sessile and motile biota.

In the eastern Mediterranean algae-dominated rocky habitats, including *Cystoseira* meadows, have been decimated by large populations of herbivorous fish introduced through the Suez Canal. The two voracious grazers, *Siganus luridus* and *S. rivulatus* have transformed lush rocky reefs into "barrens" (Giakoumi, 2014; Sala *et al.*, 2011; Vergés *et al.*, 2014).

The individual and cumulative impacts of these invasions adversely affect the conservation status of native species and critical habitats, as well as the structure and function of ecosystems and the availability of natural resources (Galil, 2007). Some species are noxious, poisonous, or venomous and pose clear threats to human health (Galil *et al.*, 2015).

3.3.4.4 The Black and Azov Seas

OVERVIEW OF THE SYSTEM

The Black and Azov Seas are connected to the Mediterranean Sea by the Bosporus and the Dardanelles Straits and the Sea of Marmara. The area of the Black Sea

Figure **3 36** **Number of alien species recorded in the coastal waters of the Mediterranean Sea in 1957, 1987, and 2017.**

Introductions through the Suez Canal account for the larger numbers in the Levantine Basin. Source: Galil *et al.* (2017), unpublished.
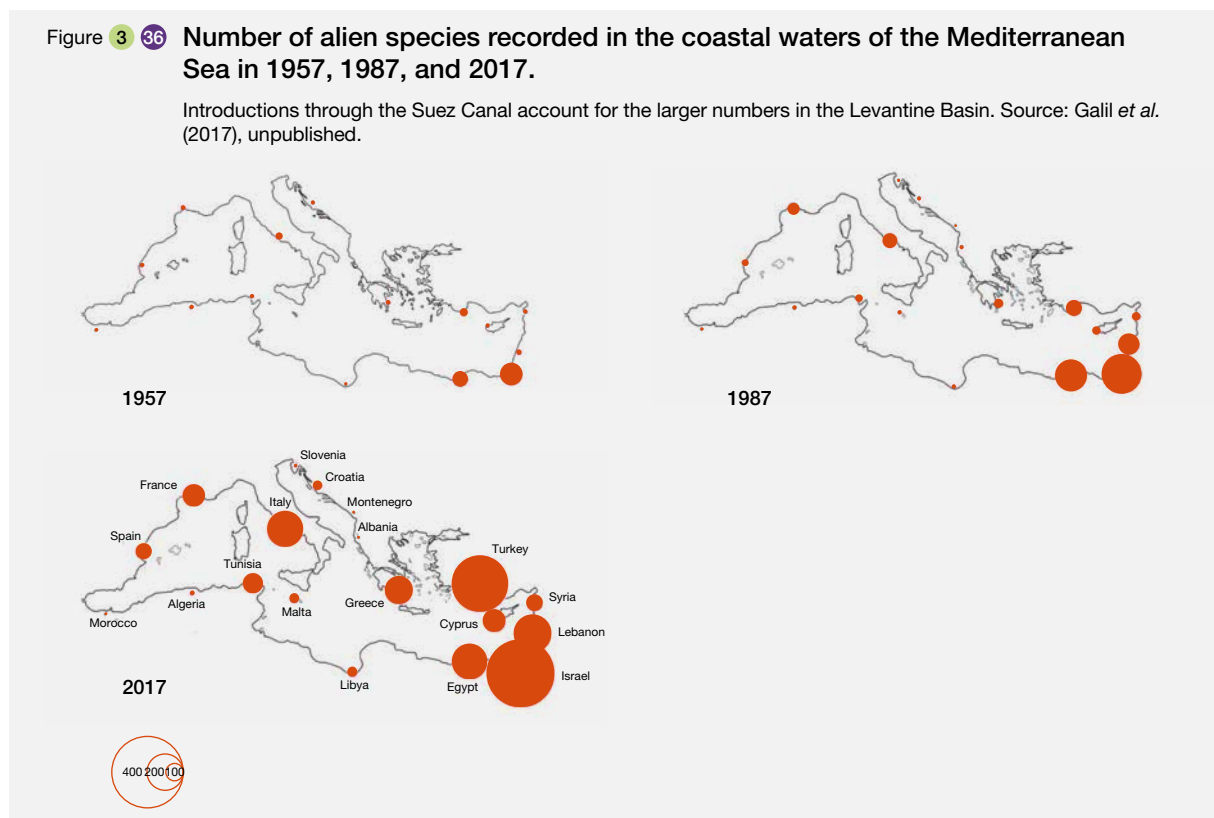
is 422,000 km, with a maximum depth of 2,210 m and the mean depth is 1,240 m (Dobrovolskii and Zalogin, 1982). It is very stratified (Vershinin, 2003, 2016), with about 90% of its volume as anoxic water, saturated with hydrogen sulphide accumulated from decaying organic matter. The thin oxygen rich upper layer is about 10-15% of total water volume and only about 100-150 m thick, but supports most of the unique biodiversity of the Black Sea (BSC, 2008; Filippov, 1968; Murray *et al.*, 1989; Yakushev, 1999). The deeper waters are inhabited mostly by protozoa, bacteria, and some multi-cellular invertebrates, though overall knowledge about its biodiversity is very limited (BSC, 2008). Recent publications estimate the number of Black Sea species at about 5,000 (Gomoiu *et al.*, 2012).

Two major rivers flow into the Sea of Azov: the Don and Kuban, and salinity is at its lowest (about 1‰) near the mouth of the Don (Kotlyakov, 2004). Flora and fauna are composed of different biogeographic groups with a predominance of eurythermic and euryhaline species. Only for the last 6,000-7,000 years the Black Sea has been connected to the Mediterranean basin and freshwater organisms gave place to marine life. Relicts contribute less than 5% of current species, whereas about 85% of the current species originate from the Mediterranean. Now there are about 700 species of phytoplankton, 150 of zooplankton, 300 macroalgae, 1,500 benthic invertebrates and about 180 fish species and three marine mammals in the Black Sea (Vinogradov, 1958; Sorokin, 2002; Vershinin, 2003).

PAST AND CURRENT TRENDS

During the 1980s and early 1990s, the Black Sea ecosystem was in a severely degraded condition, being rated with highest concern in five out of seven environmental categories, and the worst of any of the European seas (Stanners & Boudreau, 1995). The deterioration of this ecosystem was the result of two main drivers: a) eutrophication caused by increase of phosphate and nitrate input from large rivers leading to changes in the silicon/phosphorous and silicon/nitrogen balance (Nesterova & Terenko, 2009); and b) invasion by the ctenophore *Mnemiopsis leidyi*. This ctenophore, a competitor of planktivorous fishes, reached very high biomass levels (>1 kg m^{-2}) (Kideys, 2002), devastating the food chain of the entire Black Sea basin. After the ctenophore bloom, there were sharp decreases in anchovy catch and in the biomass of non-gelatinous zooplankton across the Black Sea which lead to a simplification of the food web that consisted mainly of phytoplankton, gelatinous zooplankton and ctenophores and bacteria **(Figure 3.37)** (Shiganova *et al.*, 2000; Stelmakh *et al.*, 2012; Vinogradov *et al.*, 1995).

Extinction of about half of the native bivalve species was brought about by the invasion of the Pacific gastropod *Rapana venosa*, starting in 1947. Black Sea populations of *Ostrea edulis* and *Flexopecten ponticus* are now on the brink of extinction (Sorokin, 2002; Vershinin, 2016; Zaitsev & Mamaev, 1997). The populations of predators such as dolphins, mackerel and tuna have declined because of

Figure **3** **37** **Total fish catches in the Black Sea, according to FAO data (ton/year).**
Source: Living Black Sea (2016).
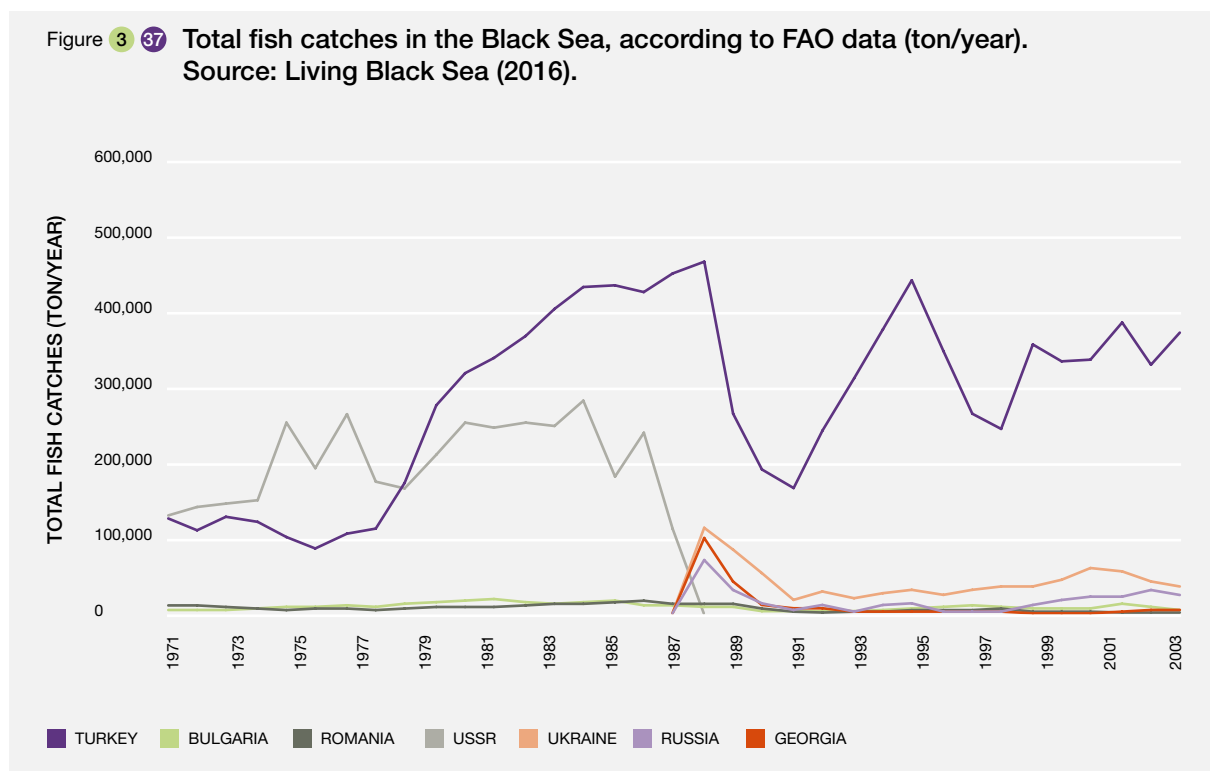
pollution and overfishing. Fishing has been refocused on the sprats *Sprattus and Clupeonella*, whose population had also dramatically decreased by the early 1990s (Tokarev & Shulman, 2007).

Since the mid-1990s there have been some signs of ecosystem recovery. Western Black Sea coastal waters improved due to reduced nutrient inputs, especially phosphorus (Kresin *et al.*, 2008), mainly due to the economic recession after the dissolution of the Soviet Union. This led to fewer microalgal blooms, recovery of some algal populations, increasing plankton biodiversity, decreasing opportunistic and gelatinous species, re-appearance of some native fodder zooplankton and fish species, and increasing edible zooplankton biomass (Ogus, 2008). After 1992, several eutrophication indices also improved in the eastern and deep Black Sea, indicating a more widespread recovery of the Sea (Kideys, 2002). Then, the ctenophore *Beroe ovata*, a specialized predator of *Mnemiopsis* was also introduced into the Black Sea leading to a sharp decline of *Mnemiopsis* followed by a sharp decline of *Beroe* itself. The *Mnemiopsis* population crash and reduction of eutrophication led to increases in non-gelatinous zooplankton, egg densities of anchovy, as well as increases in the biomass of two native gelatinous cnidarians (*Rhizostoma pulmo* and *Aurelia aurita*) and anchovy landings. In the early 2000s the concentration of zooplankton returned to the level before the invasion of *Mnemiopsis leidyi*. In 2004 in the north-eastern part of the Sea the number of species was comparable with numbers before the invasion of *Mnemiopsis*. The total number of fish roe and especially fish larvae, however, remains below the level of the 1960s (Tishkov, 2009).

In the Azov Sea in 1950 to 1970 the construction of storage reservoirs and implementation of water management led to a significant decrease in river inflow (Bespalova, 2016) and subsequent increase in salinity (Kuksa, 1994). There was a migration of Black Sea species to the Azov Sea and the native freshwater and brackish water ecosystems changed, with a decrease of commercial fish spawning in the estuary systems. Pollution by heavy metals, organochlorine pesticides, and petroleum hydrocarbons increased, leading to the reduction of productivity (Bespalova, 2016; Kotlyakov, 2004). Annual migration of *Mnemiopsis leidyi* led to a decrease in zooplankton biomass (Khrustalev *et al.*, 2001; Mirzoyan *et al.*, 2002) that caused damage to the anchovy and sprat populations, resulting in the loss of commercial catch of these species.

The first Black Sea Red Book (Dumont *et al.*, 1999) included 160 endangered species. Of those, sturgeons are the most endangered, along with species that inhabit shallow coastal waters such as turbot, sharks, seals, shrimp and oysters. Several marine mammals and seabirds were also considered to be threatened when their population

size and distribution was assessed, with the potential to become extinct in the near future (Eremeev *et al.*, 2011). The habitats at risk include some in the water column, lagoons, estuaries and deltas, and wetlands and saltmarshes. In a recent assessment of 63 shallow water habitat types in the Black Sea, 86% of the habitats were considered data deficient (Gubbay *et al.*, 2016). Excluding those, 67% of habitats were classified as threatened, including 11% as critically endangered.

Phytoplankton and zooplankton ecological communities are currently recovering, but the communities of higher trophic level species (benthonic species, fish) have not yet recovered. Commercial stocks of anchovy are at a relatively high level, and stocks have recovered, but populations of the majority of anadromous and catadromous fishes, such as sturgeons **(Table 3.4)** are still low and 70% of the industrial fish catching consists of small pelagic fishes (Lukoyanov, 2013).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Increased temperature of the upper mixed and the cold intermediate layers due to climate change contributes to naturalization of thermophilic species from the Mediterranean Sea and thins the upper, oxygen-rich layer of water in the Black Sea. Increase of temperature also causes increased evaporation from the seawater surface and reduced inflow from rivers.

Invasions as a result of introduction from ballast water have caused profound changes in the Black Sea. There are 156 species naturalized in this basin (Shiganova, 2000; Shiganova *et al.*, 2000). Invasive species from the coastal Atlantic waters of North America, belonging to eurybiont marine organisms, have the greatest influence.

Twenty countries in Western, Central and Eastern Europe discharge industrial and household wastewaters into the Black Sea basin. Moreover, the main pressure falls on the north-west shallow part of the Black Sea, where the main spawning grounds and habitats of algae and benthic species, are located. Drainage of agricultural lands and increase in mineral fertilizer flows led to the eutrophication of waters and changes in the structure of communities. Nutrients coming from the Danube river remained significant, but stable, in recent years (EEA, 2015c). Rice agriculture has a strong impact on Azov Sea biodiversity (water balance and pollution of seawater). Water pollution by oil and oil-products killed marine animals in the Azov and Black Seas (Diagelets *et al.*, 2014).

Fish stocks have deteriorated dramatically over the past three decades. The diversity of commercial fish caught

Table ③ ④ **Average fish catching in the Azov district in the 20th century, ton/year.**

Fish	Natural regime	After river regulation	
	1930-1940	1975-1982	1988-1989
Anadromous:			
Acipenser guldenstadti	463	73	71
A. stellatus	684	17	7
Huso huso	276	15	5
Alosa caspia tanaica	1,508	97	47
Vimba vimba	233	15	11
Pelecus cultratus	3,696	475	376
Catadromous and freshwaters:			
Lucioperca lucioperca	10,224	432	410
Abramis brama	20,353	912	1,960
Rutilus rutilus heckeli	770	18	11.6
Cyprinus carpio	895	2	6
Silurus glanis	1,200	2	0
Esox lucius	70	0	4
Others	2,730	2	20
Total:	43,102	2,060	3,033

has decreased over this period from about 26 species to only six, although the volume of fish caught has actually increased, after a near collapse in 1990. This is almost entirely due to significant anchovy fishing by Turkey, accounting for almost 80% of the total catch. Illegal fishing is also increasing, affecting biodiversity as well as the fishing industry (EEA, 2015c). Fishing gear is also responsible for a decrease in non-target species. For example, dolphins are being stranded in lost or abandoned fishnets, even inside marine protected areas (Nicolaev *et al.*, 2013; Radu & Anton, 2014; Zaharia *et al.*, 2014).

3.3.4.5 Arctic Ocean

OVERVIEW OF THE SYSTEM

The Barents, the White, the Kara, the Laptev, the East-Siberian, the Chukchi, and the Bering Seas together form the Arctic Seas of Europe and Central Asia. The region is a part of the Arctic biogeographic realm except some areas on its south-western and south-eastern margins which are temperate (Spalding *et al.*, 2007). The most distinctive feature of the region is its ice-associated ecosystems.

PAST AND CURRENT TRENDS

As the Eurasian Arctic Ocean is among the less studied marine regions of the world (Jorgensen *et al.*, 2016) and monitoring data are sparse, the majority of observed variations are for the Barents, the White, the western Kara, the Bering, and the Chukchi Seas. While studies that speculate or attempt to forecast impacts of current climate change on Arctic marine biota are numerous, documented impacts are much more scarce (Wassman *et al.*, 2011).

The generally observed and well documented trend of northward species' range shifts (including invasive species) has been defined in the western and the eastern parts of Eurasian Arctic as a processes of "Atlantification" and "Pacification" respectively (Fossheim *et al.*, 2015b; Jørgensen *et al.*, 2016a). In particular, the invasive snow crab is rapidly spreading in the eastern Barents and the Kara Sea (Pavlov & Sundet, 2011; Zalota & Spiridonov, 2015), and other "warm-water" decapods are shifting north-eastward from respective biogeographical borderlines drawn decades earlier (Zimina *et al.*, 2015). Consequences of this process could be unpredictable and different for the different ecosystems. For example, in the Chukchi Sea more nutritious copepods with

high fat content could increase; while in the Barents Sea less nutritious boreal copepods could replace their Arctic relatives (CAFF, 2017). At the same time there are observations showing increasing primary and secondary productivity in the Barents Sea (Dalpadado *et al.*, 2014).

There are also different trends in species and abundance of Arctic fish in the northern Barents Sea (Johannesen *et al.*, 2017). Overall there was a negative trend in the number of Arctic fish species from 2004-2015 but, while some species declined across the area, others declined only in the southern part and increased in the north, indicating displacement, while others did not show any significant change.

There are also changes in Arctic vertebrates' demography, abundance, distribution, phenology and community structure related to these processes (McRae *et al.*, 2012) **(Figure 3.38)**. Several marine mammal species are currently recovering from commercial exploitation (see also paragraph 3.4.3), which could mask reductions in carrying capacity associated with habitat loss in the short-term (Laidre *et al.*, 2015).

There is limited evidence of a decrease in benthic species biomass and diversity with increased pelagic grazing and recycling in the water column across the region (Kędra *et al.*,

2015). In contrast, there are observations showing increase in biomass and diversity of the benthic communities in the Chukhchi Sea where Pacific species of polychaetes, crustaceans, mollusks, and bryozoans have been found in recent years (Sirenko & Gagaev, 2007), later research conducted in this region showed that, despite the presence of Pacific species in the area (e.g. northward shift and increased biomass of Walleye Pollock were observed in the Bering and the Chukchi Seas; Overland & Stabeno, 2004), local benthic communities remained relatively stable (Sirenko, 2009).

Shrinking of multi-year ice cover and related increases of open waters and shelf seas caused a major decline in the productivity of sea-ice algae (Pabi *et al.*, 2008; Wassman *et al.*, 2011). Shifts in range and seasonal movement patterns have altered predator-prey relationships, resulting e.g. in changes in diet of sea birds (Meltofte *et al.*, 2013). Some arctic species have to travel more and expend more energy to find food. This can affect the condition of individuals and populations (CAFF, 2017). In the Barents Sea, the Chukchi Sea, and the Bering Sea, ecosystems are transforming from mostly ice-associated to more pelagic systems with changes in functional diversity (Wiedmann *et al.*, 2014) and structure of food webs (Kortsch *et al.*, 2015).

Figure **3** **38** **Index of abundance of Arctic vertebrate species from 1970 to 2007 grouped by high, low and sub-Arctic. Source: Mc Rae *et al.* (2012).**

Legend:
HIGH ARCTIC   LOW ARCTIC   SUB ARCTIC
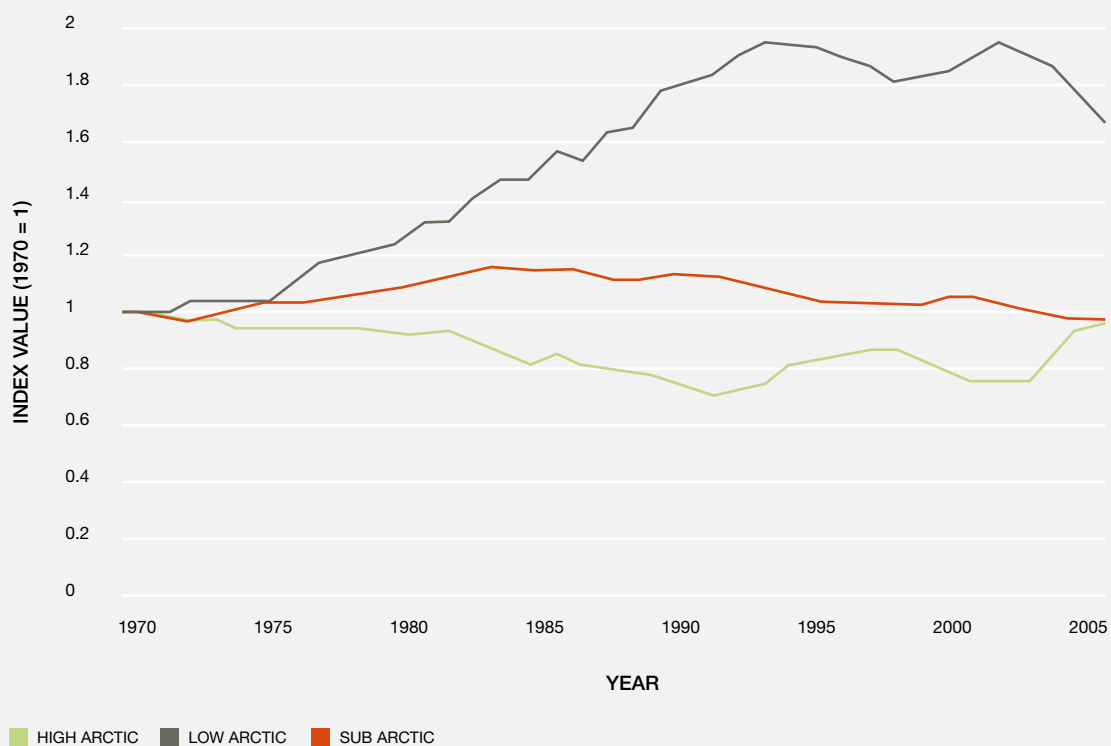
ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

The primary driver of the observed biodiversity change in the Arctic marine ecosystems is ongoing climate change, and in particular warming (Huntington *et al.*, 2005) and the decrease of sea ice. Current trends show that some species that are dependent on sea ice for reproduction, resting or foraging, are experiencing a reduction in range as sea ice retreats earlier and the open water season is prolonged (CAFF, 2017). This has been shown for many species, such as ducks breeding on the Siberian tundra and wintering at sea, which have now shortened their migration in response to declines in winter sea ice cover. Changes in sea ice conditions are probably also linked to changes of abundance and health of marine mammals, such as declines in the abundance of hooded seals, reduced body condition of Barents Sea harp seals, and changes in prey composition of bearded seals. Early sea ice retreat also reduces suitable breeding and pup rearing habitat for ringed seals. This negatively affects polar bears, which feed on ringed seals, as these conditions make them much more difficult to catch. The bears are thus shifting to prey on ground-nesting seabirds nests (Prop *et al.*, 2015), potentially causing a decline on these bird populations.

Multi-year sea ice is disappearing and is being replaced by first-year sea ice. This is expected to cause shifts in ice algal communities with cascading effects on the ice-associated ecosystem. Decline in ice amphipod abundance was already seen around Svalbard since the 1980s, coinciding with declining sea ice conditions (CAFF, 2017).

Although climate change and its effects are the major drivers of change in the Arctic (Wassman *et al.*, 2011), other drivers are also contributing **(Box 3.3; Table 3.5)**. For many years both local communities and international fleets have harvested several species of fish, seabirds and marine mammals and some stocks of fishes, large whales and seals were reduced to a small fraction of their original population sizes. Their current trends are, therefore, still subject to recovery from past overexploitation, complicating the interpretation of observed trends and attribution to environmental drivers (CAFF, 2017). Sea ice has been limiting the areas for industrial-scale fisheries until now but as the ice retreats, there is potential for expansion of this activity into previously unfished areas. In the Barents Sea, declines in benthic biomass have been linked to the intensity of bottom trawling and this is likely also important in other parts of the Arctic (CAFF, 2017).

So far, there are few examples of invasive marine species becoming established in the Arctic. However, in the Barents Sea two large non-native crab species, the snow crab and the king crab, have become abundant and are affecting benthic communities (CAFF, 2017; Oug *et al.*, 2011).

Finally, population sizes and trends of many migratory Arctic birds are influenced by overharvest, disturbance, and habitat loss outside the Arctic (Meltofte *et al.*, 2013).

3.3.4.6 North West Pacific Ocean

OVERVIEW OF THE SYSTEM

The Russian Far Eastern seas, consisting of the western part of the Bering Sea, Okhotsk Sea and northern part of the Sea of Japan and the adjacent waters of the Pacific Ocean **(Figure 3.39)**, have deep basins separated from the open ocean by chains of islands: Aleutian, Kuril and Japan Islands, that stretch from the Bering Strait to the coast of the Korean Peninsula (34° to 66° N). These are young basins with extensive development of recent metamorphic, volcanic and seismic processes. Natural hazards such as landslides in the coastal zone and continental slopes, earthquakes and volcanic eruptions that can cause tsunamis are widespread. This is one of the most highly productive regions of the global ocean with record levels of primary production equivalent to 70% of all Russian marine biological resources (Antonov *et al.*, 2013) and important fishing areas with valuable marine animals and algae **(Figure 3.40)**.

In these waters, there are 37 species of marine mammals: 27 cetaceans, eight pinnipeds, the polar bear and the sea otter (Artyukhin *et al.*, 1999; Burdin *et al.*, 2009; Hunt *et al.*, 2000; Geptner *et al.*, 1976; Sokolov, 1986; Yablokov *et al.*, 1972). The pelagic fishes in Russian waters of the Far Eastern seas and the Pacific Ocean comprise about 450 species, among which 114 species are identified in the Sea of Japan, 258 species in the Sea of Okhotsk, 170 species in the Bering Sea, and 319 species in the Russian waters of the Pacific Ocean. The average density of pelagic fauna in this area was calculated from about 20 years of trawl catches between 1980 and 2009, as an average of 16.8 tons/km^2 and a total resource of about 70–80 million tons (Ivanov & Sukhanov, 2015) **(Figure 3.40)**.

The Sea of Japan is one of the most diverse seas in Europe and Central Asia. A total of 33,629 species have been reported to occur in these waters. The state of knowledge was extremely variable, with taxa containing many inconspicuous, smaller species tending to be less well known. The total number of species is estimated as 155,542, including 121,913 of identified but undescribed species reached (Fujikura *et al.*, 2010).

PAST AND CURRENT TRENDS

After the dissolution of the USSR, production of commercial fish sharply decreased but since the beginning of the 21st century fishing volume has steadily increased. In 2012

Figure **3** **39**   **General chart of the North Western Pacific area. Source: Google (n.d.).**
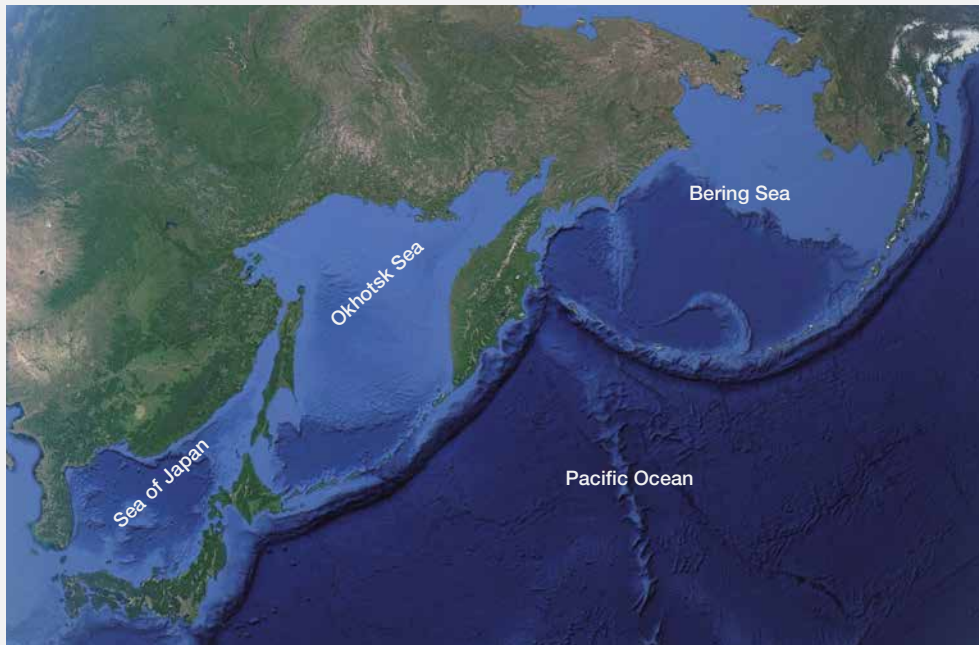
Figure **3** **40**   **Mean annual composition of the most abundant nekton species and total nekton biomass in different regions of the Far Eastern Seas: numerator — in the epipelagic layer (0–200 m), denominator — in the mesopelagic layer (200–1000 m), in frame — total biomass (106 t) and concentrations (t/km²) of nekton. Source: Shuntov & Temnykh (2013). With permission of Springer.**
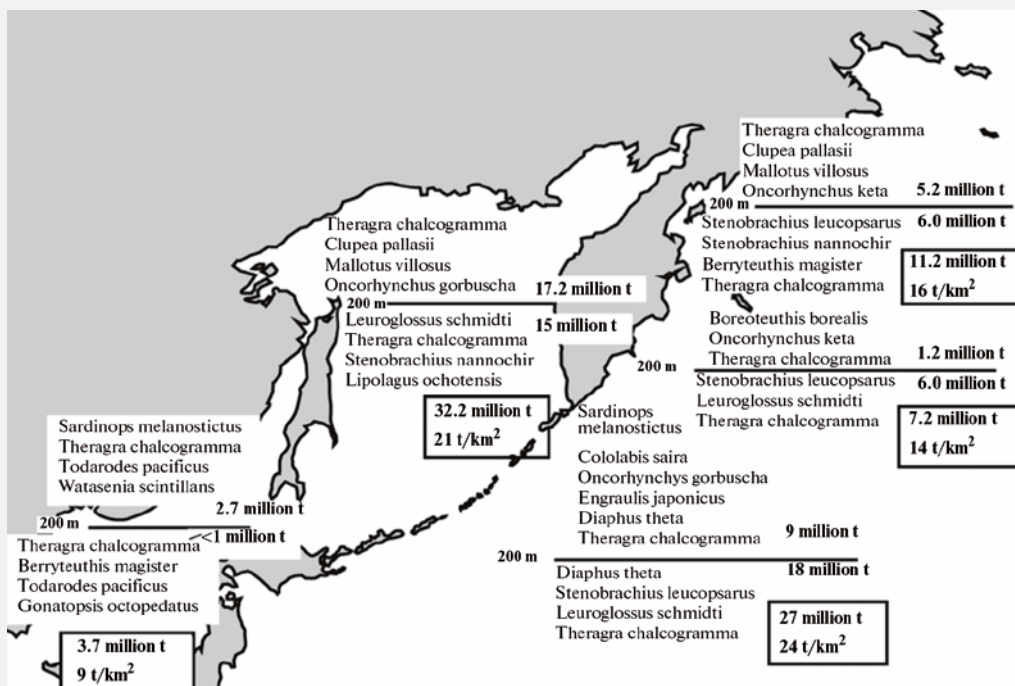
the official catch was equal to 1.7 million tons (Antonov *et al.*, 2013) **(Figure 3.41)**. The volume of poaching is unknown. From 2006 to 2012 there was significant growth in catch, mainly of pollock, cod, herring, bluefish and lemonemy (*Laemonema longipes*) and the composition of the 2012 catch can be seen in **Figure 3.41** (Shevchenko & Datsky, 2014).

From 1930 through the 1970s benthic communities of the Amur Bay have changed dramatically because of pollution: the number of polychaetes has decreased between 5-10 times, brittle stars 2-3 times, the average biomass of benthos by one third. Stocks of Gray's mussels have diminished, and the number and growth rate of scallops have drastically decreased. The stocks of commercial seaweeds (*Ahnfeltia*) decreased – from 86.5 to 40 thousand tons from 1961 to the present time (Belan, 2003). The number of polychaetes, tolerant to low oxygen conditions increased (Belan, 2003). Mass mortalities of small fish have occurred (Yablokov *et al.*, 2014).

The Okhotsk-Korean population of grey whales is one of the most vulnerable in the world. It is included into the Red List of threatened species as "critically endangered" (IUCN, 2015) and is in the Russian Red Book. The reason for its decline in the past was whaling, while in the present day intensive exploitation of oil and gas deposits on the shelf near Sakhalin Island threaten destruction of the population on its the summer-autumn feeding grounds (Adrianov, 2011). The far Eastern seas are important for the Russian economy due to the discovery of large oil and gas reserves on the Far Eastern shelf. However, after an agreement

between NGOs and an oil company, mitigation plans for the company exploitation where agreed and followed and the number of whales increased from about 115 animals in 2004 to 174 in 2015 (Martin-Mehers, 2016).

In the waters of the Gulf of Peter the Great 32 potentially harmful species of microalgae capable of producing biotoxins were discovered (Adrianov & Tarasov, 2007). Recently blooms of strains of microalgae that are highly pathogenic and highly virulent have appeared and accumulations of dangerous microorganisms in filter-feeding organisms may lead to a threat to human health (Adrianov & Tarasov, 2007) **(Figure 3.42)**.

312 invasive species were found in Peter the Great Bay, including 104 southern migrants, most of them were transported in ballast waters. In the last 12 years 19 new tropical and subtropical species were detected (Adrianov, 2011). The expansion to the north of not only individual species, but entire complexes of the southern biota is one of the consequences of climate change (IPCC, 2014b).

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Fishery is the main pressure on the North West Pacific Ocean's biological diversity. The total catch every year reaches several million tons of fish and invertebrates. Before 1990, the Soviet Union provided more than half of the world's total catch of pollock (about 2.5 million tons) (FAO, 2011). Excessive fishing of species such as crab, cod, pollock and others, and the by-catch of non-target fish lead

Figure 3 41 **The composition of marine fish catch of the Far East seas in 2012 (tonnes; %). Source: Shevchenko & Datsky (2014).**

THERAGRA CHALCOGRAMMA 69%
CLUPEA PALLASII 14%
COLOLABIS SAIRA 4%
COD 3%
PLEURONECTIDAE 3%
HEXAGRAMMIDAE 2%
ELEGINUS GRACILIS 1%
MACROURUS 1%
GOBIIDAE 1%
OTHER SPECIES 2%
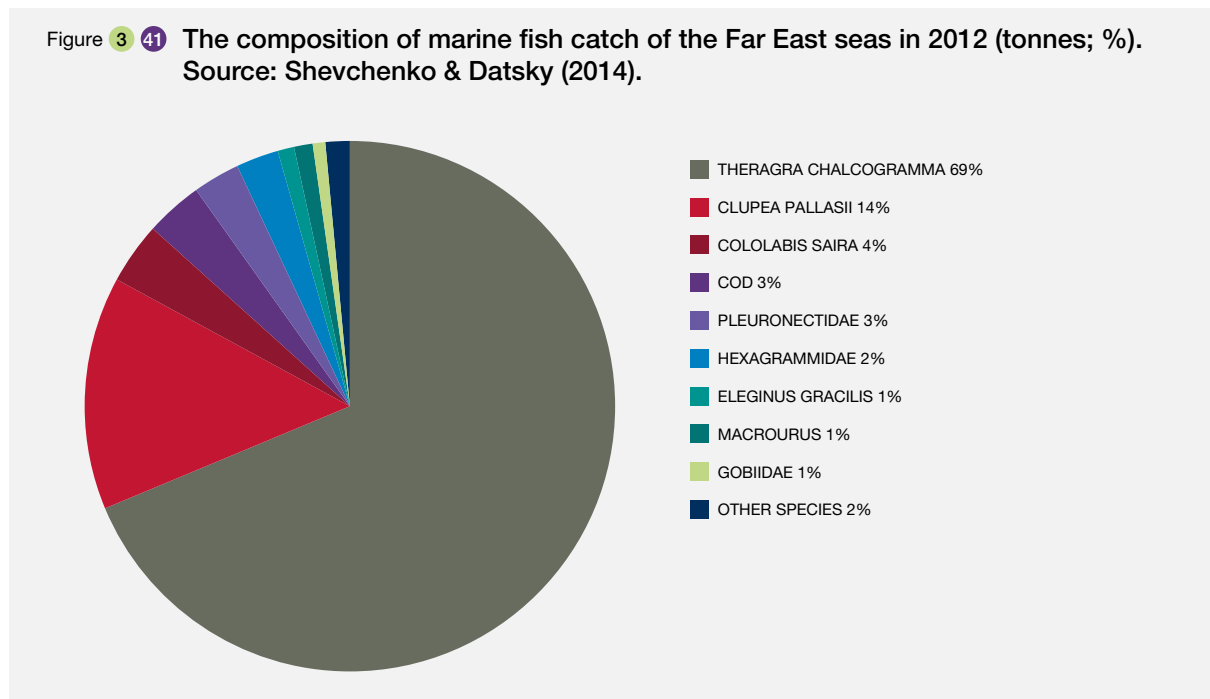
Figure **3** **42** **Harmful algal blooms in the Far East seas of Russia in 1980–2005.**
Source: Orlova *et al.* (2002).
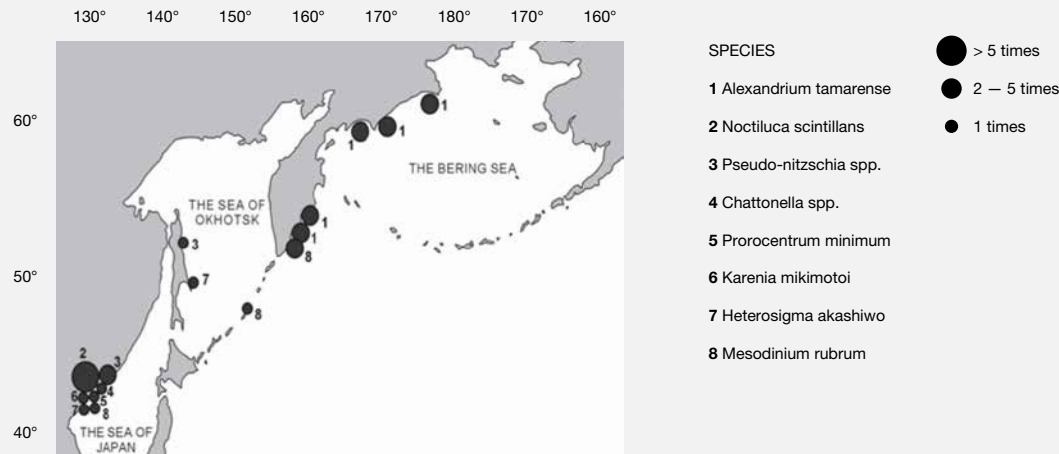
to the loss of fishing activities, e.g. the loss of Far Eastern crab fisheries (Adrianov, 2011).

A fast reduction of sea ice (4% of the sea area per decade) was recorded in the Okhotsk Sea in the period of 1957-2012 (Roshydromet, 2014). Thirty-three years of observations (1979-2011) showed that the air temperature above the water surface in the Sea of Japan had increased by 0.27°C. In the last 50 years the average temperature of surface waters in the Peter the Great Bay have increased by nearly 0.6°C and the amount of precipitation in the Far Eastern Seas has decreased (Roshydromet, 2014). This creates favourable conditions for invasive species (Adrianov, 2011). An assessment performed by PICES (The North Pacific Marine Science Organization) (Kestrup *et al.*, 2015) found 208 NIS (Non-indigenous Aquatic Species) for the North West Pacific, introduced mostly by ballast water, hull fouling, the aquatic animal and plant trade or aquaculture.

Areas of the North West Pacific are impacted by pollution by oil products from oil and gas extraction on the shelf. Draining of fuel in ports and along the transportation routes, and dumping of decommissioned ships in the coastal zones is a very significant source of pollution in these waters. Their effects on marine biota are severe, including oil films on the surface poisoning birds and other animals, and disrupting photosynthesis and oxygen exchange with the atmosphere (Yablokov *et al.*, 2014).

Excessive run-off of nutrients (nitrogen, phosphorous) from land causes eutrophication leading to hypoxia and the degradation of water ecosystems (Yablokov *et al.*, 2014). Also, marine farming of fish and invertebrates harms the ecosystems at the local level degrading habitats and increasing pollution from organic waste, leading to the

deterioration of water quality and a drop in farm productivity. The area affected by pollution can be tens of times greater than the area of the farm (Vyaznikova, 2014).

3.3.4.7 Deep-sea in Europe and Central Asia

OVERVIEW OF THE SYSTEM

The deep-sea is usually defined as those parts of the ocean deeper than 200 m (Gage & Tyler, 1991) beyond the edge of the continental shelf. It is the largest biome on earth, covering approximately 60% of the Earth's solid surface. In the Europe and Central Asia region, the deep-sea covers an area greater than 15 million km^2, encompassing 8 pelagic and 37 benthic biogeographic provinces (UNESCO, 2009) and 11 hydrothermal vent provinces (Rogers *et al.*, 2012). Due to its limited accessibility, it is the least understood, yet one of the richest ecosystems on the planet supporting a high diversity of habitats (e.g. deep-sea pelagic habitats, continental slopes, abyssal soft sediments plains, seamounts, mid ocean ridges, deep-sea canyons and trenches, and smaller habitats such as hydrothermal vents, cold seeps, or cold water coral reefs) and species, as well as a set of supporting and regulating functions and services (Thurber *et al.*, 2014).

PAST AND CURRENT TRENDS

The Millennium Ecosystem Assessment recognized more than 10 years ago that enormous deep-sea species richness remains undiscovered (MEA, 2005), and this is still true today (Mengerink *et al.*, 2014). While there is a solid

understanding of biodiversity changes in many coastal ecosystems, trends in the deep sea are poorly described (MEA, 2005), and even basic ecological information (e.g., species ranges, population subdivision, population genetic diversity, dispersal capability and demographic parameters) is lacking for the vast majority of species (Taylor & Roterman, 2017).

However, changes in biodiversity and abundance have been reported as a result of deep-sea fishing activities, oil spills, climate change, and other activities (Koslow *et al.*, 2016). Also declines in cold-water coral and deep-sea sponge abundance and community structure have been widely reported, including off Norway, in the Barents Sea, the Azores, and other regions (Clark *et al.*, 2016; Pham *et al.*, 2014a).

Recent changes in climate (5-16 years) in the deep-sea changed benthic species diversity, abundance and faunal composition (Glover *et al.*, 2010). This biodiversity loss in deep-sea ecosystems has been shown to produce exponential reductions of ecosystem functions (Danovaro *et al.*, 2008) (see Section 3.2).

Although trends are based on a very limited portion of the deep-sea (Koslow *et al.*, 2016), they indicate increased habitat degradation, and declines in biodiversity, abundance and probably ecosystem functioning (Baldrighi *et al.*, 2017). This may also mean that the achievement of important Aichi Biodiversity Targets may be compromized. Targets 5, 6 and 10 under the Strategic Goal B of the Strategic Plan for Biodiversity 2011-2020, and Target 11 under Strategic Goal C may require additional attention and management measures in this context. This may include more effective fisheries management, and an increase of protected areas in the deep-sea and other area-based conservation measures.

ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

Although humans utilized the oceans for millennia, only recently, through technological developments, deep-sea exploitation has begun. The past century has seen a significant increase in human activities that directly affect deep-sea ecosystems, including fishing, waste disposal, oil and gas extraction and bio-prospecting (Morato *et al.*, 2006; Pham *et al.*, 2014b; Ramirez-Llodra *et al.*, 2011; Sandrea & Sandrea, 2010; Synnes, 2007). Added to these pressures are indirect effects caused by global climate change (Ramirez-Llodra *et al.*, 2011).

Bottom fishing has been the major driver of past ecosystem changes in the deep-sea (Clark *et al.*, 2016). It has modified seafloor morphology and its physical properties (Puig *et al.*, 2012), produced overfishing of many stocks, and produced

extensive damage to benthic communities, many of them of Vulnerable Marine Ecosystems (VME) (Clark *et al.*, 2016; Hall–Spencer *et al.*, 2002; Pham *et al.*, 2014a).

Global landings of marine deeper water species have increased over the last 50 years (Morato *et al.*, 2006; Watson & Morato, 2013). Many of these fisheries have been overfished or depleted (reviewed in Norse *et al.*, 2012). Bailey *et al.* (2009) and Godbold *et al.* (2013) analysed scientific trawl data from 1977 to 2002 in the Porcupine region of the North East Atlantic deep sea and found a significant decrease of 36% in fish biomass in fished depths and considerably deeper.

Decline in deep-sea benthic invertebrate diversity (reviewed by Clark *et al.*, 2016) has been observed as a consequence of deep-sea fishing in the Barents Sea, and other regions.

Although evidence has been found from the geological record that past climate change has impacted deep-sea faunas, the evidence that recent climate change or climate variability has altered deep-sea benthic communities is still limited (Glover *et al.*, 2010). This mainly reflects the lack of observations and monitoring of this vast seafloor habitat.

Additionally, new industrial activities in the deep-sea are emerging, including the extraction of gas hydrates, carbon sequestration, and mining. Future deep-sea mining (Petersen *et al.*, 2016) has the potential to disturb hundreds of thousands of km^2 of seabed and pelagic environment, with uncertain consequences (Levin *et al.*, 2016). The recent discovery of microplastics in deep-sea sediments suggests that this emergent form of pollution is more far reaching than previously anticipated (Van Cauwenberghe *et al.*, 2013).

3.3.4.8 Progress towards goals of Multilateral Environmental Agreements

AICHI BIODIVERSITY TARGET 11 AND SUSTAINABLE DEVELOPMENT GOAL 14, TARGET 14.5

Subtarget "At least 10 per cent of coastal and marine areas are conserved (in marine protected areas)"

The definition of marine protected area varies significantly (e.g. Costello & Ballantine, 2015), which causes divergence in the numbers presented as percentage of marine protected area coverage both globally and regionally. In Europe and Central Asia, the coverage of marine protected areas was calculated as 4% of its marine area (within the "exclusive economic zone" of 200 nautical miles) by Brooks *et al.* (2016) and as 5.3% calculated by the present assessment with 2017 numbers from the Convention on Biological Diversity (CBD, 2017).

Within Europe and Central Asia, significant differences occur in terms of coverage both between the different regional seas and the coverage of coastal waters and off-shore, within the exclusive economic zone of coastal states. Marine protected area networks cover more than 5.9% of the European Union marine area but only about 3% of Russian Federation marine waters. On the other hand, in European Union countries more than 16% of coastal marine areas now have some form of protection but, beyond 12 nautical miles from the shore, an area representing 80% of the European Union's total sea area, only 3% are protected.

In the framework of regional agreements such as OSPAR (see below), HELCOM (see below), the Bucharest and Barcelona Conventions, and the Arctic Council there have been significant advances regarding the area covered by marine protected areas, including in "areas beyond national jurisdiction", and the integration of these marine protected areas in regional networks.

The OSPAR network comprises 448 marine protected areas, covering 5.9% of the OSPAR maritime area, including 16.7% of its coastal waters; 2.3% of the exclusive economic zones (EEZs) of OSPAR countries; and seven marine protected areas situated in areas beyond national jurisdiction covering 8.9% of this OSPAR area (OSPAR, 2017a). Marine protected area coverage also varies geographically, covering 14.7% of the Greater North Sea but only 1.9% of the Arctic OSPAR area.

The HELCOM marine protected area network from the Baltic Sea was the first in the world, already in 2010, to reach the target of conserving at least 10% of coastal and marine areas. But although today this network covers 11.8% of the Baltic Sea, protection is not evenly distributed between sub-basins or between coasts and open sea, and the aim remains to reach the 10% target in all offshore sub-basins (HELCOM, 2017e).

In the Mediterranean 1,231 marine protected areas and "other effective area-based conservation measures" now cover 7.14% of the Sea area, through a large variety of conservation designations, but with the "no-go", "no-take" or "no-fishing" zones accounting only for 0.04% (MedPAN and RAC/SPA, 2016). Coverage is very uneven in geographic terms: over 72.77% of the surface covered is located in the western Mediterranean. Designations cover 9.79% of European Union waters mostly due to the Natura 2000 at sea network. To reach the Aichi Biodiversity Target 11 of 10% of marine areas protected, an additional 71,900 km^2 (2.86% of the Mediterranean) will have to be designated. To also fulfill the representivity goal, these new designations should target currently under-represented features and subregions (MedPAN and RAC/SPA, 2016).

The extent of protected areas in the Arctic's marine environment has almost quadrupled since 1980 and represents today 4.7% of the Arctic marine area (CAFF, 2017). The marine protected area is dominated by several very large areas and some parts of the Arctic marine ecosystem are still poorly protected. In 2013, the Arctic Council adopted a resolution to identify "Areas of heightened ecological and cultural significance" similar to the Convention on Biological Diversity's "ecologically and biologically significant areas" criteria. Through this process, 98 areas were identified covering about 76% of the Arctic marine area. These areas were identified primarily on the basis of their ecological importance for fish, birds or marine mammals (CAFF, 2017). Approximately 5% of "areas of heightened ecological importance" lie within the present protected areas.

An effort to achieve the Aichi Biodiversity Target 11 of 10% has led to a significant increase in number and extent of marine protected areas of different kinds in Europe and Central Asia in recent years (e.g. in OSPAR it went from 159 in 2010 to 448 in 2016 and from 1.06% of the areas in 2010 to 5.9 in 2016 (OSPAR, 2017a) and in the Mediterranean 397 new marine protected areas were designated between 2012 and 2016). The general trend in marine protected area designation is therefore very positive. In 2017, 15 coastal nations have already more than 10% of their marine waters protected (CBD, 2017).

Global conservation targets based on area alone will, however, not optimize protection of marine biodiversity, and the emphasis should be on better marine protected area design, adequate management and compliance to ensure that they achieve their desired conservation value. Edgar *et al.* (2014) showed that the conservation benefits of marine protected area increased significantly with the accumulation of five key features: no fishing allowed, well enforced, old (>10 years), large (>100 km^2), and isolated by deep water or sand. These were also shown to be key features in the Mediterranean (Giakoumi *et al.*, 2017), although here some small but well managed marine protected areas were also effective in conservation.

Subtarget "Protected areas are ecologically representative and well connected and include areas of particular importance for biodiversity and ecosystem services"

Since there is so much difference between coverage of marine protected area in open seas and in coastal waters, ecological representativeness is still not achieved in Europe and Central Asia. In OSPAR progress was made in recent years towards an ecologically coherent and well-managed network, but further work is required to achieve this goal (OSPAR, 2017a). This network is well distributed in the Greater North and Celtic Seas, but substantial gaps remain

in Arctic Waters and the wider Atlantic Ocean. Also 19 of the 54 OSPAR listed features (i.e. species or habitats) are already protected by more than one marine protected area in those parts of the North East Atlantic where they are considered to be at risk. This includes all five listed invertebrates, three of the seven bird species, one of the two reptile species, one of the three marine mammal species, five of the 22 fish species and four of the 15 types of habitat.

The HELCOM assessment of ecological coherence (HELCOM, 2016) showed that the areal representation of different types of broad-scale habitats and the replication of a set of indicative species and biotope were at an acceptable level for supporting a coherent marine protected area network. However, connectivity, which measures how well the network supports the migration and dispersal of species, is not yet optimal.

Subtarget "Protected areas are effectively and equitably managed"

For many of the marine protected areas in waters of Europe and Central Asia, management plans either do not exist; or knowledge on the implementation of protective measures or the effectiveness of these measures to reach the sites' conservation targets is insufficient (MedPAN and RAC/SPA, 2016). Only a small percentage is known to have reached or to be moving towards the objectives they were set up to attain. The resources needed to adequately implement the existing regulations and to manage pressures inside and outside of marine protected areas are still very often not in place.

Information on management is available for 61% of OSPAR marine protected areas, with a further 16% partially documented. But management measures have been implemented for only 12% of OSPAR marine protected areas, with partial action for a further 54%. The situation is similar for monitoring, implemented only for about 14% of these marine protected areas (OSPAR, 2017a). So only 11% of OSPAR marine protected areas were found to be moving towards or have achieved their conservation objectives.

Implementation of management actions for OSPAR marine protected areas in "areas beyond national jurisdiction" have started by OSPAR member countries, but successful management requires cooperation with international organisations with competence for the management of human activities, such as fishing, shipping and deep-sea mining. A mechanism to help cooperation between the relevant organisations has been started between OSPAR and the Northeast Atlantic Fisheries Commission, referred to as "the collective arrangement" (OSPAR, 2017a). On-going negotiations within the United Nations on the conservation and sustainable use of marine biological diversity in areas

beyond national jurisdiction (so called "BBNJ process") is expected to result in a new implementing agreement under the United Nations Law of the Sea that will finally allow marine protected areas in "areas beyond national jurisdiction" to be adequately managed.

HELCOM is now working towards the development of a method to assess the management effectiveness of HELCOM marine protected areas and of the network. Such an assessment will determine the environmental positive effects of the marine protected area management (HELCOM, 2017e).

Many sites of the current system of marine protected area and "other effective area-based conservation measures" in the Mediterranean Sea do not have regulations in place to curb existing pressures or enough means to enforce them. Information about management measures and their effectiveness in maintaining or restoring biodiversity is also lacking. Resources allocated to management are not sufficient for the requirements, thereby compromising successful conservation (MedPAN and RAC/SPA, 2016).

EUROPEAN UNION MARINE STRATEGY FRAMEWORK DIRECTIVE

Progress towards the European Union Marine Strategy Framework Directive goals

The European Union Marine Strategy Framework Directive, approved in 2008, has as its main objective to achieve good environmental status in all waters of the European Union by 2020 (EEA, 2015c). This status is described through 11 descriptors including: biodiversity, non-indigenous species, commercially exploited fish, food-webs, eutrophication, sea-floor integrity, hydrographical conditions; contaminants in the environment, contaminants in seafood, marine litter, and energy, all relevant for determining the status of marine biodiversity and ecosystem functioning. The Directive aims to maintain or restore biodiversity and to attain a marine environment that is healthy, clean, and productive in all the European Union Seas and Ocean areas, and those it shares with its neighbors. Its implementation should also make significant contributions to achieving the goals of the European Union Biodiversity Strategy for the marine environment.

The first assessment of Europe's seas at European Union-wide scale (EEA, 2015c) used data from the first Marine Strategy Framework Directive and Habitats Directive's reporting completed in 2012 and other sources. 80% of the species and habitats assessments under the Marine Strategy Framework Directive were categorized as "unknown" status, but a more complete picture is available for the marine habitats and species protected by the Habitats Directive. Even among assessments of the conservation status of

species and habitat types of conservation interest, only 7% of marine species and 9% of marine habitat types show a "favourable conservation status". Moreover 27% of species and 66% of assessments of habitat types show an "unfavourable conservation status" and the remainder are categorized as "unknown". Additionally, 58% of the assessed commercial stocks did not have "good environmental status", while the status of 40% of commercial fish stocks was not assessed due to lack of data.

There are many "unknowns" when it comes to European Union member State reporting and in commercial fish stock statistics data from mandatory reporting. This highlights the difficulty associated with obtaining data to assess the health status of even the seas that are under European Union responsibility, where relatively rich information exists. However, by comparing information available from European, regional, and national sources, a common pattern of change can be seen: ecological extinctions are being observed across species belonging to different functional groups including species such as monk seals in the Black Sea, bluefin tuna in the eastern North Sea, sharks in the Mediterranean Sea and North East Atlantic Ocean and habitat-forming species like oysters in the North Sea and sea grasses in the Baltic and Mediterranean Seas. Even if there are a few examples of species where the declining trends appear to be halted, such as for bluefin tuna (*Thunnus thynnus*) in certain areas (EEA, 2015c), patterns of degradation are observed across all of the ecosystem components, and across all of the information sources considered. The observed loss of biodiversity affects ecosystem functioning and may cause irreversible loss of ecosystem resilience, putting in jeopardy ecosystem health. Based on different assessments considered the European Union's marine ecosystems could therefore not be considered to be in a healthy state, as would be the objective of the Marine Strategy Framework Directive.

The European Environment Agency (2015e) considered in addition that European Union marine areas could also not be considered clean, even though some improvements in eutrophication are already visible, for example in the Black and Baltic Seas. It stated, however, that they could be considered productive, thus fulfilling one of the three main goals of the Marine Strategy Framework Directive.

Even if this Directive is only valid in the European Union, member States are required to use existing regional cooperation structures to co-ordinate among themselves and to make every effort to coordinate their actions with those of third countries in the same region or subregion. This cooperation has been taking place through OSPAR, HELCOM, the Barcelona and Bucharest Conventions for more than 30 years, and is also done in the framework of the Arctic Council.

Box ③ ❶ **Summary of past and current trends in biodiversity and ecosystems and their attribution to direct drivers of change.**

The table and figure of this box summarize past and current trends in biodiversity and ecosystems for terrestrial and inland surface water units of analysis and marine areas in Europe and Central Asia and the attribution of these trends to direct drivers of change. **Table 3.5** presents the assessed information in terms of trends in areal extent and biodiversity status. Biodiversity status summarizes the biodiversity information assessed in Sections 3.3 and 3.4. **Figure 3.43** summarizes the trend information on biodiversity status.

Table ③ ⑤ **Summary of past and current trends in biodiversity and ecosystems in terms of spatial extent and biodiversity status for terrestrial and inland surface water units of analysis and in terms of biodiversity status for marine systems, and summary of the attribution of these trends to direct drivers of change.**

🔴 HIGH IMPACT 🟡 MODERATE IMPACT 🟢 NO OR MARGINAL IMPACT

Unit of analysis	Indicator	GENERAL TREND Past — ECA WE	CE	EE	CA	Present — ECA WE	CE	EE	CA	CLIMATE CHANGE Past — ECA WE	CE	EE	CA	Present — ECA WE	CE	EE	CA
TERRESTRIAL																	
Snow and ice-dominated systems	Extent	↘				↘				🟡				🔴			
Snow and ice-dominated systems	Extent	↘	↘	↘	↘	↘	↘	↘	↘	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴
Snow and ice-dominated systems	Biodiversity status	↘				↘				🟡				🔴			
Snow and ice-dominated systems	Biodiversity status	↘	↘	↘	↘	↘	↘	↘	↘	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴
Tundra	Extent	↕				↕				🟡				🔴			
Tundra	Extent	↕	·	↕	·	↕	·	↕	·	🟡	·	🟠	·	🔴	·	🔴	·
Tundra	Biodiversity status	↘				↘				🟡				🔴			
Tundra	Biodiversity status	↘	·	↘	·	↘	·	↘	·	🟡	·	🟠	·	🔴	·	🟠	·
Alpine and subalpine systems	Extent	↓				↘				🟡				🟡			
Alpine and subalpine systems	Extent	↓	↓	↓	↓	↘	↘	↘	↘	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
Alpine and subalpine systems	Biodiversity status	↘				↘				🟡				🟡			
Alpine and subalpine systems	Biodiversity status	↘	↘	↘	↘	↘	↘	↘	↘	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
Temperate and boreal forests and woodlands	Extent	↘				↗				🟢				🟡			
Temperate and boreal forests and woodlands	Extent	↘	↘	↘	↘	↗	↗	↕	↕	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🔴
Temperate and boreal forests and woodlands	Biodiversity status	↘				↘				🟢				🟡			
Temperate and boreal forests and woodlands	Biodiversity status	↘	↘	↘	↘	↘	↘	↘	↘	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🔴
Mediterranean forests, woodland and scrubs	Extent	↓				↓				🟡				🔴			
Mediterranean forests, woodland and scrubs	Extent	↓	↓	↓	↓	↓	↓	↓	↓	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🟡
Mediterranean forests, woodland and scrubs	Biodiversity status	↘				↘				🟡				🟡			
Mediterranean forests, woodland and scrubs	Biodiversity status	↘	↘	↘	↘	↘	↘	↘	↘	🟡	🟡	🟡	🟡	🔴	🔴	🟡	🟡
Tropical and subtropical dry and humid forests	Extent	↓				↕				🟢				🟢			
Tropical and subtropical dry and humid forests	Extent	↓	↓	↓	↓	↔	↕	↕	↕	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
Tropical and subtropical dry and humid forests	Biodiversity status	↓				↕				🟡				🔴			
Tropical and subtropical dry and humid forests	Biodiversity status	↓	↓	↓	↓	↕	↕	↕	↕	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴
Temperate grasslands	Extent	↓				↕				🟢				🟡			
Temperate grasslands	Extent	↓	↓	↓	↓	↘	↘	↕	↗	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🔴
Temperate grasslands	Biodiversity status	↓				↕				🟢				🟡			
Temperate grasslands	Biodiversity status	↓	↓	↓	↓	↓	↘	↕	↕	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🔴
Deserts	Extent	↗				↗				🟡				🔴			
Deserts	Extent	↕	·	↗	↗	↕	·	↗	↗	🟡	·	🟡	🟡	🔴	·	🔴	🔴
Deserts	Biodiversity status	↘				↘				🟡				🔴			
Deserts	Biodiversity status	↘	·	↘	↘	↘	·	↘	↘	🟡	·	🟡	🟡	🔴	·	🔴	🔴
Permafrost peatlands	Extent	↔				↔				🟢				🟢			
Permafrost peatlands	Extent	↔	·	↔	·	↔	·	↔	·	🟢	·	🟢	·	🟢	·	🟢	·
Permafrost peatlands	Biodiversity status	↔				↘				🟢				🟡			
Permafrost peatlands	Biodiversity status	↔	·	↔	·	↘	·	↘	·	🟢	·	🟢	·	🟡	·	🟡	·
Boreal peatlands	Extent	↓				↘				🟢				🟡			
Boreal peatlands	Extent	↓	·	↓	·	↘	·	↘	·	🟢	·	🟢	·	🟡	·	🟡	·
Boreal peatlands	Biodiversity status	↓				↘				🟢				🟡			
Boreal peatlands	Biodiversity status	↓	·	↓	·	↘	·	↘	·	🟢	·	🟢	·	🟡	·	🟡	·
Temperate peatlands	Extent	↘				↔				🟢				🟢			
Temperate peatlands	Extent	↘	↘	↘	·	↔	↔	↔	·	🟢	🟢	🟢	·	🟢	🟢	🟢	·
Temperate peatlands	Biodiversity status	↘				↔				🟢				🟢			
Temperate peatlands	Biodiversity status	↘	↘	↘	·	↔	↔	↔	·	🟢	🟢	🟢	·	🟢	🟢	🟢	·

ECA=Europe and Central Asia, WE=Western Europe, CE=Central Europe, EE= Eastern Europe, CA=Central Asia. ↑/↓ denote strong and consistent increase/decrease in the indicator; ↗/↘ denote moderate and consistent increase/decrease in the indicator; ↔ stable indicator; ↕ variable trend in the indicator.

LAND USE CHANGE								POLLUTION								OVEREXPLOITATION								INVASIVE ALIEN SPECIES							
Past				Present				Past				Present				Past				Present				Past				Present			
ECA				ECA				ECA				ECA				ECA				ECA				ECA				ECA			
WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA

Box 3 1

Table 3 5

Unit of analysis	Indicator	GENERAL TREND — Past ECA WE	CE	EE	CA	GENERAL TREND — Present ECA WE	CE	EE	CA	CLIMATE CHANGE — Past ECA WE	CE	EE	CA	CLIMATE CHANGE — Present ECA WE	CE	EE	CA
TERRESTRIAL																	
Forest-steppe, steppe and other southern peatlands	Extent	↓				↘				●				●			
Forest-steppe, steppe and other southern peatlands	Extent	↓	↓	↓	↓	↘	↘	↘	↘	●	●	●	●	●	●	●	●
Forest-steppe, steppe and other southern peatlands	Biodiversity status	↓				↘				●				●			
Forest-steppe, steppe and other southern peatlands	Biodiversity status	↓	↓	↓	↓	↘	↘	↘	↘	●	●	●	●	●	●	●	●
Agricultural areas	Extent	↗				↔											
Agricultural areas	Extent	↔	↗	↑	↑	↔	↗	↘	↘								
Agricultural areas	Biodiversity status	↓				↘				●				●			
Agricultural areas	Biodiversity status	↓	↘	↘	↓	↘	↘	↕	↕	●	●	●	●	●	●	●	●
Urban areas	Extent	↓				↘				●				●			
Urban areas	Extent	↓	↓	↓	↓	↘	↘	↘	↘	●	●	●	●	●	●	●	●
Urban areas	Biodiversity status	↓				↘				●				●			
Urban areas	Biodiversity status	↓	↓	↓	↓	↘	↘	↓	↓	●	●	●	●	●	●	●	●
Subterranean habitats	Extent	↘				↓				●				●			
Subterranean habitats	Extent	↘	↘	↘	↘	↘	↓	↓	↓	●	●	●	●	●	●	●	●
Subterranean habitats	Biodiversity status	↘				↓				●				●			
Subterranean habitats	Biodiversity status	↘	↘	↘	↘	↘	↓	↓	↓	●	●	●	●	●	●	●	●
INLAND SURFACE WATER																	
Freshwater	Extent	↓				↘				●				●			
Freshwater	Extent	↓	↓	↓	↓	↘	↕	↓	↓	●	●	●	●	●	●	●	●
Freshwater	Biodiversity status	↓				↘				●				●			
Freshwater	Biodiversity status	↓	↓	↓	↓	↘	↕	↓	↓	●	●	●	●	●	●	●	●
Aral Sea	Extent	↓				↘				●				●			
Aral Sea	Extent	•	•	•	↓	•	•	•	↘	•	•	•	●	•	•	•	●
Aral Sea	Biodiversity status	↓				↘				●				●			
Aral Sea	Biodiversity status	•	•	•	↓	•	•	•	↘	•	•	•	●	•	•	•	●
Caspian Sea	Extent	↕				↔				●				●			
Caspian Sea	Extent	•	•	↕	↕	•	•	↔	↔	•	•	●	●	•	•	●	●
Caspian Sea	Biodiversity status	↘				↘				●				●			
Caspian Sea	Biodiversity status	•	•	↘	↘	•	•	↘	↘	•	•	●	●	•	•	●	●
Saline lakes	Extent	↘				↘				●				●			
Saline lakes	Extent	↘	↘	↘	↕	↘	↘	↘	↓	●	●	●	●	●	●	●	●
Saline lakes	Biodiversity status	↘				↘				●				●			
Saline lakes	Biodiversity status	↘	↘	↘	↘	↘	↘	↘	↘	●	●	●	●	●	●	●	●
MARINE																	
North East Atlantic	Biodiversity status	↘				↘				●				●			
Baltic Sea	Biodiversity status	↘				↓				●				●			
Mediterranean Sea	Biodiversity status	↓				↓				●				●			
Black and Azov Seas	Biodiversity status	↓				↘				●				●			
Arctic Ocean	Biodiversity status	↕				↘				●				●			
North West Pacific Ocean	Biodiversity status	↘				↘				●				●			
ECA deep-sea	Biodiversity status	↕				↘				●				●			
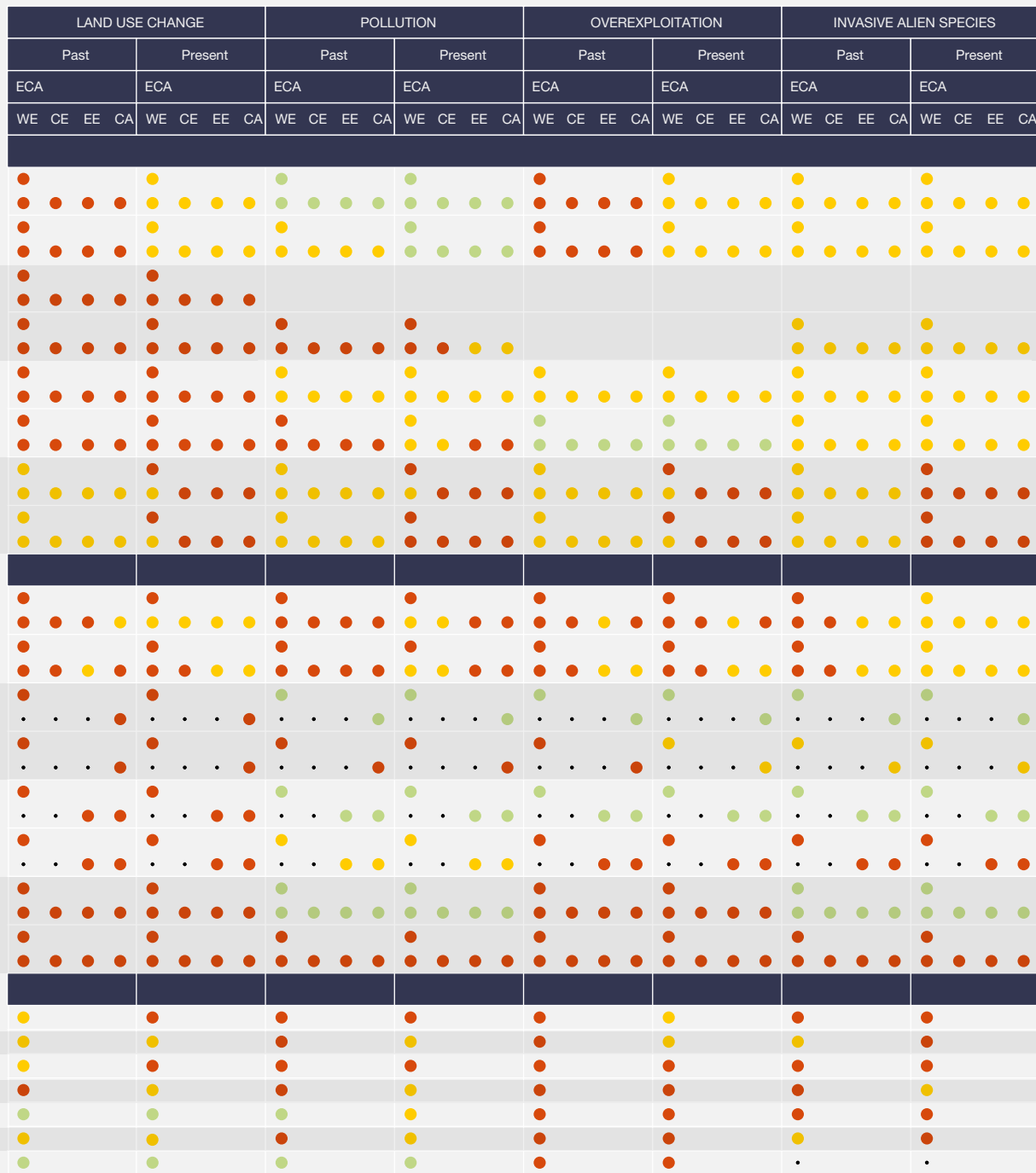
Box 3 1

Figure 3 43 **Summary graph of the assessment of past (~1950–2000) and current (~2001–2017) trends in biodiversity status of marine, inland surface water and terrestrial ecosystems for the four subregions and the whole of Europe and Central Asia.**

The figure summarizes the trends in biodiversity status of the assessed units of analysis (habitat types). Biodiversity status represents the expert assessment of available indicators of habitat intactness, species richness and the status of endangered species. The trends are presented by unit of analysis and subregion for terrestrial and inland surface-water ecosystems, and by sea or ocean area for marine ecosystems. WE=Western Europe, CE=Central Europe, EE= Eastern Europe, CA=Central Asia, ECA=Europe and Central Asia.

		PAST					PRESENT				
		WE	CE	EE	CA	ECA	WE	CE	EE	CA	ECA
TERRESTRIAL	Agroecosystems										
	Alpine and subalpine systems										
	Boreal peatlands										
	Deserts										
	Forest-steppe, steppe and other southern peatlands										
	Mediterranean forests and scrubs										
	Permafrost peatlands										
	Snow and ice-dominated systems										
	Subterranean habitats										
	Temperate and boreal forests and woodlands										
	Temperate grasslands										
	Temperate peatlands										
	Tropical and subtropical dry and humid forests										
	Tundra										
	Urban ecosystems										
INLAND SURFACE WATER	Aral Sea										
	Caspian Sea										
	Inland surface water										
	Saline lakes										

MARINE	North East Atlantic	Baltic Sea	Mediterranean Sea	Black and Azov Seas	Arctic Ocean	North West Pacific Ocean	ECA deep-sea
PAST							
PRESENT							

Legend:

- Strong and consistent increase in indicator
- Moderate and consistent increase in indicator
- Strong and consistent decrease in indicator
- Moderate and consistent decrease in indicator
- Stable indicator
- Variable trend in indicator
- Not applicable

Confidence level:
- Well established
- Established but incomplete/unresolved
- Inconclusive

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3.4 PAST AND CURRENT TRENDS BY TAXONOMIC GROUP

3.4.1 Introduction

Europe and Central Asia hosts more than 10% of the world's vascular plant species, and about 25% of animal and plant groups comprehensively assessed by IUCN are unique to this region. Between 20 and 120 species have gone extinct regionally and an additional 44 to 67 have gone extinct globally since the 1500s[7] (data summarized from the IUCN Red List of Threatened Species - Species Information System, March 2017). These numbers are an under-estimation considering that only about 86,000 species have been assessed by the IUCN, less than 4% of species of plants and animals described today (estimated to be 2.3 millions according to Jenkins *et al.*, 2013). In addition to the extinctions recorded at large scale, numerous extinction events were recorded at the country level. The following statistics are based on a subset of taxonomic group that has been comprehensively assessed[8]. There is a high risk of extinction for 13% of species occurring in Europe and Central Asia in these selected groups and for which data is available (94% of the 2,493 species in these taxonomic groups). 13.5% of the species in the region are endemic, and 27.9% of these species are threatened. The Central and Western European subregions hold the highest percentages of species threatened (13.3%) and endemic (10.6%), and the highest percentage of endemics threatened (35.1%), with these percentages primarily driven by the many threatened endemic species in the Mediterranean hotspot and the Macaronesian Islands **(Figure 3.44)**.

Eastern Europe and Central Asia have lower percentages of species threatened (<10%) and endemic (<5%), and a smaller proportion of endemics threatened (<10%). For mammals, birds, and amphibians, global assessments of extinction risk against the Red List Categories and Criteria have been undertaken multiple times over the last three decades to derive Red List Indices as indicators of the rate at which species groups are sliding towards extinction, and these can be combined with species distribution data to produce geographically downscaled Red List Indices (i.e.,

regional contributions towards the global Red List Index; Rodrigues *et al.* 2014). Specifically, changes in aggregate extinction risk of all regions' and subregions' species can be calculated, showing how adequately species are conserved relative to their potential contribution to global species conservation. The contribution to increasing global extinction risk varies among the subregions, with Central and Western Europe contributing the most, followed by Central Asia and Eastern Europe **(Figure 3.45)**.

Below we discuss status and trends for most major taxonomic groups. These trends and their attribution to different direct drivers are summarized in **Table 3.11**. Insufficient data were available to assess status and trends of marine species except for mammals, birds and fishes. Status and trends in community composition and biomass stocks of marine plankton are dealt with in the marine units of analyses section, whereas the lack of status and trends of other taxonomic groups, including non-planktonic marine invertebrates, algae and protozoans, are discussed in the knowledge gaps section.

3.4.2 Birds

Status and trends

There are an estimated 887 extant bird species in Europe and Central Asia, 25 endemic (BirdLife International, 2016), and 71 threatened with extinction (categories vulnerable, endangered and critically endangered; BirdLife International, 2016). Analysis of changes of categories in the IUCN Red List between 1988 and 2008 suggests that Eastern Europe was the subregion with the greatest declines (the most changes towards higher threat categories), and Central Asia was the subregion with the smallest declines (Brooks *et al.*, 2016). No species within the region has gone extinct since 1980, but three species are possibly extinct or nearing extinction in the Western, Central and Eastern European subregions (BirdLife International, 2016).

Areas of high bird richness include Russia, Turkey, the Mediterranean, Israel, the Black Sea and the Caucasus (BirdLife International, 2015, 2016; **Figure 3.46 A**). The highest rates of endemism, and highest numbers of threatened species (Jenkins *et al.* 2013; BirdLife International, 2016; **Figure 3.46 B**) are found in the Mediterranean and Macaronesian islands, as well as the Caucasus (BirdLife International, 2015, 2016), and Central Asia.

There is strong evidence for a moderate *overall decline of bird populations* in the region (BirdLife International, 2017). A recent report (BirdLife International, 2015) shows that out of the 533 species breeding in the EU-27 countries, 153 have declined since 2001, while 136 show a long-term decline (since 1980, **Table 3.6**). Most of the large-scale,

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7. The lower value are the documented number of extinctions, the upper value is obtained by including also all species classified by IUCN as possibly extinct.

8. Mammals, birds, chameleons, amphibians, sharks and rays, selected bony fish groups (angelfishes and butterflyfishes, tarpons and ladyfishes, parrotfishes and surgeonfishes, groupers, wrasses, tunas and billfishes, hagfishes, sturgeon, blennies, pufferfishes, seabreams, porgies, picarels), freshwater caridean shrimps, cone snails, freshwater crabs, freshwater crayfish, lobsters, reef-building corals, conifers, seagrasses, and plant species occurring in mangrove ecosystems. Species assessed by IUCN in other taxonomic groups may not be a random sample, but likely a subset of species deemed at higher risk of extinction, therefore extrapolating their extinction risk to all species may bias the percentage of species endangered.

long-term research studies (Gregory *et al.*, 2007; Jørgensen *et al.*, 2016b; Reif *et al.*, 2008; Vickery *et al.*, 2014) as well as many smaller studies (e.g. Vilkov 2013) also report declines in either species richness or populations. However, different species groups and regions exhibit different trends, and knowledge gaps exist. Notably, population sizes are unknown for many species, particularly in Eastern Europe and Central Asia (BirdLife International, 2017).

A large proportion of species in decline are associated with marine habitats (BirdLife International, 2015). Terrestrial species show contrasting trends among functional groups. Decline is strongest for migratory birds (BirdLife International, 2008; Vickery *et al.* 2014) and habitat

specialists (Le Viol *et al.*, 2012). The latter, coinciding with an increased frequency of generalist species, leads to a decrease in functional diversity. This trend, often referred to as "biotic homogenization", is maybe the typical change in terrestrial avian communities across groups and locations (Le Viol *et al.*, 2012).

Genetic diversity is often studied at the population level (Eeva *et al.*, 2006; Liu *et al.*, 2013) and no clear large-scale trend patterns have been detected as this is still a young field of exploration. Possible threats to avian genetic diversity include habitat fragmentation, hybridization with feral (Randi, 2008) or introduced or invasive species (Muñoz-Fuentes *et al.*, 2007).

Figure **3** **44** Overview of extinction risk of species in the Europe and Central Asia region. Source: IUCN (2017c).
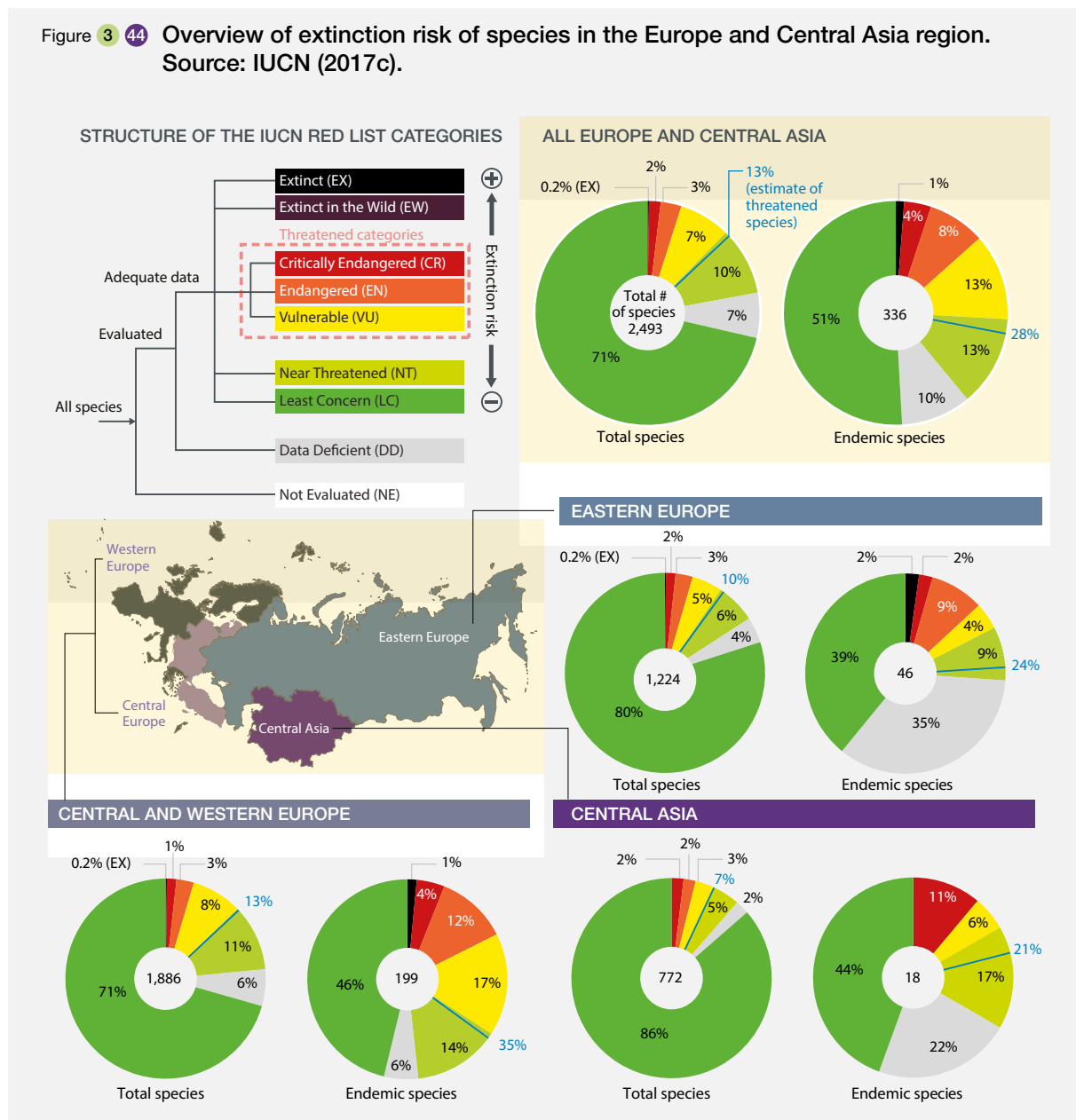
Figure 3 45 **Red List Indices of species extinction risk weighted by the fraction of each species' distribution occurring within Europe and Central Asia and its subregions.**

The position on the vertical axis indicates the aggregate extinction risk facing species in the region overall, while the slope indicates how rapidly this extinction risk is changing. For the region as a whole, the risk of extinction of species has increased over the last 20 years. Species in the Central and Western Europe subregions are least well-conserved relative to the region's potential contribution to global species conservation, and are declining fastest in status. Source: Data from Brooks *et al.* (2016).
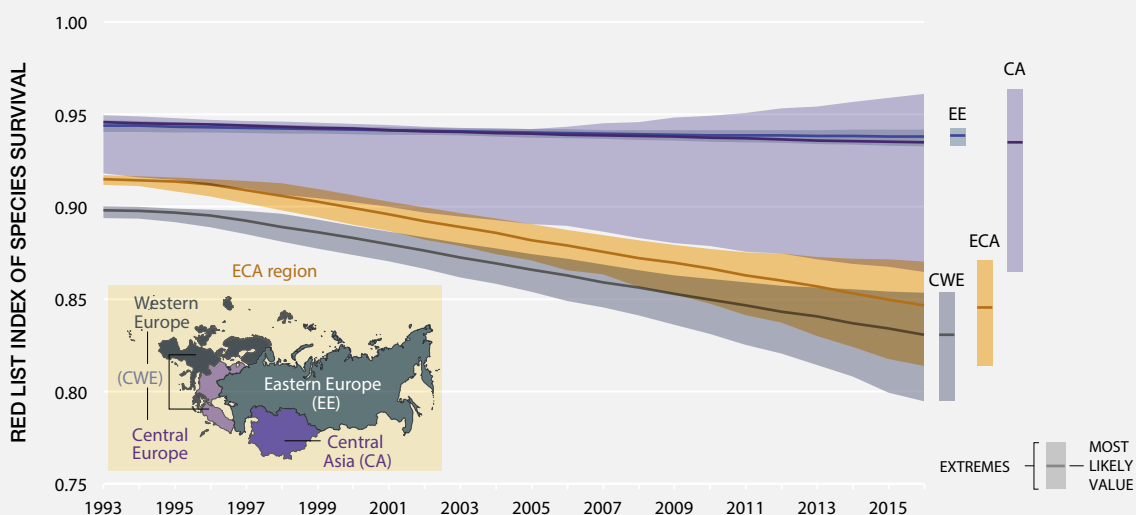

Table 3 6 **Long-term and recent trends of bird species breeding in EU-27 countries (EEA, 2015a). Short-term covers the time period 2001-2012, and long-term the period 1980-2012. The total number of species is 456.**

Trend	Long-term	Short-term
Declining	136	153
Increasing	150	133
Stable	49	96
Fluctuating	6	12
Uncertain	55	23
Unknown	79	58

Drivers of change

Exploitation (hunting, poaching, and bycatch from fisheries) was found to be the largest threat to vulnerable or endangered species by the IUCN (BirdLife International, 2015). Although the exact numbers of birds killed are difficult to evaluate due to lack of data, the order of magnitude in the entire Mediterranean region is several millions of birds killed each year (Arizaga & Laso, 2015; Brochet *et al.*, 2016; Casas *et al.*, 2009; Sokos *et al.*, 2013), while hunting and poaching seem also to be significant in Central Asia (BirdLife International, 2016; Chemonics International, 2001a).

Land and water use is an important driver as it affects multiple species at once. As such it is often reported both in scientific literature and indigenous and local knowledge sources (Roué and Molnár 2016). Overall, decreases in the extent of specific habitats and urban expansion contribute to biotic homogenization (Le Viol *et al.*, 2012; McKinney, 2006). Recent agricultural changes have had a dramatic effect on bird diversity (Donald *et al.* 2001, also see section on Agricultural areas). Amongst forest birds, several changes have been documented, mostly showing a decrease in old forest specialists, deciduous forest

specialists, and cavity-nesters. All these changes can be related to the intensification of forestry practices, which often entail dense monocultures that are harvested before structural elements can benefit many bird species (Gil-Tena *et al.*, 2007; Lõhmus *et al.*, 2016; Sirkiä *et al.*, 2010; Smith *et al.*, 2008). However, no large-scale consensus on land use related trends in forest birds seems to exist (Gregory *et al.*, 2007; Ram *et al.*, 2017).

There is clear evidence that bird communities are locally affected by pollution from industrial activities (Eeva *et al.*, 2012) or pesticide use in agricultural fields, directly (Wegner *et al.*, 2005) or indirectly (Hallmann *et al.*, 2014). Light pollution in urban environments has been shown to affect the timing of reproductive events (Dominoni & Partecke, 2015), but there is not yet any clear evidence of an impact on abundance or community composition.

Invasive alien species and invasive native species (e.g. rats, domestic cats), have been shown to threaten the
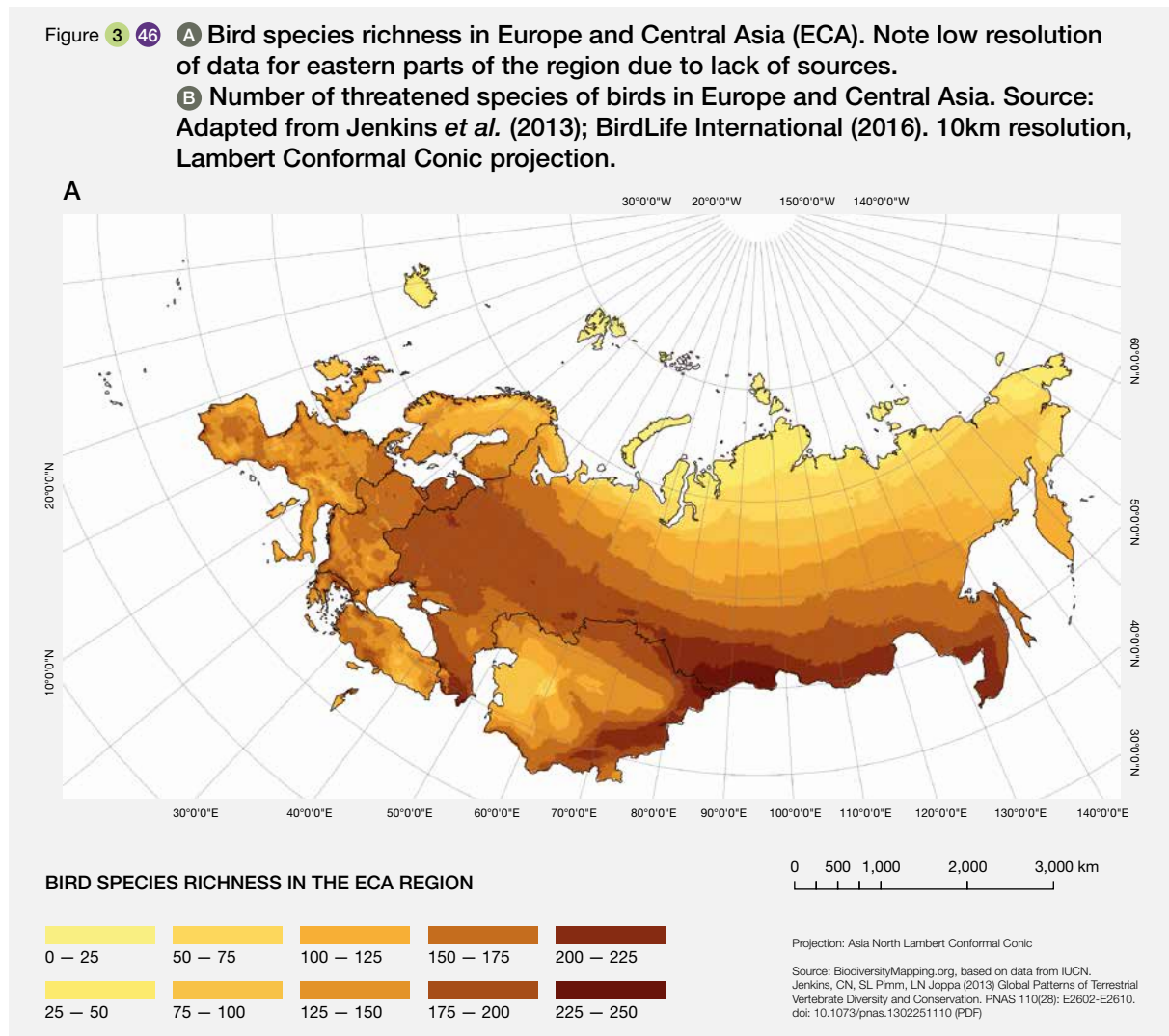
reproductive success of many birds, particularly colonial seabirds (BirdLife International, 2015), and have been linked with declines of some species (e.g. Skorka *et al.*, 2010).

Climate-driven community changes and range expansions or contractions have been reported in many studies (Estrada *et al.*, 2016), and both scientific studies and reports from indigenous herders suggest that local bird declines have been caused by climate change (Roué & Molnár, 2017; Vilkov, 2013). However, evidence of direct impacts of climate change on population decline remains weak.

Other important drivers include direct mortality caused by power lines and wind turbines, although the consequences of population decline are only documented for a few, rare species (BirdLife International, 2015).

In many cases, it is the combination of drivers that put bird species at risk. Seabirds, for instance, have declined strongly due to a multiplicity of threats. Conservation

Figure **3** **46** **A** Bird species richness in Europe and Central Asia (ECA). Note low resolution of data for eastern parts of the region due to lack of sources.
B Number of threatened species of birds in Europe and Central Asia. Source: Adapted from Jenkins *et al.* (2013); BirdLife International (2016). 10km resolution, Lambert Conformal Conic projection.

A

BIRD SPECIES RICHNESS IN THE ECA REGION

0 — 25	50 — 75	100 — 125	150 — 175	200 — 225
25 — 50	75 — 100	125 — 150	175 — 200	225 — 250

Projection: Asia North Lambert Conformal Conic

Source: BiodiversityMapping.org, based on data from IUCN.
Jenkins, CN, SL Pimm, LN Joppa (2013) Global Patterns of Terrestrial
Vertebrate Diversity and Conservation. PNAS 110(28): E2602-E2610.
doi: 10.1073/pnas.1302251110 (PDF)

B

NUMBER OF THREATENED BIRD SPECIES IN THE ECA REGION

Number of bird species per 10 km²

HIGH: 12   LOW: 1

0   500 1,000   2,000   3,000 km

Projection: Asia North Albers Equal Area Conic

Source: BiodiversityMapping.org, based on data from IUCN.
Jenkins, CN, SL Pimm, LN Joppa (2013) Global Patterns of Terrestrial
Vertebrate Diversity and Conservation. PNAS 110(28): E2602-E2610.
doi: 10.1073/pnas.1302251110 (PDF)

Figure ③ ㊼ **Species richness of reptiles across Europe and Central Asia. Source: IUCN (2017c).**

REPTILE SPECIES RICHNESS

High: 62   Low: 0

0   500 1,000   2,000   3,000 km

Projection: Asia North Lambert Conformal Conic

Source: Data from IUCN 2017, published in Roll, Uri,
et al. Nature ecology & evolution (2017).

279
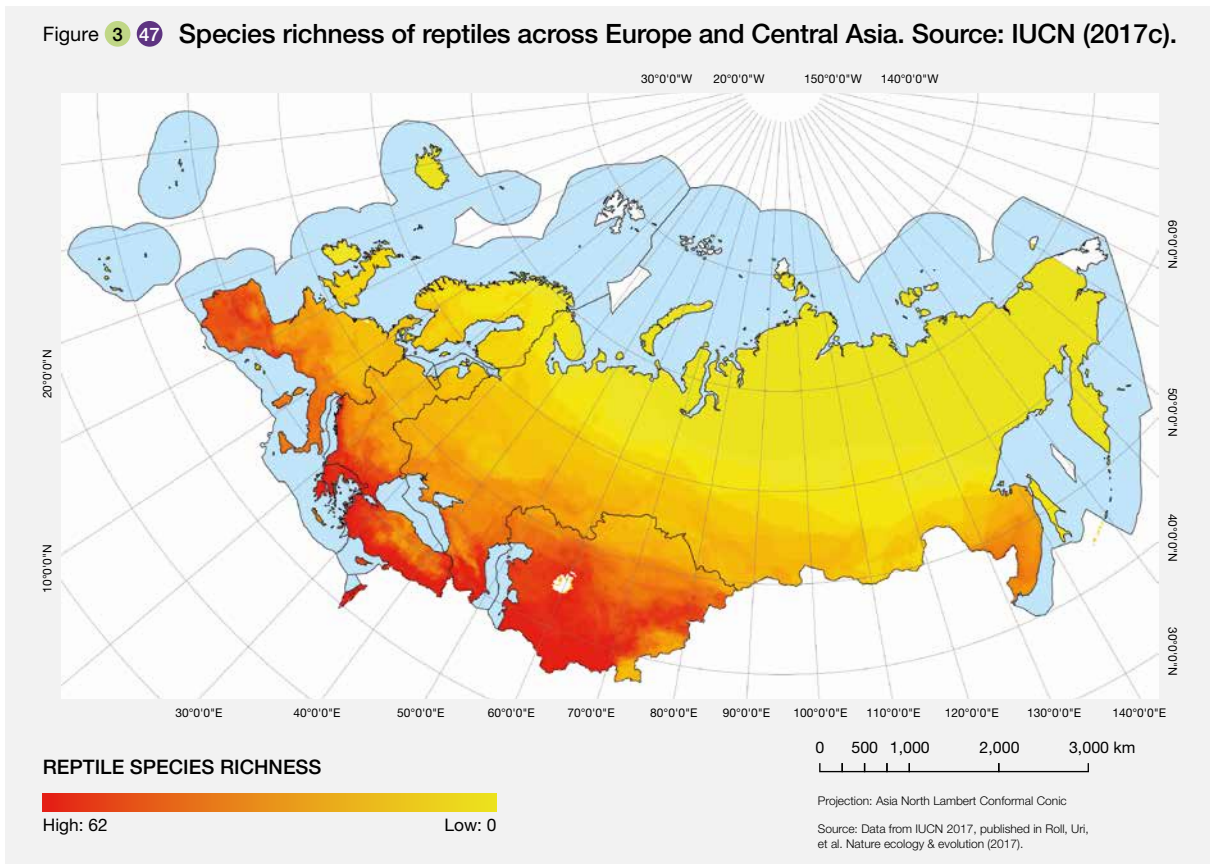
efforts reducing multiple pressures (e.g. the European Union Habitats Directive, national legislations) have been shown to have positive effects on bird populations (Gamero, 2016).

3.4.3 Mammals

Status and trends

There are 538 species of mammals in the IUCN database that are extant in the region. Of these, 66 are threatened with extinction (categories vulnerable, endangered and critically endangered). Their number could be up to 124 (23% of the total) if all data deficient species where found to be vulnerable or worse (IUCN, 2016). Globally, the net annual change in IUCN extinction risk categories for mammals from 1996 to 2008 has been -13, meaning that, on average, 13 species moved one category closer to extinction (Brooks *et al.*, 2016). The Europe and Central Asia contribution to the global trend is -0.47, which is equivalent to having one species endemic to the region moving one category closer to extinction every two years (Brooks *et al.*, 2016).

A notable decline in Europe and Central Asia in recent decades is that of the *Saiga tatarica*, an antelope inhabiting the steppes and semi-desert regions in Russia, Kazakhstan, Uzbekistan and Western Mongolia, which deteriorated from vulnerable in 1996 to critically endangered in 2008. This followed a greater than 95% decline in population size from approximately one million in the early 1990s to an estimated

50,000 by 2008, primarily owing to poaching. An epidemic of pasteurellosis in 2015 caused a further population collapse by 50% in two weeks, with an estimated mortality of >70% (Milner-Gulland & Singh, 2016).

However, species that have received conservation attention are generally improving their conservation status. There are 87 mammal species in the Annex II, IV and V of the European Union Habitats Directive. European Union member States are required to take steps towards their conservation and report on the conservation status of these species every six years. Species in these annexes generally improved their conservation status between 2006 and 2012 relative to the previous six years **(Table 3.7)**. Mammal species had more populations with stable or genuinely improved conservation status than otherwise in all biogeographic areas in the European Union except Boreal, and Marine Mediterranean (EEA, 2015a).

Remarkable recoveries due to conservation efforts include the one of the European bison, *Bison bonasus*, which was extinct in the wild after World War I and reduced to a captive population of 54 animals. Conservation efforts started in 1929 with a captive breeding programme followed by reintroductions in Białowieża National Park in Poland; Russia and several other locations in Europe. Today there are more than 2,700 wild bison, in several populations, mostly stable or increasing in numbers. Other remarkable recoveries are that of the European beaver (*Castor fiber*), the European hamster (*Cricetus cricetus*) (EEA, 2015a) and large terrestrial carnivores (Chapron *et al.*, 2014). Among the latter group, are once critically endangered large felids

Table ③ ⑦ Number of mammal species in each biogeographic area of European Union countries whose conservation status was stable, or genuinely improved or worsened between the 2001-2006 assessment period and the 2007-2012 period. Total indicates the total number of species or biogeographic region assessments of mammals in the European Union and include also assessments with non-genuine changes (e.g. because of taxonomic revisions or improved knowledge), or unknown or unreported trends. The biogeographic areas are Alpine (ALP); Atlantic (ATL); Boreal (BOR); Continental (CON); Macaronesian (MAC); Mediterranean (MED); Pannonian (PAN); Marine Atlantic (MATL); Marine Baltic (MBAL); Marine Mediterranean (MMED). No genuine changes were recorded for the Macaronesian and Marine Macaronesian regions and are not reported here. Species of the Black Sea and Steppic region were only assessed in 2012 and are excluded here. Species with non-genuine changes in assessment, are not reported here.

	ALP	ATL	BOR	CON	MAC	MED	PAN	MATL	MBAL	MMED
Stable	9	11	3	27	3	11	13	3	2	1
Improved	8	17	1	15	3	5	1	2	3	0
Worsened	14	15	5	22	1	5	13	1	1	1
Total	364	213	111	416	14	222	119	156	22	75

such as the Iberian lynx (*Lynx pardinus*) whose population tripled from 2002 (52 mature individuals) to 2012 (156), the Amur tiger (*Panthera tigris altaica*), from 20-30 animals in the 1930s to 500 and stable in 2016, and the Amur leopard (*Panthera pardus orientalis*), whose population has doubled since 2000. Among marine mammals, the Baltic seal (*Pusa hispida ssp. botnica*) rebounded from 3,000 individuals in the 1970s affected by hunting pressure and impaired fertility due to organochlorine pollution, to over 25,000 today thanks to hunting regulations afforded by the European Union and national legislations, habitat protection and improved water quality (Härkönen, 2015).

Drivers of change

National and international legislation affording legal protection and law enforcement are the main drivers of large carnivore recoveries in Western and Central Europe (Chapron *et al.*, 2014). Habitat protection and law-enforcement by government and non-government agencies are the main drivers in Eastern Europe (Government of the Russian Federation, 2015).

The main threats to terrestrial mammal species in the region are land-use change (including changes to intense cropland and pastures, logging, and extractive activities), affecting 186 species; followed by hunting and trapping, affecting 123 species; and invasive species, affecting 73 species; it should be noted that these threats are not mutually exclusive (Joppa *et al.*, 2016). Nearly all marine mammals are impacted by persistent organic pollutants, especially polychrorinated byphenils (PCBs), despite being banned by the Stockholm Convention in 2004, their concentrations in sediments and in the marine food-chains have remained high, due to low compliance to the Convention requirements of safe storage and elimination of PCB stockpile and limited decontamination of sediments, landfills, building and equipment (Stuart-Smith & Jepson, 2017). As a result, high PCB concentrations in European cetaceans from 1990 to 2012 were associated with long-term population declines and low or zero rates of reproduction, consistent with severe PCB-induced population-level effect (Jepson *et al.*, 2016). Climate change is an emergent threat for mammals that is potentially overlooked in the region (Pacifici *et al.*, 2015 **Table 3.11**).

3.4.4 Reptiles

Status and trends

Reptile species richness across the region follows a latitudinal gradient. It is highest in southern Turkey and along the eastern Mediterranean coast to Israel, with further hotspots in parts of the Iberian peninsula and southern France, the Balkans, southern Transcaucasia,

the southern deserts of Central Asia and southern and far east Russia (**Figure 3.47**, Sillero *et al*., 2014; Roll *et al*., 2017). At the subregional level, species richness is highest across Western Europe, with 213 species recorded and 212 assessed **(Table 3.8)**. This is due to the subregion combining separate faunas: the Macaronesian fauna of Portugal and Spain, the western Mediterranean fauna and the fauna of the eastern Mediterranean of Israel.

Species richness of small-range endemics is highest in the Caucasus, southern Balkan Peninsula, central and southern Iberian Peninsula, southern Turkey, and southern Central Asia. There are also a number of important refugia, i.e. places supporting a relict population of a previously more widespread species. These are both mesophyllic (Caucasian Black Sea coast of Russia, Georgia, Turkey & Southeast Azerbaijan & southern Far East Russia; Tuniyev, 1990, 1997) and xerophyllic (Spain, Portugal, Italy, Greece, Turkey, Armenia, Azerbaijan, Georgia, Russia and Central Asia; Tuniyev, 1995).

Areas of high diversity at the level of genera and families are: the Balkan Peninsula for turtles; south Turkey and Kopet Dag for skinks; Central Asia for agamas; south Mediterranean and southern Central Asia for geckos; the Caucasus, southern Balkan Peninsula and Iberian Peninsula, Mediterranean and Aegean Sea islands for lacertids; southern Central Asia for boas; and the Caucasus and north-east Turkey for vipers.

In this assessment we compiled a dataset of all 408 extant species of reptiles occurring in Europe and Central Asia from the Reptile Database (Uetz, 2017) and IUCN Red List of Threatened Species (IUCN, 2017c). Of these, 289 have published assessments of extinction risk on the IUCN Red List. Sixty-three species are assessed as threatened with extinction (categories vulnerable, endangered and critically endangered; **Table 3.8**). Thus between 21.7% (assuming that no data deficient species are threatened with extinction) and 26.6% (assuming that all data deficient species are threatened with extinction) of species within the region are threatened with extinction. Best estimates of extinction threat generally assume that data deficient species fall into non-data deficient categories in the same proportions as non-data deficient species (IUCN, 2017a), indicating here that about 22.8% of reptile species in Europe and Central Asia are threatened with extinction. This level of threat is similar to the one of reptiles globally (18.8% - Böhm *et al.*, 2013) and across Europe (Western, Central and Eastern Europe, including the Russian Federation up to the Urals and excluding the Caucasus) (19.7% - Cox & Temple, 2009; χ^2=2.31, df=2, p=0.315). However, recent studies suggest that globally data deficient reptiles are neither widespread nor common, suggesting there may be an underestimation of extinction risk (Meiri *et al.*, 2018).

Extinction threat is lowest for snakes and highest for turtles and tortoises **(Table 3.8)** which is comparable to global patterns (Böhm *et al.*, 2013). Extinction risk across all assessed species is highest in Western Europe. More than one third of reptiles endemic to Europe and Central Asia subregions are at risk of extinction and this threat is highest across Central Asia **(Table 3.8)**. Not all species have been assessed yet for the IUCN Red List, however, there are a number of ongoing assessments.

One Canary Island endemic, *Gallotia auaritae,* is listed as possibly extinct or likely extinct (Martin, 2009; Mateo Miras & Martínez-Solano, 2009). There is evidence for at least two extinctions from Europe and Central Asia: the Persian toad agame *Phrynocephalus persicus* is thought to have gone extinct from Azerbaijan and now solely exists outside Europe and Central Asia in Iran (Anderson *et al.*, 2009). In Israel, the Nile crocodile (*Crocodylus niloticus*) was lost in the early 20th century, probably due to hunting (Dolev Pervolutzki, 2004;

Masterman, 1921). Of the more speciose genera, those with most threatened species include the narrow-endemic vipers (genus *Vipera sensu lato*, 22 species, 45% threatened, six not evaluated), toad-headed agamas of Eastern Europe and Central Asia (genus *Phrynocephalus*, 13 species, 31% threatened, three not evaluated); species of mostly Mediterranean wall lizards, often very common, but with small ranges (genus *Podarcis*, 23 species, 30% threatened, three not evaluated); and the Caucasian – Asia Minor rock lizards (genus *Darevskia*, 26 species, 23% threatened, three not evaluated).

Compared with data on extinction risk, data on reptile population trends are sparse. Deriving trends from IUCN Red List data is difficult since not all species have yet been assessed and many have only ever been assessed once. Only one species has a documented change in global extinction risk, the globally distributed leatherback sea turtle, *Dermochelys coriaceae*, critically endangered in 2004

Table ③ ⑧ **Global IUCN Red List status of reptiles occurring within the Europe and Central Asia assessment region, for species with a published assessment (Total = 289). N is the number of species recorded in the assessment region. IUCN categories: DD: data deficient; LC: least concern; NT: near threatened; VU: vulnerable; EN: endangered; CR: critically endangered. Source: IUCN (2017c).**

Group	DD	LC	NT	VU	EN	CR	Total	N	% threatened	% lower bound	% upper bound
Reptiles	14	186	26	21	25	17[1]	289	*408*	22.9	21.8	26.6
Lizards	6	120	17	13	16	12[1]	184	*246*	23.0	22.3	25.5
Snakes	7	63	7	4	7	4	92	*141*	17.7	16.3	24.0
Amphisbaenians	0	2	0	0	0	0	2	*5*	-	-	-
Turtles/tortoises	1	1	2	4	2	1	11	*16*	70.0	63.6	72.7
By region											
Western Europe	3	105	18	11	14	9[1]	160	*212*	21.7	21.3	23.1
Central Europe	4	88	11	6	9	5	123	*156*	16.8	16.3	19.5
Eastern Europe	2	55	9	6	4	2	78	*119*	15.8	15.4	18.0
Central Asia	5	51	1	4	2	3	66	*109*	14.5	13.4	20.9
Endemics											
Endemic ECA	7	52	17	11	17	14[1]	118	*145*	37.8	35.6	41.5
Western Europe[2]	2	25	11	4	8	6[1]	56	*71*	33.3	32.1	35.7
Central Europe[2,3]	3	16	5	2	6	4	36	*45*	36.4	33.3	41.7
Eastern Europe[3,4]	0	11	5	3	4	2	25	*28*	36.0	36.0	36.0
Central Asia[4]	3	9	0	3	1	3	18	*21*	46.7	38.9	55.6

1. *Gallotia auaritae*, endemic to the Canary Islands, is listed as critically endangered (possibly extinct)
2. Four species endemic to Western Europe and Central Europe
3. Eleven species endemic to Central and Eastern Europe
4. Two endemic species shared between Eastern Europe and Central Asia

and vulnerable in 2013. Of the 289 species with published IUCN Red List assessments, 98 species show declining populations and only five show an increasing trend across their global range: three least concern species (*Cyrtopodion scabrum, Hemidactylus turcicus, Podarcis siculus*) and two critically endangered species of the Canary Island endemic *Gallotia* (*Gallotia bravoana*, *G. intermedia*), which have been subject to conservation action (control of predators). Populations for 119 species are considered stable, and the status of the remaining 61 is unknown.

The Living Planet database currently contains 66 population time series representing 23 species of reptiles for Europe and Central Asia (LPI, 2016). These are exclusively from Western and Central Europe (49 and 17 time series, representing 22 and three species, respectively). Most Central European population time series focus on marine turtles in Turkey and Cyprus. In Western Europe, data are also available for snakes and lizards. The loggerhead turtle (*Caretta caretta*) is increasing across available time series, while the few time series available for *Testudo hermanni* (not threatened on the IUCN Red List), three species of vipers (*Vipera aspis*, least concern; *V. berus*, not evaluated; *V. ursinii*, vulnerable) and *Hierophis viridiflavus* indicate declining population trajectories. Increases in sea turtle populations have been noted in other parts of the eastern Mediterranean too, for example in Israel (Casale & Margaritoulis, 2010).

Other data sources suggest declines for *Testudo kleinmanni* in Israel, the only country in Europe and Central Asia where this species is thought to occur (Dolev Pervolutzki, 2004). There is also direct evidence from the literature that some snake populations are in decline in specific Western European localities (e.g., UK: *Coronella austriaca*; Italy: *Vipera aspis, Vipera ursinii;* France: *Vipera aspis, Hierophis viridiflavus, Zamenis longissimus*; Reading *et al.*, 2010).

Annexes II, IV and V of the European Union Habitats Directive list 91 reptile species and 7 subspecies. Most species were only assessed once for the European Union Habitats Directive or did not have enough information for a conclusive definition of their status. In many cases it is therefore not possible to determine a trend **(Table 3.9)**. Only few genuine changes in conservation status were recorded between the two reporting periods. However, only one species and one subspecies were recorded to have a worsening status between the two assessment periods of 2001-2006 and 2007-2012: *Podarcis lilfordi* in the Mediterranean, though in places this species is still very common; and *Lacerta vivipara pannonica*. In terms of spatial planning, however, a recent study suggests that the Natura 2000 network mostly covers widespread reptile species, while narrow-range endemics are under-represented in Natura 2000 and national protected area networks (Abellán & Sánchez-Fernández, 2015).

Drivers of change

The main threats to reptiles in Europe and Central Asia, according to the IUCN Red List, are agriculture, residential/commercial development, and biological resource use **(Figure 3.48)**. These threats primarily cause habitat fragmentation and loss.

The major threat of habitat loss affects in particular relic forest species, and species of the steppe and semi-desert ecosystems, which are often not able to persist on agricultural and other transformed lands. *Eremias pleskei* (Armenia, Azerbaijan, Turkey and Iran) is listed as critically endangered, based on a population decline of more than 80% over ten years. Its natural sandy habitat has virtually disappeared due to human disturbance (Tuniyev *et al.*, 2009). For habitat specialists, such as the critically endangered *Phrynocephalus horvathi* and *Acanthodactylus*

Table **3** **9** **Number of reptile species in each biogeographic area of European Union countries whose conservation status was stable, or genuinely improved or worsened between the 2001-2006 assessment period and the 2007-2012 period. The biogeographic areas are Alpine (ALP); Atlantic (ATL); Boreal (BOR); Continental (CON); Mediterranean (MED); Pannonian (PAN); Marine Atlantic (MATL); Marine Mediterranean (MMED). Species of the Black Sea and Steppic area were only assessed in 2012 and are excluded here. Non-genuine changes were mainly due to taxonomic revisions or improved knowledge.**

	ALP	ATL	BOR	CON	MED	PAN	MATL	MMED
Stable	2	4	1	10	4	5	0	2
Improved	2	2	0	0	0	1	0	1
Worsened	5	2	3	0	1	2	0	1
Non-genuine changes /Unknown/ Not Assessed	86	34	10	112	128	34	22	22

Figure **3 48** **Main threats affecting reptiles in Europe and Central Asia according to species assessments published in the IUCN Red List (all species, grey; endemic species, orange). Source: IUCN (2017c).**
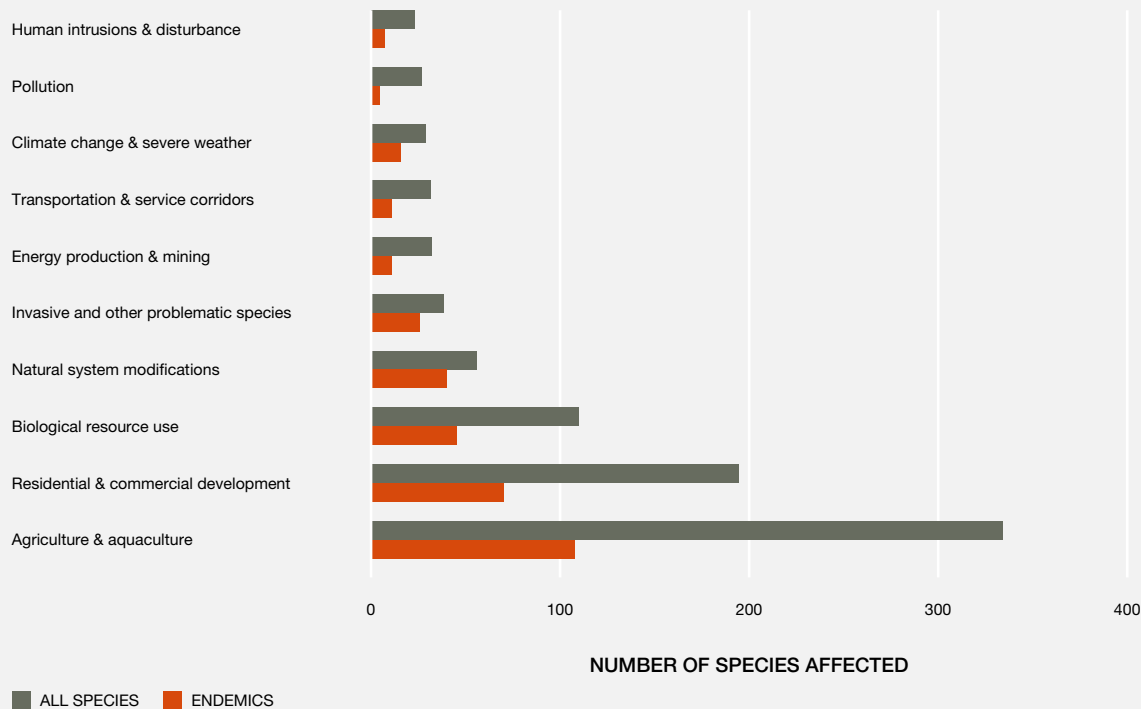
beershebensis which are found on highly specific soils, habitat conversion can have a major impact (Ananjeva & Agasyan, 2009; Werner et al., 2006). The disappearance of steppe vipers of the "ursinii-renardi" complex throughout most of the previously occupied habitats in Europe and Central Asia is associated with ploughing of steppes for agriculture (Tuniyev, 2016). Dam building has been detrimental to species such as Rafetus euphraticus in Turkey, causing drastic habitat alteration (Taskavak et al., 2016).

Significant threats include the illegal capture of commercially valuable species for the pet trade (all representatives of the vipers and turtles, and some species of lizards) in Turkey, the Caucasus and Central Asia. Trionyx triunguis softshell turtles have been reported as bycatch and have been killed, and nests destroyed, by fishermen who may perceive them as competitors; they are also affected by pollution, resulting in a listing of the Mediterranean subpopulation in Israel, Lebanon, Syria and Turkey as critically endangered (European Reptile & Amphibian Specialist Group, 1996). There are also reports of reptile poaching in Israel, which affects species such as Uromastyx aegypticus (Yom-Tov, 2003). Prosecution of snakes continues in the area, especially in Turkey, the Caucasus and southern regions of Russia, and is associated with low levels of environmental education.

Invasive predator species play a particularly important role for island species, such as the Canary Island genus Gallotia (four of the eight species are critically endangered). Climate change is likely to play a major role in the region in the future. Climate change has led to an increase in summer temperatures and length of the dry summer period in the western Caucasus, resulting in a reduction of habitats of mesophylic Colchis reptile species (Darevskia derjugini) and an increase in the number of eastern Mediterranean snakes (Hierophis caspius, Platiceps najadum) on the Black Sea Coast (Tuniyev, 2012).

Other threats, such as pollution, are less prominent in the IUCN Red List data; however, a recent risk evaluation of pesticide use to protected European reptiles suggests that ten species, including all six Habitats Directive Annex II turtles, are at above-average pesticide risk (Wagner et al., 2015).

3.4.5 Amphibians

Status and trends

Europe and Central Asia is highly diverse with, for example, thirty-five percent of the world's newt and salamander species (26 species of the family Salamandridae) present

in Europe, extending from Iceland in the west to the Urals in the east and from Franz Josef Land in the north to the Mediterranean in the south.

A total of 74 amphibian species are known in Western, Central and Eastern Europe, with the highest numbers occurring in France, Italy, Spain and the Balcans (20-30 species each) (Corbett, 1989). Fifty-nine percent of amphibian species (Temple & Cox, 2009) have declining populations. In the western Palearctic (i.e. European region and part of Asia with Turkey and the Caucasian region), species richness decreases with increasing latitude for amphibians and reptiles (Meliadou & Troumbis, 1997).

Amphibians represent the third most endangered group of vertebrates in the European Union, with 23% of species (19 species out of the 83 assessed) considered as threatened (Temple & Cox, 2009) **(Figure 3.49)**. According to the Habitats Directive, more than two-thirds of the amphibian species assessed by European Union countries by biogeographical region (104) have an unfavourable conservation status. About 59% of European amphibian
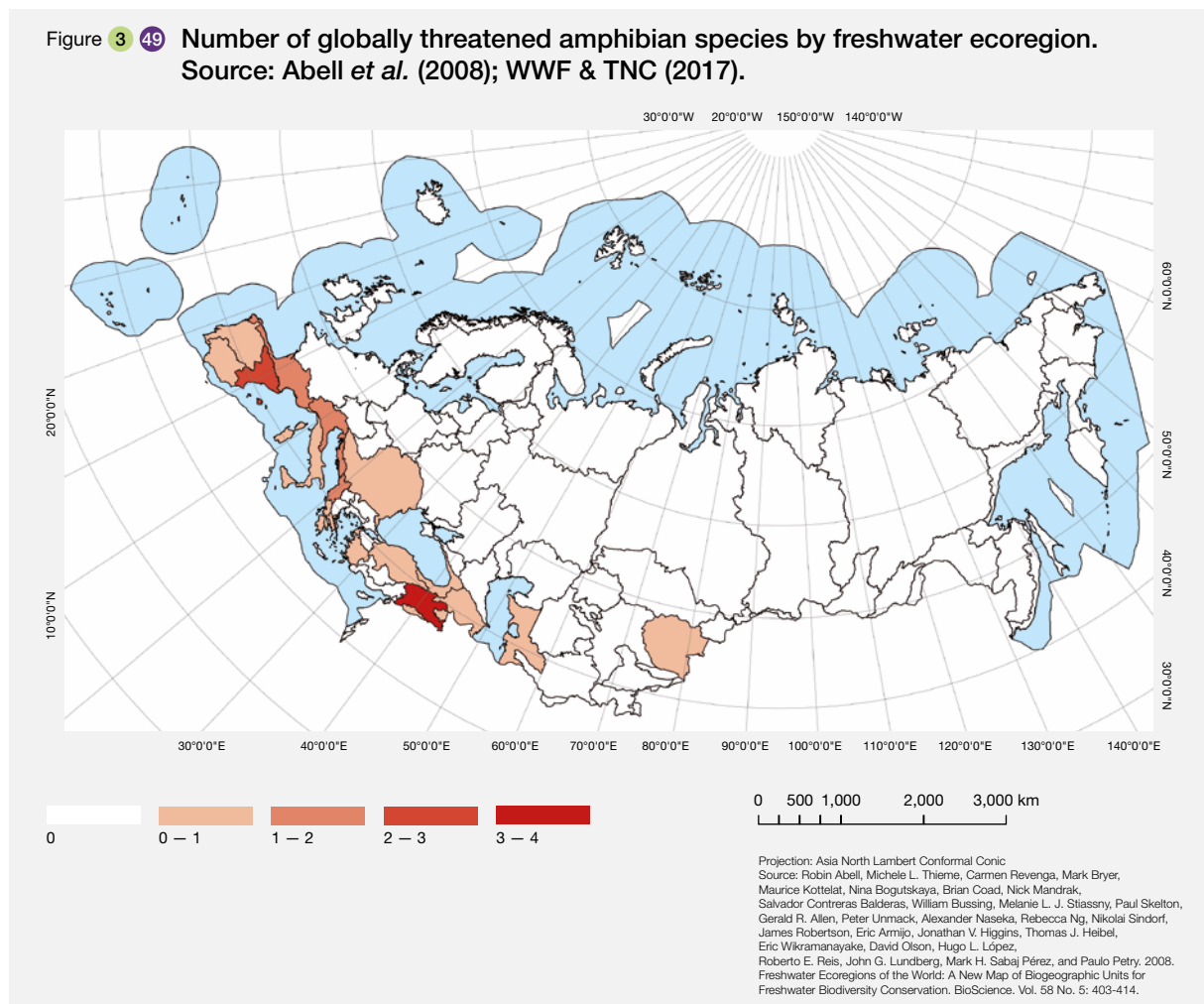
populations are declining with a further 36% stable and only 2% on the increase. These declines seem to have worsened over the past 25 years and amphibians are now more threatened than either mammals or birds (Beebee & Griffiths, 2005).

The recent Red list of European amphibians (Temple & Cox, 2009) has highlighted that about 23% of European amphibians (85 species in total) are threatened and show declining populations. This is even more significant given that 74% of European amphibians are endemic (only found in Europe) and that these endemic species tend to be more threatened within Europe.

Drivers of change

The three main causes for amphibians decline in the region are: 1) that fewer habitats available for these species, and what remains is often in small and isolated patches; much of the habitat has become less suitable through destruction or transformation, e.g. urbanization with roads, drainage and water pollution (Hamer & Mcdonnell, 2008) and with the

Figure **3** **49** **Number of globally threatened amphibian species by freshwater ecoregion. Source: Abell *et al.* (2008); WWF & TNC (2017).**

Projection: Asia North Lambert Conformal Conic
Source: Robin Abell, Michele L. Thieme, Carmen Revenga, Mark Bryer, Maurice Kottelat, Nina Bogutskaya, Brian Coad, Nick Mandrak, Salvador Contreras Balderas, William Bussing, Melanie L. J. Stiassny, Paul Skelton, Gerald R. Allen, Peter Unmack, Alexander Naseka, Rebecca Ng, Nikolai Sindorf, James Robertson, Eric Armijo, Jonathan V. Higgins, Thomas J. Heibel, Eric Wikramanayake, David Olson, Hugo L. López, Roberto E. Reis, John G. Lundberg, Mark H. Sabaj Pérez, and Paulo Petry. 2008. Freshwater Ecoregions of the World: A New Map of Biogeographic Units for Freshwater Biodiversity Conservation. BioScience. Vol. 58 No. 5: 403-414.

loss of areas managed by traditional means (Hartel *et al.*, 2010), more intense fish farming and recreational activities; 2) Climate changes, which threaten species particularly in areas where water and humid habitats are already scarce and expected to become even drier (Araújo *et al.*, 2006); 3) Introduction of alien species, including the chytrid fungus, which is a particularly virulent disease affecting the skin and nervous system of adult amphibians and the mouthparts of their larvae, and responsible for amphibian declines worldwide (fatal for many species) (Duffus & Cunningham, 2010; European Commission, 2009). These three factors may also interact to exacerbate each other. In addition, there is rising concern that the impact of pesticides on amphibians has been underestimated and that pesticides could locally be a cause of amphibian population declines (Brühl, *et al.* 2013). While amphibians are generally declining, in the absence of the above mentioned drivers they can be well represented in traditionally managed landscapes by stable populations and species rich communities (Hartel *et al.*, 2010).

3.4.6 Fishes

3.4.6.1 Marine fishes

Status and trends

There are considerably more species of fish in all marine areas surrounding Europe and Central Asia than those known to consumers from markets. For example, reported species richness is around 100 in the Caspian Sea (Mitrofanov & Mamilov, 2015), 833 in the Far Eastern seas of Russia (Volvenko, 2014), 650 in the Mediterranean Sea (United Nations, 2016), 200 in the Black Sea (Bologa & Sava, 2012), and 100 in the Baltic Sea (HELCOM, 2009). Species richness tends to be comparatively higher in coastal areas, along the continental slope, and towards the south **(Figure 3.50)**. Due to the high mobility of fish and the open nature of marine waters, there are intense, complex, competitive interactions within fish communities, which naturally leads to large differences in the population biomasses of different species (Fung *et al.*, 2013).

In Europe and Central Asia, 26% of marine fish species have known trend data. Of those, 72% are stable, 26% have declining populations and 2% have been increasing over the last decade (IUCN, 2017c). In a comprehensive assessment of threats to European marine fish species, Nieto *et al.* (2015) found that 59 species (7.5%) were threatened. All 15 critically endangered species amongst these are Chondrichthyes (sharks, rays, and similar). The low resilience of these organisms is due to their life-history traits (slow reproduction and small number of offspring). Indeed, poor conservation status is most common for Chondrichthyes and other species with large body size,

which also infers slow reproductive rates (Fernandes *et al.*, 2017b). Among the largest species, many migrate over large distances. Of species with assessed stock, including those considered overfished, Fernandes *et al.* (2017) found only a small proportion to be threatened. Considering trends in the sizes of species populations, 8.4% were found to be declining, mainly due to overfishing, but also coastal development, energy production and mining, and pollution. Increasing trends were found for 1.7% of populations. For about 69% of marine fish species data for European Union waters is insufficient to estimate trends (Nieto *et al.*, 2015).

Good data on trends is available for the North East Atlantic shelf seas, which permits application of trend analyses that take into account that fish populations can naturally fluctuate over wide ranges (Greenstreet *et al.*, 2012). These reveal recovery of a statistically significant number of fish species classed as sensitive (based on their recruitment pattern) in the Celtic Sea, but not yet in the North Sea (OSPAR, 2017b). Yet, in both of these highly fished areas the number of recovering species has increased over time (OSPAR, 2017b) as a result of changes in fisheries management.

Considering the strong relationship between conservation status and body size (Fernandes *et al.*, 2017) and the slow recovery dynamic of overall fish community size structure (Fung *et al.*, 2013), the state of marine fish communities can be assessed based on the "typical length" (Lynam & Rossberg, 2017) of fish caught in surveys. Using this measure, OSPAR (2017b) showed that demersal fish communities continue to deteriorate in some parts of North East Atlantic shelf, e.g. in the southern parts of the North Sea and along the continental slope **(Figure 3.51)**, while in other areas recovery can be observed. This illustrates the surprisingly localized impact of varying exploitation patterns on the status of marine fish communities. For pelagic fish communities, trends in either direction tend to be less apparent (OSPAR, 2017b). For the Baltic Sea, good status of piscivores and of cyprinids/mesopredators (in terms of total biomass) is reported by (HELCOM, 2017a).

For status and trends of fish biodiversity in the Mediterranean Sea, indigenous and local knowledge offers important information that is unavailable from scientific surveys. Combined survey data and interviews with local fishermen in the Spanish Mediterranean Sea and Gulf of Cadiz, Coll *et al.* (2014) documented overall declines in abundances and maximum sizes of fish. Potential extirpations, notably of Chondrichthyes, were reported as well. Small fish were reported to have proliferated, potentially due to a trophic cascade effect. A meta-analysis by Vasilakopoulos *et al.* (2014) of 42 stocks of nine species in 1990–2010 covering the entire European Mediterranean and Black Seas comes to similar conclusions: exploitation rates have been increasing, and stocks are shrinking and are being harvested too early in their lifecycle. In the Black

Figure **3 50** Species richness of European marine fishes. Source: Nieto *et al.* (2015).
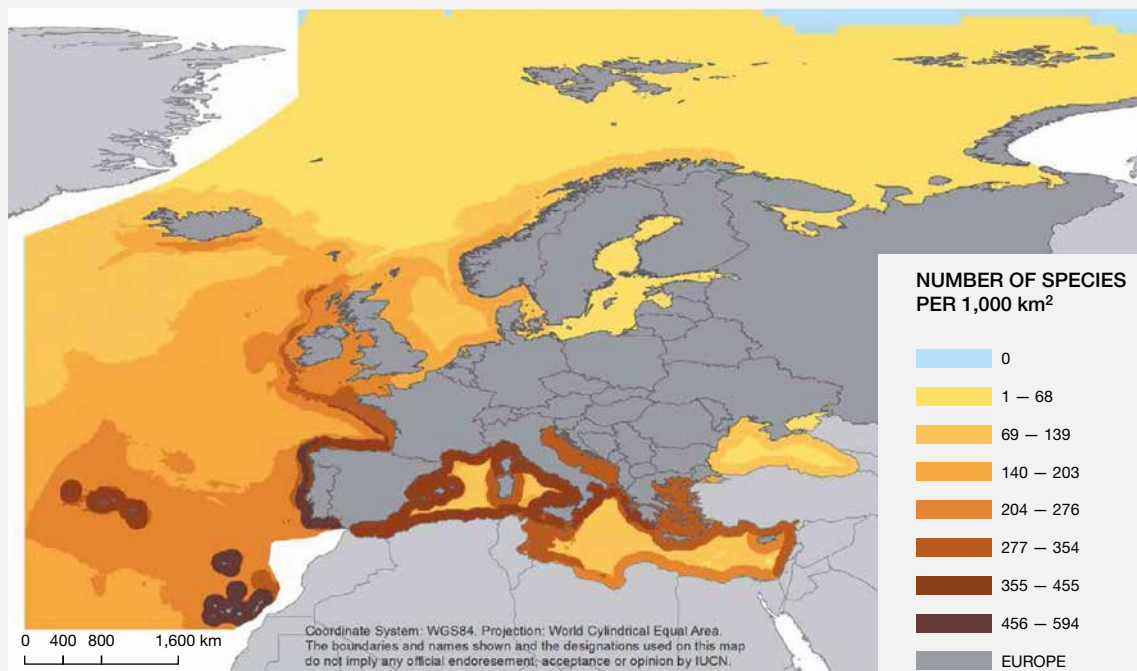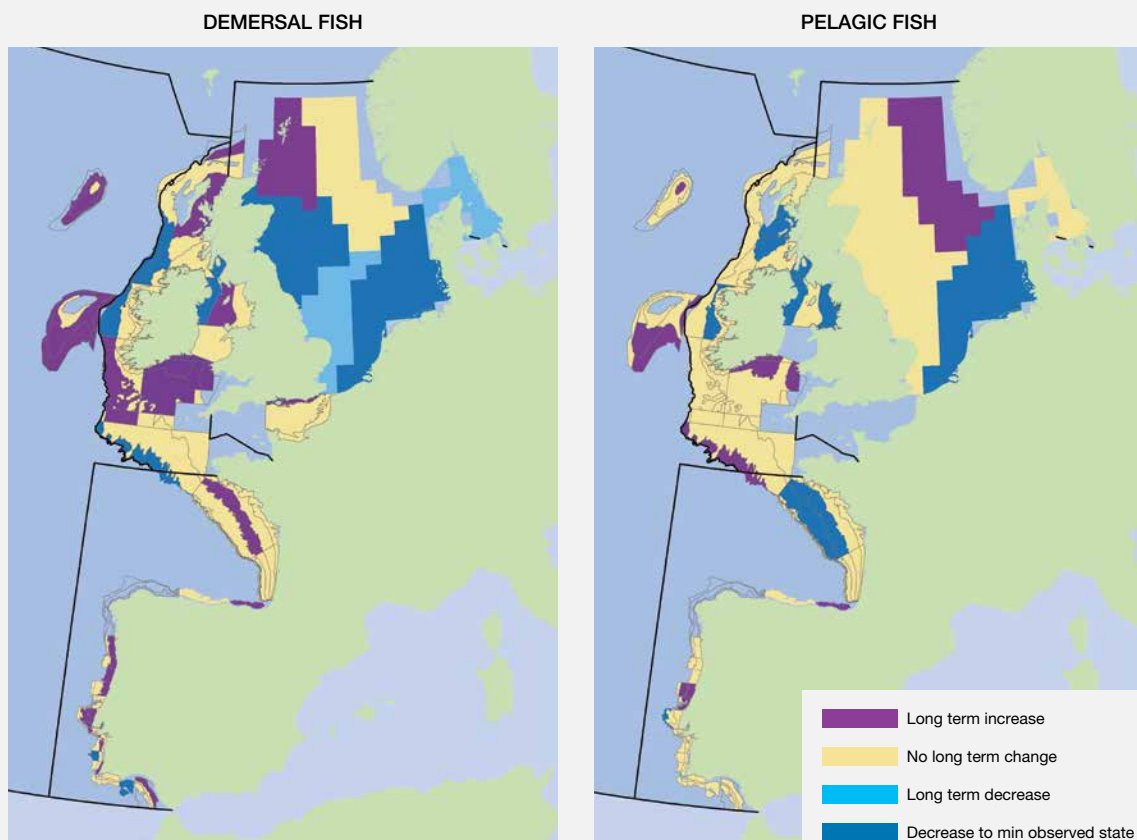

Figure **3 51** Spatial pattern of trends in the "typical length" of fish caught in surveys. Source: OSPAR (2017).

Sea, two sturgeon species were recently declared extinct (Yankova *et al.*, 2014).

For the North-West Pacific a digital database covering the years 1977-2010 is available (Volvenko, 2014), but coverage has been argued not to be sufficient even to reveal specific trends. In data from pelagic trawl surveys, Ivanov and Sukhanov (2015) document a pronounced decline of pelagic fish biomass and diversity in the Russian Waters of Far Eastern Seas from the 1980s to the mid 1990s, and a pronounced recovery in the subsequent period until 2009, without providing a clear attribution.

Drivers of change

Overfishing is still the main threat to marine fish across Europe and Central Asia. Throughout the region, the expansion of industrial fishing after the Second World War and the resulting over-exploitation of fish led to pressures on biodiversity at community level, except in the Arctic Ocean where only specific stocks appear to be affected (CAFF, 2013). However, during the last few decades changes in management practices have led to improvement in the status of stocks and release of pressures on fish-communities as a whole, especially throughout the North East Atlantic Shelf Seas. In other parts of Europe and Central Asia, institutional barriers to coordinated action and the relatively high costs involved in regular stock assessments have so far prevented demonstrable recovery of fish communities.

Other drivers are also responsible for the negative trend identified, especially different forms of pollution in enclosed seas (Black, Mediterranean, Baltic, Caspian and Aral Seas); coastal developments degrading and sometimes extirpating coastal habitats important as nurseries; energy production; and mining. These are sometimes exacerbated by climate change. In the Black Sea, for example, ecosystem disruptions by eutrophication and invasive species continue to impact fish communities (Bologa & Sava, 2012). In rivers feeding the Caspian and Aral Seas construction of dams has led to drastic reductions in the abundance and extinction of some migratory fish (Mitrofanov & Mamilov, 2015).

3.4.6.2 Freshwater fishes

Status and trends

The European Union contains 546 native species of freshwater fish of which, according to IUCN assessments, at least 37% are threatened and 4% are considered near threatened (Freyhof & Brooks, 2011). This is currently the second most threatened taxonomic group assessed, after freshwater molluscs. The highest diversity of fish species

can be found in the Danube River with 103 species, followed by the Volga River with 88 species **(Figure 3.52)**. Southern Europe is the region with the highest number of local endemic species, with natural ranges limited to one or few streams, springs or rivers, and several of them have only recently been discovered. They are therefore still not well known to conservationists and national or regional governments (Freyhof & Brooks, 2011). Central Asia is home to approximately 120 fish species of which 30 are on the Red List (Karimov *et al.*, 2009; Milner-Gulland *et al.*, 2006). Several fish species naturally entered the floodplains from the north (Siberia) and west (Western Asia). Many Eurasian fish species have formed sub-species in Central Asia (e.g. Amudarya trout, Aral roach, Aral asp, Samarkand khramulya, Aral bream) and contribute to high endemic diversity (e.g. Aral Sea basin) (Berg, 1949; Nikolsky, 1971; Turdakov, 1963).

There are no other groups of freshwater fishes in Europe and Central Asia that show higher threat levels than anadromous species (e.g. sturgeons, herrings of the genus *Alosa*, salmonids and some whitefishes of the genus *Coregonus* and *Stenodus*) (Freyhof & Brooks, 2011). Trends also highlight a crisis with, for example, a sixfold decline in Baltic salmon catches between 1990 and 2009 (Mannerla *et al.*, 2011).

Although these figures are at a European level and such detailed data are difficult to access for Central Asia, it is expected that these trends and the observed decline of about 17% of European freshwater fishes populations are also true in Central Asia. In Europe, only 1% of freshwater fish species populations are on the increase, against 17% declining and 6% considered stable (Freyhof & Brooks, 2011). However, there is a lack of reliable data on trends, and therefore the actual percentage of species that is declining is probably largely underestimated. In fact, population trends for 76% of all fish species in Western Europe, Central Europe and western parts of Eastern Europe still remain unknown because almost no population trend data exist from most countries (Freyhof & Brooks, 2011). Thus, monitoring data for freshwater fish species diversity and abundance is urgently needed in order to accurately measure population trends and improve the accuracy of future Red List assessments. The highest number of threatened freshwater fish species is found in the south of the European subregions **Figure 3.53**).

Villéger and co-authors (2014) have also shown that among current European fish assemblages, functional homogenization (reduction in diversity of functional traits over space and time) exceeds taxonomic homogenization (reduction in species diversity) six-fold. In addition, non-native species originating from other parts of Europe played a stronger role in this homogenization process than non-native species from outside Europe, while extinction did not play a significant role.

Figure **3** **52** **Distribution of freshwater fish species richness across Europe and Central Asia. Source: Abell *et al.* (2008); WWF & TNC (2017).**

NUMBER OF FISH SPECIES BY FRESHWATER ECOREGION

1 — 9   10 — 19   20 — 27   28 — 33   34 — 39

40 — 46   47 — 56   57 — 68   69 — 83   84 — 114

0   500 1,000   2,000   3,000 km

Projection: Asia North Lambert Conformal Conic
Source: Abell, R., M.L. Thieme, C. Revenga, M. Bryer, M. Kottelat, N. Bogutskaya, B. Coad, N. Mandrak, S. Contreras Balderas, W. Bussing, M.L.J. Stiassny, P. Skelton, G. R. Allen, P. Unmack, A. Naseka, R. Ng, N. Sindorf, J. Robertson, E. Armijo, J.V. Higgins, T.J. Heibel, E. Wikramanayake, D. Olson, H.L. López, R.E. Reis, J.G. Lundberg, M.H. Sabaj Pérez, and P. Petry. 2008.
"Freshwater Ecoregions of the World: A New Map of Biogeographic Units for Freshwater Biodiversity Conservation."
BioScience 58: 403-414. Digital media at: www.feow.org

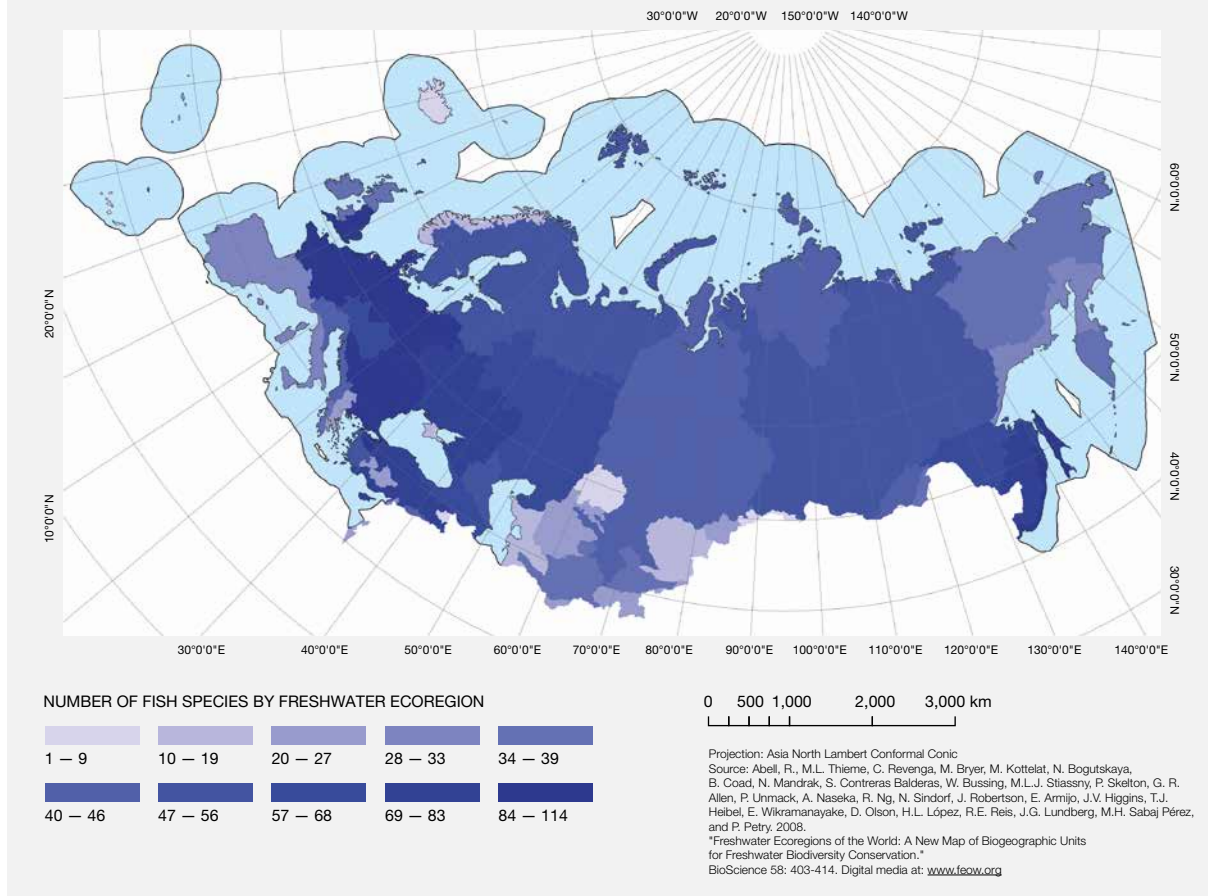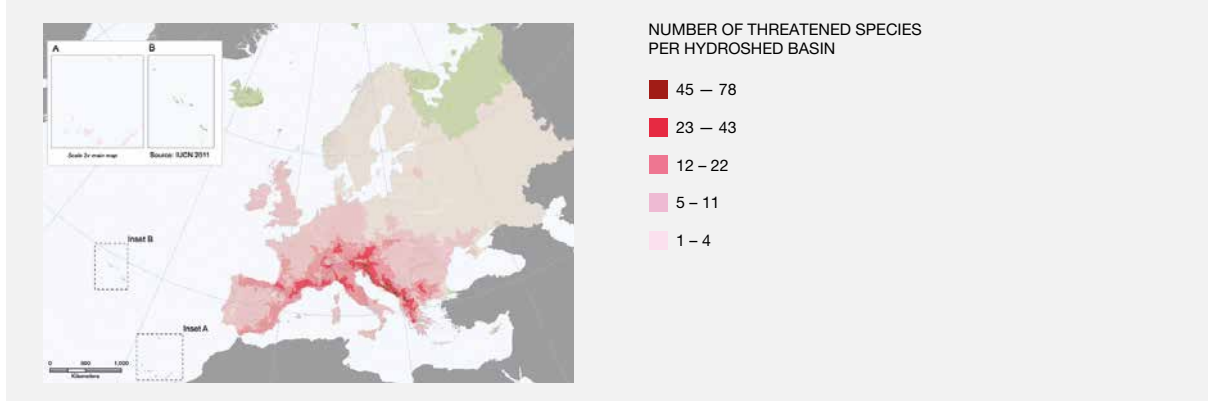Figure **3** **53** **Distribution of threatened freshwater fish in Europe (as defined by the European Environment Agency, including Russia up to the Urals). Source: Freyhof & Brooks (2011).**

NUMBER OF THREATENED SPECIES
PER HYDROSHED BASIN

45 — 78
23 — 43
12 – 22
5 – 11
1 – 4

Drivers of change

A main threat for freshwater fish species in Europe and Central Asia is the destruction or modification of their habitat. This includes a change in the river continuum with the construction of dams and weirs that fragment populations. This has direct consequences for the

remixing of upstream-downstream genetic pools and for free seasonal migrations. In addition, it leads to a deep modification of flow patterns transforming lotic habitat into lentic ones and, as a result, changing species assemblages, functional diversity and homogenization of freshwater fish communities. Water abstraction is one of the most

important threats to European freshwater fishes, especially in the Mediterranean basin where illegal water abstraction is widespread (Freyhof & Brooks, 2011). Many countries in southern parts of Western and Central Europe still lack effective enforcement of legislation that could limit the damages of excessive water abstraction to biodiversity. The increased frequency and intensity of droughts are worsening the situation.

Another important threat is pollution of industrial, agricultural and domestic origin (e.g. hormone distruptors from polimery and paint industries that cause reproductive disorders, in particularly in aquatic organisms). In lakes, for example, the percentage of land used for agriculture in the catchment (which leads to anthropogenically enhanced productivity) is associated with several changes in fish communities such as increase in species richness and abundance and a decrease in their community average body size (Brucet *et al.*, 2013). At least eight of the 13 globally extinct species of European freshwater fishes were victims of water pollution and lake eutrophication, mainly during the late 19th and in the 20th centuries (Freyhof & Brooks, 2011). However, due to European Union regulation, the water quality of rivers and lakes has improved in recent decades and this has helped to improve conditions for many fish species. In Central Asia and the Caucasus, however, about one third of untreated sewage goes directly into regional rivers. Pollution as a result of change in land use is still relevant in these regions, in particular the increase in siltation due to agricultural practice and destruction of riparian vegetation, which used to act as an important buffer zone to freshwater ecosystems.

Climate change is also affecting fish populations, particularly in the coldest and the most arid parts of Europe and Central Asia. Jeppesen and co-authors (2012) published long-term (10–100 years) series of fish data from 24 European lakes. Along with a temperature increase of about 0.15–0.3°C per decade, considerable changes have occurred in either fish assemblage composition, body size or age structure during recent decades, with a shift towards dominance of warm water species. These changes took place despite a general reduction in nutrient loading. Similar responses to warming were found in river fish (Daufresne *et al.*, 2009). Arctic charr has been particularly affected. In the arid conditions of Central Asia, agriculture relied on the extensive use of irrigation. From the 1950s to the 1980s, about 40 reservoirs (total water volume more than 57 km³), more than 150,000 irrigation canals, more than 100,000 drainage canals and 10 lakes for residual water storage (with an area of about 7,000 km²) were created. These large-scale constructions impacted local fish communities. Dams on the rivers blocked passage to spawning areas for migratory fishes. As a result, fringebarbel, sturgeon and Aral barbell vanished from local waters. All fish populations in the floodplain (such as common carp, asp, sabrefish,

bream, roach, pike-perch) have established new stocks in all newly constructed man-made reservoirs and lakes. Also, the abundance of riverine fishes such as shovelnoses (three species), pike-asp, zarafshon dace and minnow dramatically decreased due to a change in flow and a reduction of turbidity in the river sections downstream of the reservoirs (Berg, 1949; Kamilov, 1973; Nikolsky, 1938; Turdakov, 1963).

Another key threat in Central Asia is water salinization (Jeppesen *et al.*, 2015). For example, in the three decades from 1961 to 1991 the Aral Sea's salt concentration increased from 10.2 ppt to 35 ppt (Pavlovskaya, 1995). Freshwater fishes cannot adapt to these levels of salinity and many therefore became extinct. The discharge of drainage waters from irrigated fields and industries has also led to salinization and chemical pollution of rivers. Parts of many Central Asian rivers have been contaminated by phenols, oil products, heavy metals, pesticides and nitrogen compounds (Pavlovskaya, 1995).

In recent years there have been many examples of alien pathogen and parasite introductions in Europe and Central Asia and their dramatic effects on aquatic wildlife and biodiversity, with several having a direct impact on fish biodiversity and ecosystem services (Peeler *et al.*, 2011). For example, *Anguillicola crassus*, a parasitic nematode, directly impacted wild populations of the European eel, *Anguilla anguilla*. The most severe of all, identified in the last decade as a major threat to European fish diversity (Gozlan *et al.*, 2005), is the rosette agent, a generalist fungal-like pathogen introduced along with the Asian gudgeon (*Pseudorasbora parva*) and responsible for the rapid decline of endemic fish species across Europe and Central Asia. This pathogen and its host have caused the decline and extinction of native population across Europe - some of them endemic or not yet even described. Most of these introductions across the region occured via the aquaculture trade, fisheries or ornamental purposes (Boll *et al.*, 2016; Gozlan, 2016).

3.4.7 Terrestrial Invertebrates

Status and trends

The diversity of terrestrial invertebrates in Europe and Central Asia is unevenly explored, with a substantial lack of knowledge for most taxa, especially for below-ground (soil) fauna. Above-ground terrestrial invertebrates are generally better known, with described insect species numbering in the order of 100,000 in Europe[9], about 80,000 for Kazakhstan (The Fifth National Report on Progress in

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9. The countries included in this checklist are listed here http://insectoid.info/checklist/insecta/europe/

Implementation of the Convention on Biological Diversity 2014), and about 30,000 for the Russian Far East (Lelej & Storozhenko, 2010). Scientific knowledge of certain groups is rapidly increasing. For example, the number of described fly (Diptera) species in the Palearctic was 29,579 according to a catalogue published in 1992 (Soós *et al.*, 1992) and increased to 44,894 in 2009 (Pape *et al.*, 2009), an increase of about 15,000 species new to science or to the region. Heteroptera species numbered 9,365 in 2006, an almost 10% increase compared to 1995 (Aukema *et al.*, 2013). Bumblebee species numbers increased from 23 to 33 in the 170,500 km^2 large Tuva Republic (Russia) based on a survey in 2013 (Kupianskaya *et al.*, 2014). For several speciose taxa, there is no information even on species presence, even though some of these include taxa with extreme importance for ecosystem functions, like Hymenoptera (with many parasitoid species), or most soil organisms, contributing to biological control and pollination, or soil fertility, respectively. Despite their extremely high species richness, and importance for ecosystems services, only a very small proportion of species is assessed by the IUCN Red List **(Table 3.10)**.

Trends are known for certain groups, such as butterflies. Major declines of butterfly populations occurred in the 1950s-1970s due to agricultural intensification in Western Europe but one third of species are still declining (van

Swaay *et al.*, 2010). Bees (honeybees and wild bees including bumblebees) have been recently evaluated as pollinators by IPBES (2016b). Many wild bee species have been declining in Western Europe. For example, 50% of bee species are threatened in some European countries, while data for other regions are currently insufficient to draw conclusions (IPBES, 2016b). Better taxonomic coverage extists for terrestrial invertebrates of community interest according to the Habitats Directive and monitored throughout the European Union. One quarter of these species (arthropods, molluscs and others) have deteriorating conservation status (EEA, 2015d). A recent meta-analyses found a 77% decline in flying insect biomass across 63 protected sites in Germany from 1987 to 2016, likely due to agricultural intensification in the surrounding fields, with protected sites therefore acting as ecological traps (Hallmann *et al.*, 2017). This analysis suggests that the extent of insect decline in Europe has been greatly underestimated.

In Europe alone, the update of the database of invasive species[10] (Roques *et al.*, 2009), lists 1,590 terrestrial arthropod species of non-European origin established in Europe, including 1,390 insects, 47 spiders, 102 mites, 34 myriapods and 17 crustaceans (Kenis & Branco, 2010).

10. Delivering Alien Invasive Species Inventories for Europe http://www.europe-aliens.org/

Table **3** **10** **Number and trends of red listed species, and the major drivers of change for five groups with diverse ecology. The area covered is Western Europe, Central Europe, and part of Eastern Europe (continent of Europe).**

	Number of species	Increasing (%)	Stable (%)	Decreasing (%)	Unknown (%)	Major drivers
Terrestrial molluscs (Cuttelod *et al.*, 2011)	246	0.6	39.8	6.3	53	Urbanization, agriculture, recreation and other human activities, change in fire regime, roads and shipping lanes
Bees (Nieto *et al.,* 2014)	1,942	0.7	12.6	7.7	79	Agricultural expansion and intensification, livestock farming and ranching, pollution (agricultural and forestry effluents), residential and commercial development (urban sprawl), fire and fire suppression, climate change
Butterflies (Swaay *et al.*, 2010)	482	4	55	31	10	Agricultural intensification, abandonment, climate change (including droughts), change of woodland management, tourism and recreation
Saproxylic beetles (Nieto & Alexander, 2010)	436	2.3	26.8	13.8	57.1	Logging and wood harvesting
Grasshoppers, Crickets, Bush-crickets (Hochkirch *et al.*, 2016)	1,082	2.2	7.6	30.2	59	Livestock grazing, arable farming, increasing wild fire frequency, urbanization and infrastructure, touristic development

Local ecological knowledge on invertebrates is scarce, including their status and trends over the last decades. Some culturally salient invertebrate species have, however, functioned as important keystone species in the lives of certain communities (Marian, 1903; Ulicsni *et al.*, 2016). Indigenous and local knowledge can be a valuable information source in understudied regions for those species that migrate northwards as a consequence of climate change. Some of these species (e.g. mosquitos and ticks) may have (or already have) a strong but yet undocumented impact on local wild and domestic livestock.

Drivers of change

Environmental changes may rapidly disrupt biotic interactions (insect-insect, plant-insect, invertebrate-nutritional source). Species involved in species-specific interactions (e.g. pollination, foraging) are particularly sensitive to environmental changes. The extinction of a butterfly species may be locally explained by the extinction of its host plant. A parallel decline in pollinators and insect-pollinated plants in Western Europe favoured wind-pollinated plants, and contributes to global homogenization (Biesmeijer *et al.*, 2006, Carvalheiro *et al.*, 2013). Beyond independent taxon-based extinctions, the possible cascading effects of species loss are often neglected, which are considered likely to greatly contribute to general homogenization and species loss (Kearns *et al.*, 1998; Koh, 2004).

Honeybees suffer from colony collapse disorder, which also affects the production of colonies (Breeze *et al.*, 2014; Kovács-Hostyánszki *et al.*, 2016). Many of the environmental threats to bee diversity are associated with intensified agriculture (shifting agricultural practice linked to pollution, pesticides and the increasing intensification of farming), as well as change in land use and climate (Nieto *et al.*, 2014; Goulson *et al.*, 2008). Similar trends (sensitivity to agricultural intensification, change in land use and climate) were also observed in other kinds of insects acting as pollinators (IPBES, 2016b). Many wild bees and butterflies have been declining in abundance, occurrence and diversity at local and regional scales, as it has been recorded in Western Europe (IPBES, 2016b).

3.4.8 **Freshwater invertebrates**

Status and trends

No assessment has been performed on freshwater invertebrates for the whole of Europe and Central Asia except molluscs and dragonflies. In the interest of highlighting the magnitude of threat facing freshwater invertebrates, the next paragraph reports some global statistics. Note that the trends for the world and Europe and Central Asia are not necessarily similar, as exemplified by

comparig the global trends with European ones in the next two paragraphs.

The great majority of freshwater animals are invertebrates, mostly insects (60%) and crustaceans (10%) with molluscs being the most diverse but also threatened group of animals, with at least 43.7% of the species (373 species) considered as threatened (Cuttelod *et al.*, 2011). In the Red List assessment, IUCN experts have included 7,482 species divided in odonates, molluscs, crabs and crayfish as these taxonomic groups have received extensive attention. Therefore, these groups represent the best available dataset to quantify the extinction risk among freshwater invertebrates. It includes assessments of 1,280 species of freshwater crabs, 590 species of crayfish, 1,500 species of freshwater molluscs (30% of all known species) and 1,500 species of dragonflies and damselflies (26% of all known species). However, the precise level of threat is unknown as there is a high number of species (2,504), which have a data deficient status. Therefore, the level of threat is between 23% and 56% depending on whether we assume that no species or all data deficient species are threatened. Currently, 131 species are classified as extinct with an additional four as extinct in the wild. The most threatened groups are gastropods (from 33%-68%, respectively assuming no data deficient species are threatened or all of them are), bivalves (26%-49%), crayfish (24%-47%) (Richman *et al.*, 2015), crabs (16%-65%) and dragonflies (9%-44%) (Cumberlidge *et al.*, 2009). Due to a high proportion of range-restricted species living in highly specialized habitats subject to pollution (including sedimentation) or habitat destruction, freshwater gastropods have the highest percentage of threatened species (51%). This results in 3% of gastropods and 5% of bivalves being classified as extinct with the greatest number of extinctions reported for molluscs with more than that reported for birds, mammals and amphibians.

Concerning Europe (Europe as defined by IUCN including Western and Central Europe, and Eastern Europe up to the Urals and the Caucasus region), the most threatened group among those that are well monitored is gastropods (45-70% of species threatened depending on whether or not data deficient species are considered threatened) (Cuttelod *et al.*, 2011), followed by bivalves (20–26%) (Cuttelod *et al.*, 2011), and dragonflies (15-19%) (Kalkman *et al.*, 2010). Distribution and population of many widespread species of molluscs have been declining since the 1880s, with the greatest losses between 1920 and 1960 due to habitat change and degradation (Cuttelod *et al.*, 2011). Many species of European dragonflies have shown a dramatic decline in distribution and abundance since the second half of the 20th century (Kalkman *et al.*, 2010; Sahlén *et al.*, 2004), particularly in the south of Europe due to the dessication of their habitats. Overall, 24% of assessed populations are declining (only 12% of species

have not been assessed). At least in parts of Europe, some of the species of dragonflies considered threatened have recovered since the 1990s as result of improved water management (Kalkman *et al.*, 2010). The number of Plecoptera species decreased due to water quality degradation and physical alteration of streams and rivers, particularly those inhabiting lowland rivers of industrialized Central European countries (Fochetti & Tierno De Figueroa, 2008). *Taeniopteryx araneoides* (Klapálek) and *Oemopteryx loewi* (Albarda), once common in large Central European rivers, are now extinct (Zwick, 2004). These are among the very few documented cases of extinction in insects. Although some invertebrate species have been lost in British rivers since 1800 (four out of 30 stoneflies, three out of 37 dragonflies, three out of 193 cais, and six out of 386 water beetles), the diversity of invertebrate communities has overall increased in recent decades largely due to improvements in wastewater treatment (Moss, 2015). Family level richness increased on average by nearly 20% from 1991 to 2008, particularly in urban catchments, with a widespread shift towards taxa of well-oxygenated and less polluted waters.

Drivers of change

Water pollution, including nitrates and phosphates from agricultural sources, are the main threat to freshwater invertebrates (e.g. Cuttelod *et al.*, 2011). Habitat modifications linked to change of flow patterns as a result of dam construction and, specifically in Europe, as a result of water abstraction for domestic supplies and crop irrigation, threaten about 26% of freshwater invertebrate species. In addition, habitat modifications due to change in land use, including decline of riparian macrophytes as a result of floodplain drainage, for example for housing development projects, are responsible for 19% of threatened freshwater species. A review by Stendera and co-authors (2012) showed an overall decreasing trend in abundance, richness and diversity of invertebrates due to all these stressors, predominantly land use, eutrophication, and habitat destruction.

Alien species introduced as a result of human activities were also found to have a role in causing a decrease and change in invertebrate community structure. For example, invasions of amphipod species from Ponto-Caspian rivers were enabled by the creation of canal networks interconnecting the major Eastern and Western European river systems since the late 1700s and later enhanced by intentional transfers of potential fish food organisms to hydropower reservoirs. The rate and range of the invasions have dramatically increased since the late 1980s and in the 2000s across these three subregions and many river communities are undergoing major change with the aggressive expansion of *Dikerogammarus villosus* (Väinölä *et al.*, 2008). Another example is the North American

euryhaline *Gammarus tigrinus*, which was introduced to Britain and then intentionally to Germany in 1957 to replace locally extinct native species and has since then broadly occupied river, lake, and estuarine habitats in Europe (Väinölä *et al.*, 2008). Some *Mysids* autochthonous in the Ponto-Caspian region are also currently invading some aquatic ecosystems of Northern Europe (Leppäkoski *et al.*, 2002). The impact of these species on native lacustrine and riverine ecosystems can be severe, including a reduction in zooplankton abundance, with concomitant negative effects on higher consumers (Ketelaars *et al.*, 1999). However, at least for molluscs, though invasive species are now widely present and have had an impact on some species, their presence impacts less than 5% of the threatened species (Cuttelod *et al.*, 2011). In addition, the introduction of diseases along with the introductions of alien crayfish species has also been a major issue with *Aphanomyces astaci*, the crayfish plague, responsible for the severe decline of the native European crayfish, *Astacus astacus.*

The effects of climate change on macroinvertebrates vary depending on the region and the taxon group (Domisch *et al.*, 2011; Jähnig *et al.*, 2012) and some studies at national scale have confirmed that, in England, for example, improved water quality through positive management better explained assemblages than increased winter temperatures (Durance & Ormerod, 2009). At a local scale Brown and co-authors (2007) found that a lower contribution of meltwater (from snow and glaciers) to streams significantly increased macroinvertebrate diversity, although some cold adapted taxa decreased in abundance. Some groups such as Trichoptera are potentially more at risk than others by changes in climate across Europe (Hering *et al.*, 2009). Recently it has become evident that many dragonflies of temperate regions are responding, both in distribution and phenology, to global climate change (Kalkman *et al.*, 2008). The ranges of common and widespread southern species are expanding in Europe but there is as yet no strong evidence that northern species are decreasing as a result of the rising temperatures, as might be expected. There is evidence that ranges are changing for Odonata (Moss, 2015), bugs (Hickling *e al.*, 2006), Plecoptera, and aquatic beetles (Heino, 2002), and Diptera (Burgmer *et al.*, 2007).

Lake zooplankton has provided good examples of climate change effects on invertebrates. There is evidence of direct and indirect (through changes in hydrology) effects on seasonality, community composition, parasitism, grazing and production. For example, in the lake Muggelsee, in Berlin, zooplankton species with high thermal tolerances or rotifers that grow quickly at high temperatures have become more common (Wagner & Adrian, 2011). The trend towards warm springs and summers has also affected the population dynamics of several cyclopoid copepods whose growth

phase was prolonged both in spring and autumn (Gerten & Adrian, 2002). Predatory Cladocera as well as filter feeders have also been affected by warming. In Lake Maggiore, Italy, there was a more than 10-fold increase in the mean annual population density of *Bythotrephes longimanus* between 1987 and 1993, due to warmer winter and spring temperatures (Manca & DeMott, 2009). *Bythotrephes* remained abundant and increased even more during the following ten years, as water temperature continued to increase. *Daphnia hyalina galeata*, the dominant grazer, and a prey of *Bythotrephes*, decreased sharply as *Bythotrephes* increased. Temperature increase in a series of Russian lakes was also associated with a shift from copepods to cladocerans, resulting in the highly unsaturated fatty acid content of the community falling and thus providing food of reduced quality for fish (Gladyshev *et al.*, 2011) irrespective of timing.

Acidification of surface waters was a severe environmental problem, particularly in northern Europe, during the second half of the last century causing freshwater biodiversity loss. International action plans have led to chemical recovery of some surface waters due to decreased acid deposition, but acidification problems persist in some lakes and rivers. Long-term studies (1988-2007) have shown an overall weak recovery of invertebrate species as a response to chemical recovery in boreal lakes (Angeler & Johnson, 2012). In the Vosges mountains (France), Guerold and co-authors (2000) found a high reduction in diversity for many aquatic species, and among them Molluscs, Crustaceans and Ephemeroptera disappeared totally from strongly acidified streams. In addition, there is evidence that acidification has simplified some invertebrate communities in UK streams and probably made them more vulnerable to climate effects, which conversely might offset biological recovery from acidification (Moss, 2015).

3.4.9 Vascular plants

Status and trends

Of the estimated 32,000 vascular plant species occurring in Europe and Central Asia, IUCN evaluated 2,483 (approx. 8%) in the Red List of Threatened Species. Of these, 810 (32.6%) are threatened (270 critically endangered, 287 endangered and 253 vulnerable). Another 166 are listed as near threatened. Four species are extinct and four species extinct in the wild (likely strongly underestimated). There is a remarkably high percentage of species with unknown population trend (approx. 46%). About one fifth of the evaluated plants (19.6%) show a declining population trend, whereas about one third (31.6%) is stable. Only a very small proportion (2.5%) has increasing population sizes. However, these percentages might be biased, as it is likely that more threatened than un-threatened species have

been evaluated by IUCN. Especially the total percentage of species with increasing population sizes is likely larger, as many generalists tend to expand their range sizes (Bilz *et al.*, 2011; IUCN, 2017b).

At the national level, all occurring species have often been evaluated in Red Lists and the average proportions of extinct and endangered species are often quite high (e.g. in densely populated regions), reflecting the local decline of species richness and of population sizes (Lozano, 2000; Bornand *et al.*, 2016; Broggi & Waldburger, 1984; Cheffings & Farrell, 2005; Conti *et al.*, 1992; Curtis *et al.*, 1988; Icelandic Natural History Institute, 1996; Lilleleht, 1998; Ludwig & Schnittler, 1996; Marhold & Hindák, 1999; Millaku *et al.*, 2013; Niklfeld, 1999; Olivier *et al.*, 1995; Oltean *et al.*, 1994; Parfenov *et al.*, 1987; Phitos *et al.*, 1995; Procházka, 2000; Rakonczay, 1989; Rassi *et al.*, 2010; Latvian Academy of Science, 1997; Shelyak-Sosonka, 1996; Silic, 1996; Sugar, 1994; Vangjeli *et al.*, 1995; Vangjeli *et l.*, 1997; Velchev, 1984; Weeda *et al.*, 1990; Westling, 2015; Wind & Pihl, 2004; Wraber *et al.*, 1989; Zarzycki & Kaźmierczakowa, 2001).

Europe as defined by IUCN (West and Central Europe, Eastern Europe up to the Ural and Caucasus region) harbors more than 20,000 vascular plant species (Euro+Med, 2017). Of these, 1,826 species have been evaluated for the European Red List of Vascular Plants, comprising species listed as priority for conservation in multilateral environmental agreements (Habitats Directive, Bern Convention, CITES, EU Wildlife Trade Regulation), crop wild relatives and aquatic plants. About one third (467 species; 26%) is threatened with extinction. 45% and 10% of the MEA-listed species are listed as threatened or near threatened, respectively, 12% and 5% of the crop wild relatives, and 7% and 7% of the aquatic species. The percentage of species with an unknown population trend is notable, as this has been determined for only half of the crop wild relative species (48%), approx. one third of the policy species (37%) and about one fifth (19%) of the aquatic plants. Of the evaluated plants, 38% of the policy species, 16% of the aquatic plants and 11% of the crop wild relative species are declining, while the populations of 22% of the species listed in multilateral environmental agreements, 39% of the crop wild relatives species, and 64% of the aquatic plants are stable. However, population trend analyses are often based on survey data from only a small part of the species range or on subjective assessments based on known threats or habitat decline. Moreover, these percentages might be biased as probably more threatened than unthreatened species have been evaluated (Bilz *et al.*, 2011). Sixty-four species are known to have gone extinct (Silva *et al.*, 2008). Currently 6,190 endemic taxa (164 species groups, 5,191 species, 835 subspecies) are listed for Europe and about 50% of them are in danger of extinction. About 3,000 taxa are considered as local endemics, only occurring in one country

or one archipelago. Particularly high numbers of endemic taxa are found in the Mediterranean and the Macaronesian Islands (Blondel *et al.*, 2010; Bruchmann, 2011; Cañadas *et al.*, 2014).

Eastern Europe, and more particularly Russia, harbors about 11,400 vascular plant species (Chandra & Idrisova, 2011), 676 of them are considered threatened (Govenment of the Russian Federation, 2015). Only 53 species are evaluated in the IUCN Red List (IUCN, 2017b).

Central Asian countries harbor at least 7,000 vascular plant species. Endemism is particularly high, ranging from <1% to 15% depending on the country (Chemonics International, 2001a, 2001b, 2001c, 2001d, 2001e, 2001f; Nowak *et al.*, 2011) and especially high in the mountains of the Caucasus region. IUCN lists only 38 species as threatened (IUCN, 2017b), which very likely is strongly underestimated.

Drivers of change

Major threats to the diversity of vascular plants in the region are related to habitat destruction and degradation. Habitat loss is the primary cause of risk for 83% of endangered plant species (Silva *et al.*, 2008). Particularly vulnerable are species with small distribution ranges (e.g. endemic species), specialized habitat and/or microhabitat requirements, narrow environmental tolerances and poor dispersal and competitive ability (Bilz *et al.*, 2011; IUCN, 2017b; Pauli *et al.*, 2012). The intensification of agriculture is suggested to have the most severe impacts **(Table 3.11)** (Allan *et al.*, 2014; Bilz *et al.*, 2011; Government of the Russian Federation, 2015; Werger & van Staalduinen, 2012). Land-use intensification promotes generalist species while specialists are decreasing, leading to large-scale homogenization and loss of ecosystem functions (Gossner *et al.*, 2016; Soliveres, Manning, *et al.*, 2016; Soliveres, van der Plas, *et al.*, 2016; van der Plas *et al.*, 2016b).

While the abandonment of intensive land-use regimes can lead to a recovery of grassland ecosystems (Brinkert *et al.*, 2016; Kämpf *et al.*, 2016), the abandonment of traditional non-intensive land-use regimes, can also lead to the disappearance of plant species with the growth of shrubland or forest, especially in mountain or steppe regions (MacDonald *et al.*, 2000; Mathar *et al.*, 2015; Orlandi *et al.*, 2016; Stöcklin *et al.*, 2007).

Recreational human activities, invasive alien species, pollution (e.g. fertilizer, pesticides), habitat fragmentation, habitat loss and overexploitation are also major threats (Bilz *et al.*, 2011; IUCN, 2017b; Government of the Russian Federation, 2015; Sekercioglu *et al.*, 2011; Silva *et al.*, 2008). Islands with high proportions of endemic species are particularly vulnerable to invasive alien species, especially the Macaronesian and the Mediterranean islands

(Bruchmann, 2011; Celesti-Grapow *et al.*, 2016; IUCN, 2017b; Silva *et al.*, 2008). However, studies of the impact of invasive alien species on the diversity of native species are largely missing across Europe and Central Asia and statements on negative impacts often anecdotal (Künzi *et al.*, 2015).

Numerous vascular plant species are used for medicinal, ornamental and cultural purposes as well as in traditional agriculture (IPBES, 2016b), in some cases causing overexploitation, i.e. East-Mediterranean orchids used for salep production (Ghorbani *et al.*, 2014).

3.4.10 Bryophytes

Status and trends

Bryophytes are photosynthetic non-vascular plants that reproduce by spores. Despite the wide range of substrates colonized by bryophytes as a group, many species are restricted to narrow ecological niches with specific requirements concerning substrates and habitat persistence. Bryophytes constitute an important component of vegetation, biodiversity and biomass in various ecosystems (e.g. forest, wetland, mountain, tundra) and thereby make essential contributions to ecosystem functions (e.g., soil stabilization, water retention, carbon sinks in peatlands).

Across Europe and Central Asia, only 14 bryophyte species have been evaluated in the IUCN Red List of Threatened Species (IUCN, 2017b). In Europe, nearly 2,000 bryophyte species occur (1,342 mosses, 494 liverworts and hornworts), representing around 10% of the worlds' bryophyte diversity. Fifty-one per cent of these are endangered (693 moss and 242 liverwort and hornwort taxa; Hodgetts, 2015). A checklist for Eastern Europe and northern Asia (including Central Asia) includes 1,302 moss species and complements the European checklist (Ignatov *et al.*, 2006). Although globally and across Europe and Central Asia, only very few bryophyte species have become extinct (Hallingbäck & Hodgetts, 2001), locally or on the country scale many species are endangered or have even become extinct. However, data on population trends are largely missing. Existing trend analyses are often based on survey data from only small parts of the species range or on subjective assessments. This calls for further investigation, especially in less surveyed countries.

Drivers of change

As bryophytes are sensitive to changes, habitat destruction or degradation can eradicate local bryophyte populations leading to decreasing range sizes (Hallingbäck & Hodgetts, 2001; Hodgetts, 2015; Akatov *et al.*, 2012; Natcheva *et al.*, 2006; Sabovljevit *et al.*, 2001). For example, deforestation and the replacement of natural forests in combination

with short forestry rotation cycles causes a general lack of over-mature trees and deadwood. This can reduce species richness and change community composition. In particular, habitat specialists, such as old-growth forest species, are then replaced by habitat generalists (Bardat & Aubert, 2007; Hallingbäck & Hodgetts, 2001; Hofmeister *et al.*, 2015; Paillet *et al.*, 2010; Sabovljevit *et al.*, 2001; Vanderpoorten *et al.*, 2004).

In non-forested ecosystems, bryophytes profit from non-intensive management regimes, habitat heterogeneity and low competition (Bergamini *et al.*, 2001; Hejcman *et al.*, 2010; Möls *et al.*, 2013; Müller *et al.*, 2012; Takala *et al.*, 2014; Zechmeister & Moser, 2001). Large-scale habitat conversion, peatland drainage, peat extraction and land-use intensification over recent decades has led to habitat degradation and homogenization at the landscape level. This has greatly reduced the extent of high-quality bryophyte habitats in line with a drastic decline of bryophyte diversity and a persistent loss of bryophyte species, even after applying different regeneration methods (Bergamini *et al.*, 2009; Hallingbäck & Hodgetts, 2001; Hedberg *et al.*, 2012; Hodgetts, 1992; Sabovljevit *et al.*, 2001; Shustov, 2015).

In particular the application of fertilizer promotes competitive vascular plant and bryophyte species that suppress species adapted to poor soil conditions (Alatalo *et al.*, 2015b; Aude & Ejrnæs, 2005; Bergamini & Pauli, 2001; Hallingbäck & Hodgetts, 2001; Heino *et al.*, 2005; Hejcman *et al.*, 2010; Müller *et al.*, 2012; Van Der Wal *et al.*, 2005; Virtanen *et al.*, 2000).

While the abandonment of intensive land-use regimes can lead to the recovery of grassland ecosystems (Brinkert *et al.*, 2016; Kämpf *et al.*, 2016), the abandonment of traditional non-intensive land-use regimes in grasslands, can also lead to the development of shrubland or forest ecosystems. This can result in the loss of bryophyte diversity (Takala *et al.*, 2012).

Environmental pollution can have severe effects on bryophyte diversity, population sizes, regional species pools and bryophyte performance, for example, SO_2 deposition (Bates & Farmer, 1992; Hallingbäck & Hodgetts, 2001; Akatov *et al.*, 2012; Sabovljevit *et al.*, 2001; Zotz & Bader, 2009; Zvereva & Kozlov, 2011), high nitrogen deposition in large parts of Western and Central Europe (Armitage *et al.*, 2014; Bobbink *et al.*, 2010; Field *et al.*, 2014; Kumpula *et al.*, 2012; Phoenix *et al.*, 2012), and various other pollutants (Sabovljevit *et al.*, 2001; Zvereva & Kozlov, 2011).

Climate warming might lead to expanding distribution ranges of warmth-loving bryophyte species northwards and to higher altitudes, but might also consistently negatively affect the abundance and diversity of bryophytes with a particular future threat for oceanic bryophytes across

Western and Central Europe (Bergamini *et al.*, 2009; Delgado & Ederra, 2013; Hodd *et al.*, 2014; Zotz & Bader, 2009). Warming experiments further suggest a future productivity increase and shrub encroachment in tundra regions with consistently negative effects on abundance and diversity of bryophytes (well established; Alatalo *et al.*, 2015b; Cornelissen *et al.*, 2001; Elmendorf *et al.*, 2012; Lang *et al.*, 2012; Pajunen *et al.*, 2011; Virtanen *et al.*, 2013; Walker *et al.*, 2006).

Data on the impact of invasive species on bryophyte diversity is largely missing (but see Hallingbäck & Hodgetts, 2001). The rapid colonization of sand dunes and heathlands in 21 European countries by the invasive moss *Campylopus introflexus* suppresses other species (Essl & Lambdon, 2009; Essl *et al.*, 2013).

A relatively minor threat is overexploitation (e.g. use bryophytes for commercial, scientific or private purposes). However, collecting by bryologists has led to the extinction of one Portuguese species (Hallingbäck & Hodgetts, 2001).

3.4.11 Lichens

Status and trends

Lichens are symbiotic associations between mycobiontic (fungi) and photobiontic (algae) partners. They are an important component of vegetation and biodiversity in various ecosystems and contribute to ecosystem functions (e.g. biogeochemical cycling, carbon storage, food-webs; Cornelissen *et al.*, 2007; Curtis *et al.*, 2005; Edwards *et al.*, 1960; Gerson & Seaward, 1977; Pettersson *et al.*, 1995; Seaward, 2008). Despite the wide range of substrates colonized by lichens as a group, many lichen species are restricted to narrow ecological niches with specific requirements concerning substrate or habitat variables (Nash, 2008a).

Global estimates for lichen species numbers range from 13,500 (Hawksworth *et al.*, 1996) to 25,000 (Wirth & Hauck, 2013). In Europe (all 3 subregions, but excluding Russia) around 7,000 species occur (Feuerer, 2013), Russia harbors 3,388 species (Urbanavichus, 2010). Across Europe and Central Asia, only five lichen species have been evaluated in the IUCN Red List of Threatened Species (IUCN, 2017b). National red lists across the region often comprise only parts of the occurring lichen flora and a comprehensive supra-national red list, applying the IUCN criteria, is completely missing. However, the proportion of nationally endangered or extinct species is generally high (Aptroot *et al.*, 1998; Cieslinski *et al.*, 2003; Liška *et al.*, 2012; Nascimbene *et al.*, 2013a; Randlane *et al.*, 2008; Scheidegger & Clerc, 2002; Serusiaux, 1989; Timdal, 2015; Türk & Hafellner, 1999; Westling, 2015; Wirth *et al.*, 2011; Woods & Coppins,

2012; Zamin *et al.*, 2010). Lichens were not considered in the Natura 2000 programme and the Global Strategy for Plant Conservation of the Convention on Biological Diversity (Nascimbene *et al.*, 2013b). This indicates the general need to fill this gap in line with the 2020 Aichi Biodiversity Targets.

Knowledge on endemic lichen species is scarce. An attempt was made by the Arctic Council, listing 133 lichen species which were never found outside Panarctic countries. Of these, 61 lichen species only occur in Europe and Central Asia (Kristinsson *et al.*, 2010). Moreover, 34 lichen species were so far recorded only from the British Isles (Woods & Coppins, 2012) and 12 from the Madeira archipelago (Carvalho *et al.*, 2008). In addition, data on bryophyte population trends are largely missing. Existing trend analyses are often based on survey data from only small parts of the species range or on subjective assessments. This calls for the need of further investigation, especially in less surveyed countries.

Drivers of change

Lichens are very sensitive to changes in their environment. Therefore, pollution, environmental, land-use and climatic changes, and habitat destruction can eradicate local lichen populations leading to a decline in range size. For example, deforestation and the replacement of natural forests with plantations, in combination with short forestry rotation cycles, cause a general lack of over-mature trees and deadwood, and lack of forest structure. This can lead to homogenous lichen communities and the isolation of dispersal or establishment-limited species, reducing the species richness and the genetic diversity of lichens (Cornelissen *et al.*, 2001; Ellis, 2012, 2015; Hauck *et al.*, 2013; Hofmeister *et al.*, 2015; Nascimbene *et al.*, 2013a; Paillet *et al.*, 2010; Scheidegger & Werth, 2009; Wolseley, 1995). In non-forested ecosystems, lichens profit from non-intensive management regimes, habitat heterogeneity and low competition. Large-scale conversion and land-use intensification over recent decades has led to habitat degradation and homogenization at the landscape level in line with a drastic decline of lichen diversity (Boch *et al.*, 2016; Dengler *et al.*, 2014; Gossner *et al.*, 2016; Hölzel *et al.*, 2002; Kamp *et al.*, 2011; Korotchenko & Peregrym, 2012; Mathar *et al.*, 2015; Akatov *et al.*, 2012; Shustov, 2015; Stofer *et al.*, 2006; The Russian Academy of Sciences, 2014; Werger & van Staalduinen, 2012; Wirth *et al.*, 2011; Wolseley, 1995). The abandonment of traditional non-intensive land-use regimes in grasslands is leading to the loss of soil-dwelling lichens (Hauck, 2009; Leppik *et al.*, 2013).

Environmental pollution can have severe effects on lichen diversity, population sizes, regional species pools and lichen performance. For example, sulphate deposition eradicated the lichen diversity in large parts of Europe (Bates & Farmer,

1992; Gilbert, 1992; Hauck, 2009; Hauck *et al.*, 2013; Insarov & Insarova, 2013; Kirschbaum *et al.*, 2006; Akatov *et al.*, 2012; Nash, 2008b; Purvis, 2015; Purvis *et al.*, 2010; Sedelnikova, 1988; Zotz & Bader, 2009). In addition, the high nitrogen deposition in large parts of Europe promotes nitrophytic species to the detriment of acidophytic ones (Hauck, 2010; Insarov *et al.*, 2010; Russian Academy of Sciences, 2008; Liška *et al.*, 2012; Lisowska, 2011; van Herk, 2001); increases the growth of competing species such as vascular plants; and suppresses soil-dwelling lichens (Armitage *et al.*, 2014; Britton & Fisher, 2010; Field *et al.*, 2014; Phoenix *et al.*, 2012).

Climate-warming might lead to expanding distribution ranges of warmth-loving lichen species northwards, but also might consistently negatively affect the abundance and diversity of lichens (Aptroot & van Herk, 2007; Davydov *et al.*, 2013; Insarov & Schroeter, 2002; Zotz & Bader, 2009), for example by productivity increase and shrub encroachment in tundra regions (Alatalo *et al.*, 2015a; Cornelissen *et al.*, 2001; Elmendorf *et al.*, 2012; Lang *et al.*, 2012; Pajunen *et al.*, 2011; Virtanen *et al.*, 2013; Walker *et al.*, 2006) or the replacement of lichen-rich forests (Andreev *et al.*, 2014).

Data on the impact of invasive species on lichen diversity is largely missing. However, the invasive moss *Campylopus introflexus* is causing a decline of lichen abundance and diversity in sand dunes and heathlands of 21 European countries (Biermann & Daniels, 1997; Essl & Lambdon, 2009; Hassel & Soderstrom, 2005; Ketner-Oostra & Sýkora, 2004; Sparrius & Kooijman, 2011). Moreover, the replacement of native forests by stands of non-native tree species negatively affects lichen diversity, for example *Robinia pseudoacacia* stands (Nascimbene *et al.*, 2015). The invasive box tree moth (*Cydalima perspectalis*) is depleting natural European box (*Buxus sempervirens*) forests in the Caucasus region (Russian Forest Protection Centre, n.d.). As many rare epiphyllous lichen species are growing on the evergreen leaves of the European box (Vězda, 1983), this severely threatens their populations. In addition, epidemic tree pests, such as the current large-scale European ash borer, a species of jewel beetle (*Agrilus planipennis*) across Europe threatens many lichen species, as ash is the host tree of a large number of specialized and threatened epiphytic lichens (Ellis *et al.*, 2014; Ellis *et al.*, 2012; Jönsson & Thor, 2012; Lõhmus & Runnel, 2014; Marmor *et al.*, 2017; Rigling *et al.*, 2016).

3.4.12 Fungi

Fungi contribute a large share of terrestrial species richness and are key players in ecosystem processes (Peay *et al.*, 2016). Estimates of the global number of fungal species range between 2.2 to 3.8 million, of which 120,000

Table ③ ⑪ **Summary of past and current trends in the biodiversity of different taxa in Europe and Central Asia and of the attribution of these trends to direct drivers of change (3.4.2-3.4.12).**

● HIGH IMPACT ● MODERATE IMPACT
● NO OR MARGINAL IMPACT

| TAXON		GENERAL TREND Past ECA				GENERAL TREND Present ECA				CLIMATE CHANGE Past ECA				CLIMATE CHANGE Present ECA			
		WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA
Mammals		↘				↘				🟢				🟡			
		↘	↘	↘	↘	↘	↓	↓	↘	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🟡
Birds		↘				↘				🟢				🟡			
		↘	↘	↓	↘	↘	↘	↓	↘	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🟡
Reptiles		↑				↑				🟡				🟡			
		↗	↗	↗	↗	↔	↗	↗	↔	🟢	🟢	🟡	🔴	🟡	🟡	🟡	🔴
Amphibians		↓				↘				🟢				🟢			
		↓	↓	↕	↔	↘	↕	↕	↔	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
Marine fishes	Arctic Ocean	↔				↔				🟢				🟢			
	North East Atlantic pelag./demer.	↔	↓			↔	↕			🟢				🟡			
	Mediteranean	↘				↘				🟢				🟢			
	Black Sea	↘				↔				🟢				🟢			
	Caspian Sea	↘				↕				🟢				🟢			
	North West Pacific pelag./demer.	↘				↗											
Freshwater fishes										🟢				🟢			
										🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
Terrestrial invertebrates		↘				↓				🟡				🟡			
		↓	↘	↕	↕	↓	↓	↘	↕	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴
Freshwater invertebrates		↓				↘				🟢				🟡			
		↓	↓	↓	↓	↘	↕	↓	↓	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🟡
Vascular plants		↘				↘				🟢				🟡			
		↘	↘	↘	↘	↘	↘	↘	↘	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🟡
Bryophytes		↘								🟢				🟢			
		↘	↘	↘	↘					🟢	🟢	🟢	🟢	🟡	🟡	🟡	🟡
Lichens		↘				↘				🟢				🟢			
		↘	↘	↘	↘	↘	↘	↘	↘	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢

currently are described and accepted species. Fungi are, for practical reasons, often divided into macro- and microfungi. The overwhelming number are microfungi, i.e. species without sporocarps like molds and yeast or sporocarps smaller than 1 mm. These are not dealt with here, similar to microorganisms, due to insufficient knowledge of their distribution and ecology and lack of IUCN Red List assessments. Macrofungi (phyla Basidiomycota and Ascomycota), have visible sporocarps (> 1 mm), constitute about 30% of known fungi, and are undergoing extinction risk assessments according to the categories and criteria of IUCN (Dahlberg & Mueller, 2011). Due to their largely hidden mycelial nature and frequently sporadic and short-lived sporocarps, fungi are more poorly understood and appreciated than plants and animals. Hence, fungi have largely been invisible to the conservation community and policymakers and often overlooked in national and international nature conservation actions. During the last decades, however, the knowledge has significantly increased of the status and trends for fungi, how human activities affect fungal diversity and how to counteract

threats (Dahlberg & Mueller, 2011; Heilmann-Clausen *et al.*, 2015).

Status and trends

Macrofungal checklists exist for most European countries and for most Russian regions, but have varying degrees of completeness (Senn-irlet *et al.*, 2007). However, there is no combined checklist for Europa or Central Asia. Species richness of macrofungi in Europe has been estimated to be at least 15,000 (Dahlberg *et al.*, 2010) and 8,000 in Russia (Kovalenko *et al.*, 2005, Svetasheva, pers. com). The total species richness of fungi in Europe (Western and Central Europe including Turkey but excluding Isarel), is considered to exceed 75,000 – 100,000 (Senn-irlet *et al.*, 2007). In 2005, the number of known fungi in Russia was 11,000 and the total number of fungi exceeded 25,000 (Kovalenko *et al.*, 2005). Only twenty-five macrofungal species have been globally assessed for extinction risk according to the IUCN Red List categories and criteria (IUCN, 2017c), but the list is growing thanks to a dedicated Red List Initiative

ECA=Europe and Central Asia, WE=Western Europe, CE=Central Europe, EE= Eastern Europe, CA=Central Asia. ↑/↓ denote strong and consistent increase/decrease in the indicator; ↗/↘ denote moderate and consistent increase/decrease in the indicator; ↔ stable indicator; ↕ variable trend in the indicator.
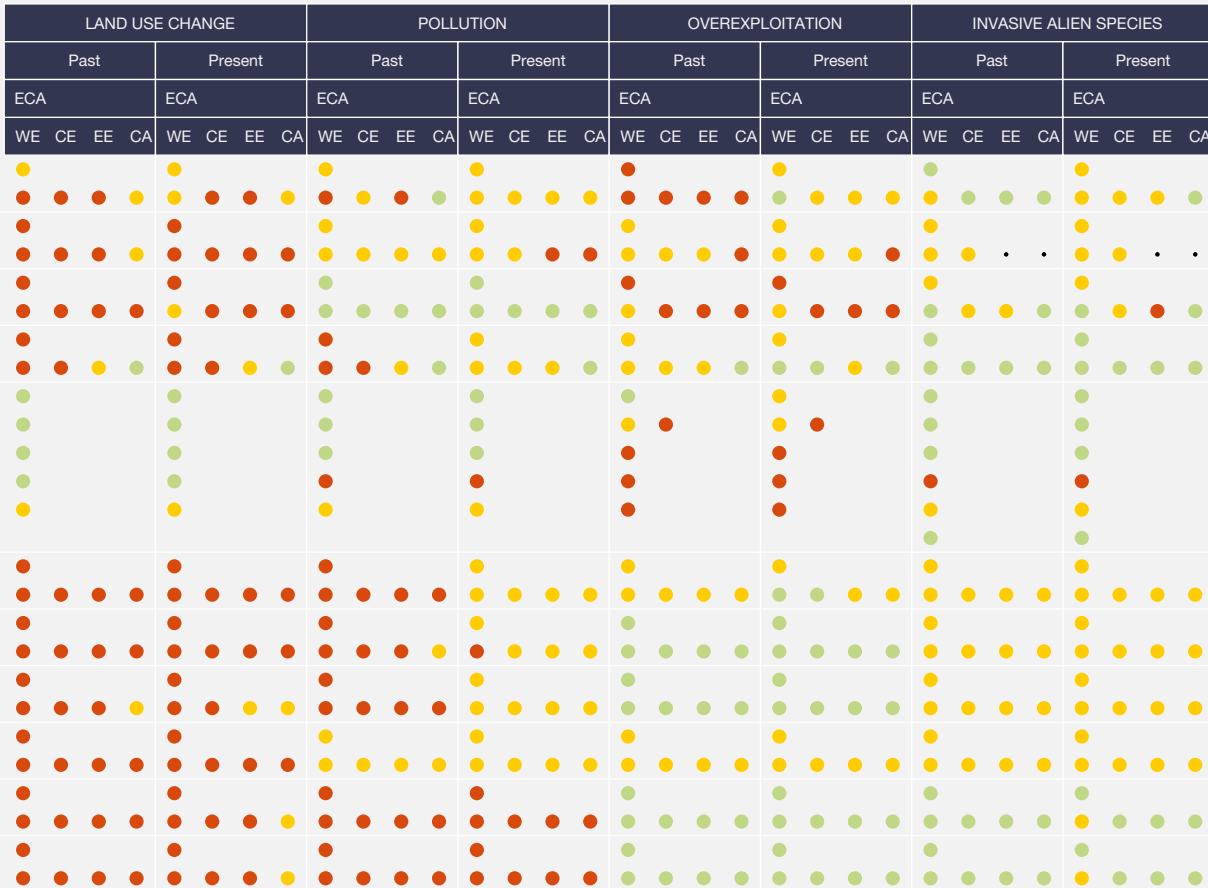
for fungi[11]. Of these, 13 species are distributed in Europe and Central Asia, of which 10 are distributed in geographic Europe (including geographically European Russia west of the Ural mountains, but not Turkey and Israel) and three in the whole region. Ten of the 13 species are threatened (one EN and nine VU). At least 33 national fungal Red Lists exist in Western and Central Europe, which are widely used for management and conservation actions across Europe (Dahlberg *et al.*, 2010). Similarly, Russia has a national Red Data Book (2008) with 24 listed species of fungi and in addition, 82 of the 85 regions in Russia have regional Red Data Books, which in total include 700 macrofungal species (Svetasheva, 2017). In total, 5,500 macrofungal species are red-listed in at least one European country, of which at least 1,664 species are considered to qualify as red-listed also at the European level (Dahlberg *et al.*, 2010). In European countries with comprehensive fungal red-list assessment, about 20% of known species are red-listed and 10% categorized as threatened (e.g. in Finland, Germany,

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11. http://iucn.ekoo.se/en/iucn/welcome

the Netherlands, Norway and Sweden; Tingstad *et al.*, 2017). These figures imply that about 5% of the European and Central Asian macrofungi would be threatened with extinction (categories vulnerable, endangered and critically endangered) if such a red-list assessment would take place.

The lack of fungi in global and European Red Lists have hindered the inclusion of fungi in international conservation agreements such as in the Annex II, IV and V of the European Union Habitats Directive. Forest ecosystems are by far the most species-rich habitats for macrofungi. Natural and extensively used European and Asian forests provided different conditions and dynamics to fungal diversity than the managed forests of today (Nordén *et al.*, 2014). Accordingly, about 75% of the nationally and globally threatened macrofungi are dependent on woodlands, mainly as associates with coarse dead wood or as ectomycorrhizal fungi with particular habitat requirements, and restricted to old-growth forests conditions. The persistence of threatened woodland fungi is determined by a combination of stand level factors together with factors related to the

surrounding landscape matrix such as proximity and extent of intensively managed forests and old growth forest habitats (Jönsson *et al.* 2017). Other habitats of large importance for fungal conservation are semi-natural grassland and natural steppe, containing some of the most threatened species, and totalling about 10-20% of national and globally threatened species. These habitats have dramatically declined throughout Europe and Asia due to conversion to arable crops, tree plantations and scrublands (Emanuelsson, 2010). Many grassland fungal species have evolved in nutrient poor and stable conditions, and disappear when artificial fertilizers are applied and decline due to atmospheric deposition of nitrogen (Arnolds, 2001). Furthermore, some types of wetland, e.g. mires and alkaline fens, are important habitats for about 5% of nationally threatened fungi in Europe. These species are sensitive to any change of hydrological regime and eutrophication (Fraiture & Otto, 2015; Svetasheva, 2015). Alcaline fens are of high conservation priority due to extensive past drainage (Šefferová Stanová *et al.*, 2008).

There is strong evidence of a decline of ectomycorrhizal fungi due to eutrophication and linked to the level of nitrogen deposition in Europe (e.g. Arnolds, 2010; Dahlberg *et al.*, 2010).

Drivers of change

The major threats to threatened macrofungi in the region are (i) habitat decline and degradation due to intensified land use of forests, semi-natural grasslands and steppe, (ii) land-use change of forests, semi-natural grasslands and steppe, followed by (iii) eutrophication and (iv) effects of invasive pathogens on native tree species (Senn-Erlet *et al.* 2007; Dahlberg *et al*., 2010). Climate change is an emergent threat likely to directly and indirectly affect fungal diversity (Heilmann-Clausen *et al.*, 2015).

The invasion of the alien fungal pathogens Dutch elm disease and ash decline have been devastating for the distribution of elm and ash in Europe and caused declines in fungal diversity associated with these trees (Brasier & Buck, 2001; Landolt *et al.*, 2016). Ecological impacts of alien invasive pathogens are projected to continue to increase in the future due to trade and climate (Santini *et al.*, 2013).

Long-term Pan-European studies imply climate to drive community changes and range expansion, so far manifested by increased fungal fruiting periods (e.g. Kauserud *et al.*, 2012). Forest management has a potential to compensate negative effects of climate change by increasing set-aside forests to prevent the decline of old-forest species under climate change (Mair *et al.*, 2017). Climate is also affecting the distribution of invasive tree pathogens native to Europe that may become negative for native tree species, e.g. the northerly range expansion of the pathogen *Diplodia* to Scots pine (Oliva *et al.*, 2013). Furthermore, climatic change

increasingly fosters alien tree species, e.g. *Acer negundo* and *Robinia pseudacacia* to invade forests and grasslands, thereby changing fungal communities and driving threatened species out of these habitats (Kleinbauer *et al.*, 2010).

3.4.13 Progress towards Multilateral Environmental Agreements for species conservation

European Union Biodiversity Strategy

Target 1 of the European Union Biodiversity Strategy calls for halting the "deterioration in the status of all species and habitats covered by European Union nature legislation (Habitats and Birds Directives), and achieving a significant and measurable improvement in their status so that, by 2020, compared with current assessments:

(a) 100% more habitat assessments and 50% more species assessments under the Habitats Directive show [a favourable or] an improved conservation status [with respect to the last reporting period at the time of adoption of the European Union Biodiversity Strategy to 2020: that is the 2001-2006 reporting period];

(b) 50% more species assessments under the Birds Directive [with respect to 2001-2006 as with the Habitats Directive] show a secure or improved status".

For the Birds Directive, the baseline was 52% of the 447 species naturally occurring in the European Union having a secure status. In the last reporting period (2007-2012), this figure was unchanged, and 8.5% were assessed as threatened but improving. Therefore, there is still a 17.5% shortfall in the percentage of species that should be secure or improving with respect to 2001, for the European Union target to be met (EEA, 2015a).

An additional 17% of the bird species naturally occurring in the European Union were assessed as threatened, and 15% were assessed as near-threatened or declining or having depleted populations. The remaining 16% of the species had unknown population status. There are no discernible geographic patterns in these status and trends, but there are ecosystem-level and taxonomic differences: grassland, heathland and coastal species, petrels, shearwaters and galliforms have a higher proportion of threatened, near-threatened and declining species than other groups (EEA, 2015a) (Section 3.4.2). Moreover, short-term declining trends are more prevalent among bird species in all marine ecosystems than species in other ecosystems (EEA, 2015a).

For the Habitats Directive, the baseline in 2001 initially assessed 15% of species as being favourable but, when further data became available, a retrospective analysis

corrected this baseline to 23%. This means that, for the European Union biodiversity target 1 to be met, 35% of species assessments should be favourable or improving by 2020 (150% of 23%). Overall, 118 monitored species of plants and animals in the European Union have unfavourable conservation status but improving trends, 572 have unfavourable conservation status and deteriorating trends and 905 have unfavourable status and stable trends (EEA, 2015a).

Overall, in the 2007-2012 reporting period, 23% of the assessment were still favourable, 60% were unfavourable and 17% had unknown conservation status. Looking at trends of unfavourable species, 4% of the species assessments were unfavourable but improving 20% were unfavourable stable, 21% unfavourable and deteriorating and 14% unfavourable with unknown trends. There is therefore a 8% shortfall in species assessments that should be favourable or improving with respect to 2001 for the European Union target to be met (EEA, 2015a).

The terrestrial and freshwater species faring worst in terms of status and trends are slightly more prevalent in the Pannonian and Steppic biogeographic regions of Central Europe (Hungary, part of Slovakia and Czech Republic, part of Romania) and the Continental, Atlantic and Mediterranean biogeographic regions (all of Western and Central Europe part of European Union , except Hungary, Scandinavia, and the Baltic Countries) (EEA, 2015a). The Macaronesian islands stand out by having the highest number of unfavourable but improving population assessments (12.1%) followed by Boreal and Atlantic regions (9% and 6.8% of assessment, respectively).

Assessing progress towards the European Union Biodiversity Strategy for marine species is marred by uncertainty in status and trends (Section 3.4.6), over half of the assessments having unknown trends. The exception is the Baltic Marine Bioregion, for which all trends are considered known and 60% are improving.

The main drivers of recent past population declines across all realms are agriculture (use of biocides and chemicals affected 73% of assessed populations, intensification 42%, modification of cultivation practices 36%); reduction of habitat connectivity (55%); pollution of surface waters (56%); invasive alien species (46%); human induced changes in hydraulic conditions (43%); and forestry (removal of dead trees (39%), clearance (38%), logging of natural and plantation forests (38%) (EEA, 2015a).

Across all species and realms, 99% of the favourable assessments for species in the 2007–2012 period were already favourable in the 2001–2006 period; this means that only 0.4% (11 assessments) truly changed from unfavourable to favourable (EEA, 2015a). At this rate, European Union Biodiversity target 1 will not be met for species.

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) Strategic Vision for 2020 includes Goal 3 "Contribute to significantly reducing the rate of biodiversity loss and to achieving relevant globally agreed goals and targets by ensuring that CITES and other multilateral instruments and processes are coherent and mutually supportive". CITES is legally binding and regulates trade in live plants and animals, their parts and products derived from them. Species subject to regulations are listed in three Appendices[12]. 529 species in Appendices of CITES occur in Europe and Central Asia. Of the 334 species with known population trends, 74% are declining **(Table 3.12)**. Importantly, 206 of these species continue to be threatened by direct large-scale overexploitation and 23 of these are endemic of Europe and Central Asia. It was not possible to track the trade flows of these species, however 17 of these are endemic, and therefore their unsustainable harvest occurs within the region. These are nearly 50% of the 40 endemic species listed in CITES annexes. This suggest that countries in Europe and Central Asia are moving away from achieving the CITES vision for 2020[13].

Aichi Biodiversity Targets

Here we report on progress towards Aichi Biodiversity Targets 12 and 13, the only ones exclusively focusing on species. Aichi Biodiversity Target 12 calls for halting species extinctions and improving the conservation status of threatened species by 2020. The indicators identified to monitor progress towards this target are the Red List Index and the Living Planet Index, although any credible measure of population trends or conservation status can be used to assess progress at national or regional scale. The Red List Index for Europe and Central Asia is declining and the Living Planet Index, only available for selected terrestrial vertebrates, is slightly declining since 2004 **(Figure 3.54)**. Our independent review of the conservation status of all reported taxa in Europe and

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12. Those in Annex I are particularly threatened and their commercial trade is banned; those in Annex II are those for which permits are needed for their international trade; those in Annex III are species included at the request of a Party that already regulates trade in the species and that needs the cooperation of other countries to prevent unsustainable or illegal exploitation; these species also require permits. Some species, including the gray wolf, are in Annex I in some countries and in Annex II in other.

13. Number obtained by intersecting IUCN data on direct threats to species with population trends from the IUCN Red List Database version 2017.1 (IUCN, 2017c) on the subset of species listed in the CITES Annexes and whose range overlap with the Europe and Central Asia region. The list of threats considered where: Hunting & collecting terrestrial animals (threat code 5.1.1: target species, threat code 5.1.4: motivation unknown or unrecorded), Gathering terrestrial plants (threat code 5.2.1: target species, threat code 5.2.4: motivation unknown or unrecorded), Logging and wood harvesting (threat code 5.3.2: target species, large scale harvest, threat code 5.3.5: motivation unknown or unrecorded), Fishing & harvesting aquatic resources (threat code 5.4.2: target species, large scale harvest; threat code 5.46: motivation unknown or unrecorded).

Table ③ ⑫ Trends in CITES-listed species in Europe and Central Asia. Data obtained from analysing IUCN assessment data retrieved in September 2017 (IUCN, 2017c). Species lists for CITES were obtained by querying https://www.speciesplus.net.

	Increasing	Stable	Declining	Unknown
Appendix I	11	6	23	7
Appendix II	15	50	216	183
Appendix I and II	0	3	0	1
Appendix III	1	1	8	4

Figure ③ �54 **Trends of the Living Planet Index for Europe and Central Asia for terrestrial vertebrates.**

The Living Planet Index is the geometric mean of the rate of change in population abundance of vertebrate species populations since 1970. Source: LPI (2016). The Living Planet Index is based on the population abundance of 2,707 populations of 392 species monitored within Europe and Central Asia between 1970 and 2012. The black line shows the index values and the shaded areas represent the 95 per cent confidence limits surrounding the trend. The trend indicates a 10% increase (range: -17 to +45 per cent) between 1970 and 2012 and a steady decline since 2004.
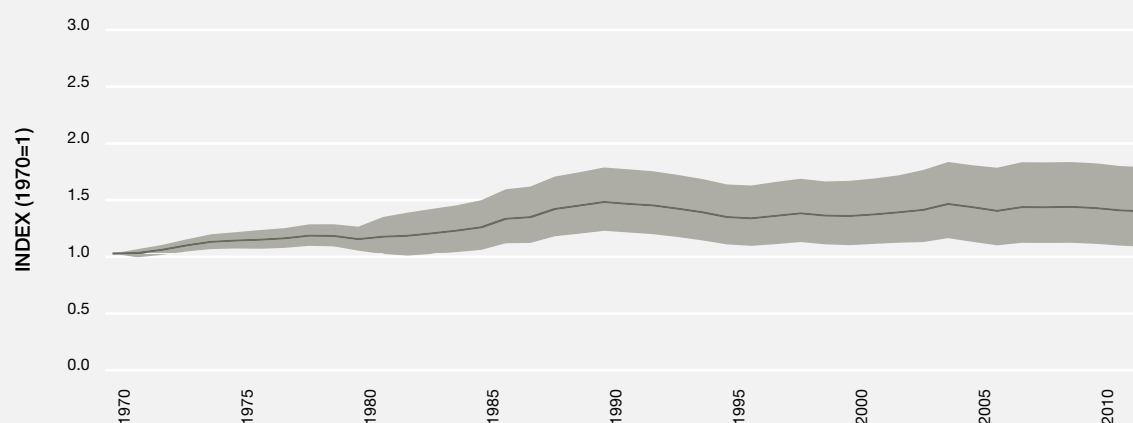
Central Asia **(Table 3.11)** confirms the trends reported by these two indicators, which, unlike our review, are taxonomically biased towards vertebrates and selected plant groups. There are notable exceptions to these general trends. For instance, the conservation status of large mammalian carnivores and bird species that have benefited from conservation attention has improved in the last two decades (Sections 3.4.2, 3.4.3). Nevertheless, 44.4% of the species extant in Europe and Central Asia with known population trends in the IUCN Red List are declining (over a total of 5,244 species extant in the region and with known trends of July 2017), 50.2% are stable and only 5.3% are increasing.

For marine species these figures are 436 decreasing, 410 stable, and 59 increasing, respectively, i.e. 48.2%, 45.3% and 6.5%; for terrestrial species 42%, 51.7%,

and 6.3%; and for freshwater species 50.2%, 7.3% and 42.5%. Note, however, that population trends are assessed throughout a species range which could extend outside the region[14]. These results combined suggest that, despite decelerating trends in extinction risk, countries in Europe and Central Asia are not on track to meet Aichi Biodiversity Target 12.

Aichi Biodiversity Target 13 calls for the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives to be maintained by 2020. The indicator chosen for animals is the trend in conservation status of domestic breeds. In 2005, 2,228 domesticated breeds of mammals and 976 domesticated breeds of

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14. Data obtained by analyzing population trends and geographic range from IUCN (2017b)

birds were recorded for Europe and the Caucasus by the Food and Agriculture Organization of the United Nations. Of these, a minimum of 50.7% and a maximum of 74.6% were either extinct or at risk of extinction, depending on how many species with unknown trends were assumed to be at risk. In 2015 a further 540 mammal breeds and 426 avian breeds were listed in Europe and Central Asia. The minimum number of breeds extinct or at risk had decreased to 45.3% but the maximum number increased to 80.4% or, put in a different way, the number of certainly safe breeds decreased by 5.8% in 10 years (FAO, 2007, 2015a).

For plant species, the indicators are more complex. A common proxy is the number of crop varieties grown in a country or region. However, this is not always correlated with genetic diversity. While genetic erosion was reported in several countries in Europe and Central Asia, a recent meta-analysis found that, overall, there appears to have been no substantial reduction in genetic diversity as a result of crop breeding in the twentieth century (van de Wouw *et al.*, 2010). In addition, the threat of hybridization of ornamental species with domestic congeners seems not to be high (Klonner *et al.*, 2017). On the other hand, several local crop varieties were lost due to replacement by higher-yielding crops, for instance all local maize and wheat varieties in Albania (FAO, 2010). As the latest FAO report on genetic diversity of cultivated plants and wild relatives puts it, "convincing evidence may be lacking for genetic erosion in farmer varieties on the one hand and released varieties on the other hand, far greater consensus exists on the occurrence of genetic erosion as a result of the total shift from traditional production systems depending on farmer varieties to modern production systems depending on released varieties" (FAO, 2010). Based on these conclusions and those of the FAO reports on domestic animal breeds it appears that, despite efforts to protect rare domestic breeds and germoplasms of cultivated plant varieties, Aichi Biodiversity Target 13 is not on track to be met for Europe and Central Asia.

Convention on Migratory Species

The Convention on Conservation of Migratory Species of Wild Animals is more commonly known as the Convention on Migratory Species (CMS). Its Strategic Plan for Migratory Species (2015-2023), mirrors the Strategic Plan for Biodiversity 2011-2020. Its target 8 is, "[by 2023] the conservation status of all migratory species, especially threatened species, has considerably improved throughout their range".

To report on progress towards this target for Europe and Central Asia, we intersected information from the IUCN Red List database, reporting global population trends for over

12,000 species in Europe and Central Asia, with the list of species in Appendices I[15] and II[16] of the CMS.

There are 371 migratory bird species listed in the annexes of the Convention occurring in Europe and Central Asia. 150 of them have declining trends, 111 are stable, 67 increasing and 43 have unknown trends. Among the long-distance migrants, most engage in various Afro-Palearctic flyways. The majority of these species have long-term population declines, especially over the period 1970-1990, in particular those that winter in open savannas and breed on agricultural land (Vickery *et al.*, 2014). More recently, Sahelian-wintering birds have shown some sign of recovery, whereas birds wintering in less arid parts of sub-Saharan Africa have shown a continued decline (Vickery *et al.*, 2014).

Migrating ungulates have not fared better. Six out of eight have declining trends, including the saiga antelope which has twice suffered population collapses since the early 1990s, due to hunting and infectious diseases (Section 3.4.3). Of the 42 migratory bat species in Europe and Central Asia, 15 are declining, nine are stable, one is improving and 17 have unknown trends.

Among marine species listed in the appendices of the Convention on Migratory Species, all three sea-turtles in Europe and Central Asia - loggerhead, green and leatherback - have declining population trends. Twenty-three out of 27 cetaceans have unknown trends. Of the remaining four, three are increasing (blue, humpback and bowhead whale) and one, the Indo-Pacific humpback dolphin, is declining.

Twelve of 13 migratory sharks and rays have overall population declines, while the great white shark has unknown trends in Europe and Central Asia.

The only bony fishes listed in the Convention appendices from Europe and Central Asia are 14 sturgeon fishes, of which 13 are declining, while the Syr darya shovelnose sturgeon has unknown trends. A 15th species of the same family occurring in Europe and Central Asia, the Siberian

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15. Appendix I comprises migratory species that have been assessed as being in danger of extinction throughout all or a significant portion of their range. Source: http://www.cms.int/en/page/appendix-i-ii-cms Parties that are a Range State to a migratory species listed in Appendix I shall endeavour to strictly protect them by: prohibiting the taking of such species, with very restricted scope for exceptions; conserving and where appropriate restoring their habitats; preventing, removing or mitigating obstacles to their migration and controlling other factors that might endanger them.

16. Appendix II covers migratory species that have an unfavourable conservation status and that require international agreements for their conservation and management, as well as those that have a conservation status which would significantly benefit from the international cooperation that could be achieved by an international agreement. The Convention encourages the Range States to species listed on Appendix II to conclude global or regional Agreements for the conservation and management of individual species or groups of related species. Source: http://www.cms.int/en/page/appendix-i-ii-cms

sturgeon *Acipenser baerii*, is not listed by the Convention despite being migratory, and is also declining. There are no migratory invertebrates listed in the Convention appendices.

Overall, these results show that Europe and Central Asia countries are moving away from achieving Convention on Migratory Species targets **(Table 3.13)**.

3.5 FUTURE DYNAMICS OF BIODIVERSITY AND ECOSYSTEMS

3.5.1 Terrestrial systems

3.5.1.1 Species distribution and conservation status

Short term projections of the impact of climate change on plants, mammals and birds to 2020 indicate widespread contractions in suitable climatic ranges spanning from 10% to 55% depending on climate scenario and taxonomic group considered (Casazza *et al.*, 2014; Thuiller *et al.*, 2011). Extrapolations of trends in farmland bird abundance to 2020 assuming business-as-usual socio-economic trends and full implementation of the Common Agricultural Policy in the European Union also show overall declines across the region, as well as national declines for 15 out of 26 countries considered (Scholefield *et al.*, 2011).

Few studies investigated projections for a period relevant to the lifespan of the Sustainable Development Goals (2030). Disaggregated results of species richness intactness (ratio of species native to a pristine community extant in a given location) of plant and animals for the region from Newbold *et al.* (2015), report an 8% decline by 2035 under two alternative scenarios of land use, compatible with relative concentration pathways scenarios IMAGE 2.6 (w/m^2 of radiating forcing), and AIM 6.0 (w/

m^2). For 2030, Verboom *et al.* (2007) found a 4% decline in relative richness under the 4 Special Report on Emissions Scenarios (SRES).

Combined effects of land-use and climate change under business-as-usual scenarios for the second part of the 21st century, are projected to cause widespread range shift and contraction and local population declines across animal and plant species. On average, ranges of mammalian carnivore and ungulate species in Europe (excluding the Russian Federation) are expected to contract by 8% assuming that all species can adapt locally to climate change (therefore declining exclusively due to habitat loss); by 15% if they are allowed to track suitable climatic conditions by dispersing at their maximum physiological dispersal; or by 24% if it is assumed that they cannot disperse (Rondinini and Visconti, 2015). Under these conditions, range shifts and contractions are predicted by 2050 for two-thirds of European breeding birds (Barbet-Massin *et al.*, 2012), for tree species in France (Cheaib *et al.*, 2012) and for alpine plants in Europe with about 50% average reduction in range size by 2100 (Dullinger *et al.*, 2012; Engler *et al.*, 2011).

On average, across all plant and animal groups, local richness and mean species abundance are projected to continue to decline throughout the region, under business-as-usual socio-economic scenarios **(Figure 3.55, Figure 3.56).** Declines are widespread throughout Europe and Central Asia with the exception of the arid parts of Central Asia and the Russian Federation which are less suitable to agricultural expansions and therefore are not projected to incur further habitat loss **(Figure 3.55)**.

Extinction risk prognoses assessed through IUCN Red List criteria, are projected to deteriorate for one to eight species of large mammals in Western and Central Europe (out of 27 investigated), depending on the assumption made with regards to ability to track climate change (Rondinini & Visconti, 2015; Visconti *et al.*, 2016).

Overall, these results provide evidence that, under business-as-usual socio-economic trends and in absence of new policies for conservation of biodiversity and ecosystem

Table ③ ⑬ Trends in species listed in appendices of the Convention on Migratory Species in Europe and Central Asia. Data obtained from analysing IUCN assessment data retrieved in September 2017 (IUCN, 2017c). Species lists for the Convention were obtained by querying https://www.speciesplus.net.

	Increasing	Stable	Declining	Unknown
Appendix I	5	0	13	4
Appendix II	64	118	158	76
Appendix I and II	5	4	30	7

Figure **3** **55** Bivariate map showing spatial pattern in species richness (shades of blue) and local mean percentage changes in extent of suitable habitat between 2010 and 2050 (shades of red, d-ESH in the caption) for all mammalian terrestrial carnivore and ungulate species under a business-as-usual scenario, with land use and climate change and assuming that species cannot disperse to track climate change **A** and species can disperse one mean dispersal distance per generation **B**. Source: Visconti *et al.* (2016).
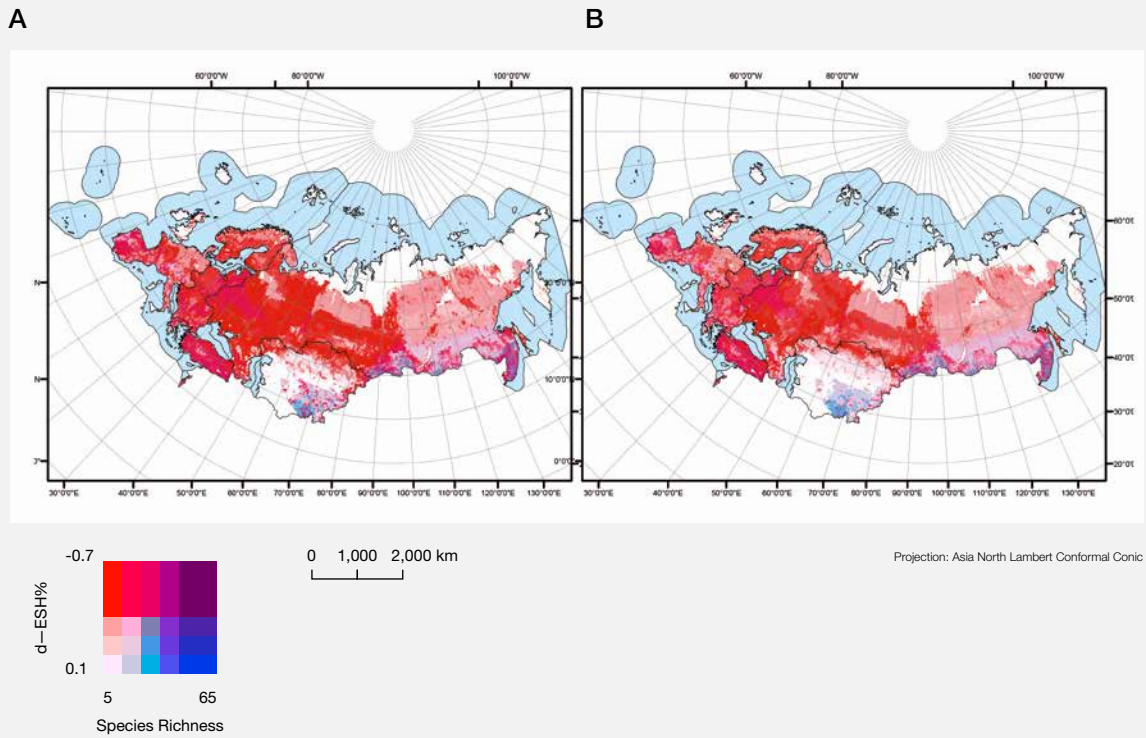
Figure **3** **56** **Trends in mean species abundance across animal and plant groups for four future scenarios for Europe and Central Asia and the world.**

BL: baseline; GT: global technology, DS: decentralized solutions, CC: consumption change. Storylines for each scenarios are in Chapter 5. Source: PBL (2014).

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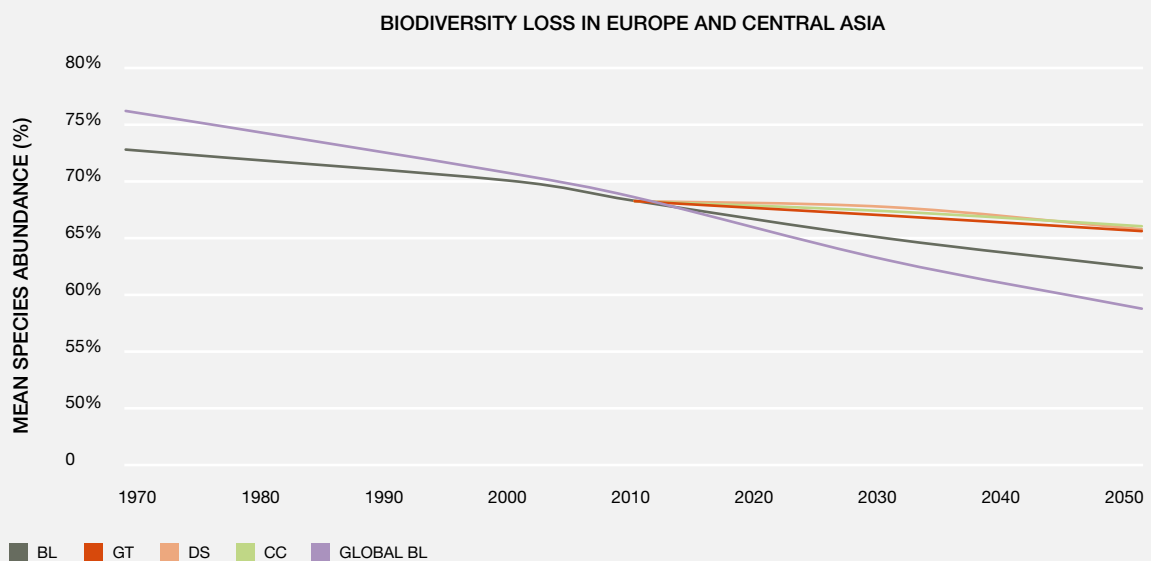
services, the Convention on Biological Diversity vision to halt the loss of biodiversity, will not be met by 2050 and beyond for Europe and Central Asia. Normative scenarios that aim to meet these targets have been produced (PBL, 2010, 2012, 2014). These studies showed that policies to mitigate climate change that involve replacing intensive forestry with reduced-impact logging, and increasing yields to spare land from cultivation, can together stem biodiversity losses expected under baseline patterns of consumption and production (see also Chapter 5 on normative scenarios designed to meet biodiversity goals).

3.5.1.2 Community composition

Local taxonomic richness of native species (alpha diversity) across plants, fungi and animal taxa in the terrestrial environment is expected to decline across all of Europe and Central Asia under business-as-usual scenarios of habitat loss (ignoring other drivers of change), except for boreal forests in Fennoscandia and Russia and for the arid regions of central Asia which are not projected to incur agricultural expansion (Newbold *et al.*, 2015). Similar richness patterns are found in freshwater environments (below).

Species range shift, ecological filtering through loss of native vegetation, and the introduction of new species are projected to result in increased temporal turnover of species across most terrestrial ecosystems (Barbet-Massin & Jetz, 2014; Newbold *et al.*, 2015; Verboom *et al.*, 2007). Similarly, local functional diversity is also expected to increase, at least for birds across all subregions of Europe and Central Asia, as a result of climate-driven range shifts (Barbet-Massin & Jetz, 2014). Climate-driven range shifts, and species introductions are likely to lead to declines in beta (i.e. between-site) diversity across the region, with resulting spatial biotic homogenization. For instance, beta taxonomic diversity of plant species in the French Alps is expected to decline by 10-23% by 2050, depending on the climatic model applied (Thuiller *et al.*, 2014a). Beta phylogenetic diversity in Europe for birds and mammals is expected to decrease by 32% and 30% by 2080 under BAU socio-economic scenarios, as a consequence of climate-induced range shifts, expansions and contractions (Thuiller *et al.*, 2011).

3.5.1.3 Ecosystem extent, function and structure

Within Europe and Central Asia, the extent of coniferous forests is expected to be maintained or even increase. Meanwhile, tundra, other Alpine ecosystems, Mediterranean ecosystems, and broad-leaved and mixed forests are expected to substantially contract, because of climate and land-use change (Benito Garzon *et al.*, 2008;

Lehsten *et al.*, 2015; Verboom *et al.*, 2007). Increasing water deficit (aridification) may lead to range contractions of some tree species, especially those with limited migration ability, such as European beech (Saltré *et al.*, 2015). A rapid upward shift of mountain vegetation belts by ca. 500 m and treeline positions of ca. 2,500 m a. s. l. by the end of this century is also predicted (Schwörer *et al.*, 2014).

Alpine, Scandinavian, and Icelandic glaciers are projected to retreat. The range of losses depends of climate modelling scenario and varies from 20% to 90% from the 2006 ice volume (IPCC, 2014b).The extent of tundra in the region is limited northward by the ocean and by a small area of Arctic desert. Shrinking of the tundra belt due to loss of permafrost, most active in Siberia and in the southern Arctic (IPCC, 2014a), with subsequent replacement by coniferous forests is expected by the end of the 21st century (Lindner *et al.*, 2010, Kharuk *et al.*, 2006).

It is likely that aridification will reduce the geographical ranges of broadleaved forests, and that Euro-Siberian conifers at medium and high elevations will be displaced by Mediterranean sclerophyll species. Mediterranean mountains might lose their key role as refugia for cold-adapted species and this may have a disproportionate impact on phylogenetic diversity (Barbet-Massin *et al.*, 2012; Benito Garzon *et al.*, 2008; Ruiz-Labourdette *et al.*, 2012; Thuiller *et al.*, 2011, 2014a).

Mediterranean-type ecosystems will suffer from rising temperature, rainfall change (declining in most cases), increased drought, and increased fire frequency (IPCC, 2014b).

Increased seasonal thawing of permafrost due to climate warming may alter the hydrological and thermal regime of polygon and palsa peatlands, as well as their spatial structure (Minayeva & Sirin, 2009, 2010; Minayeva *et al.*, 2017b). However, many forecasts of the effect of climate change are ambiguous. Climate change may lead to permafrost degradation in the southern parts of the Asian territory of Russia, whereas forest is likely to expand into in the forest tundra. Fires on peatlands and other paludified habitats have already become more frequent from the tundra to the steppe (Minayeva *et al.*, 2013).

The carbon stored in natural vegetation is likely to increase under business-as-usual scenarios of climate change (Friend *et al.*, 2014). However, changes in plant respiration and release in soil carbon will be such that there will be a net release of soil carbon in forest and grassland ecosystems (Wolf *et al.*, 2012). The potential standing stock of plant biomass in Russia is predicted to increase in response to elevated precipitation (Shuman & Shugart, 2009).

3.5.1.4 Emerging drivers of change

Russian tundra is expected to be further fragmented, polluted and degraded by projected transport systems, settlements and industrial sites (Government of the Russian Federation, 2013). A warmer climate and longer period of open sea water will make territories of Polar Islands more available for tourism which can become a negative factor of disturbance for animals and birds (Bagin *et al.*, 2011).

3.5.2 Freshwater systems

3.5.2.1 Species distribution and conservation status

Freshwater molluscs, most aquatic insects, headwater fishes and crustaceans are expected to contract their ranges due to climate change with greater than 2℃ warming by 2070 (IPCC Assessment Report 4, scenarios A1B and A2), while aquatic macrophytes, dragonflies and downstream fishes have the potential to expand their range, assuming they are able to disperse and that no other threats will impede their expansion (Alahuhta *et al.*, 2011; Capinha *et al.*, 2013; Cordellier *et al.*, 2012; Domisch *et al.*, 2011). Stenothermal species (with narrow thermal ranges, such as Arctic charr, *Salvelinus alpinus*) will probably shift range or become locally extinct, whereas eurythermal species (with a wide thermal tolerance, such as common carp, *Cyprinus carpio*) will likely be able to adapt to new thermal regimes. At high latitudes, cold-adapted species, such as salmonids, and amongst them notably the northernmost freshwater fish species, Arctic charr, will likely experience major population reductions, a continuation of current trends (Brucet *et al.*, 2010; Moss, 2015).

In a large analysis of projected bioclimatic envelopes for 323 freshwater plants, 470 fishes, 659 molluscs, 133 odonates, 54 amphibians, five crayfish and four turtles across 18,783 European catchments Markovic *et al.* (2014) found that in Europe under the climate change scenario A1B for 2050, 6% of common and 77% of rare species are predicted to lose more than 90% of their current range and 59% of all freshwater species are predicted to lose habitat suitability across more than 50% of their current range. They forecasted that nine molluscs and eight fish species should experience 100% range loss. As the most species-rich group, molluscs are particularly vulnerable due to the high proportion of rare species and their relatively limited ability to disperse. Furthermore, around 50% of molluscs and fish species will have no protected area coverage given their projected distributions. Dragonflies might be able to shift or even expand their ranges, assuming they are able to disperse to track suitable climate.

Caddiesflies (order Trichoptera) are among the most sensitive taxa to climate change. About 20% of the Trichoptera species in most southern European ecoregions and about 10% in high mountain range possess characteristics that make them vulnerable to climate change (Hering *et al.*, 2009).

Macroinvertebrate communities are central to ecological assessments of river and stream ecological quality under the Water Framework Directive. Systems by which these assessments are made could be upset by effects of climate change (Hassall *et al.*, 2010). For example, range shifts in Odonata could change scores derived from the Biological Monitoring Working Party (BMWP) system that is used and have effects consequently on conservation monitoring and assessments (Moss 2015). The Plecoptera are particularly crucial, since they have been allocated some of the highest BMWP scores and have been shown to be "cold-adapted" and to decline in species richness with increasing temperature (Heino *et al.*, 2009).

Many southern countries in Europe, such as Portugal, Spain, Italy, Greece and Turkey are home to high numbers of endemic and threatened species. The consumption of freshwater is expected to increase in the coming years, both as a result of increasing demand and climate change, posing a threat to freshwater habitats and species (Freyhof & Brooks, 2011). This is also true for the Crimean Peninsula where a highly endemic fish fauna is restricted to a few small streams, from which water is already extracted in large and unsustainable amounts.

3.5.2.2 Community composition

Under scenarios of strong climatic impacts (e.g. SRES A1B and A2), freshwater ecosystems are projected to undergo large changes in community structures and therefore loss of ecological integrity. Local species richness in freshwater systems is projected to decline for most taxa due to climate change, but this is expected to be partially compensated by colonisation of new species; species turnover for instance is projected to increase for freshwater stream fishes in France by about 60% by 2080 (Buisson *et al.*, 2008), and aquatic plants and dragonflies local richness is expected to increase in Western Europe assuming unlimited dispersal (Markovic *et al.* 2014). Floating invasive alien plant species are projected to become more prevalent in the region (Meerhoff *et al.*, 2012; Moss, 2015).

Global warming and associated changes in water level and salinity will likely seriously affect community composition in lakes and ponds (Brucet *et al.*, 2009, 2012; Jeppesen *et al.*, 2012, 2015) with some effects already being observed. For example, complex changes in fish community structure

Box ③ ② 21ˢᵗ century scenarios for mountain ecosystems.

Trends in future climate, land use and invasion projections for mountain systems

Similar to other regions of the world, mountain systems in Europe and Central Asia are projected to warm at a higher rate than other areas (Rangwala *et al.*, 2013). Climate models predict an average temperature change for mountain ranges worldwide of 2-3°C by 2070 and 3-5°C by the end of the century (Nogués-Bravo *et al.*, 2007), with greater increases for mountains in northern latitudes than in temperate and Mediterranean climates, with severe impacts expected on biodiversity. Additional threats on biodiversity are represented by invasive species, predicted to increasingly invade mountains under climate change (e.g. Pauchard *et al.*, 2009; Petitpierre *et al.*, 2015) and by land-use change and pollution (Yoccoz *et al.*, 2010). Biological responses to ongoing global changes were already evidenced, and these trends are expected to intensify in the future (Pereira *et al.*, 2010), with complex biophysical dynamics in mountain systems (Bugmann *et al.*, 2007).

Vegetation

Both mechanistic and correlative modelling approaches predict an advance of the treeline, and a consequent reduction of the alpine and nival areas (Körner, 2012; Pellissier *et al.*, 2013). Currently, however, the main driver of upward treeline shifts is land abandonment (Gehrig-Fasel *et al.*, 2007), which shows the importance of considering land-use changes in combination with climate change. Most models project strong changes in composition and structure of temperate and Mediterranean mountain forests, affecting biodiversity and ecosystem services, such as protection against rockfalls and avalanches (Elkin *et al.*, 2013). 21ˢᵗ century climate change scenarios predict a massive reduction of high-elevation grassland plant diversity and high community turnover, possibly changing the structures of current natural ecosystems (Engler *et al.*, 2011), but first extinctions may only be observed in several decades (e.g. 40 years at high elevation in the Swiss Alps; Engler *et al.* 2009). For the whole European Alps, Dullinger *et al.* (2012) predicted a range reduction around 44-50% for 150 high-mountain species, including several endemics, with possible delays in extinctions (extinction debt). Species that already occur near mountain tops with no possible escape upward have a greater risk of extinction, as predicted for Europe (e.g. Dirnböck *et al.*, 2011; Dullinger *et al.*, 2012; Engler *et al.*, 2011; Randin *et al.*, 2009; Thuiller *et al.*, 2005), Spain (Felicísimo *et al.*, 2011), or Norway (Wehn *et al.*, 2014). On the other hand, mountain systems that have pronounced microclimatic variations may allow species to persist locally (Randin *et al.*,

2009; Scherrer & Körner, 2011; Trivedi *et al.*, 2008). The melting of permanent snow and ice may also provide new potential habitats at higher elevations than currently found, although the formation of soils may take several hundred years (Engler *et al.*, 2011; Guisan & Theurillat, 2001). In the lower alpine areas, losses of grasslands are to be expected by upward shift of treelines (Dirnböck *et al.*, 2003; Körner, 2012; Pellissier *et al.*, 2013), with a 2.2 degree warming leading to an upward shift of the treeline of about 400 m, to a reduction of the lower alpine zone of more than 20% and of the upper alpine and nival zones of more than 50% (Körner, 2012; see Theurillat & Guisan, 2001 for 3.3 degree warming). Counteracting these trends in alpine habitat losses would require the maintenance of large summer farms (Dirnböck *et al.*, 2003). Model simulations show that pasture-woodland systems on lower elevation mountains (e.g. Jura mountain in Western Europe), in particular, may suffer from increased drought, resulting in progressive shifts from Norway spruce to beech under moderate warming, or to Scots pine under extreme warming.This may require changes in silvopastoral practices, such as intensifying pasturing and moving to mixed herds (e.g. cattle, horses, sheep, and goats) to prevent forest encroachment and the loss of species-rich open grasslands and forest-grassland ecotones (Peringer *et al.*, 2013). Also using simulations combining land-use and climate change scenarios for the Larch in the French Alps, Albert *et al.* (2008) conclude that ongoing and future agri-environmental policies have to be quickly adapted to protect biodiversity and ecosystem services provided by subalpine grasslands.

Much fewer modelling studies exist that examine the effects of pollution on plant species and vegetation in mountains of Europe and Central Asi. In the Jizera Mts of Northern Bohemia, ongoing nitrogen deposition results in an unbalanced nutrition of Norway spruce, causing crown defoliation that may ultimately decrease the upper optimal limit for the young spruce stands (Lomský *et al.*, 2012), but positive effects of nitrogen deposition combined with climate warming were also observed in other mountains (Hauck *et al.*, 2012), making prediction of pollution effects on vegetation still uncertain.

More studies exist on invasions by exotic plants in mountain areas. Although mountains areas were long considered as more preserved than lowlands from biological invasions (Pauchard *et al.*, 2009), recent modelling studies predict increasing threats by invasive alien species in mountains of the region under climate change, sometimes combined with land-use change (Cervenkova & Münzbergová, 2009; Hof, 2015; Kašák *et al.*, 2015; Petitpierre *et al.*, 2015; Simpson & Prots, 2013).

may be expected owing to the direct and indirect effects of temperature, and indirect effects of eutrophication, water-level changes and salinisation on fish metabolism, biotic interaction and geographical distribution (Jeppesen *et al.*, 2010). Local extinctions and changes in community

composition are likely in the coldest and the most arid regions, after the expansion of the warm adapted species. Fish species richness will likely increase in many continental lakes owing to a poleward expansion of warm-tolerant species.

Enhanced salinization may also promote changes in fish assemblages leading to a greater importance of small-bodied or planktivorous species, and therefore, a strengthening of eutrophication effects (Brucet *et al.*, 2010; Jeppesen *et al.*, 2010).

Several studies have reported projected impacts on community composition of invasive alien species, in isolation or in combination with climate change. For example the Louisiana red swamp crayfish *Procambarus clarkia*, a highly invasive species, is projected to expand its range throughout Europe in the coming decades (Ellis *et al.*, 2012), the African clawed frog *Xenopus laevis* is expected to become invasive in Europe (Ihlow *et al.* 2016), as is the Asian gudgeon *Pseudorasbora parva*, which has been predicted to expand its invasive range throughout Europe and Central Asia with significant ecological implications for its fish diversity (Fletcher *et al.*, 2016). In some instances, the extent of overlap between native species and their invasive alien competitors is projected to increase, this is the case of the native depressed river mussel (*Pseudanodonta complanata*) and its invasive competitor *Dreissenia polimorpha*. In other cases, climate change can partially reduce the overlaps between invasive and native species. This is the case for the invasive *Pacifastacus leniusculus*, which is projected to lose suitable habitat due to climate more than the native white-clawed crayfish *Austropotamobius pallipes* (Gallardo & Aldridge, 2013). Most of these patterns also emerge with lower emission scenarios (e.g. SRES B1 and B2 climate scenarios) but with less dramatic change (Capinha *et al.*, 2013; Cordellier *et al.*, 2012; Sauer *et al.*, 2011).

An increase in species richness at warmer temperature is predicted for phytoplankton and periphyton in shallow lakes, while the opposite is true for macroinvertebrates and zooplankton (Brucet *et al.*, 2012; Jeppesen *et al.*, 2012; Meerhoff *et al.*, 2012). Another study (Shurin *et al.*, 2010) suggested that potential impacts of global change on lake zooplankton biodiversity will depend on the relative magnitudes and interactions between shifts in chemistry and temperature. The study shows that temporal fluctuations in the chemical environment tend to exclude zooplankton species whereas temperature variability tends to promote greater richness. Thus, increasing frequency of extreme events and greater ranges of variability may be as or more important than changes in average conditions as drivers of zooplankton community diversity.

3.5.2.3 Ecosystem functioning

In inland waters, total biomass stock of planktonic autotrophs has been projected to either remain stable or increase under business-as-usual climate projections for the 21st century (Elliot *et al.*, 2005; Markensten *et al.* 2010, Arheimer *et al.*, 2005). Mooij *et al.* (2007) predict that

cyanobacteria blooms will increase productivity despite related declines in diatoms and green algae. Cyanobacteria being a poor food source for zooplankton, these and higher trophic levels are likely to decline as a result of climate change. Moreover, due to reduced critical nutrient loading and eutrophication, temperate lakes (with temperature varying between 2 and 22 degrees) are likely to switch from the clear to the turbid state in a 3 degree-warming scenario.

Changes in important functional traits are expected in the future due to global warming. For example, the body size of fish and zooplankton is expected to decrease under higher temperature with negative consequences for the functioning of the food web and the biodiversity of aquatic ecosystems (Daufresne *et al.*, 2009; Emmrich *et al.*, 2014; Meerhoff *et al.*, 2012). Global warming is also expected to affect other fish life-history traits (e.g. shorter life span, earlier and less synchronized reproduction), as well as the feeding mode (i.e. increased omnivory and herbivory); behaviour (i.e. stronger association with littoral areas and a greater proportion of benthivores); and winter survival (Jeppesen *et al.*, 2010). The increased dominance of smaller fish and omnivory will lead to stronger predation by fish on zooplankton and weaker grazing pressure of zooplankton on phytoplankton in warmer lakes (Jeppesen *et al.*, 2014). This will have negative consequences for the ecological status of shallow lakes. Importantly, changes in fish communities that occur with global warming partly resemble those triggered by eutrophication. This implies a need for lower nutrient thresholds to obtain clear-water conditions and good ecological status in the future (Jeppesen *et al.*, 2010; Meerhoff *et al.*, 2012).

Increased salinity due to global warming, water abstraction and pollution may also have negative consequences for the ecosystem structure, function, biodiversity and ecological state of lakes, temporary and permanent ponds, wetlands and reservoirs (Brucet *et al.*, 2009; Cañedo-Argüelles *et al.*, 2016; Jeppesen *et al.*, 2015).

3.5.2.4 Emerging drivers of change

Aquaculture is growing worldwide, already providing more than 50% of the fish and other aquatic organisms on the market. Development of aquaculture, which is now mainly focused on intensive technologies, including integrated agriculture-aquaculture multi-trophic farming, pond culture, cage-culture, recirculating aquaculture systems (RAS) technologies (Karimov, 2011; Thorpe *et al.*, 2011) might have contrasting effects on biodiversity. On one hand aquaculture might substitute the demand for natural fish and other aquatic species and will promote the conservation of biodiversity. On the other hand, aquaculture has historically been the source of invasions in some parts of the region, specifically in Eastern Europe and Central Asia. Lack of

adequate management, development of aquaculture and use of genetically modified organisms can further increase invasions of alien species and threaten biodiversity and/or endemic species.

The Brönmark & Hansson (2002) review on environmental threats to lakes and ponds predicted that biodiversity in fresh waters will, in most parts of the world, have decreased considerably by the year 2025. Changes in biodiversity may in turn affect freshwater ecosystem processes such as primary productivity, detritus processing and nutrient transport at the water-sediment interface. In addition, loss of species at higher trophic levels may have strong repercussions down the food chain (Brönmark & Hansson, 2002). Furthermore, these authors suggested that "old" problems such as eutrophication, acidification and contamination, may become less of a problem in the future, whereas "new" threats such as global warming, UV radiation, invasive alien species and endocrine disruptors most likely will increase in importance.

3.5.3 Marine systems

3.5.3.1 Species distribution and conservation status

Direct and indirect impacts of climate change on species distribution and abundance have been predicted for all marine systems and virtually all taxonomic groups investigated.

Climate change effects on Arctic and sub-Arctic marine mammal and bird species will vary by life history, distribution, and habitat specificity with some major negative effects on ice-obligate species (such as hooded seal, narwhal and ivory gull; Moore & Huntington, 2008); some species coming to the region seasonally may benefit from ice loss (killer whale, grey whale) (Larsen *et al.*, 2014). It is projected that polar bear number will decrease dramatically with approximately two-thirds of the world's polar bears extirpated by the middle of the 21st century under A1B scenario (Amstrup *et al.*, 2008; Larsen *et al.*, 2014). There is a risk that Arctic shelf species might become locally extinct due to shortage of climatically suitable shelf habitat (Fossheim *et al.*, 2015a).

In the North East Atlantic, pelagic ecosystems and taxa are projected to display higher modifications than demersal communities, a pattern explained in some regions by the influence of regional topography (e.g. North Sea; Weinert *et al.* 2016). This does not mean that demersal species are not affected by the projected changes, only that rates are variable. For instance, marine fish in the North Sea have projected poleward shifts which can be up to two times higher than the observed current rate of shift (Cheung *et*

al., 2016). Benthic communities of the North Sea were also shown to be strongly impacted under the IPCC AR4 scenario A1B, with latitudinal northward shift projected in 2099 for 64% of the 75 species examined by Weinert *et al.* (2016). Seabirds, which are often faithful to breeding colonies, are also expected to show important changes in their distribution in the North East Atlantic. For example, the ranges of 65% or 70% of 23 seabirds from the British Isles are expected to shrink by 2100 under two emission scenarios (IPCC AR4 climate change scenario A1B and A2 respectively) and under the hypothesis of unlimited dispersal; this value increases to 100% (and all of them lose at least 25% of their range) with no dispersal (Russell *et al.*, 2015).

Less information is available on projected impacts of fisheries in the region. For the Atlantic cod and the European seabass, under a scenario of an increase in demand of 5.6% per year, a decline of the spawning stock sizes of the North Sea cod by 97% is predicted toward by 2050, compared with a scenario with a stable demand (Quaas *et al.*, 2016). Cascading effects are also projected along the trophic network: by 2040, climate change, in particular summer warming, is projected to lower the abundance of the copepod *Calanus finmarchicus* which is used as a prey by cod in the North East Atlantic (Kamenos, 2010).

Some catch species will also have reduced survival and fertility due to direct and indirect impact of climate change. For instance, Baltic Sea cod eggs require certain environmental conditions regarding oxygen (>2 ml/l oxygen) and salinity (> than 11 g/kg). Physical and chemical changes in the Baltic will reduce cod reproductive potential by 75% by 2100 (Neumann, 2010).

3.5.3.2 Community composition

Species turnover is projected across all marine systems in the region and across a large range of marine habitats and taxa. Reductions in sea ice in the central Arctic are likely to enhance invasion of benthic taxa from the Pacific to the Atlantic due to more freely flowing currents (Hunt *et al.*, 2016, Renaud *et al.*, 2015). The Chukchi and the Barents Seas along with the western part of the Kara Sea are the most likely locations for the expansion of some boreal benthic species and communities (Renaud *et al.*, 2015).

The advection of zooplankton to the Arctic Basin along the Eurasian shelf is projected to cease during the 21st century, as revealed by models based on climate scenario A1B, with increased participation of the species of temperate origin in the communities of the Eurasian Arctic Seas (Wassmann *et al.*, 2015). In particular, for the Barents Sea by 2059, zooplankton of Atlantic origin will increase and zooplankton of Arctic origin will decrease under moderate climate change (SRES B2 scenario, Ellingsen *et al.*, 2008).

Boreal fish species replacing Arctic species are known to be opportunistic generalists, and their expansion is known to alter the structure of Arctic food webs and is predicted to increase the connectivity between benthic and pelagic habitats. As a result, more densely connected and less modular Arctic marine food-webs are expected to emerge (Kortsch *et al.*, 2015).

Models of fish invasions have shown that the rate of spread of non-native species in the Barents Sea are five times higher than the global average, with the central Barents Sea fish community spreading northwards and Arctic community retreating. This shift appears to be taking place at a speed at >159 km per decade.

For some marine alien species already introduced in the North East Atlantic, like the American clam, *Ensis directus* (Raybaud *et al.*, 2014), and the Pacific oyster, *Crassostrea gigas* (Jones *et al.*, 2013), expansion of their current range is projected with high level of confidence by the end of the 21st century, under medium to severe climate change scenarios.

In the North East Atlantic, 21st century scenarios of moderate (e.g. IPCC RCP 4.5, 550 ppm B1) to severe climatic change (e.g. IPCC RCP 6.0 or RCP 8.5, 720 ppm A1B), are projected to generate important changes in marine community structure, population abundance, and species range and richness (Beaugrand *et al.*, 2015; Blois *et al.*, 2013; Cheung *et al.*, 2009; Garciá Molinos *et al.*, 2016; Jones & Cheung, 2015). These scenarios establish with high confidence that communities are modified because of the joint effect of loss of species and colonization by new species (i.e. species turnover). In addition, the projections highlight that expansion of species ranges are prevailing over species loss or range contraction, thus leading to a transient net local increase in richness, particularly around the 40-30°N line of latitude **(Figure 3.57)**.

3.5.3.3 Ecosystem extent and function

Across all marine systems and habitats, 21st century climate change and ocean acidification are projected to induce changes in extent and functioning of ecosystems. Most of the Eurasian Arctic Seas lie within today's seasonal ice zone. The general trend of "borealization" of the region is expected to continue (Fossheim *et al.*, 2015), inducing habitat gains and losses and a large species turnover; changes in phenology and production; substantial food web reorganizations; and changes in ecosystem functioning (Kortsch *et al.*, 2015; Larsen *et al.*, 2014).

In the Baltic Seas, maximum sea-ice cover is expected to decline by 75% under high climate change (SRES A1B) and by half under the most optimistic scenarios of climate change (B1) by the end of the 21st century. Melting sea ice

will decrease water salinity and the resulting warming and changes in water density are projected to promote instability in water stratification thereby reducing the areas of suboxic water (with < 2 ml/L of oxygen) (Neumann, 2010).

Kelp forest ecosystems (*Laminaria hyperborea*) are expected to expand to northern territories under all plausible climate change scenarios. This, coupled with significant loss of suitable habitats, is projected at low latitude range margins, including in areas where long-term persistence was inferred (e.g. north-western Iberia) (Assis *et al.*, 2016b), might have important consequences on the genetic diversity, and adaptive potential, of these habitat-structuring species (Assis *et al.*, 2018). A significant loss of maerl beds, dominated by coralline algae, is also predicted to occur by 2100 in the North East Atlantic, due to elevated pCO_2 (Brodie *et al.*, 2014).

Species range changes, phenological reactions, and variations in production, is expected to cause the Eurasian Arctic Seas ecosystem structure and functions to change (Larsen *et al.*, 2014).

Though primary production on ocean shelves is expected to increase (Hunt *et al.*, 2016), so far no unidirectional changes in the primary production in the individual Eurasian Arctic Seas have been observed. Reliable trends in its variation (increasing) were ascertained for the Barents, and Kara seas (Vetrov & Romankevich, 2011). There are two peaks in primary production in the Arctic Seas: spring ice algal peak and consecutive phytoplankton bloom. The role of the first one is expected to diminish; the timing of maximum phytoplankton production is expected to change and to influence the variability in time-lags between ice algal and phytoplankton peak production (from 45 to 90 days; Ji *et al.*, 2013, Kędra *et al.*, 2015). The frequency of mismatch between peak in demand from marine grazers and supply of their food, will increase. This will alter trophic flows throughout the food chain (Ji *et al.*, 2013). The spatio-temporal mismatch between the breeding season and the peak in food availability will potentially have a negative impact on seabird populations (Grémillet & Boulinier, 2009).

Phenological changes due to climate change and chemical changes have been already observed and further projected in the future. For instance, the decrease of anadromy prevalence of Arctic char (over 50% to the end of 21st century with high-levels of global warming, under the IPCC AR4, A2 emission scenario) because of the increase of lake and terrestrial catchments productivity (Finstad & Hein, 2012). Seasonal Cyanobacteria blooms in the Baltic are projected to to start earlier and last a month longer by the end of the 21st century (Neumann, 2010). Invasive species like the Pacific oyster, *Crassotrea gigas*, have also shown phenological changes. Specifically, reproductive effort and spawning periods are changing as a response to increased seawater and phytoplankton concentration (Thomas *et al.*, 2016).

Figure 3 57 **Differences between current (2006) and projected (2100) species richness (ΔRichness) based on models of 12,796 marine species from 23 phyla for IPCC RCP 4.5 Ⓐ and RCP 8.5 Ⓑ climate change scenarios.**

Source: García Molinos *et al.* (2016). Reprinted by permission from Macmillan Publishers Ltd: Nature Climate Change, copyright (2016).
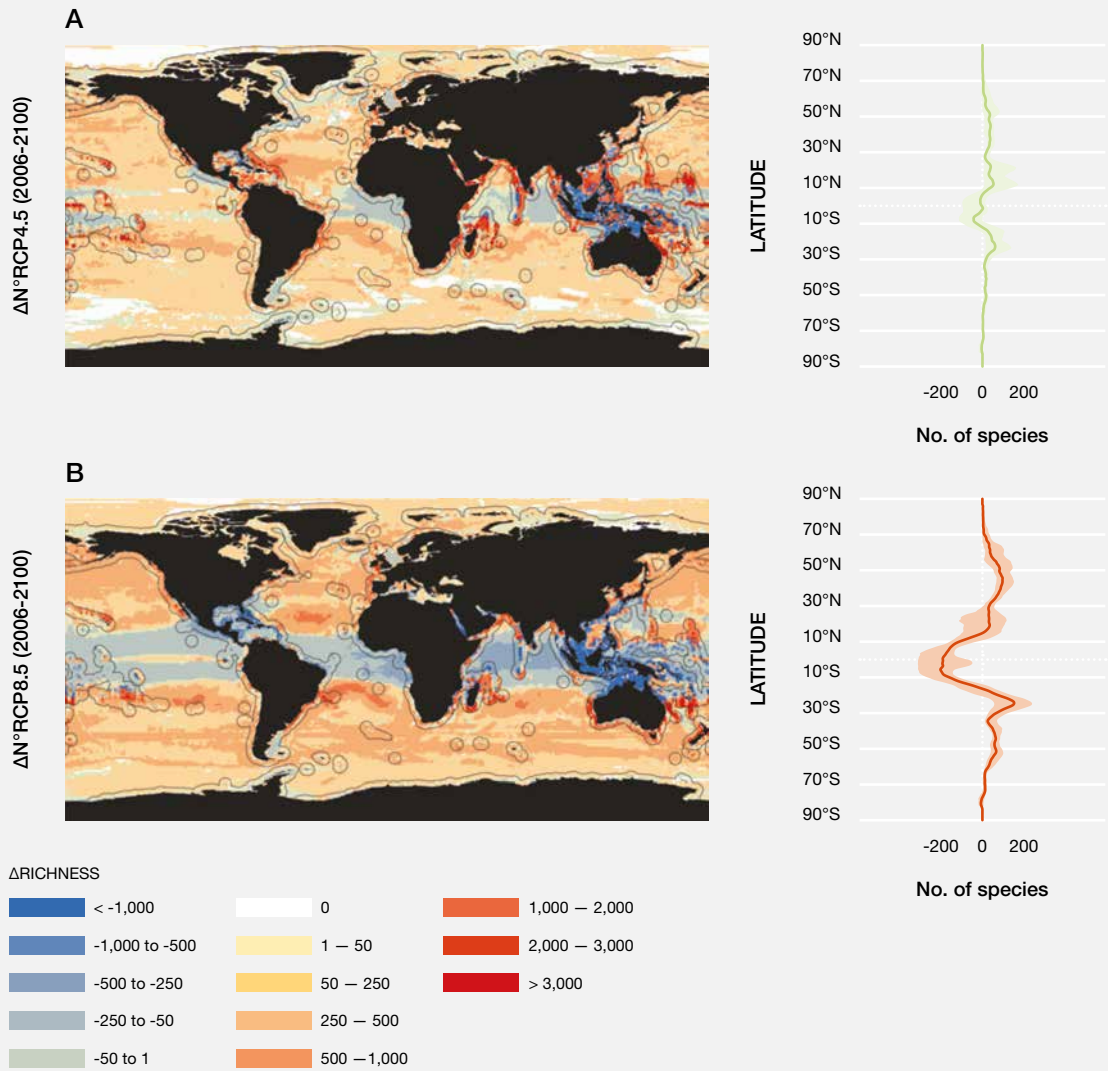
In the North East Atlantic and the North Sea, the projected general trends point to accelerating changes in ecosystem functioning, notably due to the effect of climate change on nutrient availability, and changes in timing of phytoplankton production, phytoplankton and zooplankton biomass, with cascading effects on the trophic network (Friocourt *et al.*, 2012 and examples in Soto, 2001). For instance, larval cod survival probability is declining by 22-44% in the North Atlantic, notably because of starvation effect due to food limitations (Kristiansen *et al.*, 2014). And the growth and weight of adult cod is also projected to be declining under IPCC Assessment Report scenario RCP 8.5 (highest green-house emission scenario for this assessment), because of physiological constraints (Butzin & Pörtner,

2016). Physiological processes as well as metabolic pathways will thus be modified as a response to climate change and ocean acidification. Responses may, however, be very different across taxa: for instance, autotrophs like seagrasses and many macroalgae are expected to display higher growth and photosynthetic rates under elevated pCO_2 (Koch *et al.*, 2013), whereas calcareous algae like maerl are likely to suffer from ocean acidification (Brodie *et al.*, 2014). Particularly well documented are changes in breeding phenology and success and the timing of migration of seabirds of the North East Atlantic (e.g. effect on breeding phenology; Frederiksen *et al.*, 2004). Among other documented changes are migration patterns. For instance, migration patterns of the North East Atlantic

mackerel are projected to change under moderate and high climate change scenarios (RCP 4.5 and 8.5, respectively). The outcome of these scenarios is that this living natural resource could expand in the near future.

3.5.3.4 Emerging drivers of change

Discovery of gas and oil fields across Europe and Central Asia, especially in the Arctic circle and the far north-east of the region (Sakhalin shelf and Kamchatka) pose a threat to terrestrial and marine biodiversity (Kontorovich *et al.*, 2013).

Enormous amounts of manganese, copper, nickel and cobalt are found on or beneath the seafloor (World Ocean Review, 2014). Demand for these resources are set to increase since they are needed for developing clean technologies, such as making wind turbines or hybrid cars. Deep-sea mining has not yet begun, mostly for technical reasons, but there has been an increase in the number of applications for mining contracts and it is estimated that by the end of 2017 there will be about 27 projects worldwide (Wedding *et al.*, 2015). Research to determine the impacts of deep sea mining has shown that deep-sea mining cannot be done without directly destroying habitats and species, resulting in biodiversity loss (Vanreusel *et al.*, 2016) and indirectly degrading large volumes of the water and seabed area with the polluted sediment plume it generates (Van Dover *et al.*, 2017). This mining requires enormous areas: a single 30-year operation license to mine metal-rich nodules will involve an area about the size of Austria. Most mining-induced loss of biodiversity in the deep sea will not recover for decades or centuries, given the very slow rates of recovery of many deep-sea species and ecosystems (Vanreusel, *et al.* 2016).

Shipping is expected to double by 2050, emphasizing the need for alternative shipping routes. Alternative routes are essential to minimize impacts caused by the increased threats from shipping accidents and oil spills (Kotta *et al.*, 2016).

With projected sea-ice declines, large swaths of Arctic Ocean will be opened up to shipping and fisheries (Jørgensen *et al.*, 2016a; Mullon *et al.*, 2016). This will cause additional pressure on the biodiversity of the region, speeding introductions of boreal fauna (Renaud *et al.*, 2015), and possibly reducing bottom complexity. Changes in advection are projected to accelerate the transboundary pollution effects increasing the number of contaminants in the food web (Jørgensen *et al.*, 2016a).

The continuing enlargement of the Suez Canal will allow greater cohorts of deeper living biota to enter the Mediterranean Sea, enhancing the risk of establishment and spread (Galil *et al.*, 2017). Increase in commercial shipping and recreational boating will enhance the introduction and secondary spread of non-native biota.

3.6 KNOWLEDGE GAPS

Knowledge gaps concern a) the full geographic (and temporal) coverage of past, current, and future trends of some ecosystem types and some taxa across Europe and Central Asia, b) patterns and underlying mechanisms of the biodiversity – ecosystem service relationship, and c) consideration of indigenous and local knowledge for all ecosystem types and taxa.

Geographic gaps

Overall, we found large gaps in knowledge on habitat extent and intactness, and species conservation status and trends for Eastern Europe and Central Asia. For instance, there is no systematic monitoring of plant and animal species across the range of these subregions. This is of particular concern given the size of these subregions and the diversity of habitat and species there. Outside the European Union long-term monitoring data is available almost exclusively for protected areas, which poses the risk of underestimating overall biodiversity trends in these regions.

Role of drivers

Information on future trends in biodiversity was predominantly focused on the impact of climate change, especially on plants and vertebrate species. There were very few studies investigating the impact of land-use change and even fewer investigating future projected impacts of pollution, invasive species, fishing and other drivers of change.

It was often impossible to quantify the relative role of drivers of change in determining trends in species and ecosystems. This was due to lack of synthetic studies on this subject and the limited ability to meta-analyze the literature to provide this evidence. Therefore, the attribution of drivers to trends was based on the qualitative expert assessment of the authors rather than on quantitative empirical evidence from experimental or quasi-experimental studies.

Marine systems

Most marine systems are hidden to human eye and therefore lack of visibility, knowledge gaps, and lack of concerted actions are regularly pointed out for marine systems (e.g. Allison & Bassett, 2015; Mccauley *et al.*, 2016).

Nevertheless, the rate of description of new marine species has been increasing, since 1955, at a higher rate than for terrestrial species (Appeltans *et al.*, 2012). Still, it is estimated

that between one-third and two-thirds of marine species are still to be described, with estimates of the total number falling in the range of 0.7 to 1 million (as compared to the 226,000 species currently described). Under-estimation of marine diversity is not restricted to remote and under-studied locations. It also holds in Europe and Central Asia, with the increasing discovery of cryptic species (i.e. species that are not, or are hardly, distinguished according to morphological criteria). This underestimation of marine diversity implies that the trends are incomplete for most marine taxa.

An important gap in knowledge regarding current as well as future changes is genetic responses to environmental changes. Only few taxa, among them fishes and algae, have been studied so far (e.g. Araújo *et al.*, 2016; Assis *et al.*, 2016a; Hutchinson *et al.*, 2003; Nicastro *et al.*, 2013), but these studies indicate changes in genetic diversity and genetic structure of marine species. Integration of a genetic component is of paramount importance for conservation of genetic resources as well as for modelling of future trends in marine biodiversity (Arrieta *et al.*, 2010; Gotelli & Stanton-Geddes, 2015).

Until recently, scant attention was paid to marine ecosystems and most marine taxa in conservation policies (e.g. see Habitats Directive and species lists in the European Union). Only a small number of species and few habitat types are included in Annex I of the Habitats Directive (EEA, 2015a). The gap in knowledge is exemplified by the large percentage of species in the "unknown" category in the first assessment of "good environmental status" in light of the newer Marine Strategy Framework Directive (2008) in the European Union **(Figure 3.58)**.

Most long-term marine datasets (since the 1950s) concern pelagic ecosystems (e.g. Beaugrand *et al.* 2002), intertidal rocky shores (e.g. Mieszkowska *et al.*, 2006), or specific taxa or taxonomic groups (in particular fishes, marine mammals or seabirds). Almost no data are available to document changes in subtidal rocky areas although they are rich in biodiversity and support key engineer species, for instance in subtidal kelp forests (Smale *et al.*, 2013).

Open ocean plankton communities are also poorly known. It is estimated that, in each litre of seawater, there are on average 10 billion organisms, including viruses, prokaryotes, unicellular eukaryotes, and metazoans.

The most notable knowledge gap in marine biodiversity for Europe and Central Asia is the lack of data on status and trends of biodiversity in deep-sea areas (>200 m) despite canyons, seamounts and other important deep-sea habitats and ecosystems being present in Europe and Central Asia Seas and Oceans. Less than 1% of the deep-sea floor (UNEP, 2007; Rogers *et al.*, 2015) and 0.4-4% of known seamounts (Kvile *et al.*, 2014) have been sampled. Those that are known are mainly areas with sandy bottoms that can be trawled. This highlights significant gaps in basic knowledge, including lack of baseline data on biodiversity, abundance and biomass and its spatial and temporal variations. New habitat types and species are still being discovered on almost every deep-sea scientific cruise.

Some progress in addressing these knowledge gaps is signified by recent marine assessments. For instance, an assessment of data available and surveys needed was recently reviewed for kelp in the North East Atlantic (Araújo

Figure ③ ㊺ **Knowledge and categorization of "good environmental status" in marine ecosystems of the European Union. Source: ETC/ICM (2014).**
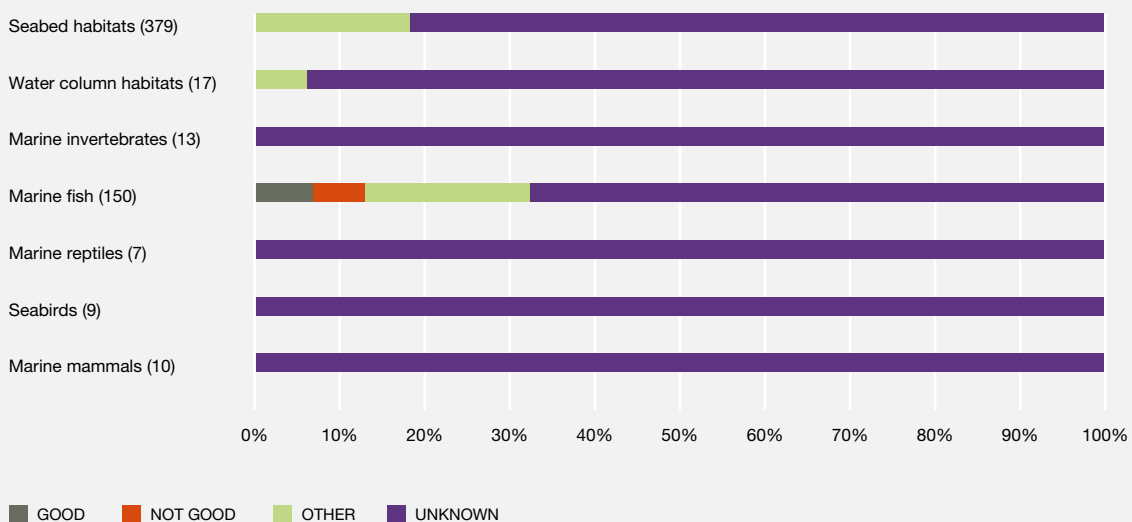
et al., 2016). The results from Tara Oceans and Malespina cruises and Ocean Sampling Day program, which collected genetic, morphological, and physico-chemical samples from stations around the world (about 35,000 biological samples and about 13,000 contextual measure taken a three different depths just for Tara Oceans) is now being analysed by a large international team of scientists. Metagenomes and meta-barcodes from stations are being built as well as quantitative and high-resolution image databases, and the first global studies are being published (e.g TARA Ocean (https://www.embl.de/tara-oceans/start/). IUCN recently coordinated an assessment dedicated to the Anthozoans of the Mediterranean Sea, which include, for instance, iconic species like the red coral (Otero *et al.*, 2017).

Freshwater systems

The chemical status of 40% of Europe's surface waters remains unknown (EEA, 2015d), considering that good chemical status was only achieved for all surface bodies in five of the 27 European Union member States, it is likely that the environmental conditions of some of these water bodies are poor.

Agricultural areas

Overall information on biodiversity trends in agricultural areas decreases from west to east. In particular, studies on biodiversity and agriculture for Eastern Europe and Central Asia often focus on drivers of biodiversity in agricultural areas rather than biodiversity trends (Smelansky, 2003), while biodiversity is surveyed for semi-natural ecosystems rather than more productive agroecosystems in these countries. Capacity building for monitoring biodiversity in agricultural areas in the eastern part of the region is thus needed.

The level of knowledge on biodiversity trends in agricultural areas and main direct drivers has increased substantially during the last decade. However, most studies have used species richness or abundance (and genetic diversity for animal breeds and plant varieties) as indicators of biodiversity. Promoting a stronger focus on functional diversity in future studies and monitoring schemes may be the best way to complement previous approaches. To better understand and predict biodiversity trends in agricultural areas in Europe and Central Asia, it will be necessary: (i) to reinforce the knowledge basis on the demography and population dynamics of species (including the role of behaviour, density-dependent effects, and extinction debt); (ii) to account for small-scale spatio-temporal effects and scale up biodiversity changes and trends from local to national and regional levels; and (iii) to detail the effects of changes in agricultural practices (characteristics of the varieties grown, harvesting techniques, types of pesticides used, etc.) to a greater extent (Kleijn *et al.*, 2011).

Urban areas

The data available for urban areas are mostly for the larger and more easily observed taxa, such as vascular plants, birds and mammals. There is good data for bats, and reasonably good data on amphibians, reptiles and some insect taxa, including butterflies. The small amount of data available on taxa more difficult to observe and distinguish, such as Syrphids and other Diptera, suggest high levels of diversity and numerous rare and threatened species (Kelcey, 2015). Thus, more surveying of such taxa would generate valuable new knowledge on urban biodiversity.

Taxonomic gaps

While birds are arguably the most studied and best known group in Europe and Central Asia, there is still one species, the large-billed reed-warbler, *Acrocephalus orinus* listed as being data deficient by the IUCN and therefore having unknown extinction risk, and there are also 79 species with unknown population trends in the European Union (EEA, 2015a). Long-term trends are rarely available. Low capacity or difficult access means that regions such as Caucasus, the Arctic part of Europe, Romania, Croatia, the Faroe Islands and the Azores are underrepresented in bird conservation status assessments (BirdLife International, 2015).

More substantial knowledge gaps exist for other terrestrial vertebrate groups. There are, respectively, 55 mammals, 11 reptiles and three amphibians that are classified as data deficient by the IUCN. In addition, population trends are unknown for 100 of 1,026 bird species extant in the region and assessed by IUCN as well as 263 of 537 mammals, 7 of 129 amphibians and 56 of the 268 species of reptiles (IUCN, 2017c).

There are at least 100,000 species of insects known in Europe, and an unknown number of earthworms, arachnids, snails and other invertebrate species. However, it is plausible that several hundreds of thousands of species of invertebrates occur in Europe and Central Asia. Despite this extremely high diversity, and importance for ecosystem services, only a very small proportion is listed in the IUCN Red List. More specifically, there are only 2,132 species of terrestrial invertebrates in the IUCN Red List that are extant in the Europe and Central Asia region. The majority of these are European bees, which include 1,965 species (Nieto *et al.*, 2014). Moreover, almost nothing is known about species, trends and threats for this taxonomic group from Central Asia.

There are no meaningful trends in geographic extent or population size of freshwater species available for Europe and Central Asia. Therefore, a table of trends and importance of drivers was impossible to produce. Of particular concern is the lack of data for freshwater

invertebrates, for which even current status is available only for a minority of species (EEA, 2010). For example, several freshwater crab species have data deficient status according to the IUCN Red List, which highlights the need to increase monitoring efforts globally but also in Europe and Central Asia.

Similarly, almost a quarter of all European freshwater molluscs are data deficient and many might prove to be threatened once enough data become available to evaluate their extinction risk. However, the number of data-deficient species may well increase, since 76% of freshwater fishes and 83% of freshwater molluscs have unknown population trends (Cuttelod *et al.*, 2011). Data are also deficient for many other freshwater invertebrate groups (Balian *et al.*, 2008). This is owing to several reasons such as lack of taxonomic information, knowledge gaps in geographical coverage of data and lack of long-term data. These gaps need to be assessed urgently, by fostering taxonomic research and monitoring and by making proprietary databases and databases under pay-wall freely and openly available.

Biases across taxonomic groups in marine systems are also largely documented (McCauley *et al.*, 2015; Poloczanska *et al.*, 2013) **(Figure 3.59)**. For instance, no extinction of marine animal species has been documented in the past

five decades (IUCN, 2017b), but only a small fraction of described marine mammals has been evaluated and 17 that were assessed were determined to be data deficient (IUCN, 2017c; McCauley *et al.*, 2015). This is exemplified by the extensive work carried out by Brooks *et al.* (2016) in which marine taxa are not included, except for decapods. This is not surprising, since trend data are not available even for 69% of the best-known group of marine organisms, the European marine fish species.

Availability of regional information on marine plankton and invertebrates is varied across Europe and Central Asia, with certain systems having more information on biodiversity status available (e.g. the North East Atlantic (OSPAR, 2017); the Mediterranean (Coll *et al.*, 2010a); and the Baltic (Ojaveer *et al.*, 2010). Most often, information remains descriptive: existence, abundance, geographical distributions of species for instance, but little meta-information is available yet to discern conservation status. OSPAR (2008) lists five marine invertebrate species as threatened or declining in the North Atlantic and North Sea since 2003, as well as a series of habitats formed by marine invertebrates (e.g. mussel beds, deep sea sponge aggregations). In the Mediterranean, while much information is available, marine invertebrate knowledge is often considered to be limited, with new species still being described. There is also a high proportion of endemic

Figure **3** **59** **Number of papers examining past and current trends in marine communities and ecosystems (total 73) in the Atlantic.**

The total number of species examined (total 440) in these papers (per taxonomic group) is indicated by the bars.
Source: Data extracted from raw data compiled by Poloczanska *et al.* (2013).
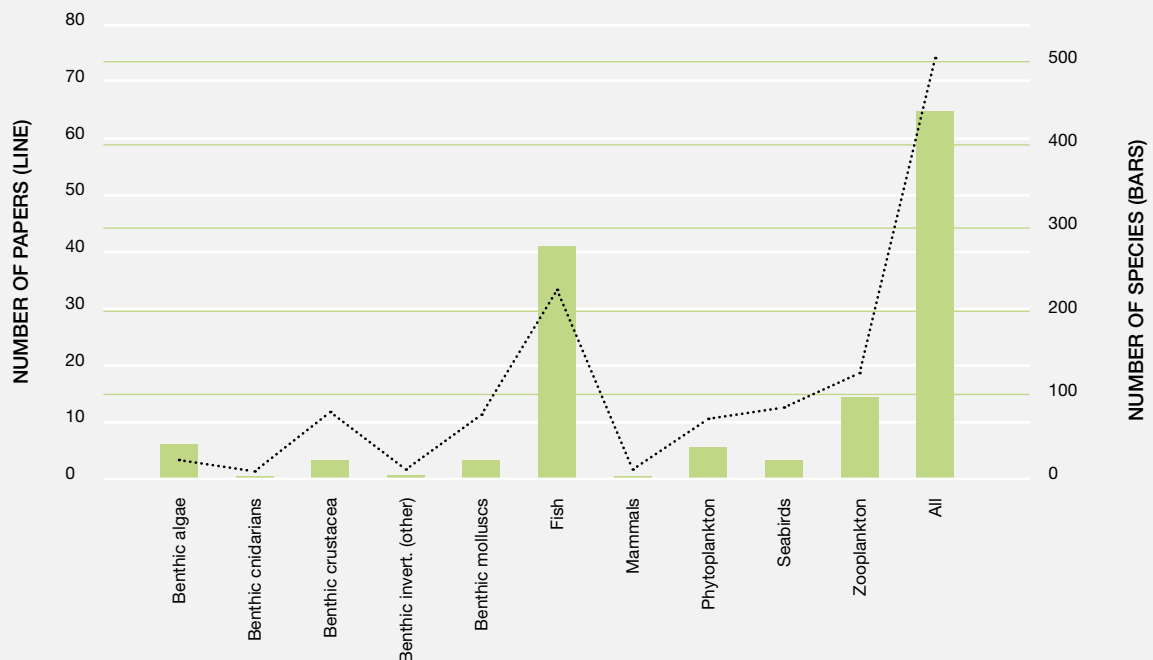
species in the Mediterranean, especially sponges and mysids (Coll *et al.*, 2010a). Mediterranean anthozoans have been reviewed in detail by IUCN, showing that 13% of them are threatened while almost half lack sufficient data for assessing risk of extinction (Otero *et al.*, 2017).

Marine microbes may represent more than 90% of the ocean's biomass, are the major drivers of its biogeochemical cycles (Danovaro *et al.*, 2017), and can be found in the whole water column up to 2,000 metres below the seafloor. Although there has been an exponential increase in research on marine archaea, bacteria and viruses, and evidence that archaea and viruses may increase in importance with depth (Danovaro *et al.*, 2015) their biodiversity and functioning is still largely unknown.

At least 7,000 species of lichens are known to occur in Europe (excluding Russia), while across the whole of Europe and Central Asia only five lichen species have been assessed in the IUCN Red List and have known conservation status (IUCN, 2017b).

Less than 10% of all species of vascular plants known to occur in the region have been assessed by the IUCN Red List (2,483 species for an estimated >30,000 for the region) (IUCN, 2017c). Among those assessed, 46.2% have unknown population trends. These also include species of conservation concern, such as 20% of the species included in the European Red List of Vascular Plants; (Bilz *et al.*, 2011). These knowledge gaps are caused by lack of field data, difficulties in accessing data for some countries, and uncertain taxonomy. Processes threatening vascular plants are also unknown for several species.

The number of fungus species in Europe exceeds 75,000, 15,000 of which are macrofungi (Senn-irlet *et al.*, 2007). Currently there are no regional or continental data on status and trends of fungi.

We were unable to assess status and trends in diversity, biomass and community composition of soil and freshwater micro-organisms: Protozoa, Bacteria, Rotifera, Nematoda, Tardigrada, despite the key role of these organisms in soil formation, nutrient and carbon cycling, and water retention (Orgiazzi *et al.*, 2016).

Relationship between biodiversity and ecosystem function and services

For some ecosystem services, there is insufficient data to evaluate the relationship between biodiversity and ecosystem service provision. For example, the effects of fish diversity on fisheries yield and the effects of biodiversity on flood regulation are inconclusive (Cardinale *et al.*, 2012). Additionally, ecosystem services provided by taxa other than plants are only beginning to be studied. Finally, the majority of studies reviewed focused on taxonomic diversity at the community level (i.e. species richness or diversity), rather than on intraspecific, functional phylogenetic diversity.

REFERENCES

Abasov, M. M., Ponomaryov, V. L., Nesterenkova, A. E., Loginov, A. N., & Fedosov, S. A. [Абасов, М. М., Пономарёв, В. Л., Нестеренкова, А. Э., Логинов, А. Н., & Федосов, С. А.]. (2016). Разработка мер интегрированной защиты самшита от самшитовой огнёвки [Integrated protection measures of *Buxus* against *Cydalima perspectalis*]. Сборник Научных Трудов Государственного Никитского ботанического сада [*Proceedings of Nikitsky Botanical Garden*], *142*, 102–113.

Abdul Malak, D. Livingstone, S. R., Pollard, D., Polidoro, B. A., Cuttelod, A., Bariche, M., Bilecenoglu, M., Carpenter, K. E., Collette, B. B., Francour, P., Goren, M., Kara, M. H., Massutí, E., Papaconstantinou, C., & Tunesi, L. (2011). *Overview of the conservation status of the marine fishes of the Mediterranean Sea*. Gland, Switzerland: IUCN.

Abdulla, A., Gomei, M., Maison, E., & Piante, C. (2008). *Status of marine protected areas in the Mediterranean Sea - A collaborative study by IUCN, WWF and MedPan*. Malaga, Spain: IUCN.

Abdurakhmanov, G. M., Krivolutsky, D. A., Mjalo, E. G., & Ogureeva, G. N. [Абдурахманов, Г. М., Криволуцкий, Д. А., Мяло, Е. Г., & Огуреева, Г. Н.]. (2003). Биогеография [*Biogeography*]. Moscow, Russian Federation: Publishing Centre "Academia".

Abell, R., Thieme, M. L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., Coad, B., Mandrak, N., Balderas, S. C., Bussing, W., Stiassny, M. L. J., Skelton, P., Allen, G. R., Unmack, P., Naseka, A., Ng, R., Sindorf, N., Robertson, J., Armijo, E., Higgins, J. V., Heibel, T. J., Wikramanayake, E., Olson, D., López, H. L., Reis, R. E., Lundberg, J. G., Sabaj Pérez, M. H., & Petry, P. (2008). Freshwater ecoregions of the world: A new map of biogeographic units for freshwater biodiversity conservation. *BioScience*, *58*(5), 403. https://doi.org/10.1641/B580507

Abellán, P., & Sánchez-Fernández, D. (2015). A gap analysis comparing the effectiveness of Natura 2000 and national protected area networks in representing European amphibians and reptiles. *Biodiversity and Conservation*, *24*(6), 1377–1390. https://doi.org/10.1007/s10531-015-0862-3

Adrianov, A. V., & Tarasov, V. G. [Адрианов, А. В., & Тарасов, В. Г.]. (2007). Современные проблемы экологической безопасности морских акваторий Дальнего Востока РФ [Modern problems of ecological safety of the marine areas of the Russian Far East]. In N. A. Dashko & V. G. Tarasov [Н. А. Дашко & В. Г. Тарасов] (Eds.), Динамика морских экосистем и современные проблемы сохранения биологического потенциала морей России [*Dynamics of marine ecosystems and modern problems of preservation of biological potential of Russian seas*] (pp. 177–194). Vladivostok, Russian Federation: Dalnauka.

Adrianov, A. V. [Адрианов, А. В.]. (2011). Экологическая безопасность дальневосточных морей России [Ecological safety of the Russian Far East seas]. Вестник Российской Академии Наук [*Herald of the Russian Academy of Sciences*], *81*(2), 111–119.

Aerts, R., & Heil, G. (Eds.). (2013). *Heathlands: patterns and processes in a changing environment* (*Vol. 20*). Dordrecht, The Netherlands: Springer Science & Business Media.

Airoldi, L., Beck, M. W. (2007). Loss, status and trends for coastal marine habitats of Europe. In R. N. Gibson, R. J. A. Atkinson, & J. D. M. Gordon, *Oceanography and marine biology, Volume 45* (pp. 345–405. Boca Raton, USA: CRC Press.

Akatov V. V., Akatova T. V., Trepet S.A., & Sirotiuk E. A. [Акатов, В. В., Акатова, Т. В., Трепет, С. А., & Сиротюк, Э. А.]. (2003). Туризм – новая угроза видовому разнообразию территории Всемирного природного наследия Западного Кавказа [Tourism – a new threat to species diversity in the territory of world natural heritage site western Caucasus]. In V. V. Kovalev, E. A. Sirotiuk, V. V. Akatov, & S. A. Trepet. [В. В. Ковалев, Э. А. Сиротюк, В. В. Акатов, & С. А. Трепет] (Eds.), Туризм в горных регионах: путь к устойчивому развитию [*Tourism in mountain regions: the way to sustainable development*] (pp. 121–130). Maikop, Russian Federation: Caucasus Reserve.

Akatov, V. V., Akatova, T. V., Varzareva, V. G., Dvoretskaya, E. V., Zagurnaya, Yu. S., Eskina, T. G., Ignatov, M. S., Ignatova, E. A., Kijashko, A. A., Konstantinova, N. A., Kuranova, N. G., Litvinskaya, S. A., Nagalevsky, M. V., Otte, V., Rezchikova, O. N., Sirotjuk, E. A., Timukhin, I. N., Tuniev, B. S., Urbanavichine, I. N., Urbanavichus, G. P., Chich, S. K., & Shadzhe, A. E. [Акатов, В. В., Акатова, Т. В., Варзарева, В. Г., Дворецкая, Е. В., Загурная, Ю. С., Ескина, Т. Г., Игнатов, М. С., Игнатова, Е. А., Кияшко, А. А., Константинова, Н. А., Куранова, Н. Г., Литвинская, С. А., Нагалевский, М. В., Отте, Ф., Резчикова, О. Н., Сиротюк, Э. А., Тимухин, И. Н., Туниев, Б. С., Урбанавичине, И. Н., Урбанавичюс, Г. П., Чич, С. К., & Шадже, Е. А.]. (2012). Красная книга Республики Адыгея. Редкие и находящиеся под угрозой исчезновения объекты животного и растительного мира. Часть 1. Введение. Растения и грибы. Издание второе. [*The Red Data Book of the Republic of Adygheya. Rare and threatened representatives of the regional fauna and flora. Part 1. Introduction. Vegetabilia and Mycota. Second edition*] A. S. Zamotaylov, E. A. Sirotyuk, T. V. Akatova, & O. N. Lipka [А. С. Замотайлов, Э. А. Сиротюк, Т. В. Акатова, & О. Н. Липка] (Eds.). Maykop, Russian Federation: Katchestvo.

Akhani, H., Djamali, M., Ghorbanalizadeh, A., & Ramezani, E. (2010). Plant biodiversity of Hyrcanian relict forests, N Iran: an overview of the flora, vegetation, palaeoecology and conservation. *Pakistan Journal of Botany*, *42*, 231–258.

Akzhygitova, N. I., Brekle, S.-W., Winkler, G., Volkova, E. A., Wucherer, W., Kurochkina, L. J., Makulbekova, G. B., Ogar, N. P., Rachkovskaya, E. I., Safronova, I. N., & Khramtsov, V. N. [Акжигитова, Н. И., Брекле, З.-В., Винклер, Г., Волкова, Е. А., Вухрер, В., Курочкина, Л. Я., Макулбекова, Г. Б., Огар, Н. П., Рачковская, Е. И., Сафронова, И. Н. & Храмцов, В. Н.]. (2003). In E. I. Rachkovskaya, E. A. Volkova, & V. N. Khramtsov [Е. И. Рачковская, Е. А. Волкова, & В. Н. Храмцов] (Eds.) Ботаническая география Казахстана

и Средней Азии (в пределах пустынной области) *[Botanical geography of Kazakhstan and Middle Asia (desert region)]*. St. Petersburg, Russian Federation.

Aladin, N., Chida, T., Cretaux, J.-F., Ermakhanov, Z., Jollibekov, B., Karimov, B., Kawabata, Y., Keyser, D., Kubota, J., Micklin, P., Mingazova, N., Plotnikov, I., & Toman, M. (2017). Current status of Lake Aral – challenges and future opportunities. In *Proceedings of the 16ᵗʰ World Lake Conference "Lake ecosystem health and its resilience: Diversity and risks of extinction", November 7-11th, 2016, Bali – Indonesia*.

Aladin N. V., & Plotnikov I. S. [Аладин, Н. В., & Плотников, И. С.]. (2008). Современная фауна остаточных водоемов, образовавшихся на месте бывшего Аральского моря [Modern fauna of residual water bodies formed on the place of the former Aral Sea]. Труды Зоологического Института РАН *[Proceedings of the Zoological Institute of the Russian Academy of Sciences]*, *312*(1/2), 145–154.

Alahuhta, J., Heino, J., & Luoto, M. (2011). Climate change and the future distributions of aquatic macrophytes across boreal catchments. *Journal of Biogeography*, *38*, 383–393. https://doi.org/10.1111/j.1365-2699.2010.02412.x

Alatalo, J. M., Jägerbrand, A. K., & Čuchta, P. (2015a). Collembola at three alpine subarctic sites resistant to twenty years of experimental warming. *Scientific Reports*, *5*, 18161. https://doi.org/10.1038/srep18161

Alatalo, J. M., Jägerbrand, A. K., & Molau, U. (2015b). Testing reliability of short-term responses to predict longer-term responses of bryophytes and lichens to environmental change. *Ecological Indicators*, *58*, 77–85. https://doi.org/10.1016/j.ecolind.2015.05.050

Albert, C. H., Thuiller, W., Lavorel, S., Davies, I. D., & Garbolino, E. (2008). Land-use change and subalpine tree dynamics: Colonization of *Larix decidua* in French subalpine grasslands. *Journal of Applied Ecology*, *45*(2), 659–669. https://doi.org/10.1111/j.1365-2664.2007.01416.x

Alcantara, C., Kuemmerle, T., Prishchepov, A. V., & Radeloff, V. C. (2012). Mapping abandoned agriculture

with multi-temporal MODIS satellite data. *Remote Sensing of Environment*, *124*, 334–347. https://doi.org/10.1016/j.rse.2012.05.019

Aleksandrova, V. D. (1970). The vegetation of the tundra zones in the USSR and data about its productivity. In W. A. Fuller & P. G. Kevan (Eds.), *Proceedings of the conference on productivity and conservation in northern circumpolar lands. Edmonton, Alberta 15 to 17 October 1969.*

Aleksandrova, V. D. [Александрова В. Д.] (1983). Растительность полярных пустынь СССР *[The vegetation of the Polar Deserts of the USSR]*. Leningrad, USSR: Наука [Science].

Aleynikov, A. A., Aleynikova, A. M., Bocharnikov, M. V., Glazov, P. M., Golovlev, P. P., Golovleva, V. O., Gruza, G. V., Dobrolyubova, K. O., Evina, A. I., Lipka, O. N., Zhbanova, P. I., Zamolodchikov, D. G., Zenin, E. A., Kalashnikova, Yu. A., Kozhin, M. N., Kokorin, A. O., Krylenko, I. V., Krylenko, I. N., Kushcheva, Yu. V., Miklyayev, I. A., Miklyayeva, I. M., Nikiforov, V. V., Pavlova, A. D., Postnova, A. I., Pukhova, M. A., Rankova, E. Ya., Stishov, M. S., Sutkaytis, O. K., Uvarov, S. A., Fomin, S. Yu., & Khokhlov, S. F. [Алейников А. А., Алейникова А. М., Бочарников М. В., Глазов П. М., Головлев П. П., Головлева В. О., Груза Г. В., Добролюбова К. О., Евина А. И., Жбанова П. И., Замолодчиков Д. Г., Зенин Е. А., Калашникова Ю. А., Кожин М. Н., Кокорин А. О., Крыленко И. В., Крыленко И. Н., Кущева Ю. В., Липка О. Н., Микляев И. А., Микляева И. М., Никикфоров В. В., Павлова А. Д., Постнова А. И., Пухова М. А., Ранькова Э. Я., Стишов М. С., Суткайтис О. К., Уваров С. А., Фомин С. Ю. & Хохлов С. Ф.]. (2014). In O. N. Lipka [О. Н. Липка] (Ed.), Остров Вайгач: природа, климат и человек *[Vaigach Island: nature, climate and people]*. Moscow, Russian Federation: WWF Russia. Retrieved from http://www.wwf.ru/resources/publ/book/964

Alikhonov, B. (Ed.). (2011). *Uzbekistan's nature. Devoted to 20ᵗʰ anniversary of the State independence of the Republic of the Uzbekistan*. Tashkent, Uzbekistan: Chinor ENK.

Aljes, M., Heinicke, T., & Zeitz, J. (2016). Peatland ecosystems in Kyrgyzstan: Distribution, peat characteristics and

a preliminary assessment of carbon storage. *Catena*, *144*, 56–64. https://doi.org/10.1016/j.catena.2016.04.021

Allan, E., Bossdorf, O., Dormann, C. F., Prati, D., Gossner, M. M., Tscharntke, T., Blüthgen, N., Bellach, M., Birkhofer, K., Boch, S., Böhm, S., Börschig, C., Chatzinotas, A., Christ, S., Daniel, R., Diekötter, T., Fischer, C., Friedl, T., Glaser, K., Hallmann, C., Hodac, L., Hölzel, N., Jung, K., Klein, A. M., Klaus, V. H., Kleinebecker, T., Krauss, J., Lange, M., Morris, E. K., Müller, J., Nacke, H., Pašalić, E., Rillig, M. C., Rothenwöhrer, C., Schall, P., Scherber, C., Schulze, W., Socher, S. A., Steckel, J., Steffan-Dewenter, I., Türke, M., Weiner, C. N., Werner, M., Westphal, C., Wolters, V., Wubet, T., Gockel, S., Gorke, M., Hemp, A., Renner, S. C., Schöning, I., Pfeiffer, S., König-Ries, B., Buscot, F., Linsenmair, K. E., Schulze, E.-D., Weisser, W. W., & Fischer, M. (2014). Interannual variation in land-use intensity enhances grassland multidiversity. *Proceedings of the National Academy of Sciences of the United States of America*, *111*(1), 308–313. https://doi.org/10.1073/pnas.1312213111

Allan, E., Manning, P., Alt, F., Binkenstein, J., Blaser, S., Blüthgen, N., Böhm, S., Grassein, F., Hölzel, N., Klaus, V. H., Kleinebecker, T., Morris, E. K., Oelmann, Y., Prati, D., Renner, S. C., Rillig, M. C., Schaefer, M., Schloter, M., Schmitt, B., Schöning, I., Schrumpf, M., Solly, E., Sorkau, E., Steckel, J., Steffen-Dewenter, I., Stempfhuber, B., Tschapka, M., Weiner, C. N., Weisser, W. W., Werner, M., Westphal, C., Wilcke, W., & Fischer, M. (2015). Land use intensification alters ecosystem multifunctionality via loss of biodiversity and changes to functional composition. *Ecology Letters*, *18*(8), 834–843. https://doi.org/10.1111/ele.12469

Allen, H. D. (2014). *Mediterranean ecogeography*. New York, USA: Routlege. Retrieved from http://www.amazon.co.uk/Mediterranean-Ecogeography-Series-Harriett-Allen/dp/0582404525

Allison, E. H., & Bassett, H. R. (2015). Climate change in the oceans: Human impacts and responses. *Science*, *350*(6262), 778–782. https://doi.org/10.1126/science.aac8721

Alomar, C., Deudero, S., Andaloro, F., Castriota, L., Consoli, P., Falautano, M., & Sinopoli, M. (2016). *Caulerpa cylindracea* Sonder invasion modifies trophic niche in infralittoral rocky benthic community. *Marine Environmental Research*, *120*, 86–92. https://doi.org/10.1016/j.marenvres.2016.07.010

Alsterberg, C., Roger, F., Sundbäck, K., Juhanson, J., Hulth, S., Hallin, S., & Gamfeldt, L. (2017). Habitat diversity and ecosystem multifunctionality — The importance of direct and indirect effects. *Science Advances, 3*(2): e1601475. https://doi.org/10.1126/sciadv.1601475

Altermatt, F., Alther, R., Fišer, C., Jokela, J., Konec, M., Küry, D., Machler, E., Stucki, P., & Westram, A. M. (2014) Diversity and distribution of freshwater amphipod species in Switzerland (Crustacea: Amphipoda). *PLoS ONE 9*(10): e110328. https://doi.org/10.1371/journal.pone.0110328

Altukhov, Yu. P. [Алтухов, Ю. П.]. (2003). Genetic processes in populations [Генетические процессы в популяциях]. Moscow, Russian Federation: РТС "Academkniga".

AMAP. (2012). *Arctic climate issues 2011: Changes in Arctic snow, water, ice and permafrost*. Retrieved May 22, 2016, from http://www.amap.no/documents/doc/arctic-climate-issues-2011-changes-in-arctic-snow-water-ice-and-permafrost/129

Amstrup, S. C., Marcot, B. G., & Douglas, D. C. (2008). A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. In E. T. DeWeaver, C. M. Bitz, & L.-B. Tremblay (Eds.), *Arctic sea ice decline: observations, projections, mechanisms, and implications* (pp. 213–268). Washington DC, USA: American Geophysical Union. https://doi.org/10.1029/180GM14

Ananjeva, N., & Agasyan, A. (2009). *Phrynocephalus horvathi*. The IUCN Red List of Threatened Species. https://doi.org/10.2305/IUCN.UK.2009.RLTS.T164759A5923724.en

Andersen, J. H., Dahl, K., Goke, C., Hartvig, M., Murray, C., Rindorf, A., Skov, H., Vinther, M., & Korpinen, S. (2014). Integrated assessment of marine biodiversity status using a prototype indicator-based assessment tool. *Frontiers in Marine Science*, *1*(55), 1–8. https://doi.org/10.3389/fmars.2014.00055

Andersen, J. H., Halpern, B. S., Korpinen, S., Murray, C., & Reker, J. (2015). Baltic Sea biodiversity status vs. cumulative human pressures. *Estuarine, Coastal and Shelf Science*, *161*, 88–92. https://doi.org/10.1016/j.ecss.2015.05.002

Anderson, B. J., Armsworth, P. R., Eigenbrod, F., Thomas, C. D., Gillings, S., Heinemeyer, A., Roy, D. B., & Gaston, K. J. (2009), Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology, 46*, 888–896. https://doi.org/10.1111/j.1365-2664.2009.01666.x

Anderson, S., Turiyev, B., & Bafti, S.S. (2009). *Phrynocephalus persicus*. The IUCN Red List of Threatened Species. https://doi.org/10.2305/IUCN.UK.2009.RLTS.T164647A5915480.en

Andreev, M. P., Ahti T., Voitsekhovitch, A. A., Gagarina, L. V., Himelbrant D. E., Davydov E. A., Konoreva L. A., Kuznetsova E. S., Makry T. V., Nadeina O. V., Randlane T., Saag A., Stepanchikova I. S., & Urbanavichus G. P. [Андреев, М. П., Ахти, Т., Войцехович, А. А., Гагарина, Л. В., Гимельбрант, Д. Е., Давыдов, Е. А., Конорева, Л. А., Кузнецова, Е. С., Макрый, Т. В., Надеина, О. В., Рандлане, Т., Сааг, А., Степанчикова, И. С., & Урбанавичюс, Г. П.]. (2014). Флора лишайников России: биология, экология, разнообразие, распространениеи методы изучения лишайников *[The lichen flora of Russia: biology, ecology, diversity, distribution and methods on studying lichens]*. M. P. Andreev & D. E. Himelbrant (Eds.). Moscow, Russian Federation: KMK.

Angeler, D. G., & Johnson, R. K. (2012). Temporal scales and patterns of invertebrate biodiversity dynamics in boreal lakes recovering from acidification. *Ecological Applications*, *22*(4), 1172–1186. http://doi.org/10.1890/11-1474.1

Anker, Y., Rosenthal, E., Shulman, H., & Flexer, A. (2009). Runoff geochemical evolution of the hypersaline lower Jordan valley basin. *Israel Journal of Earth Sciences*, *58*(1), 41–61. https://doi.org/10.1560/IJES.58.1.41

Antonchikov, A. N. (2005). A review of the conservation status of steppe birds of the northern part of the Eastern Palearctic. In G. Bota, M. B. Morales, & S. Manosa (Eds.), *Ecology and Conservation of Steppe-land birds*. Barcelona, Spain: Lynx Edicions.

Antonov, N. P., Klovatch, N. V., Orlov, A. M., Datsky, A. V., Lepskaya, V. A., Kuznetsov, V. V., Yarzhombek, A. A., Abramov, A. A., Alekseev D. O., Moiseev S. I., Evseeva N. A., & Sologub D. O [Антонов Н. П., Кловач Н. В., Орлов А. М., Датский А. В., Лепская В. А., Кузнецов В. В., Яржомбек А. А., Абрамов А. А., Алексеев Д. О., Моисеев С. И., Евсеева Н. А., & Сологуб Д. О.]. (2013). Рыболовство в Дальневосточном рыбохозяйственном бассейне в 2013 г. [Fishing in the Russian Far East fishery basin in 2013]. Труды ВНИРО *[Proceedings of VNIRO]*, *160*, 133–211.

Appeltans, W., Ahyong, S. T., Anderson, G., Angel, M. V., Artois, T., Bailly, N., Bamber, R., Barber, A., Bartsch, I., Berta, A., Błażewicz-Paszkowycz, M., Bock, P., Boxshall, G., Boyko, C. B., Brandão, S. N., Bray, R. A., Bruce, N. L., Cairns, S. D., Chan, T.-Y., Cheng, L., Collins, A. G., Cribb, T., Curini-Galletti, M., Dahdouh-Guebas, F., Davie, P. J. F., Dawson, M. N., De Clerck, O., Decock, W., De Grave, S., de Voogd, N. J., Domning, D. P., Emig, C. C., Erséus, C., Eschmeyer, W., Fauchald, K., Fautin, D. G., Feist, S. W., Fransen, C. H. J. M., Furuya, H., Garcia-Alvarez, O., Gerken, S., Gibson, D., Gittenberger, A., Gofas, S., Gómez-Daglio, L., Gordon, D. P., Guiry, M. D., Hernandez, F., Hoeksema, B. W., Hopcroft, R. R., Jaume, D., Kirk, P., Koedam, N., Koenemann, S., Kolb, J. B., Kristensen, R. M., Kroh, A., Lambert, G., Lazarus, D. B., Lemaitre, R., Longshaw, M., Lowry, J., Macpherson, E., Madin, L. P., Mah, C., Mapstone, G., McLaughlin, P. A., Mees, J., Meland, K., Messing, C. G., Mills, C. E., Molodtsova, T. N., Mooi, R., Neuhaus, B., Ng, P. K. L., Nielsen, C., Norenburg, J., Opresko, D. M., Osawa, M., Paulay, G., Perrin, W., Pilger, J. F., Poore, G. C. B., Pugh, P., Read, G. B., Reimer, J. D., Rius, M., Rocha, R. M., Saiz-Salinas, J. I., Scarabino, V., Schierwater, B., Schmidt-Rhaesa, A., Schnabel, K. E., Schotte, M., Schuchert, P., Schwabe, E., Segers,

H., Self-Sullivan, C., Shenkar, N., Siegel, V., Sterrer, W., Stöhr, S., Swalla, B., Tasker, M. L., Thuesen, E. V., Timm, T., Todaro, M. A., Turon, X., Tyler, S., Uetz, P., van der Land, J., Vanhoorne, B., van Ofwegen, L. P., van Soest, R. W. M., Vanaverbeke, J., Walker-Smith, G., Walter, T. C., Warren, A., Williams, G. C., Wilson, S. P., & Costello, M. J. (2012). The magnitude of global marine species diversity. *Current Biology*, *22*(23), 2189–2202. https://doi.org/10.1016/j.cub.2012.09.036

Aptroot, A., & van Herk, C. M. (2007). Further evidence of the effects of global warming on lichens, particularly those with *Trentepohlia* phycobionts. *Environmental Pollution*, *146*(2), 293–298. https://doi.org/10.1016/j.envpol.2006.03.018

Aptroot, A., van Herk, C. M., van Dobben, H. F., van den Boom, P. P. G., Brand, A. M., & Spier, L. (1998). Rode lijst van Nederlandse korstmossen [Red List of the lichens of the Netherlands]. *Buxbaumiella*, *46*, 1–101.

Araújo, M. B., Thuiller, W., & Pearson, R. G. (2006). Climate warming and the decline of amphibians and reptiles in Europe. *Journal of Biogeography*, *33*(10), 1712–1728. https://doi.org/10.1111/j.1365-2699.2006.01482.x

Araújo, R. M., Assis, J., Aguillar, R., Airoldi, L., Bárbara, I., Bartsch, I., Bekkby, T., Christie, H., Davoult, D., Derrien-Courtel, S., Fernandez, C., Fredriksen, S., Gevaert, F., Gundersen, H., Le Gal, A., Lévêque, L., Mieszkowska, N., Norderhaug, K. M., Oliveira, P., Puente, A., Rico, J. M., Rinde, E., Schubert, H., Strain, E. M., Valero, M., Viard, F, & Sousa-Pinto, I. (2016). Status, trends and drivers of kelp forests in Europe: an expert assessment. *Biodiversity and Conservation*, *25*(7), 1319–1348. https://doi.org/10.1007/s10531-016-1141-7

Arctic Council. (2013). *Arctic Resilience Interim Report 2013*. Stockholm, Sweden: Stockholm Environment Institute.

Arheimer, B., Andréasson, J., Fogelberg, S., Johnsson, H., Pers, C. B, & Persson, K. (2005). Climate Change Impact on Water Quality: Model Results from Southern Sweden. *Ambio, 34*(7), 559-566. https://doi.org/10.1579/0044-7447-34.7.559

Arianoutsou, M. (2001). Landscape changes in Mediterranean ecosystems of Greece: implications for fire and biodiversity issues. *Journal of Mediterranean Ecology*, *2*, 165–178.

Arizaga, J., & Laso, M. (2015). A quantification of illegal hunting of birds in Gipuzkoa (north of Spain). *European Journal of Wildlife Research*, *61*(5), 795–799. https://doi.org/10.1007/s10344-015-0940-6

Armitage, H. F., Britton, A. J., van der Wal, R., & Woodin, S. J. (2014). The relative importance of nitrogen deposition as a driver of Racomitrium heath species composition and richness across Europe. *Biological Conservation*, *171*, 224-231. https://doi.org/10.1016/j.biocon.2014.01.039

Arnolds, E. (1991). Decline of ectomycorrhizal fungi in Europe. *Agriculture, Ecosystems and Environment*, *35*(2–3), 209–244. https://doi.org/10.1016/0167-8809(91)90052-Y

Arnolds, E. (2001). The future of fungi in Europe: threats, conservation and management. In D. Moore, M. Nauta, S. Evans, & M. Rotheroe (Eds.), *Fungal conservation, issues and solutions* (pp. 64–80). Cambridge, UK: Cambridge University Press.

Arnolds, E. (2010). The fate of hydnoid fungi in The Netherlands and northwestern Europe. *Fungal Ecology*, *3*(2), 81–88. http://doi.org/10.1016/j.funeco.2009.05.005

Arntzen, J.W., Denoël, M., Miaud, C., Andreone, F., Vogrin, M., Edgar, P., Crnobrnja Isailovic, J., Ajtic, R., & Corti, C. (2009). *Proteus anguinus*. The IUCN Red List of Threatened Species. http://dx.doi.org/10.2305/IUCN.UK.2009.RLTS.T18377A8173419.en

Arrieta, J. M., Arnaud-Haond, S., & Duarte, C. M. (2010). What lies underneath: conserving the oceans' genetic resources. *Proceedings of the National Academy of Sciences of the United States of America*, *107*(43), 18318–24. https://doi.org/10.1073/pnas.0911897107

Artemov, I. A., Bliakharchuk, T. A. , Bykov, N. I., Vasilchenko, A. A. , Vinogradov, V. V., Vlasenko, V. I., Gurov, A. V., Insarov, G. E., Kariakin, I. V., Knorre, A. A., Koshkarov, A. D., Koshkarova, V. L., Kuprianov, A. N., Larin, S. I., Lysanova, G. I., Mikhailov, A. Yu., Mikhailov, N. N., Ostanin, O. V., Paltsyn, M. Yu., Parfenova, E. I., Smirnov, M. N., Sukhova, M. G., Timoshkin, V. B., Kharlamova, N. F., Chebakova, N. M., Cherhykh, D. V., Shishkin, A. S., & Shmakin A. B. [Артемов, И. А., Бляхарчук, Т. А., Быков, Н. И., Васильченко, А. А., Виноградов, В. В., Власенко, В. И., Гуров, А. В., Инсаров, Г. Э., Карякин, И. В., Кноре, А. А., Кошкаров, А. Д., Кошкарова, В. Л., Куприянов, А. Н., Ларин, С. И., Лысанова, Г. И., Михайлов, А. Ю., Михайлов, Н. Н., Останин, О. В., Пальцын, М. Ю., Парфенова, Е. И., Смирнов, М. Н., Сухова, М. Г., Тимошкин, В. Б., Харламова, Н. Ф., Чебакова, Н. М., Черных, Д. В., Шишкин, А. С., & Шмакин, А.Б. (2013). In N. N. Mikhailov [Н. Н.Михайлов] (Ed.), Изменение климата и биоразнообразие в российской части Алтае-Саянского экорегиона [Climate change and biodiversity in the Russian part of the Altay-Sayan ecoregion]. Krasnoyarsk, Russian Federation: UNDP.

Artyukhin, Y. B., & Burkanov, V. N., [Артюхин, Ю. Б., & Бурканов, В. Н.] (1999). Морские птицы и млекопитающие Дальнего Востока России: полевой определитель *[Marine birds and mammals of the Russian Far East: a field guide]*. Moscow, Russian Federation: ACT [AST].

Assis, J., Araújo, M. B., & Serrão, E. A. (2018). Projected climate changes threaten ancient refugia of kelp forests in the North Atlantic. *Global Change Biology*, *24*(1), e55-e66. https://doi.org/10.1111/gcb.13818

Assis, J., Coelho, N. C., Lamy, T., Valero, M., Alberto, F., & Serrão, E. A. (2016a). Deep reefs are climatic refugia for genetic diversity of marine forests. *Journal of Biogeography*, *43*(4), 833–844. https://doi.org/10.1111/jbi.12677

Assis, J., Lucas, A. V, Bárbara, I., & Serrão, E. A. (2016b). Future climate change is predicted to shift long-term persistence zones in the cold-temperate kelp Laminaria hyperborea. *Marine Environmental Research, 113*, 174-182. https://doi.org/10.1016/j.marenvres.2015.11.005

Asykulov T., & Chordonova N. [Асыкулов, Т., & Чордонова, Н.]. (2015). Состояние орехово-плодовых лесов Кыргызстана [Status of walnut-fruit forests of Kyrgyzstan]. Наука, Новые Технологии И Инновации *[Science, New Technologies and Innovations]*, 10, 4–7.

Aude, E., & Ejrnæs, R. (2005). Bryophyte colonisation in experimental microcosms: The role of nutrients, defoliation and vascular vegetation. *Oikos*, *109*(2), 323–330. https://doi.org/10.1111/j.0030-1299.2005.13268.x

Augustine, D. J., & Mcnaughton, S. J. (1998). Ungulate effects on the functional species composition of plant communities: Herbivore selectivity and plant tolerance. *The Journal of Wildlife Management*, *62*(4), 1165–1183. https://doi.org/10.2307/3801981

Aukema, B., Rieger, C., & Rabitsch, W. (2013). *Catalogue of the Heteroptera of the palaearctic region, Volume 6: Supplement*. Amsterdam, The Netherlands: The Netherlands Entomology Society.

Aunins, A., & Martin, G. (2014). *Biodiversity assessment of MARMONI project areas*.

Azovsky, A. I. [Азовский, А. И.]. (1989). Нишевая структура сообщества морских псаммофильных инфузорий. I. Расположение ниш в пространстве ресурсов [Niche community structure of marine psammophilous ciliates. I. Location of niches in space of resources]. Журнал общей биологии *[Journal of General Biology]*, 50(3), 329–341.

Babai, D., & Molnár, Z. (2014). Small-scale traditional management of highly species-rich grasslands in the Carpathians. *Agriculture, Ecosystems & Environment*, *182*, 123–130. https://doi.org/https://doi.org/10.1016/j.agee.2013.08.018

Badino, G. (2004). Cave temperatures and global climatic change. *International Journal of Speleology*, *33*, 103–113. https://doi.org/10.5038/1827-806X.33.1.10

Bagella, S., Gascón, S., Filigheddu, R., Cogoni, A., & Boix, D. (2016). Mediterranean temporary ponds: new challenges from a neglected habitat.

Hydrobiologia, *782*(1), 1–10. https://doi.org/10.1007/s10750-016-2962-9

Bagin, A. M., Melnikov, B. P., & Tambiev, S. B. (2011). *Diagnostic analysis of the environment of the Arctic zone of the Russian Federation* (extended summary). B. A. Morgunov (Ed.). Moscow, Russian Federation: Scientific World.

Bai, Y. G., Broersma, K., Thompson, D., & Ross, T. J. (2004). Landscape-level dynamics of grassland-forest transitions in British Columbia. *Journal of Range Management*, *57*(1), 66–75. https://doi.org/10.2307/4003956

Bailey, D., Collins, M., Gordon, J. D. M., Zuur, A., & Priede, I. G. (2009). Long-term changes in deep-water fish populations in the northeast Atlantic: a deeper reaching effect of fisheries? *Proceedings of the Royal Society B: Biological Sciences, 276,* 1965-1969. https://doi.org/10.1098/rspb.2009.0098

Balata, D., Piazzi, L., & Bulleri, F. (2015). Sediment deposition dampens positive effects of substratum complexity on the diversity of macroalgal assemblages. *Journal of Experimental Marine Biology and Ecology*, *467*, 45–51. https://doi.org/10.1016/j.jembe.2015.03.005

Baldrighi, E., Giovannelli, D., D'Errico, G., Lavaleye, M., & Manini, E. (2017). Exploring the relationship between macrofaunal biodiversity and ecosystem functioning in the deep sea. *Frontiers in Marine Science*, *4*, 198. https://doi.org/10.3389/fmars.2017.00198

Balian, E. V., Segers, H., Lévêque, C., & Martens, K. (2008). The freshwater animal diversity assessment: An overview of the results. *Hydrobiologia*, *595*(1), 627–637. https://doi.org/10.1007/s10750-007-9246-3

Bálint, M., Domisch, S., Engelhardt, C. H. M., Haase, P., Lehrian, S., Sauer, J., Theissinger, K., Pauls, S. U., & Nowak, C. (2011) Cryptic biodiversity loss linked to global climate change. *Nature Climate Change, 1*(6), 313–318. https://doi.org/10.1038/NCLIMATE1191

BalticSTERN. (2013). *State of the Baltic Sea: background paper*.

Balushkina, E. V., Golubkov, S. M., Golubkov, M. S., Litvinchuk, L. F., & Shadrin, N. V. [Балушкина, Е. В., Голубков, С. М., Голубков, М. С., Литвинчук, Л. Ф., & Шадрин, Н. В.]. (2008). Влияние абиотических и биотических факторов на структурно-функциональную организацию экосистем соленых озер Крыма [Effect of abiotic and biotic factors on the structural and functional organization of the saline lake ecosystems in Crimea]. Журнал Общей Биологии *[Journal of General Biology]*, *70*(6), 504–514.

Balvanera P., Siddique I., Dee L., Paquette A., Isbell F., Gonzalez A., Byrnes J., O'Connor M. I., Hungate B. A., & Griffin J. N. (2014). Linking biodiversity and ecosystem services: Current uncertainties and the necessary next steps. *BioScience, 64*(1), 49–57. https://doi.org/10.1093/biosci/bit003

Bannikov, A. G., & Zhirnov, L. V. (1971). The Bokharan deer in the USSR. *Oryx*, *11*(1), 50-61. http://doi.org/10.1017/S0030605300009443

Barannik, V., Borysova, O., & Stolberg, F. (2004). The Caspian Sea region: Environmental change. *Ambio*, *33*(1), 45–51. https://doi.org/10.1579/0044-7447-33.1.45

Barbet-Massin, M., & Jetz, W. (2014). A 40-year, continent-wide, multispecies assessment of relevant climate predictors for species distribution modelling. *Diversity and Distributions*, *20*(11), 1285–1295. https://doi.org/10.1111/ddi.12229

Barbet-Massin, M., Thuiller, W., & Jiguet, F. (2012). The fate of European breeding birds under climate, land-use and dispersal scenarios. *Global Change Biology*. https://doi.org/10.1111/j.1365-2486.2011.02552.x

Barceló, C., Ciannelli, L., Olsen, E. M., Johannessen, T., & Knutsen, H. (2016). Eight decades of sampling reveal a contemporary novel fish assemblage in coastal nursery habitats. *Global Change Biology*, *22*(3), 1155–1167. https://doi.org/10.1111/gcb.13047

Bardat, J., & Aubert, M. (2007). Impact of forest management on the diversity of corticolous bryophyte assemblages in temperate forests. *Biological Conservation*, *139*(1), 47–66. https://doi.org/10.1016/j.biocon.2007.06.004

Bardgett, R. D, & van der Putten, W. H. (2014). Belowground biodiversity and ecosystem functioning. *Nature, 515,* 505–511. https://doi.org/10.1038/nature13855

Barenboim, G. M., Danilov-Danilyan, V. I., Gelfan, A. N., & Motovilov, Y. G. (2013). On the problems of water quality in Russia and some approaches to their solution. In B. Arheimer, A. Collins, V. Krysanova, E. Lakshmanan, M. Meybeck, & M. Stone. (Eds.), *Understanding freshwater problems in a changing world, Volume 361* (pp. 77-86). Retrieved from https://istina.msu.ru/media/publications/article/817/fb7/24998371/RedBookr361_end.pdf

Bargmann, T., Hatteland, B. A., & Grytnes, J.-A. (2015). Effects of prescribed burning on carabid beetle diversity in coastal anthropogenic heathlands. *Biodiversity and Conservation*, *24*(10), 2565–2581. http://doi.org/10.1007/s10531-015-0945-1

Barnett, E.A., Fletcher, M.R., Hunter, K., & Sharp, E.A. (2006). *Pesticide poisoning of animals 2005: Investigations of suspected incidents in the United Kingdom. Report of the environmental panel of the advisory committee on pesticides.* London, UK: Defra.

Bassin, S., Volk, M., Suter, M., Buchmann, N., & Fuhrer, J. (2007). Nitrogen deposition but not ozone affects productivity and community composition of subalpine grassland after 3 yr of treatment. *New Phytologist*, *175*(3), 523–534. https://doi.org/10.1111/j.1469-8137.2007.02140.x

Batary, P., Baldi, A., Kleijn, D., & Tscharntke, T. (2011). Landscape-moderated biodiversity effects of agri-environmental management: a meta-analysis. *Proceedings of the Royal Society B: Biological Sciences*, *278*(1713), 1894–1902. http://doi.org/10.1098/rspb.2010.1923

Batáry, P., Dicks, L. V., Kleijn, D., & Sutherland, W. J. (2015). The role of agri-environment schemes in conservation and environmental management. *Conservation Biology*, *29*(4), 1006–1016. http://doi.org/10.1111/cobi.12536

Bates, J., & Farmer, A. (1992). *Bryophytes and lichens in a changing environment*. Oxford, England: Clarendon Press.

Beaugrand, G. (2004). The North Sea regime shift: Evidence, causes, mechanisms and consequences. *Progress in Oceanography*, *60*(2–4), 245–262. https://doi.org/10.1016/j.pocean.2004.02.018

Beaugrand, G., Edwards, M., Raybaud, V., Goberville, E., & Kirby, R. R. (2015). Future vulnerability of marine biodiversity compared with contemporary and past changes. *Nature Climate Change*, *5*(7), 695–701. https://doi.org/10.1038/nclimate2650

Beaugrand, G., Luczak, C., & Edwards, M. (2009). Rapid biogeographical plankton shifts in the North Atlantic Ocean. *Global Change Biology*, *15*(7), 1790–1803. https://doi.org/10.1111/j.1365-2486.2009.01848.x

Beaugrand, G., McQuatters-Gollop, A., Edwards, M., & Goberville, E. (2013). Long-term responses of North Atlantic calcifying plankton to climate change. *Nature Climate Change*, *3*(3), 263–267. https://doi.org/10.1038/nclimate1753

Beaugrand, G., Reid, P. C., Ibañez, F., Lindley, J. A., & Edwards, M. (2002). Reorganization of North Atlantic marine copepod biodiversity and climate. *Science*, *296*(2002), 1692–1694. https://doi.org/10.1126/science.1071329

Beebee, T. J. C., & Griffiths, R. A. (2005). The amphibian decline crisis: A watershed for conservation biology? *Biological Conservation 125*(3), 271-285. https://doi.org/10.1016/j.biocon.2005.04.009

Belan, T. A. (2003). Benthos abundance pattern and species composition in conditions of pollution in Amursky Bay (the Peter the Great Bay, the Sea of Japan). *Marine Pollution Bulletin*, *46*(9), 1111–1119. https://doi.org/10.1016/S0025-326X(03)00242-X

Belimov, G. T., & Sedalishchev, V. T. [Белимов Г. Т.*,* & Седалищев В. Т.]. (1980). Озерная лягушка (*Rana ridibunda*) (Amphibia, Anura) в водоемах Якутска [The marsh frog, *Rana ridibunda* (Amphibia, Anura). Water bodies of the city of Yakutsk]. *Vestnik Zoologii [Journal of Zoology]*, *3*, 74–75.

Belonovskaya, E. A. (1995). The human-induced transformation of the ecosystems of the Caucasus Mountains. In *EURO-MAB IV. "Mountain zonality facing global change". Conf. Papers 21* (pp. 41–57). Warszawa: IgiPZ PAN.

Benejam, L., Mello, F. T., Meerhoff, M., Loureiro, M., Jeppesen, E., & Brucet, S. (2016). Assessing effects of change in land use on size-related variables of fish in subtropical streams. *Canadian Journal of Fisheries and Aquatic Sciences*, *73*, 547–556. https://doi.org/10.1139/cjfas-2015-0025

Bengtsson, J., Ahnström, J., & Weibull, A. C. (2005). The effects of organic agriculture on biodiversity and abundance: A meta-analysis. *Journal of Applied Ecology*, *42*(2), 261–269. http://doi.org/10.1111/j.1365-2664.2005.01005.x

Benito Garzon, M., Sanchez de Dios, R., & Sainz Ollero, H. (2008). Effects of climate change on the distribution of Iberian tree species. *Applied Vegetation Science*, *11*(2), 169–178. https://doi.org/10.3170/2008-7-18348

Benton, T. G., Bryant, D. M., Cole, L., & Crick, H. Q. P. (2002). Linking agricultural practice to insect and bird populations: A historical study over three decades. *Journal of Applied Ecology*, *39*(4), 673–687. https://doi.org/10.1046/j.1365-2664.2002.00745.x

Berendse, F., Chamberlain, D., Kleijn, D., & Schekkerman, H. (2004). Declining biodiversity in agricultural landscapes and the effectiveness of agri-environment schemes. *Ambio*, *33*(8), 499–502. http://doi.org/10.1579/0044-7447-33.8.499

Berg, L. S. (1949). *Freshwater fishes of the USSR and adjacent countries, part. 2.* Moskva–Leningrad: Izdatel'stvo Akademii Nauk SSSR.

Bergamini, A., & Pauli, D. (2001). Effects of increased nutrient supply on bryophytes in montane calcareous fens. *Journal of Bryology*, *23*(4), 331–339. https://doi.org/10.1179/jbr.2001.23.4.331

Bergamini, A., Peintinger, M., Fakheran, S., Moradi, H., Schmid, B., & Joshi, J. (2009). Loss of habitat specialists despite conservation management in fen remnants 1995-2006. *Perspectives in Plant Ecology, Evolution and Systematics*, *11*(1), 65–79. https://doi.org/10.1016/j.ppees.2008.10.001

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Bergamini, A., Peintinger, M., Schmid, B., & Urmi, E. (2001). Effects of management and altitude on bryophyte species diversity and composition in montane calcareous fens. *Flora*, *196*(3), 180–193. https://doi.org/0367-2530/01/196/03-180

Bergamini, A., Ungricht, S., & Hofmann, H. (2009). An elevational shift of cryophilous bryophytes in the last century - An effect of climate warming? *Diversity and Distributions*, *15*(5), 871–879. https://doi.org/10.1111/j.1472-4642.2009.00595.x

Berseneva, L. A. [Берсенева, Л. А.]. (2006). Климаты аридной зоны Азии *[Climates of Asia's arid zone]*. In P. D. Gunin [П. Д. Гунин] (Ed.), *Biological resources and natural conditions of Mongolia: Proceedings of the joint Russian-Mongolian complex biological expedition Vol. 46*. Moscow, Russian Federation: Nauka.

Bertocci, I., Araújo, R., Oliveira, P., & Sousa-Pinto, I. (2015). Potential effects of kelp species on local fisheries. *Journal of Applied Ecology*, *52*, 1216–1226. https://doi.org/10.1111/1365-2664.12483

Bespalova, L. A. [Беспалова, Л. А.]. (2016). Экологическая диагностика и оценка устойчивости ландшафтной структуры Азовского моря *[Environmental diagnosis and assessment of the sustainability of the landscape structure of the Azov Sea]*. Rostov-on-Don, Russian Federation: Rostov University.

Bhatt, U. S., Walker, D. A., Raynolds, M. K., Comiso, J. C., Epstein, H. E., Jia, G. S., Gens, R., Pinzon, J. E., Tucker, C. J., Tweedie, C. E., & Webber, P. J. (2010). Circumpolar Arctic tundra vegetation change is linked to sea ice decline. *Earth Interactions*, *14*, 8. http://doi.org/10.1175/2010EI315.1

Bianchi, C. N., Morri, C., Chiantore, M., Montefalcone, M., Parravicini, V., & Rovere, A. (2012). Mediterranean Sea biodiversity between the legacy from the past and a future of change. In N. Stambler (Ed.), *Life in the Mediterranean Sea: A look at habitat changes* (pp. 1-55). New York, USA: Nova Science Publishers. Retrieved from https://www.novapublishers.com/catalog/product_info.php?products_id=21851

Biermann, R., & Daniels, F. (1997). Changes in a lichen-rich dry sand grassland vegetation with special reference to lichen synusiae and *Campylopus introflexus*. *Phytocoenologia*, *27*(2), 257–273. https://doi.org/10.1127/phyto/27/1997/257

Biesmeijer, J. C., Roberts, S. P. M., Reemer, M., Ohlemüller, R., Edwards, M., Peeters, T., Schaffers, A. P., Potts, S. G., Kleukers, R., Thomas, C. D., Settele, J., & Kunin, W. E. (2006). Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. *Science*, *313*(5785), 351–354. https://doi.org/10.1126/science.1127863

Bikirov, Sh. B. [Бикиров, Ш. Б.]. (2012). Лесные ресурсы Западного Тянь-Шаня [Forest resources of the western Tian-Shan]. Хвойные Бореальной Зоны *[Conifers of the Boreal Zone]*, *33*(3–4), 220–223.

Bilandžija, H., Čuković, T., & Puljas, S. (2014). *Protokol praćenja stanja vrsta Congeria kusceri Bole, 1962 i Congeria jalzici Morton & Bilandžija, 2013 u Republici Hrvatskoj. [State monitoring protocol of the species Congeria kusceri Bole, 1962 and Congeria jalzici Morton and Bilandžija, 2013 in the Republic of Croatia]*.

Billeter, R., Liira, J., Bailey, D., Bugter, R., Arens, P., Augenstein, I., Aviron, S., Baudry, J., Bukacek, R., Burel, F., Cerny, M., De Blust, G., De Cock, R., Diekötter, T., Dietz, H., Dirksen, J., Dormann, C., Durka, W., Frenzel, M., Hamersky, R., Hendrickx, F., Herzog, F., Klotz, S., Koolstra, B., Lausch, A., Le Coeur, D., Maelfait, J. P., Opdam, P., Roubalova, M., Schermann, A., Schermann, N., Schmidt, T., Schweiger, O., Smulders, M. J. M., Speelmans, M., Simova, P., Verboom, J., Van Wingerden, W. K. R. E., Zobel, M., & Edwards, P. J. (2008). Indicators for biodiversity in agricultural landscapes: A pan-European study. *Journal of Applied Ecology*, *45*(1), 141–150. https://doi.org/10.1111/j.1365-2664.2007.01393.x

Bilz, M., Kell, S. P., Maxted, N., & Lansdown, R. V. (2011). *European red list of vascular plants*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/8515

Birchenough, S. N. R., Reiss, H., Degraer, S., Mieszkowska, N., Borja,

Á., Buhl-Mortensen, L., Braeckman, U., Craeymeersch, J., De Mesel, I., Kerckhof, F., Krönke, I., Parra, S., Rabaut, M., Schröder, A., Van Colen, C., Van Hoey, G., Vincx, M., & Wätjen, K. (2015). Climate change and marine benthos: a review of existing research and future directions in the North Atlantic. *Wiley Interdisciplinary Reviews - Climate Change*, *6*(2), 203–223. https://doi.org/10.1002/wcc.330

BirdLife International. (2008). *Palearctic-African migratory birds have suffered substantial declines.* Retrieved July 26, 2017, from http://datazone.birdlife.org/sowb/casestudy/palearctic-african-migratory-birds-have-suffered-substantial-declines

BirdLife International. (2015). *European red list of birds.* Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/975810

BirdLife International. (2016). *Data zone*. Retrieved from http://datazone.birdlife.org/

BirdLife International. (2017). *European birds of conservation concern: populations, trends and national responsibilities*. Cambridge, UK.

Bliss, L.C., Heal, O.W., & Moore, J.J. (Eds.). (1981). *Tundra ecosystems: A comparative analysis*. Cambride, UK: Cambridge University Press.

Bliss, L. C., & Matveyeva. (1992). Circumpolar Arctic vegetation. In F. S. Chaplin III (Ed.), *Arctic ecosystems in a changing climate* (pp. 59-90). San Diego, USA: Academic Press, Inc.

Blois, J. L., Zarnetske, P. L., Fitzpatrick, M. C., & Finnegan, S. (2013). Climate change and the past, present, and future of biotic interactions. *Science*, *341*(6145), 499–504. https://doi.org/10.1126/science.1237184

Blondel, J., Aronson, J., Bodiou, J. Y., & Boeuf, G. (2010). *The Mediterranean region. Biological diversity in space and time.* New York, USA: Oxford University Press.

Blüthgen. N., Simons, N. K., Jung, K., Prati, D., Renner, S. C., Boch, S., Fischer, M., Hölzel, N., Klaus, V.

H., Kleinebecker, T., Tschapka, M., Weisser, W. W., & Gossner, M. M. (2016). Land use imperils plant and animal community stability through changes in asynchrony rather than diversity. *Nature Communications 7,* 10697. https://doi.org/10.1038/ncomms10697

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., Emmett, B., Erisman, J.- W., Fenn, M., Gilliam, F., Nordin, A., Pardo, L., & De Vries, W. (2010). Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecological Applications*, *20*(1), 30–59. https://doi.org/10.1890/08-1140.1

Boch, S., Prati, D., Schöning, I., & Fischer, M. (2016). Lichen species richness is highest in non-intensively used grasslands promoting suitable microhabitats and low vascular plant competition. *Biodiversity and Conservation*, *25*(2), 225–238. https://doi.org/10.1007/s10531-015-1037-y

Bockheim, J. G. & Tarnocai, C. (1998). Nature, occurrence and origin of dry permafrost. *International Conference on Permafrost, Yellowknife* (*Canada*)*, Collection Nordicana, 55,* 57-63.

Boero, F. (2013). Review of jellyfish blooms in the Mediterranean and Black Sea. *Studies and Reviews. General Fisheries Commission for the Mediterranean*, 92.

Bogatov, V. V, & Fedorovskiy, A. S. (2016). Freshwater ecosystems of the southern region of the Russian Far East are undergoing extreme environmental change. *Knowledge and Management of Aquatic Ecosystems*, *417,* 34. https://doi.org/10.1051/kmae/2016021

Böhm, M., Collen, B., Baillie, J., Bowles, P., Chanson, J., Cox, N., Hammerson, G., Hoffmann, M., Livingstone, S. R., Ram, M., Rhodin, A. G. J., Stuart, S. N., van Dijk, P. P., Young, B. E., Afuang, L. E., Aghasyan, A., García, A., Aguilar, C., Ajtic, R., Akarsu, F., Alencar, L. R. V, Allison, A., Ananjeva, N., Anderson, S., Andrén, C., Ariano-Sánchez, D., Arredondo, J. C., Auliya, M., Austin, C. C., Avci, A., Baker, P. J., Barreto-Lima, A. F., Barrio-Amorós, C. L., Basu, D., Bates, M. F., Batistella, A.,

Bauer, A., Bennett, D., Böhme, W., Broadley, D., Brown, R., Burgess, J., Captain, A., Carreira, S., Castañeda, M. del R., Castro, F., Catenazzi, A., Cedeño-Vázquez, J. R., Chapple, D. G., Cheylan, M., Cisneros-Heredia, D. F., Cogalniceanu, D., Cogger, H., Corti, C., Costa, G. C., Couper, P. J., Courtney, T., Crnobrnja-Isailovic, J., Crochet, P. A., Crother, B., Cruz, F., Daltry, J. C., Daniels, R. J. R., Das, I., de Silva, A., Diesmos, A. C., Dirksen, L., Doan, T. M., Dodd, C. K., Doody, J. S., Dorcas, M. E., Duarte de Barros Filho, J., Egan, V. T., El Mouden, E. H., Embert, D., Espinoza, R. E., Fallabrino, A., Feng, X., Feng, Z. J., Fitzgerald, L., Flores-Villela, O., França, F. G. R., Frost, D., Gadsden, H., Gamble, T., Ganesh, S. R., Garcia, M. A., García-Pérez, J. E., Gatus, J., Gaulke, M., Geniez, P., Georges, A., Gerlach, J., Goldberg, S., Gonzalez, J. C. T., Gower, D. J., Grant, T., Greenbaum, E., Grieco, C., Guo, P., Hamilton, A. M., Hare, K., Hedges, S. B., Heideman, N., Hilton-Taylor, C., Hitchmough, R., Hollingsworth, B., Hutchinson, M., Ineich, I., Iverson, J., Jaksic, F. M., Jenkins, R., Joger, U., Jose, R., Kaska, Y., Kaya, U., Keogh, J. S., Köhler, G., Kuchling, G., Kumlutas, Y., Kwet, A., La Marca, E., Lamar, W., Lane, A., Lardner, B., Latta, C., Latta, G., Lau, M., Lavin, P., Lawson, D., LeBreton, M., Lehr, E., Limpus, D., Lipczynski, N., Lobo, A. S., López-Luna, M. A., Luiselli, L., Lukoschek, V., Lundberg, M., Lymberakis, P., Macey, R., Magnusson, W. E., Mahler, D. L., Malhotra, A., Mariaux, J., Maritz, B., Marques, O. A. V, Márquez, R., Martins, M., Masterson, G., Mateo, J. A., Mathew, R., Mathews, N., Mayer, G., McCranie, J. R., Measey, G. J., Mendoza-Quijano, F., Menegon, M., Métrailler, S., Milton, D. A., Montgomery, C., Morato, S. A. A., Mott, T., Muñoz-Alonso, A., Murphy, J., Nguyen, T. Q., Nilson, G., Nogueira, C., Núñez, H., Orlov, N., Ota, H., Ottenwalder, J., Papenfuss, T., Pasachnik, S., Passos, P., Pauwels, O. S. G., Pérez-Buitrago, N., Pérez-Mellado, V., Pianka, E. R., Pleguezuelos, J., Pollock, C., Ponce-Campos, P., Powell, R., Pupin, F., Quintero Diaz, G. E., Radder, R., Ramer, J., Rasmussen, A. R., Raxworthy, C., Reynolds, R., Richman, N., Rico, E. L., Riservato, E., Rivas, G., da Rocha, P. L. B., Rödel, M. O., Rodríguez Schettino, L., Roosenburg, W. M., Ross, J. P.,

Sadek, R., Sanders, K., Santos-Barrera, G., Schleich, H. H., Schmidt, B. R., Schmitz, A., Sharifi, M., Shea, G., Shi, H. T., Shine, R., Sindaco, R., Slimani, T., Somaweera, R., Spawls, S., Stafford, P., Stuebing, R., Sweet, S., Sy, E., Temple, H. J., Tognelli, M. F., Tolley, K., Tolson, P. J., Tuniyev, B., Tuniyev, S., Üzüm, N., van Buurt, G., Van Sluys, M., Velasco, A., Vences, M., Vesely, M., Vinke, S., Vinke, T., Vogel, G., Vogrin, M., Vogt, R. C., Wearn, O. R., Werner, Y. L., Whiting, M. J., Wiewandt, T., Wilkinson, J., Wilson, B., Wren, S., Zamin, T., Zhou, K., & Zug, G. (2013). The conservation status of the world's reptiles. *Biological Conservation, 157,* 372-385. https://doi.org/10.1016/j.biocon.2012.07.015

Bohn, U., Gollub, G., Hettwer, C., Neuhäuslová, Z., Raus, T., Schlüter, H., & Weber, H. (2000). *Karte der natürlichen Vegetation Europas, Maßstab 1: 2 500 000 [Map of the natural vegetation of Europe. Scale 1: 2 500 000]*. Bonn, Germany: Bundesamt für Naturschutz (BfN) / Federal Agency for Nature Conservation.

Bohn, U., Gollub, G., Hettwer, C., Neuhäuslová, Z., Raus, T., Sch lüter, H., Weber, H. & Hennekens, S. (2004). *Karte der natürlichen Vegetation Europas, Maßstab 1: 2 500 000 [Map of the natural vegetation of Europe. Scale 1: 2 500 000]*. Interactive CD-ROM: explanatory text, legend, maps. Bonn, Germany: Bundesamt für Naturschutz (BfN) / Federal Agency for Nature Conservation.

Boix, D., Kneitel, J., Robson, B. J., Duchet, C., Zúñiga, L., Day, J., Gascón, S., Sala, J., Quintana, X. D., & Blaustein, L. (2016). Invertebrates of freshwater temporary ponds in Mediterranean climates. In D. Batzer & D. Boix (Eds.), *Invertebrates in freshwater wetlands* (pp. 141-189). Cham, Switzerland: Springer. https://doi.org/10.1007/978-3-319-24978-0

Boll, T., Levi, E. E., Bezirci, G., Özuluğ, M., Tavşanoğlu, Ü. N., Çakıroğlu, A. İ., Özcan, S., Brucet, S., Jeppesen, E., & Beklioğlu, M. (2016). Fish assemblage and diversity in lakes of western and central Turkey: role of geo-climatic and other environmental variables. *Hydrobiologia*, *771*(1), 31–44. https://doi.org/10.1007/s10750-015-2608-3

Bolnick, D. I., Amarasekare, P., Araújo, M. S., Bürger, R., Levine, J. M., Novak, M., Rudolf, V. H. W., Schreiber, S. J., Urban, M. C., & Vasseur, D. (2011). Why intraspecific trait variation matters in community ecology. *Trends in Ecology and Evolution 26,*183-192 https://doi.org/10.1016/j.tree.2011.01.009

Bolnyk, S. I., & Vengerov, P. D. [Больных, С. И., & Венгеров, П. Д.]. (2011). Динамика фауны и Популяции птиц на залежных в лесостепи и степных зонах [Dynamics of fauna and the population of birds on the fallow lands in forest-steppe and steppe zones]. *Scientific statements of the Belgorod state University. Series: Natural Sciences, 15*(9), 104. Retrieved from https://cyberleninka.ru/article/v/dinamika-fauny-i-naseleniya-ptits-na-zalezhah-v-lesostepnoy-i-stepnoy-zonah

Bologa, A. S., & Sava, D. (2012). Present state and evolution trends of biodiversity in the Black Sea: decline and restoration. *Journal of the Black Sea/Mediterranean Environment*, *18*(2), 144–154. Retrieved from http://www.blackmeditjournal.org/pdf/vol18no2pdf5.pdf

Bond E. M. & Chase J. M. (2002). Biodiversity and ecosystem functioning at local and regional spatial scales. *Ecology Letters, 5*(4), 467–470. https://doi.org/10.1046/j.1461-0248.2002.00350.x

Bonn, A., Allott, T., Evans, M., Joosten, H., & Stoneman, R. (2016). *Peatland restoration and ecosystem services: Science, policy and practice*. Cambridge, UK: Cambridge University Press.

Bonnin, I., Bonneuil, C., Goffaux, R., Montalent, P., & Goldringer, I. (2014). Explaining the decrease in the genetic diversity of wheat in France over the 20th century. *Agriculture, Ecosystems and Environment*, *195*, 183–192. https://doi.org/10.1016/j.agee.2014.06.003

Boomer, I., Aladin, N., Plotnikov, I., & Whatley, R. (2000). The palaeolimnology of the Aral Sea: a review. *Quaternary Science Reviews*, *19*(13), 1259–1278. https://doi.org/10.1016/S0277-3791(00)00002-0

Borisov, V. V., Scheblykina, L. S. & Uriadova, L. P. [Борисов, В. В., Щеблыкина, Л. С., & Урядова, Л. П.]. (2014). The dynamics of the species composition

and bird habitats on abandoned arable lands with different degrees of overgrowth [Динамика видового состава и структура населения птиц Заброшенных пашен с разной степенью их зарастания]. *Bulletin of the Pskov State University. Series: Natural and Physico-mathematical Sciences, 5.* Retrieved from https://cyberleninka.ru/article/v/dinamika-vidovogo-sostava-i-struktury-naseleniya-ptits-zabroshennyh-pashen-s-raznoy-stepenyu-ih-zarastaniya

Bornand, C., Gygax, A., Juillerat, P., Jutzi, M., Möhl, A., Rometsch, S., Sager, L., Santiago, H, & Eggenberg, S. (2016). *Rote Liste Gefässpflanzen. Gefährdete Arten der Schweiz* (*Vol. 1621*) *[Red List of vascular plants. Endangered species in Switzerland (Vol. 1621)]*. Bern, Switzerland: Bundesamt für Umwelt and Geneva, Switzerland: Info Flora.

Boros, E., Ecsedi, Z., & Oláh, J. (2013). *Ecology and management of soda pans in the Carpathian Basin*. Balmazújváros, Hungary: Hortobágy Environmental Association.

Borsch, T., Dürbye, T., Gasparyan, A., Akhalkatsi, M., Henkel, M., Korotkova, N., Maharromova, E. Mansion, G. & Stevens, A.-D. (2014). *Caucasus. Plant diversity between the Black and Caspian Seas*. G. Parolly, K. Grotz, W. Lack, & N. J. Turland (Eds.). Berlin, Germany: Botanischer Garten und Botanisches Museum Berlin-Dahlem.

Boudouresque, C. F., Bernard, G., Pergent, G., Shili, A., & Verlaque, M. (2009). Regression of Mediterranean seagrasses caused by natural processes and anthropogenic disturbances and stress: A critical review. *Botanica Marina*, *52*(5), 395–418. https://doi.org/10.1515/BOT.2009.057

Brasier, C. M., & Buck, K. W. (2001). Rapid evolutionary changes in a globally invading fungal pathogen (Dutch elm disease). *Biological Invasions*, *3*(3), 223–233. https://doi.org/10.1023/A:1015248819864

Breckle, S. -W., & Wucherer, W. (2006). Vegetation of the Pamir (Tajikistan): Land use and desertification problems. In E. M. Spehn, M. Liberman, & C. Körner (Eds.), *Land use change and mountain biodiversity*

(pp. 225–238). Boca Raton, USA: CRC Press.

Breckle, S.-W., Wucherer, W., Dimeyeva, L., & Ogar, N. (Eds.). (2012). The Aralkum, a man-made desert on the desiccated floor of the Aral Sea (Central Asia). Berlin, Germany: Springer-Verlag. https://doi.org/10.1007/978-3-642-21117-1_1

Breeze, T. D., Vaissière, B. E., Bommarco, R., Petanidou, T., Seraphides, N., Kozák, L., Scheper, J., Biesmeijer, J. C., Kleijn, D., Gyldenkærne, S., Moretti, M., Holzschuh, A., Steffan-Dewenter, I., Stout, J. C., Pärtel, M., Zobel, M., & Potts, S. G. (2014). Agricultural policies exacerbate honeybee pollination service supply-demand mismatches across Europe. *PLoS ONE*, *9*(1), e82996. https://doi.org/10.1371/journal.pone.0082996

Brinkert, A., Hölzel, N., Sidorova, T. V., & Kamp, J. (2016). Spontaneous steppe restoration on abandoned cropland in Kazakhstan: grazing affects successional pathways. *Biodiversity and Conservation* *25*(12), 2543-2561. https://doi.org/10.1007/s10531-015-1020-7

Britton, A. J., Beale, C. M., Towers, W., & Hewison, R. L. (2009). Biodiversity gains and losses: evidence for homogenisation of Scottish alpine vegetation. *Biological Conservation*, *142*(8), 1728–1739. https://doi.org/10.1016/j.biocon.2009.03.010

Britton, A. J., & Fisher, J. M. (2010). Terricolous alpine lichens are sensitive to both load and concentration of applied nitrogen and have potential as bioindicators of nitrogen deposition. *Environmental Pollution*, *158*(5), 1296–1302. https://doi.org/10.1016/j.envpol.2010.01.015

Britton, A. J., Hester, A. J., Hewison, R. L., Potts, J. M., & Ross, L. C. (2017). Climate, pollution and grazing drive long-term change in moorland habitats. *Applied Vegetation Science*, *20*(2), 194–203. http://doi.org/10.1111/avsc.12260

Brochet, A.-L., Van den Bosschen, W., Jbour, S., Ndang'ang'a, P. K., Jones, V. R., Abdou, W. A. L. I., Al-Hmoud, A. R., Asswad, N. G., Atienza, J. C., Atrash, I., Barbara, N., Bensusan, K., Bino, T., Celada, C., Cherkaoui, S. I.,

Costa, J., Deceuninck, B., Etayeb, K. S., Feltrup-Azafzaf, C., Figelj, J., Gustin, M., Kmecl, P., Kocevski, V., Korbetti, M., Kotrosan, D., Mula Laguna, J., Lattuada, M., Leitão, D., Lopes, P., López-Jiménez, N., Lucić, V., Micol, T., Moali, A., Perlman, Y., Piludu, N., Portolou, D., Putilin, K., Quaintenne, G., Ramadan-Jaradi, G., Ružić, M., Sandor, A., Sarajli, N., Saveljic, D., Sheldon, R. D., Shialis, T., Tsiopelas, N., Vargas, F., Thompson, C., Brunner, A., Grimmett, R., & Butchart, S. H. M. (2016). Preliminary assessment of the scope and scale of illegal killing and taking of birds in the Mediterranean. *Bird Conservation International*, *26*(1), 1–28. https://doi.org/10.1017/S0959270915000416

Brodie, J., Williamson, C. J., Smale, D. A., Kamenos, N. A., Mieszkowska, N., Santos, R., Cunliffe, M., Steinke, M., Yesson, C., Anderson, K. M., Asnaghi, V., Brownlee, C., Burdett, H. L., Burrows, M. T., Collins, S., Donohue, P. J. C., Harvey, B., Foggo, A., Noisette, F., Nunes, J., Ragazzola, F., Raven, J. A., Schmidt, D. N., Suggett, D., Teichberg, M., & Hall-Spencer, J. M. (2014). The future of the northeast Atlantic benthic flora in a high CO_2 world. *Ecology and Evolution*, *4*(13), 2787–2798. https://doi.org/10.1002/ece3.1105

Broggi, M. F. & Waldburger, E. (1984). *Rote Liste der gefährdeter und seltenen Gefässpflanzen- arten des Fürstentums Liechtenstein [Red List of endangered and rare vascular plants. Types of the Principality of Liechtenstein]*. Liechtenstein: Naturkundliche Forschung im Fürstentum.

Brönmark, C., & Hansson, L.-A. (2002). Environmental issues in lakes and ponds: current state and perspectives. *Environmental Conservation*, *29*(3), 290–307. https://doi.org/10.1017/S0376892902000218

Brooks, T. M., Akçakaya, H. R., Burgess, N. D., Butchart, S. H. M., Hilton-Taylor, C., Hoffmann, M., Juffe-Bignoli, D., Kingston, N., MacSharry, B., Parr, M., Perianin, L., Regan, E. C., Rodrigues, A. S. L., Rondinini, C., Shennan-Farpon, Y., & Young, B. E. (2016). Analysing biodiversity and conservation knowledge products to support regional environmental assessments. *Scientific Data*, *3*, 160007. http://doi.org/10.1038/sdata.2016.7

Brown, L. E., Hannah, D. M., & Milner, A. M. (2007). Vulnerability of alpine stream biodiversity to shrinking glaciers and snowpacks. *Global Change Biology*, *13*(5), 958–966. https://doi.org/10.1111/j.1365-2486.2007.01341.x

Brucet, S., Boix, D., Gascón, S., Sala, J., Quintana, X. D., Badosa, A., Sondergaard, M., Lauridsen, T. L., & Jeppesen, E. (2009). Species richness of crustacean zooplankton and trophic structure of brackish lagoons in contrasting climate zones: North temperate Denmark and Mediterranean Catalonia (Spain). *Ecography*, *32*(4), 692–702. https://doi.org/10.1111/j.1600-0587.2009.05823.x

Brucet, S., Boix, D., Nathansen, L. W., Quintana, X. D., Jensen, E., Balayla, D., Meerhoff, M., & Jeppesen, E. (2012). Effects of temperature, salinity and fish in structuring the macroinvertebrate community in shallow lakes: implications for effects of climate change. *PLoS ONE*, *7*(2), e30877. https://doi.org/10.1371/journal.pone.0030877

Brucet, S., Boix, D., Quintana, X. D., Jensen, E., Nathansen, L. W., Trochine, C., Meerhoff, M., Gascon, S., & Jeppesen, E. (2010). Factors influencing zooplankton size structure at contrasting temperatures in coastal shallow lakes: Implications for effects of climate change. *Limnology and Oceanography*, *55*(4), 1697–1711. https://doi.org/10.4319/lo.2010.55.4.1697

Brucet, S., Pédron, S., Mehner, T., Lauridsen, T. L., Argillier, C., Winfield, I. J., Volta, P., Emmrich, M., Hesthagen, T., Holmgren, K., Benejam, L., Kelly, F., Krause, T., Palm, A., Rask, M., & Jeppesen, E. (2013). Fish diversity in European lakes: Geographical factors dominate over anthropogenic pressures. *Freshwater Biology*, *58*(9), 1779–1793. https://doi.org/10.1111/fwb.12167

Bruchmann, I. (2011). *Plant endemism in Europe: spatial distribution and habitat affinities of endemic vascular plants*.

Brühl, C. A., Schmidt, T., Pieper, S., & Alscher, A. (2013). Terrestrial pesticide exposure of amphibians: An underestimated cause of global decline? *Scientific Reports*, *3*(1), 1135. https://doi.org/10.1038/srep01135

Brunet, J., Hedwall, P. O., Holmström, E., & Wahlgren, E. (2016). Disturbance of the herbaceous layer after invasion of an eutrophic temperate forest by wild boar. *Nordic Journal of Botany*, *34*(1), 120–128. https://doi.org/10.1111/njb.01010

BSC. (2008). *State of the Environment of the Black Sea (2001 - 2006/7)*. Istanbul, Turkey: Commission on the Protection of the Black Sea Against Pollution (BSC). Retrieved from http://www.blacksea-commission.org/_publ-SOE2009.asp

Bugmann, H., Gurung, A. B., Ewert, F., Haeberli, W., Guisan, A., Fagre, D., & Kaab, A. (2007). Modeling the biophysical impacts of global change in mountain biosphere reserves. *Mountain Research and Development*, *27*(1), 66–77. https://doi.org/10.1659/0276-4741(2007)27[66:MTBIOG]2.0.CO;2

Buisson, L., Thuiller, W., Lek, S., Lim, P., & Grenouillet, G. (2008). Climate change hastens the turnover of stream fish assemblages. *Global Change Biology*, *14*(10), 2232–2248. https://doi.org/10.1111/j.1365-2486.2008.01657.x

Bukvareva, E. (2014). *The summary of the principle of optimal diversity of biosystems*.

Bukvareva, E. N., & Aleshchenko, G. M. [Букварева, Е. Н., & Алещенко, Г. М.]. (2013). Принцип оптимального разнообразия биосистем *[The principle of optimal diversity of biological systems].* Moscow, Russian Federation: KMK - Association of Scientific Publications.

Bulleri, F., & Piazzi, L. (2014). Variations in importance and intensity of competition underpin context dependency in the effects of an invasive seaweed on resident assemblages. *Marine Biology*, *162*(2), 485–489. https://doi.org/10.1007/s00227-014-2563-y

Bullock J., Jeffersen, R. G., Blackstock, T. H., Pakeman, R. J., Emmett, B. A., Pywell, R. J., Grime, J. P., & Silvertown, J. (2011). Semi-natural grasslands. In *UK National Ecosystem Assessment: Technical Report* (pp. 161-196). Cambridge, UK: UNEP-WCMC.

Burdin, A. M., Filatova, O. A., & Khoit, E. [Бурдин, А. М., Филатова, О. А., & Хойт, Э.]. (2009). Морские млекопитающие

России [*Marine mammals of Russia*]. Kirov, Russian Federation: Волго-Вятское книжное издательство [Volga-Vyatka book publishing house].

Burgmer, T., Hillebrand, H., & Pfenninger, M. (2007). Effects of climate-driven temperature changes on the diversity of freshwater macroinvertebrates. *Oecologia*, 151(1), 93–103. https://doi.org/10.1007/s00442-006-0542-9

Burkhanov, A. M., & [Бурханов, А. М.]. (2013). Обзор и анализ деятельности лесной отрасли и существующей информации о лесных экосистемах, их важности для благосостояния людей и социально-экономического развития, тенденциях в этой области, основные прямые и непрямые приводные механизмы, вызывающие изменения [*Review and analysis of forest industry activities and existing information on forest ecosystems, their importance to human well-being and socio-economic development, trends, major direct and indirect drive mechanisms causing changes*]. Bishkek, Kyrgyzstan: UNDP.

Burls, K. J., Shapiro J., Forister M. L., Hoelzer G. A. (2014). A nonlinear relationship between genetic diversity and productivity in a polyphagous seed beetle. *Oecologia 175,* 151–161. https://doi.org/10.1007/s00442-014-2893-y

Butzin, M., & Pörtner, H. O. (2016). Thermal growth potential of Atlantic cod by the end of the 21st century. *Global Change Biology*, 22(12), 4162–4168. https://doi.org/10.1111/gcb.13375

Byrnes, J. E. K., Gamfeldt, L., Isbell, F., Lefcheck, J.S., Griffin, J. N., Hector, A., Cardinal, B. J., Hooper, D. U., Dee. L. E., & Duffy, J. E. (2014). Investigating the relationship between biodiversity and ecosystem multifunctionality: challenges and solutions. *Methods in Ecology and Evolution, 5*(2), 111–24. https://doi.org/10.1111/2041-210X.12143

Cacho, O. J., Spring, D., Pheloung, P., & Hester, S. (2006). Evaluating the feasibility of eradicating an invasion. *Biological Invasions*, 8, 903–917. https://doi.org/10.1007/s10530-005-4733-9

Cadotte, M. W., Carscadden, K., & Mirotchnick, N. (2011). Beyond species: functional diversity and the maintenance of

ecological processes and services. *Journal of Applied Ecology, 48,* 1079–1087. https://doi.org/10.1111/j.1365-2664.2011.02048.x

Caesar S., Karlsson M., Forsman A. (2010). Diversity and relatedness enhance survival in colour polymorphic grasshoppers. *PLoS ONE, 5*(5), e10880. https://doi.org/10.1371/journal.pone.0010880

CAFF. (2013). *Arctic biodiversity assessment. Status and trends in Arctic biodiversity*. Conservation of Arctic Flora and Fauna (CAFF), Arctic Council.

CAFF. (2017). *State of the Arctic marine biodiversity report*. Akureyri, Iceland: Conservation of Arctic Flora and Fauna International Secretariat. Retrieved from https://oaarchive.arctic-council.org/handle/11374/1945

Callaghan, T. V., Björn, L. O. Chapin III, F. S., Chernov, Y., Christensen, T. R., Huntley, B., Ims, R. A., Johansson, M., Riedlinger, D. J., Jonasson, S., Matveyeva, N., Oechel, W., Panikov, N., Shaver, G., Elster, J., Henttonen, H., Jónsdóttir I. S., Laine K., Schaphoff S., Sitch S., Taulavuori E., Taulavuori K., & Zöckler, C. (2005). Arctic tundra and polar desert ecosystems. In C. Symon, L. Arris, & B. Heal (Eds.), *Arctic Climate Impact Assessment* (pp. 243-352). New York, USA: Cambridge University Press.

Callaghan, T. V, Björn, L. O., Chernov, Y., Chapin III, T., Christensen, T. R., Huntley, B., Ims, R. A, Johansson, M., Jolly, D., Jonasson, S., Matveyeva, N., Panikov, N., Oechel, W., Shaver, G., Elster, J., Henttonen, H., Laine, K., Taulavuori, K., Taulavuori, E., & Zöckler, C. (2004). Biodiversity, distributions and adaptations of Arctic species in the context of environmental change. *Ambio, 33*(1), 404–417. http://doi.org/10.1579/0044-7447-33.7.404

Cañadas, E. M., Fenu, G., Peñas, J., Lorite, J., Mattana, E., & Bacchetta, G. (2014). Hotspots within hotspots: Endemic plant richness, environmental drivers, and implications for conservation. *Biological Conservation*, 170, 282–291. https://doi.org/10.1016/j.biocon.2013.12.007

Cañedo-Argüelles, M., Hawkins, C. P., Kefford, B. J., Schäfer, R. B., Dyack, B. J., Brucet, S., Buchwalter, D., Dunlop,

J., Frör, O., Lazorchak, J., Coring, E., Fernandez, H. R., Goodfellow, W., González Achem, A. L., Hatfield-Dodds, S., Karimov, B. K., Mensah, P., Olson, J. R., Piscart, C., Prat, N., Ponsá, S., Schulz, C.-J., & Timpano, A. J. (2016). Saving freshwater from salts. *Science*, 351(6276), 914-916. https://doi.org/10.1126/science.aad3488

Capinha, C., Larson, E. R., Tricarico, E., Olden, J. D., & Gherardi, F. (2013). Effects of climate change, invasive species, and disease on the distribution of native European crayfishes. *Conservation Biology*, 27(4), 731–740. https://doi.org/10.1111/cobi.12043

Cardador, L., De Cáceres, M., Bota, G., Giralt, D., Casas, F., Arroyo, B., Mougeot, F., Cantero-Martíez, C., Moncunill, J., Butler, S. J., & Brotons, L. (2014). A resource-based modelling framework to assess habitat suitability for steppe birds in semiarid Mediterranean agricultural systems. *PLoS ONE, 9*(3), e92790. https://doi.org/10.1371/journal.pone.0092790

Cardinale, B. J., Srivastava, D. S., Duffy, J. E., Wright, J. P., Downing, A. L., Sankaran, M., & Jouseau, C. (2006). Effects of biodiversity on the functioning of trophic groups and ecosystems. *Nature, 443,* 989-992. https://doi.org/10.1038/nature05202

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., Narwani, A., Mace, G. M., Tilman, D., Wardle, D. A., Kinzig, A. P., Daily, G. C., Loreau, M., Grace, J. B., Larigauderie, A., Srivastava, D. S., & Naeem, S. (2012). Biodiversity loss and its impact on humanity. *Nature, 486*(7401), 59–67. https://doi.org/10.1038/nature11148

Cardinale, M., & Scarcella, G. (2017). Mediterranean Sea: A failure of the European fisheries management system. *Frontiers in Marine Science*, 4, 72. https://doi.org/10.3389/fmars.2017.00072

Cardinale, M., & Svedäng, H. (2011). The beauty of simplicity in science: Baltic cod stock improves rapidly in a "cod hostile" ecosystem state. *Marine Ecology Progress Series*, 425, 297–301. https://doi.org/10.3354/meps09098

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Diaz, S., Dietz, T., Duraiappah, A. K., Oteng-Yeboah, A, Pereira, H. M., & Whyte, A. (2009). Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences of the United States of America*, *106*(5), 1305–1312. https://doi.org/10.1073/pnas.0808772106

Carrizo, S. F., Lengyel, S., Kapusi, F., Szabolcs, M., Kasperidus, H. D., Scholz, M., Markovic, D., Freyhof, J., Cid, N., Cardosa, A. C., & Darwall, W. (2017). Critical catchments for freshwater biodiversity conservation in Europe: identification, prioritisation and gap analysis. *Journal of Applied Ecology*, *54*(5), 1209-1218. https://doi.org/10.1111/1365-2664.12842

Carstensen, J., Andersen, J. H., Gustafsson, B. G., & Conley, D. J. (2014). Deoxygenation of the Baltic Sea during the last century. *Proceedings of the National Academy of Sciences of the United States of America*, *111*(15), 5628–33. https://doi.org/10.1073/pnas.1323156111

Carvalheiro, L. G., Kunin, W. E., Keil, P., Aguirre-Gutiérrez, J., Ellis, W. N., Fox, R., Groom, Q., Hennekens, S., Van Landuyt, W., Maes, D., Van de Meutter, F., Michez, D., Rasmont, P., Ode, B., Potts, S. G., Reemer, M., Roberts, S. P. M., Schaminée, J., Wallisdevries, M. F., & Biesmeijer, J. C. (2013). Species richness declines and biotic homogenisation have slowed down for NW-European pollinators and plants. *Ecology Letters*, *16*(7), 870–878. https://doi.org/10.1111/ele.12121

Carvalho, P., Figueira, R., & Jones, M. P. (2008). List of lichens and lichenicolous fungi (Fungi). In P. Borges, C. Abreu, A. Aguiar, P. Carvalho, R. Jardim, I. Melo, P. Oliveira, C. Sérgio, A. Serrano, & P. Vieira (Eds.), *A list of the terrestrial fungi, flora and fauna of Madeira and Salvagens Archipelagos* (pp. 105–122). Funchal, Portugal: Direcção Regional do Ambiente da Madeira and Universidade dos Açores, Funchal and Angra do Heroísmo.

Casale, P., & Margaritoulis, D. (Eds.). (2010). Sea turtles in the Mediterranean: distribution, threats and conservation priorities. Gland, Switzerland: IUCN.

Casas, F., Mougeot, F., Viñuela, J., & Bretagnolle, V. (2009). Effects of hunting on the behaviour and spatial distribution of farmland birds: Importance of hunting-free refuges in agricultural areas. *Animal Conservation*, *12*(4), 346–354. https://doi.org/10.1111/j.1469-1795.2009.00259.x

Casazza, G., Giordani, P., Benesperi, R., Foggi, B., Viciani, D., Filigheddu, R., Farris, E., Bagella, S., Pisanu, S., & Mariotti, M. G. (2014). Climate change hastens the urgency of conservation for range-restricted plant species in the central-northern Mediterranean region. *Biological Conservation*, *179*, 129–138. https://doi.org/10.1016/j.biocon.2014.09.015

Cavanagh, R. D., & Gibson, C. (2007). *Overview of the conservation status of cartilaginous fishes (*Chondrichthyans*) in the Mediterranean Sea*. Gland, Switzerland: IUCN and Malaga, Spain: IUCN. http://cmsdata.iucn.org/downloads/med_shark_rep_en_1.pdf

CBD. (2017). *Protected areas.* Retrieved from https://www.cbd.int/protected/

Celesti-Grapow, L., Bassi, L., Brundu, G., Camarda, I., Carli, E., D'Auria, G., Del Guacchio, E., Domina, G., Ferretti, G., Foggi, B., Lazzaro, L., Mazzola, P., Peccenini, S., Pretto, F., Stinca, A., & Blasi, C. (2016). Plant invasions on small Mediterranean islands: An overview. *Plant Biosystems*, *150*(5), 1119–1133. https://doi.org/10.1080/11263504.2016.1218974

CEP. (2007). *Caspian environment programme transboundary diagnostic analysis revisit*. Retrieved from http://www.ais.unwater.org/ais/aiscm/getprojectdoc.php?docid=1058

CEPF. (2004). *Ecosystem profile. Caucasus biodiversity hotspot*.

CEPF. (2010a). *Ecosystem profile: Mediterranean basin biodiversity hotspot*.

CEPF. (2010b). *Caucasus biodiversity hotspot program for consolidation*.

Céréghino, R., Biggs, J., Oertli, B., & Declerck, S. (2008). The ecology of European ponds: Defining the characteristics of a neglected freshwater habitat. *Hydrobiologia*, *597*(1), 1–6. https://doi.org/10.1007/s10750-007-9225-8

Cerqueira, Y., Navarro, L. M., Maes, J., Marta-Pedroso, C., Honrado, J. P., & Pereira, H. M. (2015). Ecosystem services: the opportunities of rewilding in Europe. In H. M. Pereira & L. Navarro (Eds.), *Rewilding European Landscapes* (pp. 47–64). Springer International Publishing.

Cervenkova, Z., & Münzbergová, Z. (2009). Susceptibility of the landscape of the Giant Mountains, Czech Republic, to invasion by Rumex alpinus. In P. Pysek & J. Pergl (Eds.), *Biological Invasions: Towards a Synthesis. Proceedings of the 5th Neobiota conference (Vol. 8, pp. 75–85)*. Retrieved from http://hdl.handle.net/11104/0177613

Chandra, A., & Idrisova, A. (2011). Convention on Biological Diversity: A review of national challenges and opportunities for implementation. *Biodiversity and Conservation*, *20*(14), 3295–3316. https://doi.org/10.1007/s10531-011-0141-x

Chapron, G., Kaczensky, P., Linnell, J. D. C., Arx, M. von, Huber, D., Andrén, H., López-Bao, J. V., Adamec, M., Álvares, F., Anders, O., Balčiauskas, L., Balys, V., Bedő, P., Bego, F., Blanco, J. C., Breitenmoser, U., Brøseth, H., Bufka, L., Bunikyte, R., Ciucci, P., Dutsov, A., Engleder, T., Fuxjäger, C., Groff, C., Holmala, K., Hoxha, B., Iliopoulos, Y., Ionescu, O., Jeremić, J., Jerina, K., Kluth, G., Knauer, F., Kojola, I., Kos, I., Krofel, M., Kubala, J., Kunovac, S., Kusak, J., Kutal, M., Liberg, O., Majić, A., Männil, P., Manz, R., Marboutin, E., Marucco, F., Melovski, D., Mersini, K., Mertzanis, Y., Mysłajek, R. W., Nowak, S., Odden, J., Ozolins, J., Palomero, G., Paunović, M., Persson, J., Potočnik, H., Quenette, P.-Y., Rauer, G., Reinhardt, I., Rigg, R., Ryser, A., Salvatori, V., Skrbinšek, T., Stojanov, A., Swenson, J. E., Szemethy, L., Trajçe, A., Tsingarska-Sedefcheva, E., Váňa, M., Veeroja, R., Wabakken, P., Wölfl, M., Wölfl, S., Zimmermann, F., Zlatanova, D., & Boitani, L. (2014). Recovery of large carnivores in Europe's modern human-dominated landscapes. *Science*, *346*(6216), 1517–1519. https://doi.org/10.1126/science.1257553

Cheaib, A., Badeau, V., Boe, J., Chuine, I., Delire, C., Dufrêne, E., François, C., Gritti, E. S., Legay, M., Pagé, C., Thuiller, W., Viovy, N., & Leadley, P. (2012). Climate change impacts on tree

ranges: model intercomparison facilitates understanding and quantification of uncertainty. *Ecology Letters*, *15*(6), 533–544. https://doi.org/10.1111/j.1461-0248.2012.01764.x

Cheffings, C., & Farrell, L. (2005). *The Vascular Plant Red Data List for Great Britain*.

Chemonics International. (2001a). Biodiversity assessment for Central Asia: Regional overview.

Chemonics International. (2001b). Biodiversity assessment for Kazakhstan.

Chemonics International. (2001c). Biodiversity assessment for Kyrgyzstan.

Chemonics International. (2001d). Biodiversity assessment for Tajikistan.

Chemonics International. (2001e). Biodiversity assessment for Turkmenistan.

Chemonics International. (2001f). Biodiversity assessment for Uzbekistan.

Cheung, W. W. L., Jones, M. C., Reygondeau, G., Stock, C. A., Lam, V. W. Y., & Frölicher, T. L. (2016). Structural uncertainty in projecting global fisheries catches under climate change. *Ecological Modelling*, *325*, 57–66. https://doi.org/10.1016/j.ecolmodel.2015.12.018

Cheung, W. W. L., Lam, V. W. Y., Sarmiento, J. L., Kearney, K., Watson, R., & Pauly, D. (2009). Projecting global marine biodiversity impacts under climate change scenarios. *Fish and Fisheries*, *10*(3), 235–251. https://doi.org/10.1111/j.1467-2979.2008.00315.x

Chitanava, C. M. [Читанава, С. М.]. (2007). Флора Колхиды. Автореферат диссертации на соискание ученой степени кандидата биологических наук *[The flora of Colchis. An abstract of a PhD dissertation* (*biology)]*. Komarov Botanical Institute of the Russian Academy of Sciences. Retrieved from http://www.binran.ru/files/phd/Chitanava_Abstract_Thesis.pdf

Cieslinski, S., Czyzewska, K., & Fabiszewski, J. (2003). Czerwona lista porostów wymarłych i zagrożonych w Polsce [Red List of extinct and threatened

lichens in Poland]. *Monographiae Botanicae*, *91*.

Čížková, H., Květ, J., Comín, F. A., Laiho, R., Pokorný, J., & Pithart, D. (2013). Actual state of European wetlands and their possible future in the context of global climate change. *Aquatic Sciences*, *75*(1), 3–26. https://doi.org/10.1007/s00027-011-0233-4

Clark, M. R., Althaus, F., Schlacher, T. A., Williams, A., Bowden, D. A., & Rowden, A. A. (2016). The impacts of deep-sea fisheries on benthic communities: a review. *ICES Journal of Marine Science, 73*(Suppl.), 51-69. https://doi.org/10.1093/icesjms/fsv123

Claudet, J., Osenberg, C. W., Benedetti-Cecchi, L., Domenici, P., García-Charton, J. A., Pérez-Ruzafa, Á., Badalamenti, F., Bayle-Sempere, J., Brito, A., Bulleri, F., Culioli, J. M., Dimech, M., Falcón, J. M., Guala, I., Milazzo, M., Sánchez-Meca, J., Somerfield, P. J., Stobart, B., Vandeperre, F., Valle, C., & Planes, S. (2008). Marine reserves: Size and age do matter. *Ecology Letters*, *11*(5), 481–489. https://doi.org/10.1111/j.1461-0248.2008.01166.x

Coll, M., Carreras, M., Ciércoles, C., Cornax, M. J., Gorelli, G., Morote, E., & Sáez, R. (2014). Assessing fishing and marine biodiversity changes using fishers' perceptions: The Spanish Mediterranean and Gulf of Cadiz case study. *PLoS ONE*, *9*(1), e85670. https://doi.org/10.1371/journal.pone.0085670

Coll, M., Piroddi, C., Steenbeek, J., Kaschner, K., Ben Rais Lasram, F., Aguzzi, J., Ballesteros, E., Bianchi, C. N., Corbera, J., Dailianis, T., Danovaro, R., Estrada, M., Froglia, C., Galil, B. S., Gasol, J. M., Gertwage, R., Gil, J., Guilhaumon, F., Kesner-Reyes, K., Kitsos, M. S., Koukouras, A., Lampadariou, N., Laxamana, E., de la Cuadra, C. M. L. F., Lotze, H. K., Martin, D., Mouillot, D., Oro, D., Raicevich, S., Rius-Barile, J., Saiz-Salinas, J. I., Vicente, C. S., Somot, S., Templado, J., Turon, X., Vafidis, D., Villanueva, R., & Voultsiadou, E. (2010a). The biodiversity of the Mediterranean Sea: estimates, patterns, and threats. *PloS ONE*, *5*(8),

e11842. https://doi.org/10.1371/journal.pone.0011842

Coll, M., Shannon, L. J., Yemane, D., Link, J. S., Ojaveer, H., Neira, S., Jouffre, D., Labrosse, P., Heymans, J. J., Fulton, E. A., & Shin, Y.-J. (2010b). Ranking the ecological relative status of exploited marine ecosystems. *ICES Journal of Marine Science*, *67*(4), 769–786. https://doi.org/10.1093/icesjms/fsp261

Colloca, F., Cardinale, M., Maynou, F., Giannoulaki, M., Scarcella, G., Jenko, K., Bellido, J. M., & Fiorentino, F. (2013). Rebuilding Mediterranean fisheries: A new paradigm for ecological sustainability. *Fish and Fisheries*, *14*(1), 89–109. https://doi.org/10.1111/j.1467-2979.2011.00453.x

Comin, F., & Alonso, M. (1988). Spanish salt lakes: Their chemistry and biota. *Hydrobiologia*, *158,* 237–245. https://doi.org/10.1007/BF00026281

Conley, D. J., Carstensen, J., Aigars, J., Axe, P., Bonsdorff, E., Eremina, T., Haahti, B. M., Humborg, C., Jonsson, P., Kotta, J., Lännegren, C., Larsson, U., Maximov, A., Medina, M. R., Lysiak-Pastuszak, E., Remeikaitè-Nikienè, N., Walve, J., Wilhelms, S., & Zillén, L. (2011). Hypoxia is increasing in the coastal zone of the Baltic Sea. *Environmental Science and Technology*, *45*(16), 6777–6783. https://doi.org/10.1021/es201212r

Conservation International. (2011). Biological diversity in the Mediterranean Basin. Retrieved from http://editors.eol.org/eoearth/wiki/Biological_diversity_in_the_Mediterranean_Basin

Conti, F., Manzi, A., & Pedrotti, F. (1992). *Libro rosso delle plante d'Italia [Red book of the plants of Italy]*. Rome, Italy: WWF Italia.

Corbett, K. (1989). *Conservation of European reptiles and amphibians*. London, UK: C. Helm.

Cordellier, M., Pfenninger, A., Streit, B., & Pfenninger, M. (2012). Assessing the effects of climate change on the distribution of pulmonate freshwater snail biodiversity. *Marine Biology*, *159*, 2519–2531. https://doi.org/10.1007/s00227-012-1894-9

Cornelissen, J. H. C., Callaghan, T. V., Alatalo, J. M., Michelsen, A., Graglia,

E., Hartley, A. E., Hik, D. S., Hobbie, S. E., Press, M. C., Robinson, C. H., Henry, G. H. R., Shaver, G. R., Phoenix, G. K., Jones, D. G., Jonasson, S., Chapin, F. S., Molau, U., Neill, C., Lee, J. A., Melillo, J. M., Sveinbjornsson, B., & Aerts, R.** (2001). Global change and arctic ecosystems: Is lichen decline a function of increases in vascular plant biomass? *Journal of Ecology*, *89*(6), 984–994. https://doi.org/10.1046/j.1365-2745.2001.00625.x

Cornelissen, J. H. C., Lang, S. I., Soudzilovskaia, N. A., & During, H. J. (2007). Comparative cryptogam ecology: A review of bryophyte and lichen traits that drive biogeochemistry. *Annals of Botany*, *99*(5), 987–1001. https://doi.org/10.1093/aob/mcm030

Cornulier, T., Yoccoz, N. G., Bretagnolle, V., Brommer, J. E., Butet, A., Ecke, F., Elston, D. A., Framstad, E., Henttonen, H., Hornfeldt, B., Huitu, O., Imholt, C., Ims, R. A., Jacob, J., Jedrzejewska, B., Millon, A., Petty, S. J., Pietiainen, H., Tkadlec, E., Zub, K., & Lambin, X. (2013). Europe-wide dampening of population cycles in keystone herbivores. *Science, 340*(6128), 63-66. https://doi.org/10.1126/science.1228992

Costello, M. J., & Ballantine, B. (2015). Biodiversity conservation should focus on no-take marine reserves: 94% of marine protected areas allow fishing. *Trends in Ecology and Evolution*, *30*(9), 507–509. https://doi.org/10.1016/j.tree.2015.06.011

Costello, M. J., Coll, M., Danovaro, R., Halpin, P., Ojaveer, H., & Miloslavich, P. (2010). A census of marine biodiversity knowledge, resources, and future challenges. *PLoS ONE*, *5*(8). https://doi.org/10.1371/journal.pone.0012110

Coulson, S. J. (2015). The alien terrestrial invertebrate fauna of the high Arctic Archipelago of Svalbard: potential implications for the native flora and fauna. *Polar Research*, *34*, 27364.

Cox, N.A., & Temple, H.J., (2009). *European red list of reptiles*. Luxembourg: Office for Official Publications of the European Communities. https://doi.org/10.2779/74504

Crosa, G., Froebrich, J., Nikolayenko, V., Stefani, F., Galli, P., & Calamari, D.

(2006). Spatial and seasonal variations in the water quality of the Amu Darya River (Central Asia). *Water Research*, *40*(11), 2237–2245. https://doi.org/10.1016/j.watres.2006.04.004

Cumberlidge, N. Ng, P. K. L., Yeo, D. C. J., Magalhães, C., Campos, M. R., Alvarez, F., Naruse, T., Daniels, S. R., Esser, L. J., Attipoe, F. Y. K., Clotilde-Ba, F., Darwall, W., McIvor, A., Baillie, J. E. M., Collen, B., & Ram, M. (2009). Freshwater crabs and the biodiversity crisis: Importance, threats, status, and conservation challenges. *Biological Conservation, 142*(8), 1665-1673. https://doi.org/10.1016/j.biocon.2009.02.038

Culver, D. C., Deharveng, L., Bedos, A., Lewis, J. J., Madden, M., Reddell, J. R., Sket, B., Trontelj, P., & White, D. (2006). The mid-latitude biodiversity ridge in terrestrial cave fauna. *Ecography 29, 120-128*. https://doi.org/10.1111/j.2005.0906-7590.04435.x

Culver, D. C., & Pipan, T. (2013). Subterranean Ecosystems. *Encyclopedia of Biodiversity*, *7,* 49–62. https://doi.org/10.1016/B978-0-12-384719-5.00224-0

Culver, D. C., & Pipan, T. (2014). *Shallow subterranean habitats. Ecology, evolution, and conservation*. Oxford, UK: Oxford University Press.

Culver, D. C., & Sket, B. (2000). Hotspots of subterranean biodiversity in caves and wells. *Journal of Cave and Karst Studies*, *62*(1), 11–17.

Curtis, C. J., Emmett, B. A., Grant, H., Kernan, M., Reynolds, B., & Shilland, E. (2005). Nitrogen saturation in UK moorlands: The critical role of bryophytes and lichens in determining retention of atmospheric N deposition. *Journal of Applied Ecology*, *42*(3), 507–517. https://doi.org/10.1111/j.1365-2664.2005.01029.x

Curtis T.G.F., & McGough, H. N. (1988). *The Irish Red Data Book: 1) Vascular plants*. Dublin, Ireland: Wildlife Service Ireland Stationary Office.

Cuttelod, A., Seddon, M., & Neubert, E. (2011). *European red list of non-marine molluscs*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/84538

Daan, N., Gislason, H., Pope, J. G., & Rice, J. C. (2005). Changes in the North Sea fish community: Evidence of indirect effects of fishing? *ICES Journal of Marine Science*, *62*(2), 177–188. https://doi.org/10.1016/j.icesjms.2004.08.020

Dahlberg, A., Genney, D. R., & Heilmann-Clausen, J. (2010). Developing a comprehensive strategy for fungal conservation in Europe: current status and future needs. *Fungal Ecology*, *3*(2), 50–64. https://doi.org/10.1016/j.funeco.2009.10.004

Dahlberg, A., & Mueller, G. M. (2011). Applying IUCN red-listing criteria for assessing and reporting on the conservation status of fungal species. *Fungal Ecology*, *4*(2), 147–162. https://doi.org/10.1016/j.funeco.2010.11.001

Dahlgren, S., & Kautsky, L. (2004). Can different vegetative states in shallow coastal bays of the Baltic Sea be linked to internal nutrient levels and external nutrient load? *Hydrobiologia*, *514*, 249–258. https://doi.org/10.1023/B:hydr.0000018223.26997.b0

Dalpadado, P., Arrigo, K. R., Hjøllo, S. S., Rey, F., Ingvaldsen, R. B., Sperfeld, E., van Dijken, G. L., Stige, L. C., Olsen, A., & Ottersen, G. (2014). Productivity in the Barents Sea - Response to Recent Climate Variability. *PLoS ONE*, *9*(5), e95273. https://doi.org/10.1371/journal.pone.0095273

Danielopol, D., Griebler, C., Gunatilaka, A., & Notenboom, J. (2003). Present state and future prospects for groundwater ecosystems. *Environmental Conservation*, *30*(2), 1–27. https://doi.org/10.1017/S0376892903000109

Danilov-Danilyan, V. I., Amirkhanov, A. M., Pavlov, D. S., Sokolov, B. E., Alimov, A. F., Darevsky, I. S., Dezhkin, V. V., Ilyashenko, V. U., Mazni, L. N., Rozhnov, V. V., Scarlato, U. A., Flint, V. E., & Yablokov, A. V. [Данилов-Данильян, В. И., Амирханов, А. М., Павлов, Д. С., Соколов, В. Е., Алимов, А. Ф., Даревский, И. С., Дежкин, В. В., Ильяшенко, В. Ю., Мазни, Л. Н., Рожнов, В. В., Скарлато, Ю. А., Флинт, В. Е., & Яблоков, А. В.] (Eds.). (2001). Красная книга Российской Федерации (животные) *[Red data book of the Russian Federation (animals)]*. Moscow, Russian Federation: АСТ-Астрель [AST-Astrel]. Retrieved from http://biodat.ru/db/rb/

Danovaro, R., Company, J. B., Corinaldesi, C., D'Onghia, G., Galil, B., Gambi, C., Gooday, A. J., Lampadariou, N., Luna, G. M., Morigi, C., Olu, K., Polymenakou, P., Ramirez-Llodra, E., Sabbatini, A., Sarda, F., Sibuet, M., & Tselepides, A. (2010). Deep-sea biodiversity in the Mediterranean Sea: The known, the unknown, and the unknowable. *PLoS ONE*, *5*(8), e11832. https://doi.org/10.1371/journal.pone.0011832

Danovaro, R., Corinaldesi, C., Rastelli, E., & Dell'Anno, A. (2015). Towards a better quantitative assessment of the relevance of deep-sea viruses, bacteria and archaea in the functioning of the ocean seafloor. *Aquatic Microbial Ecology, 75,* 81-90. https://doi.org/10.3354/ame01747

Danovaro, R., Gambi, C., Dell'Anno, A., Corinaldesi, C., Fraschetti, S., Vanreusel, A., Vincx, M., & Gooday, A. J. (2008). Exponential decline of deep-sea ecosystem functioning linked to benthic biodiversity loss. *Current Biology*, *18*(1), 1–8. https://doi.org/10.1016/j.cub.2007.11.056

Danovaro, R., Rastelli, E., Corinaldesi, C., Tangherlini, M., & Dell'Anno, A. (2017). Marine archaea and archaeal viruses under global change. *F1000Research, 6,* 1241. http://doi.org/10.12688/f1000research.11404.1

Daufresne, M., Lengfellner, K., & Sommer, U. (2009). Global warming benefits the small in aquatic ecosystems. *PNAS*, *106*, 12788–12793. http://doi.org/10.1073/pnas.0902080106

Davidson, A. D., Detling, J. K., & Brown, J. H. (2012). Ecological roles and conservation challenges of social, burrowing, herbivorous mammals in the world's grasslands. *Frontiers in Ecology and the Environment*, *10*, 477–486. https://doi.org/10.1890/110054

Davies, G. M., Kettridge, N., Stoof, C. R., Gray, A., Marrs, R., Ascoli, D., Fernandes, P. M., Allen, K. A., Doerr, S. H., Clay, G. D., McMorrow, J., & Vandvik, V. (2016). Informed debate on the use of fire for peatland management means acknowledging the complexity of socio-ecological systems. *Nature Conservation*, *16*, 59–77. http://doi.org/10.3897/natureconservation.16.10739

Davydov, E. A., Insarov, G. E., & Sundetpaev, A. K. (2013). Lichen monitoring in Katon-Karagai National Park, eastern Kazakhstan, in context of climate change. *Problems of Ecological Monitoring and Ecosystem Modelling*, *25*, 428–441.

De Beaulieu, J. L., Miras, Y., Andrieu-Ponel, V., & Guiter, F. (2005). Vegetation dynamics in northern-western Mediterranean regions: instability of the Mediterranean bioclimate. *Plant Biosystems*, *139*, 114–126.

de Castro, M. C. T., Fileman, T. W., & Hall-Spencer, J. M. (2017). Invasive species in the northeastern and southwestern Atlantic Ocean: A review. *Marine Pollution Bulletin, 116*, 41-47. https://doi.org/10.1016/j.marpolbul.2016.12.048

De Frenne, P., Rodríguez-Sánchez, F., Coomes, D. A., Baeten, L., Verstraeten, G., Vellend, M., Bernhardt-Römermann, M., Brown, C. D., Brunet, J., Cornelis, J., Decocq, G. M., Dierschke, H., Eriksson, O., Gilliam, F. S., Hédl, R., Heinken, T., Hermy, M., Hommel, P., Jenkins, M. A., Kelly, D. L., Kirby, K. J., Mitchell, F. J. G., Naaf, T., Newman, M., Peterken, G., Petrík, P., Schultz, J., Sonnier, G., Van Calster, H., Waller, D. M., Walther, G.-R., White, P. S., Woods, K. D., Wulf, M., Graae, B. J., & Verheyen, K. (2013). Microclimate moderates plant responses to macroclimate warming. *Proceedings of the National Academy of Sciences of the United States of America*, *110*(46), 18561–5. https://doi.org/10.1073/pnas.1311190110

De Rigo, D., Bosco, C., San-Miguel-Ayanz, J., Houston Durrant, T., Barredo, J. I., Strona, G., Caudullo, G., Di Leo, M., & Boca, R. (2016). Forest resources in Europe: an integrated perspective on ecosystem services, disturbances and threats. In J. San-Miguel-Ayanz, D. de Rigo, G. Caudullo, T. Houston Durrant, & A. Mauri. (Eds.), *The European atlas of forest tree species* (pp. 8-19). Luxembourg: Publications Office of the European Union.

Deckers, B., Kerselaers, E., Gulinck, H., Muys, B., & Hermy, M. (2005). Long-term spatio-temporal dynamics of a hedgerow network landscape in Flanders, Belgium. *Environmental Conservation*, *32*(1), 20–29. https://doi.org/10.1017/s0376892905001840

Deharveng, L., Stoch, F., Gibert, J., Bedos, A., Galassi, D., Zagmajster, M., Brancelj, A., Camacho, A., Fiers, F., Martin, P., Giani, M., Magniez, G., & Marmonier, P. (2009). Groundwater biodiversity in Europe. *Freshwater Biology, 54,* 709-726. https://doi.org/10.1111/j.1365-2427.2008.01972.x

Dehling, D. M., Hof, C., Brändle, M., & Brandl, R. (2010). Habitat availability does not explain the species richness patterns of European lentic and lotic freshwater animals. *Journal of Biogeography*, *37*(10), 1919–1926. https://doi.org/10.1111/j.1365-2699.2010.02347.x

Delgado, V., & Ederra, A. (2013). Long-term changes (1982-2010) in the bryodiversity of Spanish beech forests assessed by means of Ellenberg indicator values of temperature, nitrogen, light and pH. *Biological Conservation*, *157*, 99–107. https://doi.org/10.1016/j.biocon.2012.06.022

Dengler, J., Bergmeier, E., Willner, W., & Chytrý, M. (2013). Towards a consistent classification of European grasslands. *Applied Vegetation Science*, *16*(3), 518–520. http://doi.org/10.1111/avsc.12041

Dengler, J., Janišová, M., Török, P., & Wellstein, C. (2014). Biodiversity of Palaearctic grasslands: A synthesis. *Agriculture, Ecosystems and Environment*, *182*, 1–14. https://doi.org/10.1016/j.agee.2013.12.015

Deudero, S., Box, A., Alós, J., Arroyo, N. L., & Marbà, N. (2011). Functional changes due to invasive species: Food web shifts at shallow *Posidonia oceanica* seagrass beds colonized by the alien macroalga *Caulerpa racemosa*. *Estuarine, Coastal and*

Shelf Science, *93*(2), 106–116. https://doi.org/10.1016/j.ecss.2011.03.017

Diagelets, E. Y., Knizhnikov, A. Y., Mnatsekanov, R. A., & Pegova, O. V. [Дягилец, Е. Ю., Книжников, А. Ю., Мнацеканов, Р. А., & Пегова, О. В.]. (2014). Люди, нефть, птицы. Рекомендации для практических мероприятий *[People, oil, birds. Recommendations for practical activities]*. Moscow, Russian Federation: WWF Russia.

Diakonov, K. N., Puzachrenko, Yu. G., Aleschenko, G. M., Sysuev, V. V., & Mamay, I. I. [Дьяконов, К.Н., Пузаченко, Ю. Г., Алещенко, Г. М., Сысуев, В. В., & Мамай, И. И.]. (2004). География, общество, окружающая среда. Том 2. Функционирование и современное состояние ландшафтов [Geography, Society and Environment. Volume II: Contemporary Landscape Processsess]. Moscow, Russian Federation: Gorodets.

Dias, E., Mendes, C., Melo, C., Pereira, D., & Elias, R. (2005). Azores central islands vegetation and flora field guide. *Quercetea, 7,* 123–173.

Dias, F. S., Miller, D. L., Marques, T. A., Marcelino, J., Caldeira, M. C., Orestes Cerdeira, J., & Bugalho, M. N. (2016). Conservation zones promote oak regeneration and shrub diversity in certified Mediterranean oak woodlands. *Biological Conservation*, *195*, 226–234. http://doi.org/10.1016/j.biocon.2016.01.009

Diaz, A., Keith, S. A., Bullock, J. M., Hooftman, D. A. P., & Newton, A. C. (2013). Conservation implications of long-term changes detected in a lowland heath plant metacommunity. *Biological Conservation*, *167*, 325–333. http://doi.org/10.1016/j.biocon.2013.08.018

Díaz, S., Lavorel, S., McIntyre, S., Falczuk, V., Casanoves, F., Milchunas, D. G., Skarpe, C., Rusch, G., Sternberg, M., Noy-Meir, I., Landsberg, J., Zhang, W., Clark, H., & Campbell, B. D. (2007). Plant trait responses to grazing - A global synthesis. *Global Change Biology*, *13*(2), 313–341. https://doi.org/10.1111/j.1365-2486.2006.01288.x

Diemont, W.H., Webb, N., & Degn, H. J. (1996). A pan European view on heathland conservation. In: *Proceedings of the National Heathland Conference, 18-20 September 1996, New Forest, Hampshire*.

Diemont, W. H., & Jansen, J. (1998). A cultural view of European heathlands. *Suffolk Natural History*, *34*, 32–34.

Diemont, W. H., Siepel, H., Webb, N. R., & Heijman, W. J. M. (Eds.). (2013). *Economy and Ecology of Heathlands*. Zeist, The Netherlands: KNNV Publishing. http://doi.org/10.1163/9789004277946

Dicks, L.V., Wright, H.L., Ashpole, J.E., Hutchison, J., McCormack, C.G., Livoreil, B., Zulka, K.P., & Sutherland, W.J. (2016) What works in conservation? Using expert assessment of summarised evidence to identify practices that enhance natural pest control in agriculture. *Biodiversity and Conservation 25,* 1383-1399. https://doi.org/10.1007/s10531-016-1133-7

Dirnböck, T., Dullinger, S., & Grabherr, G. (2003). A regional impact assessment of climate and land-use change on alpine vegetation. *Journal of Biogeography*, *30,* 401-417. https://doi.org/10.1046/j.1365-2699.2003.00839.x

Dirnböck, T., Essl, F., & Rabitsch, W. (2011). Disproportional risk for habitat loss of high-altitude endemic species under climate change. *Global Change Biology*, *17*(2), 990–996. https://doi.org/10.1111/j.1365-2486.2010.02266.x

Dixon, M. J. R., Loh, J., Davidson, N. C., Beltrame, C., Freemen, R., & Walpole, M. (2016). Tracking global change in ecosystem area: The Wetland Extent Trends index. *Biological Conservation 193,* 27–35. https://doi.org/10.1016/j.biocon.2015.10.023

Dobrovolskii, A. D., & Zalogin, B. S. [Добровольский, А. Д., & Залогин, Б. С.]. (1982). Моря СССР *[The Seas of the USSR]*. Moscow, Russian Federation: МГУ [Moscow State University].

Doğan, D. (n.d.). Global climate change and its effects in Turkey. Retrieved from https://ru.scribd.com/document/84374702/Global-Climate-Change-and-Its-Effects-in-Turkey

Dolev Pervolutzki, A. A. (2004). *Endangered species in Israel. Red list of threatened animals. Vertebrates*. Jerusalem, Israel: The Nature Reserves and Park Authority and the Society for Conservation of Nature.

Dominoni, D. M., & Partecke, J. (2015). Does light pollution alter daylength? A test using light loggers on free-ranging European blackbirds (*Turdus merula*). *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*(1667), UNSP 20140118. https://doi.org/10.1098/rstb.2014.0118

Domisch, S., Araújo, M. B., Bonada, N., Pauls, S. U., Jähnig, S. C., & Haase, P. (2013). Modelling distribution in European stream macroinvertebrates under future climates. *Global Change Biology*, *19*(3), 752–762. https://doi.org/10.1111/gcb.12107

Domisch, S., Jähnig, S. C., & Haase, P. (2011). Climate-change winners and losers: stream macroinvertebrates of a submontane region in Central Europe. *Freshwater Biology*, *56*(10), 2009–2020. https://doi.org/10.1111/j.1365-2427.2011.02631.x

Donald, P. F., Green, R. E., & Heath, M. F. (2001). Agricultural intensification and the collapse of Europe's farmland bird populations. *Proceedings of the Royal Society B: Biological Sciences*, *268*(1462), 25–29. Retrieved from https://doi.org/10.1098/rspb.2000.1325

Donald, P. F., Sanderson, F. J., Burfield, I. J., & van Bommel, F. P. J. (2006). Further evidence of continent-wide impacts of agricultural intensification on European farmland birds, 1990-2000. *Agriculture, Ecosystems and Environment*, *116*(3–4), 189–196. https://doi.org/10.1016/j.agee.2006.02.007

Doxa, A., Paracchini, M. L., Pointereau, P., Devictor, V., & Jiguet, F. (2012). Preventing biotic homogenization of farmland bird communities: The role of high nature value farmland. *Agriculture, Ecosystems and Environment*, *148,* 83–88. https://doi.org/10.1016/j.agee.2011.11.020

Drobyshev, I., Niklasson, M., & Linderholm, H. W. (2012). Forest fire activity in Sweden: Climatic controls and geographical patterns in 20th century. *Agricultural and Forest Meteorology*, *154–155*, 174–186. https://doi.org/10.1016/j.agrformet.2011.11.002

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z.-I., Knowler, D. J., Lévêque, C., Naiman, R. J., Prieur-Richard, A. H. Soto, D., Stiassny, M. L. J., & Sullivan, C. A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews*, *81*(2), 163–82. https://doi.org/10.1017/S1464793105006950

Duffus, A. L. J., & Cunningham, A. A. (2010). Major disease threats to European amphibians. *Herpetological Journal*, *20*, 117–1.

Duffy, J. E., Srivastava, D. S., McLaren, J., Sankaran, M., Solan, M., Griffin, J., Emmerson, M., & Jones, K. E. (2009). Forecasting decline in ecosystem services under realistic scenarios of extinction. In S. Naeem, D. Bunker, A. Hector, M. Loreau, & C. Perrings, *Biodiversity, ecosystem functioning and human well-being: An ecological and economic perspective* (pp. 60-77). Oxford, UK: Oxford University.

Dullinger, S., Gattringer, A., Thuiller, W., Moser, D., Zimmermann, N. E., Guisan, A., Willner, W., Plutzar, C., Leitner, M., Mang, T., Caccianiga, M., Dirnböck, T., Ertl, S., Fischer, A., Lenoir, J., Svenning, J.-C., Psomas, A., Schmatz, D. R., Silc, U., Vittoz, P., & Hülber, K. (2012). Extinction debt of high-mountain plants under twenty-first-century climate change. *Nature Climate Change*, *2*(8), 619–622. https://doi.org/10.1038/nclimate1514

Dulvy, N. K., Rogers, S. I., Jennings, S., Stelzenmüller, V., Dye, S. R., & Skjoldal, H. R. (2008). Climate change and deepening of the North Sea fish assemblage: A biotic indicator of warming seas. *Journal of Applied Ecology*, *45*(4), 1029–1039. https://doi.org/10.1111/j.1365-2664.2008.01488.x

Dumont, H., Mamaev, V. O., & Zaitsev, Y. P. (1999). *Black Sea Red Data Book*. New York, USA: United Nations Office for Project Services.

Duprè, C., Stevens, C. J., Ranke, T., Bleeker, A., Peppler-Lisbach, C., Gowing, D. J. G., Dise, N. B., Dorland, E., Bobbink, R., & Diekmann, M. (2010). Changes in species richness and composition in European acidic grasslands over the past 70 years: The contribution of cumulative atmospheric nitrogen

deposition. *Global Change Biology*, *16*(1), 344–357. https://doi.org/10.1111/j.1365-2486.2009.01982.x

Durance, I., & Ormerod, S. J. (2009). Trends in water quality and discharge confound long-term warming effects on river macroinvertebrates. *Freshwater Biology*, *54*(2), 388–405. https://doi.org/10.1111/j.1365-2427.2008.02112.x

EBCC. (2013). European Bird Census Council. http://www.ebcc.info/index.php

EBCC. (2017). *European wild bird indicators, 2017 update*. Retrieved from https://www.ebcc.info/index.php?ID=632

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., Barrett, N. S., Becerro, M. A., Bernard, A. T. F., Berkhout, J., Buxton, C. D., Campbell, S. J., Cooper, A. T., Davey, M., Edgar, S. C., Försterra, G., Galván, D. E., Irigoyen, A. J., Kushner, D. J., Moura, R., Parnell, P. E., Shears, N. T., Soler, G., Strain, E. M. A., & Thomson, R. J. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature*, *506*(7487), 216–220. https://doi.org/10.1038/nature13022

Edwards, M., & Richardson, A. J. (2004). Impact of climate change on marine pelagic phenology and trophic mismatch. *Nature*, *430*(7002), 881–884. https://doi.org/10.1038/nature02808

Edwards, R., Soos, J., & Ritcey, R. (1960). Quantitative observations on epidendric lichens used as food by caribou. *Ecology*, *41*(3), 425–431. https://doi.org/10.2307/1933317

EEA. (2002). *Europe's biodiversity - biogeographical regions and seas. EEA Report No 1/2002*. Retrieved from http://www.eea.europa.eu/publications/report_2002_0524_154909

EEA. (2004). *Biogeographical regions in Europe. The Mediterranean biogeographical region - long influence from cultivation, high pressure from tourists, species rich, warm and drying*. Retrieved from https://www.eea.europa.eu/publications/report_2002_0524_154909/biogeographical-regions-in-europe/mediterranean_biogeografical_region.pdf/view

EEA. (2010). *Assessing biodiversity in Europe — the 2010 report. Europe*. https://doi.org/10.2800/42824

EEA. (2012). *European waters — current status and future challenges: Synthesis, EEA Report No 9/2012*. http://doi.org/10.2800/63931

EEA. (2013). *The European grassland butterfly indicator: 1990–2011*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2800/89760

EEA. (2015a). *The European environment — state and outlook 2015: Synthesis report*. Copenhagen, Denmark: European Environment Agency. http://doi.org/10.2800/944899

EEA. (2015b). *Spatial analysis of marine protected area networks in Europe's seas*. http://doi.org/10.2800/406625

EEA. (2015c). *State of Europe's seas*. http://doi.org/10.2800/0466

EEA. (2015d). *The state of nature in the European Union*. https://doi.org/10.2800/603862

EEA. (2016). *European forest ecosystems - state and trends*. https://doi.org/10.2800/964893

Eeva, T., Belskii, E., Gilyazov, A. S., & Kozlov, M. V. (2012). Pollution impacts on bird population density and species diversity at four non-ferrous smelter sites. *Biological Conservation*, *150*(1), 33–41. https://doi.org/10.1016/j.biocon.2012.03.004

Eeva, T., Belskii, E., & Kuranov, B. (2006). Environmental pollution affects genetic diversity in wild bird populations. *Mutation Research*, *608*, 8–15. https://doi.org/10.1016/j.mrgentox.2006.04.021

Elkin, C., Gutiérrez, A. G., Leuzinger, S., Manusch, C., Temperli, C., Rasche, L., & Bugmann, H. (2013). A 2 degrees C warmer world is not safe for ecosystem services in the European Alps. *Global Change Biology*, *19*(6), 1827–1840. https://doi.org/10.1111/gcb.12156

Ellenberg, H., & Leuschner, C. (2010). *Vegetation Mitteleuropas mit den Alpen, 6. Auflage [Vegetation of Central Europe*

and the Alps, sixth edition]. Stuttgart, Germany: Ulmer.

Ellingsen, I. H., Dalpadado, P., Slagstad, D., & Loeng, H. (2008). Impact of climatic change on the biological production in the Barents Sea. *Climatic Change*, *87*(1–2), 155–175. https://doi.org/10.1007/s10584-007-9369-6

Elliot, J. A., Thackeray, S. J., Huntingford, C., & Jones, R. G. (2005). Combining a regional climate model with a phytoplankton community model to predict future changes in phytoplankton in lakes. *Freshwater Biology 50,* 1404–1411. https://doi.org/10.1111/j.1365-2427.2005.01409.x

Ellis, A., Jackson, M. C., Jennings, I., England, J., & Phillips, R. (2012). Present distribution and future spread of Louisiana red swamp crayfish *Procambarus clarkii* (Crustacea, Decapoda, Astacida, Cambaridae) in Britain: Implications for conservation of native species and habitats. *Knowledge and Management of Aquatic Ecosystems*, *406*, 5. https://doi.org/10.1051/kmae/2012022

Ellis, C. J. (2012). Lichen epiphyte diversity: A species, community and trait-based review. *Perspectives in Plant Ecology, Evolution and Systematics*, *14*(2), 131–152. https://doi.org/10.1016/j.ppees.2011.10.001

Ellis, C. J. (2015). Ancient woodland indicators signal the climate change risk for dispersal-limited species. *Ecological Indicators*, *53*, 106–114. https://doi.org/10.1016/j.ecolind.2015.01.028

Ellis, C. J., Coppins, B. J., & Hollingsworth, P. M. (2012). Lichens under threat from ash dieback. *Nature*, *491*(29), 672-672. https://doi.org/10.1016/j.biocon.2005.10.042

Ellis, C. J., Eaton, S., Theodoropoulos, M., Coppins, B. J., Seaward, M. R. D., & Simkin, J. (2014). Response of epiphytic lichens to 21st century climate change and tree disease scenarios. *Biological Conservation 180,* 153-164. https://doi.org/10.1016/j.biocon.2014.09.046

Elmendorf, S. C., Henry, G. H. R., Hollister, R. D., Björk, R. G., Bjorkman, A. D., Callaghan, T. V., Collier, L. S., Cooper, E. J., Cornelissen, J. H. C.,

Day, T. A., Fosaa, A. M., Gould, W. A., Grétarsdóttir, J., Harte, J., Hermanutz, L., Hik, D. S., Hofgaard, A., Jarrad, F., Jónsdóttir, J. S., Keuper, F., Klanderud, K., Klein, J. A., Koh, S., Kudo, G., Lang, S. I., Loewen, V., May, J. L., Mercado, J., Michelsen, A., Molau, U., Myers-Smith, I. H., Oberbauer, S. F., Pieper, S., Post, E., Rixen, C., Robinson, C. H., Schmidt, N. M., Shaver, G. R., Stenström, A., Tolvanen, A., Totland, Ö., Troxler, T., Wahren, C. H., Webber, P. J., Welker, J. M., & Wookey, P. A. (2012). Global assessment of experimental climate warming on tundra vegetation: Heterogeneity over space and time. *Ecology Letters*, *15*(2), 164–175. https://doi.org/10.1111/j.1461-0248.2011.01716.x

Emanuelsson, U. (2010). *The rural landscapes of Europe. How man has shaped European nature*.

Emmrich, M., Pédron, S., Brucet, S., Winfield, I. J., Jeppesen, E., Volta, P., Argillier, C., Lauridsen, T. L., Holmgren, K., Hesthagen, T., & Mehner, T. (2014). Geographical patterns in the body-size structure of European lake fish assemblages along abiotic and biotic gradients. *Journal of Biogeography*, *41*(12), 2221–2233. https://doi.org/10.1111/jbi.12366

Engler, E., Randin, C. F., Vittoz, P. Czáka, T., Beniston, M., Zimmermann, N. E., & Guisan, A. (2009). Predicting future distributions of mountain plants under climate change: does dispersal capacity matter? *Ecography 32,* 34-45. https://doi.org/10.1111/j.1600-0587.2009.05789.x

Engler, R., Randin, C. F., Thuiller, W., Dullinger, S., Zimmermann, N. E., Araújo, M. B., Pearman, P. B., Le Lay, G., Piedallu, C., Albert, C. H., Choler, P., Coldea, G., De Lamo, X., Dirnböck, T., Gégout, J. C., Gómez-García, D., Grytnes, J. A., Heegaard, E., Høistad, F., Nogués-Bravo, D., Normand, S., Puşcaş, M., Sebastià, M. T., Stanisci, A., Theurillat, J. P., Trivedi, M. R., Vittoz, P., & Guisan, A. (2011). 21st century climate change threatens mountain flora unequally across Europe. *Global Change Biology*, *17*(7), 2330–2341. https://doi.org/10.1111/j.1365-2486.2010.02393.x

ENPI-FLEG. (2015). *Analyses approaches to sustainable methods for Tugai forest rehabilitation in Azerbaijan. Final Report*.

Eremeev, V. N., Gaevskaya, A. V., Shulman, G. E., & Zagorodnyaya, J. A. [Еремеев, В. Н., Гаевская, А. В., Шульман, Г. Е., & Загородняя, Ю. А.] (Eds.). (2011). Промысловые биоресурсы Черного и Азовского морей *[Biological resources of the Black Sea and Sea of Azov]*. Sevastopol, Ukraine: EKOSI-Gidrofizika.

Erkakan, F., & Ozdemir, F. (2012). The first new cave fish species, *Cobitis damlae* (Teleostei: Cobitidae) from Turkey, *Hacettepe Journal of Biology & Chemistry*, *42*(2), 275–279.

Ermakhanov, Z. K., Plotnikov, I. S., Aladin, N. V., & Micklin, P. (2012). Changes in the Aral Sea ichthyofauna and fishery during the period of ecological crisis. *Lakes and Reservoirs: Research and Management*, *17*(1), 3–9. https://doi.org/10.1111/j.1440-1770.2012.00492.x

Eskildsen, A., Carvalheiro, L. G., Kissling, W. D., Biesmeijer, J. C., Schweiger, O., & Høye, T. T. (2015). Ecological specialization matters: Long-term trends in butterfly species richness and assemblage composition depend on multiple functional traits. *Diversity and Distributions*, *21*(7), 792–802. http://doi.org/10.1111/ddi.12340

Essl, F., & Lambdon, P. W. (2009). Alien bryophytes and lichens of Europe. In DAISIE, *Handbook of alien species in Europe* (pp. 29–41). Dordrecht, The Netherlands: Springer. https://doi.org/10.1007/978-1-4020-8280-1_3

Essl, F., Moser, D., Dullinger, S., Mang, T., & Hulme, P. E. (2010). Selection for commercial forestry determines global patterns of alien conifer invasions. *Diversity and Distributions*, *16*(6), 911–921. https://doi.org/10.1111/j.1472-4642.2010.00705.x

Essl, F., Steinbauer, K., Dullinger, S., Mang, T., & Moser, D. (2013). Telling a different story: A global assessment of bryophyte invasions. *Biological Invasions*, *15*(9), 1933–1946. https://doi.org/10.1007/s10530-013-0422-2

Estrada, A., Morales-Castilla, I., Caplat, P., & Early, R. (2016). Usefulness of species traits in predicting range shifts. *Trends in Ecology and Evolution*, *31*(3), 190–203. https://doi.org/10.1016/j.tree.2015.12.014

ETC/ICM. (2014). *Initial Assessment of European Seas based on Marine Strategy Framework Directive Article 8 reporting: Analysis of features and characteristics reported under MSFD 8.a.* Retrieved 16 June, 2015, from http://forum.eionet. europa.eu/nrc-marine-coastal-and-maritime/library/former-interest-groups-related-coastal-marine-and-maritime/nrc-marine-and-coastal-environment/background-reports-etc-icm-technical-report-1-2015-msfd-article-8-reporting

Euro+Med. (2017). Euro+Med PlantBase - the information resource for Euro-Mediterranean plant diversity. Retrieved January 31, 2017, from http://www. emplantbase.org/home.html

European Commission. (2009). European Redlist - Environment - European Commission. Retrieved September 20, 2017, from http://ec.europa.eu/environment/nature/conservation/species/redlist/amphibians/status.htm

European Commission. (2016). *European Red List of Habitats. Part 2. Terrestrial and freshwater habitats.* Luxembourg: Publications Office of the European Union. http://doi.org/10.2779/091372

European Reptile & Amphibian Specialist Group. (1996). *Trionyx triunguis (Mediterranean subpopulation).* The IUCN Red List of Threatened Species. Retrieved from https://doi.org/10.2305/IUCN. UK.1996.RLTS.T22200A9364253.en

European Union. (2007). Council Regulation (EC) No 708/2007 concerning use of alien and locally absent species in aquaculture. *Official Journal of the European Union* (*28.6.2007*), L 168/1.

Evenari, M., Shanan, L., & Tadmor, N. (1982). *The Negev: The challenge of a desert* (2nd ed.). Boston, USA: Harvard University Press.

Ewald, J., Hennekens, S. M., Conrad, S., Wohlgemuth, T., Jansen, F., Jenssen, M., Cornlis, J., Michiels, H., Kayser, J., Chytry, M., Gegout, J. C., Breuer, M., Abs, C., Walentowski, H., Starlinger, F., & Godefroid, S. (2013). Spatial and temporal patterns of Ellenberg values for nutrients in forests of Germany and adjacent regions - a survey based on phytosociological databases. *Tuexenia, 33,* 93-109.

Faber-Langendoen, D., & Josse, C. (2010). *World grasslands and biodiversity patterns: A report to IUCN ecosystem management programme.* Arlington, USA: Natureserve.

Fagúndez, J. (2013). Heathlands confronting global change: Drivers of biodiversity loss from past to future scenarios. *Annals of Botany*, *111*(2), 151–172. http://doi.org/10.1093/aob/mcs257

Faith, D. P. (1992). Conservation evaluation and phylogenetic diversity. *Biological Conservation*, *61*(1), 1–10. http://doi.org/10.1016/0006-3207(92)91201-3

Faith, D. P. (2017). A general model for biodiversity and its value. In J. Garson, A. Plutynski, & S. Sarkar (Eds.), *The Routledge handbook of philosophy of biodiversity* (pp. 69-85)*.* Abingdon, UK: Routledge.

Falasco, E., Ector, L., Isaia, M., Wetzel, C. E., Hoffmann, L., & Bona, F. (2014). Diatom flora in subterranean ecosystems: A review. *International Journal of Speleology*, *43*(3), 231–251. https://doi.org/10.5038/1827-806X.43.3.1

Falcucci, A., Maiorano, L., Boitani, & L. (2007) Changes in land-use/land-cover patterns in Italy and their implications for biodiversity conservation. *Landscape Ecology, 22,* 617-631. https://doi.org/10.1007/s10980-006-9056-4

FAO. (1989). *Arid zone forestry: A guide for field technicians. FAO Conservation Guide, 20.*

FAO. (2007). *The first report on the state of world's animal genetic resources for food and agriculture.*

FAO. (2010). *The second report on the state of the world's plant genetic resources for food and agriculture.*

FAO. (2011). *Review of the state of world marine fishery resources. FAO Fisheries and Aquaculture Technical Paper* (Vol. 569).

FAO. (2013a) *FAO statistical yearbook - World food and agriculture.*

FAO. (2013b). *State of forest resources in the Mediterranean region. State of Mediterranean forests 2013.*

FAO. (2015a). *The second report on the state of the world's animal genetic resources for food and agriculture.*

FAO. (2015b). *World reference base for soil resources 2014: International soil classification system for naming soils and creating legends for soil maps. Update 2015. World soil resources report 106.*

Favarger, C. (1972). Endemism in the montane floras of Europe. In D. H. Valentine (Ed.), *Taxonomy, Phytogeography and Evolution* (pp. 191–204). London, UK: Academic Press.

Fayvush, G., & Aleksayan, A. (2016). *Habitats of Armenia*. Yerevan: Institute of Botany.

Federal Forestry Agency [Рослесхоз]. (2013). Леса и лесные ресурсы Российской Федерации [Forests and Forest Resources of the Russian Federation]. In Ежегодный доклад «Осостоянии и использовании лесов Российской Федерации за *2011* год» *[Annual report "On stage and using of forests in the Russian Federation in 2011"]* (pp. 3–24). Moscow, Russian Federation: Rosleskhoz.

Fedoriak, M., Rudenko, S., Iaroshynska, O., & Zhukovets, E. (2012). Spiders (Aranaea) of Chernivtsi City (Ukraine). *Arachnologische Mitteilungen, 43*, 37–50.

Feest, A., Van Swaay, C., & Van Hinsberg, A. (2014). Nitrogen deposition and the reduction of butterfly biodiversity quality in the Netherlands. *Ecological Indicators*, *39,* 115-119. http://doi.org/10.1016/j.ecolind.2013.12.008

Feistel, R., Nausch, G., & Wasmund, N. (Eds.). (2008). *State and evolution of the Baltic Sea, 1952-2005: a detailed 50-year survey of meteorology and climate, physics, chemistry, biology, and marine environment*. Hoboken, USA: John Wiley & Sons.

Felicísimo, Á. M., Muñoz, J., Villalba, C. J., & Mateo, R. G. (2011). *Impactos, vulnerabilidad y adaptación al Cambio Climático de la biodiversidad española. 2. Flora y vegetación*. *[Impacts, vulnerability and adaptation to climate change of Spanish biodiversity. 2. Flora and vegetation]* Madrid, Spain: Ministry of the Environment.

Felline, S., Mollo, E., Ferramosca, A., Zara, V., Regoli, F., Gorbi, S., & Terlizzi, A. (2014). Can a marine pest reduce the nutritional value of Mediterranean fish flesh? *Marine Biology*, *161*(6), 1275–1283. https://doi.org/10.1007/s00227-014-2417-7

Fernandes, P. G., Ralph, G. M., Nieto, A., García Criado, M., Vasilakopoulos, P., Maravelias, C. D., Cook, R. M., Pollom, R. A., Kovačić, M., Pollard, D., Farrell, E. D., Florin, A.-B., Polidoro, B. A., Lawson, J. M., Lorance, P., Uiblein, F., Craig, M., Allen, D. J., Fowler, S. L., Walls, R. H. L., Comeros-Raynal, M. T., Harvey, M. S., Dureuil, M., Biscoito, M., Pollock, C., McCully Phillips, S. R., Ellis, J. R., Papaconstantinou, C., Soldo, A., Keskin, Ç., Knudsen, S. W., Gil de Sola, L., Serena, F., Collette, B. B., Nedreaas, K., Stump, E., Russell, B. C., Garcia, S., Afonso, P., Jung, A. B. J., Alvarez, H., Delgado, J., Dulvy, N. K., & Carpenter, K. E. (2017). Coherent assessments of Europe's marine fishes show regional divergence and megafauna loss. *Nature Ecology & Evolution*, *1*(7), UNSP 0200. https://doi.org/10.1038/s41559-017-0170

Ferrante, M., Conti, G. O., Fiore, M., Rapisarda, V., & Ledda, C. (2013). Harmful algal blooms in the Mediterranean Sea: Effects on human health. *EuroMediterranean Biomedical Journal, 8*(6), 25-34.

Feuerer, T. (2013). Checklists of lichens - A global information system for the biodiversity of lichens. Retrieved December 8, 2016, from http://www.lichens.uni-hamburg.de/lichens/portalpages/portalpage_checklists_switch.htm

Field, C. D., Dise, N. B., Payne, R. J., Britton, A. J., Emmett, B. A., Helliwell, R. C., Hughes, S., Jones, L., Lees, S., Leake, J. R., Leith, I. D., Phoenix, G. K., Power, S. A., Sheppard, L. J., Southon, G. E., Stevens, C. J., & Caporn, S. J. M. (2014). The role of nitrogen deposition in widespread plant community change across semi-natural habitats. *Ecosystems, 17*(5), 664-877. https://doi.org/10.1007/s10021-014-9765-5

Fijen, T. P. M., Kamp, J., Lameris, T. K., Pulikova, G., Urazaliev, R., Kleijn, D., & Donald, P. F. (2015). Functions of extensive animal dung pavements around

the nests of the black lark (*Melanocorypha yeltoniensis*). *Auk*, *132*(4), 878–892. https://doi.org/10.1642/AUK-15-38.1

Filibeck, G., Arrigoni, P. V, & Blasi, C. (2004). Some phytogeographical remarks on the forest vegetation of Colchis (western Georgia). *Webbia, 59,* 189–214. http://doi.org/10.1080/00837792.2004.10670768

Filippov, D. M., & [Филиппов, Д. М.]. (1968). Циркуляция и структура вод Черного моря *[Circulation and structure of waters of the Black Sea]*. Moscow, USSR: Наука [Science].

Finstad, A. G., & Hein, C. L. (2012). Migrate or stay: terrestrial primary productivity and climate drive anadromy in Arctic char. *Global Change Biology*, *18*(8), 2487–2497. https://doi.org/10.1111/j.1365-2486.2012.02717.x

Fletcher, D. H., Gillingham, P. K., Britton, J. R., Blanchet, S., & Gozlan, R. E. (2016). Predicting global invasion risks: a management tool to prevent future introductions. *Scientific Reports*, *6,* 26316. https://doi.org/10.1038/srep26316

Flowerdew, J.R. (1997). Mammal biodiversity in agricultural habitats. In R. C. Kirkwood (Ed.), *Biodiversity and conservation in agriculture* (pp. 25–40). Brighton, UK: British Crop Protection Council.

Fochetti, R., & Tierno De Figueroa, J. M. (2008). Global diversity of stoneflies (Plecoptera; Insecta) in freshwater. *Hydrobiologia*, *595*(1), 365–377. https://doi.org/10.1007/s10750-007-9031-3

Fominykh, A. S., & Lyapkov, S. M. (2011). The formation of new characteristics in life cycle of the marsh frog (*Rana ridibunda*) in thermal pond condition. *Zhurnal Obshchei Biologii*, *72*(6), 403–421. https://doi.org/10.1134/S2079086412030036

Ford, D., & Williams, P. (2007). *Karst Hydrogeology and Geomorphology*. https://doi.org/10.1002/9781118684986

Forsman, A. (2014). Effects of genotypic and phenotypic variation on establishment are important for conservation, invasion and infection biology. *Proceedings of the National Academy of Sciences of the United*

States of America, 111, 302–307. https://doi.org/10.1073/pnas.1317745111

Forsman A., & Wennersten, L. (2016). Inter-individual variation promotes ecological success of populations and species: evidence from experimental and comparative studies. *Ecography, 39,* 630–648. https://doi.org/10.1111/ecog.01357

Forest Europe. (2015). *State of Europe's Forests 2015.*

Fossheim, M., Primicerio, R., Johannesen, E., Ingvaldsen, R. B., Aschan, M. M., & Dolgov, A. V. (2015). Recent warming leads to a rapid borealization of fish communities in the Arctic. *Nature Climate Change, 5,* 673-677. http://doi.org/10.1038/NCLIMATE2647

Fraiture, A., & Otto, P. (2015). Distribution, ecology and status of 51 macromycetes in Europe: Results of the ECCF Mapping Programme. *Scripta Botanica Belgica*, *53*, 1–246.

Frederiksen, M. (2010). Appendix 1: seabirds in the North East Atlantic. A review of status, trends and anthropogenic impact. *TemaNord*, *587*, 47–122.

Frederiksen, M., Anker-Nilssen, T., Beaugrand, G., & Wanless, S. (2013). Climate, copepods and seabirds in the boreal Northeast Atlantic - current state and future outlook. *Global Change Biology*, *19*(2), 364–372. https://doi.org/10.1111/gcb.12072

Frederiksen, M., Harris, M. P., Daunt, F., Rothery, P., & Wanless, S. (2004). Scale-dependent climate signals drive breeding phenology of three seabird species. *Global Change Biology*, *10*(7), 1214–1221. https://doi.org/10.1111/j.1529-8817.2003.00794.x

Frederiksen, M., Krause-Jensen, D., Holmer, M., & Laursen, J. S. (2004). Long-term changes in area distribution of eelgrass (*Zostera marina*) in Danish coastal waters. *Aquatic Botany*, *78*(2), 167–181. https://doi.org/10.1016/j.aquabot.2003.10.002

Freyhof, J. (2011). *European red list of freshwater fishes*. Retrieved from http://scholar.google.com/scholar?hl%20=en&btnG=Search&q=intitle:European%20+red+list+of+freshwater+fishes%230

Freyhof, J., & Brooks, E. (2011). *European red list of freshwater fishes.* Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/85903

Friend, A. D., Lucht, W., Rademacher, T. T., Keribin, R., Betts, R., Cadule, P., Ciais, P., Clark, D. B., Dankers, R., Falloon, P. D., Ito, A., Kahana, R., Kleidon, A., Lomas, M. R., Nishina, K., Ostberg, S., Pavlick, R., Peylin, P., Schaphoff, S., Vuichard, N., Warszawski, L., Wiltshire, A., & Woodward, F. I. (2014). Carbon residence time dominates uncertainty in terrestrial vegetation responses to future climate and atmospheric CO_2. *Proceedings of the National Academy of Sciences of the United States of America*, *111*(9), 3280–5. https://doi.org/10.1073/pnas.1222477110

Friocourt, Y. F., Skogen, M., Stolte, W., & Albretsen, J. (2012). Marine downscaling of a future climate scenario in the North Sea and possible effects on dinoflagellate harmful algal blooms. *Food Additives and Contaminants - Part A Chemistry, Analysis, Control, Exposure and Risk Assessment*, *29*(10), 1630–1646. https://doi.org/10.1080/19440049.2012.714079

Fuchs, R., Herold, M., Verburg, P. H., Clevers, J. G. P. W., & Eberle, J. (2015). Gross changes in reconstructions of historic land cover/use for Europe between 1900 and 2010. *Global Change Biology*, *21*(1), 299–313. https://doi.org/10.1111/gcb.12714

Fujikura, K., Lindsay, D., Kitazato, H., Nishida, S., & Shirayama, Y. (2010). Marine biodiversity in Japanese waters. *PLoS ONE*, *5*(8), e11836. https://doi.org/10.1371/journal.pone.0011836

Fuller, R. J., Norton, L. R., Feber, R. E., Johnson, P. J., Chamberlain, D. E., Joys, A. C., Mathews, F., Stuart, R. C., Townsend, M. C., Manley, W. J., Wolfe, M. S., MacDonald, D. W., & Firbank, L. G. (2005). Benefits of organic farming to biodiversity vary among taxa. *Biology Letters*, *1*(4), 431–434. http://doi.org/10.1098/rsbl.2005.0357

Fundukchiev, S. E. (1987). *Anthropogenic transformation of the bird populations in "Golodnaya Steppe"* (Abstract of the thesis for Candidate of Biological Sciences).

Fung, T., Farnsworth, K., Shephard, S., Reid, D., & Rossberg, A. (2013). Why the size structure of marine communities can require decades to recover from fishing. *Marine Ecology Progress Series*, *484*, 155–171. https://doi.org/10.3354/meps10305

Gabriel, D., Carver, S. J., Durham, H., Kunin, W. E., Palmer, R. C., Sait, S. M., Stagl, S., & Benton, T. G. (2009). The spatial aggregation of organic farming in England and its underlying environmental correlates. *Journal of Applied Ecology*, *46*(2), 323–333. http://doi.org/10.1111/j.1365-2664.2009.01624.x

Gabriel, D., Sait, S. M., Hodgson, J. A., Schmutz, U., Kunin, W. E., & Benton, T. G. (2010). Scale matters: The impact of organic farming on biodiversity at different spatial scales. *Ecology Letters*, *13*(7), 858–869. http://doi.org/10.1111/j.1461-0248.2010.01481.x

Gabriel, D., I. Roschewitz, T. Tscharntke, and C. Thies. (2006). Beta diversity at different spatial scales: plant communities in organic and conventional agriculture. *Ecological Applications 16,* 2011-2021. http://doi.org/10.1890/1051-0761(2006)016[2011:BDADSS]2.0.CO;2

Gage, J. D., & Tyler, P. A. (1991). *Deep-sea biology: a natural history of organisms at the deep-sea floor*. Cambridge, UK: Cambridge University Press.

Gagic, V., Bartomeus, I., Jonsson, T., Taylor, A., Winqvist, C., Fischer, C., Slade, E. M., Steffen-Dewenter, I., Emmerson, M., Potts, S. G., Tscharntke, T., Weisser, W., & Bommarco, R. (2015). Functional identity and diversity of animals predict ecosystem functioning better than species-based indices. *Proceedings of the Royal Society B: Biological Sciences, 282*(1801), 20142620. http://dx.doi.org/10.1098/rspb.2014.2620

Gallardo, B., & Aldridge, D. C. (2013). The "dirty dozen": Socio-economic factors amplify the invasion potential of 12 high-risk aquatic invasive species in Great Britain and Ireland. *Journal of Applied Ecology*, *50*(3), 757–766. http://doi.org/10.1111/1365-2664.12079

Galil, B., Marchini, A., Occhipinti-Ambrogi, A., & Ojaveer, H. (2017). The enlargement of the Suez Canal—Erythraean

introductions and management challenges. *Management of Biological Invasions*, *8*(2), 141–152. https://doi.org/10.3391/mbi.2017.8.2.02

Galil, B. S. (2007). Loss or gain? Invasive aliens and biodiversity in the Mediterranean Sea. *Marine Pollution Bulletin*, *55*(7–9), 314–322. https://doi.org/10.1016/j.marpolbul.2006.11.008

Galil, B. S., Boero, F., Campbell, M. L., Carlton, J. T., Cook, E., Fraschetti, S., Gollasch, S., Hewitt, C. L., Jelmert, A., Macpherson, E., Marchini, A., McKenzie, C., Minchin, D., Occhipinti-Ambrogi, A., Ojaveer, H., Olenin, S., Piraino, S., & Ruiz, G. M. (2015). "Double trouble": the expansion of the Suez Canal and marine bioinvasions in the Mediterranean Sea. *Biological Invasions*, *17*(4), 973–976. https://doi.org/10.1007/s10530-014-0778-y

Galil, B. S., Marchini, A., & Occhipinti-Ambrogi, A. (2015). East is east and West is west? Management of marine bioinvasions in the Mediterranean Sea. *Estuarine, Coastal and Shelf Science, 201,* 7-16. https://doi.org/10.1016/j.ecss.2015.12.021

Galil, B. S., Marchini, A., Occhipinti-Ambrogi, A., Minchin, D., Narščius, A., Ojaveer, H., & Olenin, S. (2014). International arrivals: widespread bioinvasions in European Seas. *Ethology Ecology & Evolution*, *26*(2–3), 152–171. https://doi.org/10.1080/03949370.2014.897651

Gamero, A. Brotons, L., Brunner, A., Foppen, R., Fornasari, L., Gregory, R. D., Herrando, S., Hořák, D., Jiguet, F., Kmecl, P., Lehikoinen, A., Lindström, Å., Paquet, J. Y., Reif, J., Sirkiä, P. M., Škorpilová, J., van Strien, A., Szép, T., Telenský, T., Teufelbauer, N., Trautmann, S., van Turnhout, C. A. M., Vermouzek, Z., Vikstrøm, T., & Voříšek, P. (2016). Tracking progress towards EU biodiversity strategy targets: EU policy effects in preserving its common farmland birds. *Conservation Letters*, *10*(4), 395-402. https://doi.org/10.1111/conl.12292

Gamfeldt, L., Snäll, T., Bagchi, R., Jonsson, M., Gustafsson, L., Kjellander, P., Ruiz-Jaen, M. C., Fröberg, M., Stendahl, J., Philipson,

C. D., Mikusiński, G., Andersson, E., Westerlund, B., Andrén, H., Moberg, F., Moen, J., & Bengtsson, J. (2013). Higher levels of multiple ecosystem services are found in forests with more tree species. *Nature Communications, 4,* 1340. https://doi.org/10.1038/ncomms2328

Gamfeldt, L., Lefcheck, J. S., Byrnes, J. E. K., Cardinale, B. J., Duffy, J. E., & Griffin, J. N. (2015). Marine biodiversity and ecosystem functioning: what's known and what's next? *Oikos, 124,* 252-265. https://doi.org/10.1111/oik.01549

Gamfeldt, L., Hillebrand, H., & Jonsson, P. R. (2008). Multiple functions increase the importance of biodiversity for overall ecosystem functioning. *Ecology 89*(5), 1223–1231. https://doi.org/10.1890/06-2091.1

Gamfeldt, L., & Roger, F. (2017). Revisiting the biodiversity-ecosystem multifunctionality relationship. *Nature Ecology & Evolution, 1*(7), 0168. https://doi.org/10.1038/s41559-017-0168

García-Charton, J. A., Pérez-Ruzafa, A., Marcos, C., Claudet, J., Badalamenti, F., Benedetti-Cecchi, L., Falcón, J. M., Milazzo, M., Schembri, P. J., Stobart, B., Vandeperre, F., Brito, A., Chemello, R., Dimech, M., Domenici, P., Guala, I., Le Diréach, L., Maggi, E., & Planes, S. (2008). Effectiveness of European Atlanto-Mediterranean MPAs: Do they accomplish the expected effects on populations, communities and ecosystems? *Journal for Nature Conservation*, *16*(4), 193–221. https://doi.org/10.1016/j.jnc.2008.09.007

García Molinos, J., Halpern, B. S., Schoeman, D. S., Brown, C. J., Kiessling, W., Moore, P. J., Pandolfi, J. M., Poloczanska, E. S., Richardson, A. J., & Burrows, M. T. (2016). Climate velocity and the future global redistribution of marine biodiversity. *Nature Climate Change*, *6*(1), 83–88. https://doi.org/10.1038/nclimate2769

Gascon, C., Brooks, T. M., Contreras-MacBeath, T., Heard, N., Konstant, W., Lamoreux, J., Launay, F., Maunder, M., Mittermeier, R., Molur, S., Al Mubarak, A., Parr, M., Rhodin, A., Ry, A., & Vié, J.-C. (2015). The importance and benefits of species. *Current Biology*, *25*(10),

R431–R438. http://doi.org/10.1016/J.CUB.2015.03.041

Gauquelin, T., Michon, G., Joffre, R., Duponnois, R., Génin, D., Fady, B., Bou Dagher-Kharrat, M., Derridj, A., Slimani, S., Badri, W., Alifriqui, M., Auclair, L., Simenel, R., Aderghal, M., Baudoin, E., Galiana, A., Prin, Y., Sanguin, H., Fernandez, C., & Baldy, V. (2016). Mediterranean forests, land use and climate change: a social-ecological perspective. *Regional Environmental Change*, *18*(3), 623-636. http://doi.org/10.1007/s10113-016-0994-3

Gauthier, G., Bêty, J., Cadieux, M.-C., Legagneux, P., Doiron, M., Chevallier, C., Lai, S., Tarroux, A., & Berteaux, D. (2013). Long-term monitoring at multiple trophic levels suggests heterogeneity in responses to climate change in the Canadian Arctic tundra. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *368*(1624), 20120482. https://doi.org/10.1098/rstb.2012.0482

Gegechkori, A. M. (2011). The results of biogeographical study of Arcto-Tertiary refugia (Colchis And Talysh) of southern Caucasus. *Annals of Agrarian Science*, *9*(1), 10–16. http://doi.org/10.1007/s13398-014-0173-7.2

Gehrig-Fasel, J., Guisan, A., & Zimmermann, N. E. (2007). Tree line shifts in the Swiss Alps: Climate change or land abandonment? *Journal of Vegetation Science*, *18*(4), 571–582. http://doi.org/10.1111/j.1654-1103.2007.tb02571.x

Gehrig-Fasel, J., Guisan, A., & Zimmermann, N. E. (2008). Evaluating thermal treeline indicators based on air and soil temperature using an air-to-soil temperature transfer model. *Ecological Modelling*, *213*(3–4), 345–355. http://doi.org/10.1016/j.ecolmodel.2008.01.003

Gennaro, P., Piazzi, L., Persia, E., & Porrello, S. (2015). Nutrient exploitation and competition strategies of the invasive seaweed *Caulerpa cylindracea*. *European Journal of Phycology*, *50*(4), 384–394. https://doi.org/10.1080/09670262.2015.1055591

Georgiadi, A. G., Koronkevich, N. I., Milyukova, I. P., & Barabanova, E. A. (2014). The ensemble scenarios projecting

runoff changes in large Russian river basins in the 21st century. *Proceedings of the International Association of Hydrological Sciences, 364,* 210-2015. https://doi.org/10.5194/piahs-364-210-2014

Georgievsky, M. (2016). Water resources of the Russian rivers and their changes. *Proceedings of the International Association of Hydrological Sciences, 374*, 75–77. https://doi.org/10.5194/piahs-374-75-2016

Geptner, V. G., Chapskiy, K. K., Arseniev, V. A., & Sokolov, V. E., [Гептнер, В. Г., Чапский, К. К., Арсеньев, В. А. & Соколов, В. Е.]. (1976). Млекопитающие Советского Союза. Том *2/3.* Ластоногие и зубатые киты *[Mammals of the Soviet Union. Volume 2/3. Pinnipeds and toothed whales]*. Moscow, USSR: Высшая школа [Higher School].

Gerasimov, I. P., Davitaia, F. F., Budyko, M. I. & Lavrenko, E. M. [Герасимов И. П, Давитая Ф. Ф., Будыко М. И. & Лавренко Е. М.] (Eds.). (1964). Физико-географический атлас мира *[Physico-geographical atlas of the world]*. Moscow, Russian Federation: Academy of Science of the USSR and the Main Agency of Geodesia and Cartography of the USSR. Retrieved from http://geochemland.ru/uchebnye-materialy/fgam

Gerson, U., & Seaward, M. R. D. (1977). Lichen-invertebrate associations. In M. R. D. Seaward (Ed.), *Lichen ecology* (pp. 69–119). London, UK: Academic Press.

Gerten, D., & Adrian, R. (2002). Species-specific changes in the phenology and peak abundance of freshwater copepods in response to warm summers. *Freshwater Biology*, *47*(11), 2163–2173. http://doi.org/10.1046/j.1365-2427.2002.00970.x

Ghorbani, A., Gravendeel, B., Naghibi, F., & de Boer, H. (2014). Wild orchid tuber collection in Iran: a wake-up call for conservation. *Biodiversity and Conservation*, *23*(11), 2749–2760. https://doi.org/10.1007/s10531-014-0746-y

Giakoumi, S. (2014). Distribution patterns of the invasive herbivore *Siganus luridus* (Rüppell, 1829) and its relation to native benthic communities in the central Aegean Sea, northeastern Mediterranean. *Marine Ecology*, *35*(1), 96–105. https://doi.org/10.1111/maec.12059

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Giakoumi, S., Scianna, C., Plass-Johnson, J., Micheli, F., Grorud-Colvert, K., Thiriet, P., Claudet, J., Di Carlo, G., Di Franco, A., Gaines, S. D., García-Charlton, J. A., Lubchenko, J., Reimer, J., Sala, E., & Guidetti, P. (2017). Ecological effects of full and partial protection in the crowded Mediterranean Sea: a regional meta-analysis. *Scientific Reports*, *7*(1), 8940. https://doi.org/10.1038/s41598-017-08850-w

Gibert, J., & Culver, D. C. (2005). Diversity patterns in Europe. In D. C. Culver & W. B. White (Eds.), *Encyclopedia of caves* (pp. 196–201). Amsterdam, The Netherlands: Elsevier Press.

Gibert, J., & Deharveng, L. (2002). Subterranean ecosystems: A truncated functional biodiversity. *BioScience*, *52*(6), 473–481. https://doi.org/10.1641/0006-3568(2002)052[0473:SEATFB]2.0.CO;2

Gil-Tena, A., Saura, S., & Brotons, L. (2007). Effects of forest composition and structure on bird species richness in a Mediterranean context: Implications for forest ecosystem management. *Forest Ecology and Management*, *242*(2–3), 470–476. https://doi.org/10.1016/j.foreco.2007.01.080

Gilbert, O. L. (Ed.). (1989). *The ecology of urban habitats*. London, UK: Chapman & Hall.

Gilbert, O. L. (1992). Lichen reinvasion with declining air polution. In J. Bates & A. Farmer (Eds.), *Bryophytes and lichens in a changing environment* (pp. 159–177). Oxford, UK: Clarendon Press.

Gimingham, C. H. (1972). *Ecology of Heathlands*. London, UK: Chapman & Hall.

Gladyshev, M. I., Sushchik, N. N., Anishchenko, O. V., Makhutova, O. N., Kolmakov, V. I., Kalachova, G. S., Kolmakova, A. A., & Dubovskaya, O. P. (2011). Efficiency of transfer of essential polyunsaturated fatty acids versus organic carbon from producers to consumers in a eutrophic reservoir. *Oecologia*, *165*, 521–531. https://doi.org/10.1007/s00442-010-1843-6

Glantz, M. H., & Zonn, I. S. (1997). *Scientific, environmental, and political issues in the circum-Caspian region*.

Dordrecht, The Netherlands: Kluwer Academic Publishers.

Glazovsky, N. F., & Orlovsky, N. S. [Глазовский, Н. Ф., & Орловский, Н. С.]. (1996). Проблемы опустынивания и засух в СНГ и пути их решения [The problems of desertification and drought in the CIS countries and ways to solve them]. Известия РАН, Серия Географическая *[Izvestiya RAS, Geographical Series]*, *4*, 7–23.

Glazovsky, N. F. [Глазовский, Н. Ф.]. (1990). Аральский кризис: Причины возникновения и пути выхода *[The Aral crisis: causes and ways out]*. Moscow, USSR: Наука [Science].

Glover, A. G., Gooday, A. J., Bailey, D. M., Billett, D. S. M., Chevaldonné, P., Colaço, A., Copley, J., Cuvelier, D., Desbruyères, D., Kalogeropoulou, V., Klages, M., Lampadariou, N., Lejeusne, C., Mestre, N. C., Paterson, G. L. J., Perez, T., Ruhl, H., Sarrazin, J., Soltwedel, T., Soto, E. H., Thatje, S., Tselepides, A., Van Gaever, S., & Vanreusel, A. (2010). Temporal change in deep-sea benthic ecosystems. A review of the evidence from recent time-series studies. *Advances in Marine Biology, 58*, 1-95. https://doi.org/10.1016/B978-0-12-381015-1.00001-0

Gninenko, Yu. I., Shiryaeva, N. V., & Schurov, V. I. [Гниненко, Ю. И., Ширяева, Н. В., & Щуров, В. И.]. (2014). Самшитовая огневка – новый инвазивный организм в лесах российского Кавказа [*Cydalima perspectalis* - a new invasive organism in the Russian Caucasus forests]. Карантин Растений. Наука И Практика *[The Plant Quarantine. Science and Practice]*, *1*(7), 32–39.

Godbold, J. A., Bailey, D. M., Collins, M. A., Gordon, J. D. M., Spallek, W. A., & Priede, I. G. (2013). Putative fishery-induced changes in biomass and population size structures of demersal deep-sea fishes in ICES sub-area VII, northeast Atlantic Ocean. *Biogeosciences*, *10*(1), 529–539. https://doi.org/10.5194/bg-10-529-2013

Gomoiu, M.-T., Begun, T., Caraus, I., Muresan, M., Oaie, G., Opreanu, P., Secrieru, D.; Teaca, A., & Vasiliu, D. (2012). Studying the Romanian Shelf benthic community in EU FP7 HYPOX project: ecosystem recovery trends vs.

fish kill events. In *EGU General Assembly* (pp. 9474-9474).

Goodman, S., & Dmitrieva, L. (2016). *Pusa caspica*. The IUCN Red List of Threatened Species. http://dx.doi.org/10.2305/IUCN.UK.2016-1.RLTS.T41669A45230700.en

Google (n.d.). *Google Earth.* Retrieved from https://earth.google.com/web/@49.20982722,155.46660111,-802.35379665a,11719842.44755626d,35y,-0h,0t,0r

Gossner, M. M., Lewinsohn, T. M., Kahl, T., Grassein, F., Boch, S., Prati, D., Birkhofer, K., Renner, S. C., Sikorski, J., Wubet, T., Arndt, H., Baumgartner, V., Blaser, S., Blüthgen, N., Börschig, C., Buscot, F., Diekötter, T., Jorge, L. R., Jung, K., Keyel, A. C., Klein, A.-M., Klemmer, S., Krauss, J., Lange, M., Müller, J., Overmann, J., Pašalić, E., Penone, C., Perović, D. J., Purschke, O., Schall, P., Socher, S. A., Sonnemann, I., Tschapka, M., Tscharntke, T., Türke, M., Venter, P. C., Weiner, C. N., Werner, M., Wolters, V., Wurst, S., Westphal, C., Fischer, M., Weisser, W. W., & Allan, E. (2016). Land-use intensification causes multitrophic homogenization of grassland communities. *Nature, 540,* 266-269. https://doi.org/10.1038/nature20575

Gotelli, N. J., & Stanton-Geddes, J. (2015). Climate change, genetic markers and species distribution modelling. *Journal of Biogeography*, *42*(9), 1577–1585. https://doi.org/10.1111/jbi.12562

Gottfried, M., Pauli, H., Futschik, A., Akhalkatsi, M., Barančok, P., Alonso, B., Luis, J., Coldea, G., Dick, J., Erschbamer, B., Calzado, F., Rosa, M., Kazakis, G., Krajči, J., Larsson, P., Mallaun, M., Michelsen, O., Moiseev, D., Moiseev, P., Molau, U., Merzouki, A., Nagy, L., Nakhutsrishvili, G., Pedersen, B., Pelino, G., Puscas, M., Rossi, G., Stanisci, A., Theurillat, J.-P., Tomaselli, M., Villar, L., Vittoz, P., Vogiatzakis, I., & Grabherr, G. (2012). Continent-wide response of mountain vegetation to climate change. *Nature Climate Change, 2*(2), 111-115. http://doi.org/10.1038/nclimate1329

Goulson, D., Lye, G., & Darvill, B. (2008). Decline and conservation of bumble bees. *Annual Review of Entomology*, *53*,

191–208. https://doi.org/10.1146/annurev. ento.53.103106.093454

Government of Armenia. (2015). *Armenia's third national communication on climate change*. Yerevan, Armenia: Lusabats Publishing House.

Government of Azerbaijan. (2014). *Azerbaijan fifth national report*. Retrieved from https://www.cbd.int/reports/search

Government of Kazakhstan. (2015). *The fifth national report on progress in implementation of the Convention on Biological Diversity*. Retrieved from https://www.cbd.int/reports/search

Government of Kyrgyzstan [Правительство Кыргызской Республики]. (2014). Приоритеты сохранения биологического разнообразия Кыргызской Республики на период до 2024 года [Biodiversity conservation priorities of the Kyrgyz Republic untill 2024].

Government of the Russian Federation [Правительство Российской Федерации]. (2013). Стратегия развития Арктической зоны Российской Федерации и обеспечения национальной безопасности на период до 2020 года [Strategy of the Arctic zone of the Russian Federation development and national security for the period up to 2020]. Retrieved from http://government.ru/info/18360/

Government of the Russian Federation [Правительство Российской Федерации]. (2015). Пятый национальный доклад «Сохранение биоразнообразия в Российской Федерации» [*5th national report 'Conservation of biodiversity in the Russian Federation'*]. Moscow, Russian Federation: Ministry of Natural Resources and Environment of the Russian Federation.

Government of Tajikistan. (2014). Fifth national report on preservation of biodiversity of the Republic of Tajikistan. Retrieved from https://www.cbd.int/reports/search

Government of Tajikistan [Правительство Таджикистана]. (2016). Национальная стратегия и план действий по сохранению биоразнообразия до 2020 гг. [*National biodiversity strategy and action plan till 2020*]. Retrieved from https://www.cbd.int/doc/world/tj/tj-nbsap-v2-ru.pdf

Government of Turkey. (2014). *Fifth national report.* Retrieved from https://www.cbd.int/reports/search

Government of Turkmenistan [Правительство Туркменистана]. (2015). Пятый Доклад по осуществлению решений конвенции ООН о биологическом разнообразии на национальном уровне *[Fifth national report to CBD]*. Retrieved from https://www.cbd.int/reports/search

Government of Uzbekistan. (2015). *Fifth national report of the Republic of Uzbekistan on Conservation of Biodiversity*. Retrieved from https://www.cbd.int/reports/search

Gozlan, R. E. (2008). Introduction of non-native freshwater fish: is it all bad? *Fish and Fisheries*, *9*, 106–115. https://doi.org/10.1111/j.1467-2979.2007.00267.x

Gozlan, R. E. (2015). Role and impact of non-native species on inland fisheries: The Janus syndrome. In J. F. Craig (Ed.), Freshwater fisheries ecology (pp. 770-778). London, UK: Wiley-Blackwell publisher.

Gozlan, R. E. (2016). Interference of non-native species with fisheries and aquaculture. In M. Vilà & P. E. Hulme (Eds.), *Impact of biological invasions on ecosystem services,* (pp. 119–137). Cham, Switzerland: Springer International Publishing. https://doi.org/10.1007/978-3-319-45121-3

Gozlan, R. E., St-Hilaire, S., Feist, S. W., Martin, P., & Kent, M. L. (2005). Biodiversity - Disease threat to European fish. *Nature, 435*(7045), 1046-1046. https://doi.org/10.1038/4351046a

Grabherr, G., Gottfried, M., Gruber, A., & Pauli, H. (1995). Patterns and current changes in alpine plant diversity. In *Ecological Studies 113: Arctic and Alpine Biodiversity* (Vol. 113, pp. 169–181). Berlin and Heidelberg, Germany: Springer. http://doi.org/10.1007/978-3-642-78966-3_12

Grabherr, G., Gottfried, M., & Pauli, H. (2010). Climate change impacts in alpine environments. *Geography Compass, 4*(8), 1133-1153.

Grace, J. B., Anderson, T. M., Seabloom, E. W., Borer, E. T., Adler, P. B., Harpole, W. S., Hautier, Y., Hillebrand, H., Lind, E. M., Pärtel, M.,

Bakker, J. D., Buckley, Y. M., Crawley, M. J., Damschen, E. I., Davies, K. F., Fay, P. A., Firn, J., Gruner, D. S., Hector, A., Knops, J. M., MacDougall, A. S., Melbourne, B. A., Morgan, J. W., Orrock, J. L., Prober, S. M., & Smith, M. D. (2016). Integrative modelling reveals mechanisms linking productivity and plant species richness. *Nature, 529,* 390–393. http://doi.org/10.1038/nature16524

Gravel, D., Albouy, C., & Thuiller, W. (2016). The meaning of functional trait composition of food webs for ecosystem functioning. *Philosophical Transactions of the Royal Society B: Biological Sciences, 371*(1694), 20150268. http://doi.org/10.1098/rstb.2015.0268.

Greenstreet, S. P. R., Rossberg, A. G., Fox, C. J., Le Quesne, W. J. F., Blasdale, T., Boulcott, P., Mitchell, I., Millar, C., & Moffat, C. F. (2012). Demersal fish biodiversity: species-level indicators and trends-based targets for the Marine Strategy Framework Directive. *ICES Journal of Marine Science*, *69*(10), 1789–1801. https://doi.org/10.1093/icesjms/fss148

Gregory, R. D., Vorisek, P., Van Strien, A., Gmelig Meyling, A. W., Jiguet, F., Fornasari, L., Reif, J., Chylarecki, P., & Burfield, I. J. (2007). Population trends of widespread woodland birds in Europe. *Ibis, 149*(Suppl.), 78–97. https://doi.org/10.1111/j.1474-919X.2007.00698.x

Greifswald. (2010). *The Hyrcan forest - Restoration of forest landscape in Talish region, Azerbaijan. A report to KfW*.

Grémillet, D., & Boulinier, T. (2009). Spatial ecology and conservation of seabirds facing global climate change: a review. *Marine Ecology Progress Series*, *391*, 121–138. https://doi.org/10.2307/24873660

Griffin, J. N., O'Gorman, E. J., Emmerson, M. C., Jenkins, S. R., Klein, A. -M., Loreau, M., & Symstad, A. (2009). Biodiversity and the stability of ecosystem functioning. In S. Naeem, D. E. Bunker, A. Hector, M. Loreau, & C. Perrings (Eds.), *Biodiversity, Ecosystem Functioning, and Human Wellbeing: An Ecological and Economic Perspective* (pp. 78-93). Oxford, UK, Oxford University Press.

Griffiths, H. I., Krystufek, B., & Reed, J. M. (Eds.). (2004). *Balkan biodiversity: Pattern and process in the European hotspot*. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Gross, K., Cardinale, B. J., Fox, J. W., Gonzalez, A., Loreau, M., Polley, H. W., Reich, P. B., & van Ruijven, J. (2014). Species richness and the temporal stability of biomass production: a new analysis of recent biodiversity experiments. *American Naturalist, 183*(1), 1–12. https://doi. org/10.1086/673915

Grossheim, A. A. [Гроссгейм А. А.]. (1926). Эспарцеты Кавказа [Caucasus Esparset]. In Записки научно-прикладного отдела Тифлисского ботанического сада [*Scientific Papers of Applied Section of Tiflis Botanical Garden], 5,* 149–168. Tiflis, USSR: *Tiflis Botanical Garden*

Gubbay, S., Sanders, N., Haynes, T., Janssen, J. A. M., Rodwell, J. R., Nieto, A., García Criado, M., Beal, S., Borg, J., Kennedy, M., Micu, D., Otero, M., Saunders, G., & Calix, M. (2016). *European red list of habitats. Part 1. Marine habitats.* Luxembourg: Publications Office of the European Union. https://doi. org/10.2779/032638

Guerold, F., Boudot, J. P., Gilles, J., Vein, D., Merlet, D., & Rouiller, J. (2000). Macroinvertebrate community loss as a result of headwater stream acidification in the Vosges Mountains (N-E France). *Biodiversity & Conservation*, *9*, 767–783. https://doi.org/10.1023/A:1008994122865

Guidetti, P. (2006a). Marine reserves reestablish lost predatory interactions and cause community changes in rocky reefs. *Ecological Applications*, *16*, 963–976. https://doi.org/10.1890/1051-0761(2006)016[0963:MRRLPI]2.0.CO;2

Guidetti, P. (2006b). Potential of marine reserves to cause community-wide changes beyond their boundaries. *Conservation Biology*, *21*(2), 540–545. https://doi. org/10.1111/j.1523-1939.2007.00657.x

Guidetti, P., Baiata, P., Ballesteros, E., Di Franco, A., Hereu, B., Macpherson, E., Micheli, F., Pais, A., Panzalis, P., Rosenberg, A. A., Zaala, M., & Sala, E. (2014). Large-scale assessment of Mediterranean marine protected areas

effects on fish assemblages. *PLoS ONE*, *9*(4), e91841. https://doi.org/10.1371/journal.pone.0091841

Guisan, A., & Theurillat, J. -P. (2001). Assessing alpine plant vulnerability to climate change: a modeling perspective. *Integrated Assessment*, *1*(1), 307–320. https://doi.org/10.1023/A:1018912114948

Gupta, R., Kienzler, K., Martius, C., Mirzabaev, A., Oweis, T., de Pauw, E., Qadir, M., Shideed, K., Sommer, R., Thomas, R., Sayre, K., Carli, C., Saparov, A., Bekenov, M., Sanginov, S., Nepesov, M., & Ikramov, R. (2009). *Research prospectus: A vision for sustainable land management research in Central Asia. ICARDA Central Asia and Caucasus Program. Sustainable Agriculture in Central Asia and the Caucasus Series No.1.* Tashkent, Uzbekistan: CGIAR-PFU.

Gurevich E. (2009). Influence of air temperature on the river runoff in winter (the Aldan river catchment case study). *Russian Meteorology and Hydrology*, *34*(9), 628–633. https://doi.org/10.1007/10.3103/S1068373909090088

Haaland, S. (2002). *Fem tusen år med flammer. Det europeiske lyngheilandskapet. [Five thousand years of flames. The European heathland landscape]*. Bergen, Norway: Vigmostad & Bjørke.

Haase, D., & Gläser, J. (2009). Determinants of floodplain forest development illustrated by the example of the floodplain forest in the District of Leipzig. *Forest Ecology and Management, 258*(5), 887-894. https://doi.org/10.1016/j.foreco.2009.03.025

Habel, J. C., Dengler, J., Janisova, M., Török, P., Wellstein, C., & Wiezik, M. (2013). European grassland ecosystems: threatened hotspots of biodiversity. *Biodiversity and Conservation*, *22*, 2131–2138. https://doi.org/10.1007/s10531-013-0537-x

Habel, J. C., Rodder, D., Schmitt, T. & Neve, G. (2011). Global warming will affect the genetic diversity and uniqueness of *Lycaena helle* populations. *Global Change Biology, 17,* 194-205. https://doi.org/10.1111/j.1365-2486.2010.02233.x

Halada, L., Evans, D., Romão, C., & Petersen, J. E. (2011). Which habitats of European importance depend on agricultural practices? *Biodiversity and Conservation*, *20*(11), 2365–2378. https://doi.org/10.1007/s10531-011-9989-z

Hall-Spencer, J. M., Pike, J., & Munn, C. B. (2007). Diseases affect cold-water corals too: *Eunicella verrucosa* (Cnidaria: Gorgonacea) necrosis in SW England. *Diseases of Aquatic Organisms*, *76*(2), 87–97. https://doi.org/10.3354/dao076087

Hall–Spencer, J., Allain, V., & Fosså, J. H. (2002). Trawling damage to northeast Atlantic ancient coral reefs. *Proceedings of the Royal Society B: Biological Sciences*, *269*(1490), 507-511. https://doi.org/10.1098/rspb.2001.1910

Hällfors, H., Backer, H., Leppänen, J. M., Hällfors, S., Hällfors, G., & Kuosa, H. (2013). The northern Baltic Sea phytoplankton communities in 1903-1911 and 1993-2005: A comparison of historical and modern species data. *Hydrobiologia*, *707*(1), 109–133. https://doi.org/10.1007/s10750-012-1414-4

Hallingbäck, T., & Hodgetts, N. (2001). *Status survey and conservation action plan for bryophytes: Mosses, liverworts and hornworts*. Gland, Switzerland: IUCN. Retrieved from https://portals.iucn.org/library/efiles/documents/2000-074.pdf

Hallmann, C. A., Foppen, R. P. B., Van Turnhout, C. A. M., de Kroon, H., & Jongejans, E. (2014). Declines in insectivorous birds are associated with high neonicotinoid concentrations. *Nature*, *511*(7509), 341–343. https://doi.org/10.1038/nature13531

Hallmann, C. A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., Stenmans, W., Müller, A., Sumser, H., Hörren, T., Goulson, D., & De Kroon, H. (2017). More than 75 percent decline over 27 years in total flying insect biomass in protected areas. *PLoS ONE*, *12*(10), e0185809. https://doi.org/10.1371/journal.pone.0185809

Halpern, B. S., Frazier, M., Potapenko, J., Casey, K. S., Koenig, K., Longo, C., Lowndes, J. S., Rockwood, R. C., Selig, E. R., Selkoe, K. A., & Walbridge, S. (2015). Spatial and temporal changes in

cumulative human impacts on the world's ocean. *Nature Communications*, *6*, 7615. https://doi.org/10.1038/ncomms8615

Halvorsen, R., Bryn, A., Erikstad, L., & Lindgaard, A. (2015). *Natur i Norge - NiN Versjon 2.0.0. [Nature in Norway – NiN Version 2.0.0.] V. 17*. Trondheim: Artsdatabanken.

Hamel, S., Killengreen, S. T., Henden, J. -A., Yoccoz, N., & Ims, R. A. (2013). Disentangling the importance of interspecific competition, food availability, and habitat in species occupancy: recolonization of the endangered Fennoscandian arctic fox. *Biological Conservation*, *160*, 114–120. https://doi.org/10.1016/j.biocon.2013.01.011

Hamer, A. J., & McDonnell, M. J. (2008). Amphibian ecology and conservation in the urbanising world: A review. *Biological Conservation, 141*(10) 2432-2449. https://doi.org/10.1016/j.biocon.2008.07.020

Hammond, P. S., Bearzi, G., Bjørge, A., Forney, K. A., Karczmarski, L., Kasuya, T., Perrin, W., Scott, M. D., Wang, J. Y., Wells, R. S. & Wilson, B. (2008). *Phocoena phocoena* (*Baltic Sea subpopulation*). The IUCN Red List of Threatened Species. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T17031A6739565.en

Handa, I. T., Aerts, R., Berendse, F., Berg, M. P., Bruder, A., Butenschoen, O., Chauvet, E., Gessner, M. O., Jabiol, J., Makkonen, M., McKie, B. G., Malmqvist, B., Peeters, E. T., Scheu, S., Schmid, B., van Ruijven, J., Vos, V. C., & Hättenschwiler, S. (2014). Consequences of biodiversity loss for litter decomposition across biomes. *Nature 509,* 218–221. https://doi.org/10.1038/nature13247

Hanski, I. (2014). Biodiversity, microbes and human well-being. *Ethics in Science and Environmental Polititics, 14,* 19–25. https://doi.org/10.3354/esep00150

Hanski, I., von Hertzen, L., Fyhrquist, N., Koskinen, K., Torppa, K., Laatikainen, T., Karisola, P., Auvinen, P., Paulin, L., Makela, M. J., Vartiainen, E., Kosunen, T. U., Alenius, H., & Haahtela, T. (2012). Environmental biodiversity, human microbiota, and allergy are interrelated.

Proceedings of the National Academy of Sciences of the United States of America, 109(21) 8334-8339. http://doi.org/10.1073/pnas.1205624109

Harding, K. C., & Härkönen, T. J. (1999). Development in the Baltic grey seal (*Halichoerus grypus*) and ringed seal (*Phoca hispida*) Populations during the 20th Century. *Ambio*, *28*(7), 619–627.

Härdtle, W., von Oheimb, G., Gerke, A.-K., Niemeyer, M., Niemeyer, T., Assmann, T., Drees, C., Matern, A., & Meyer, H. (2009). Shifts in N and P budgets of heathland ecosystems: Effects of management and atmospheric inputs. *Ecosystems*, *12*(2), 298–310. http://doi.org/10.1007/s10021-008-9223-3

Härkönen, T. (2015). *Pusa hispida ssp. botnica*. The IUCN Red List of Threatened Species. https://doi.org/10.2305/IUCN.UK.2015-4.RLTS.T41673A66991604.en

Harkonen, T., Harding, K. C., Wilson, S., Baimukanov, M., Dmitrieva, L., Svensson, C. J., & Goodman, S. J. (2012). Collapse of a marine mammal species driven by human impacts. *PLoS ONE*, *7*(9), e43130. https://doi.org/10.1371/journal.pone.0043130

Harrison, P. A., Berry, P. M., Simpson, G., Haslett, J. R., Blicharska, M., Bucur, M., Dunford, R., Egoh, B., Garcia-Llorente, M., Geamănă, N., Geertsema, W., Lommelen, E., Meiresonne, L., & Turkelboom, F. (2014). Linkages between biodiversity attributes and ecosystem services: A systematic review. *Ecosystem Services, 9,* 191–203. https://doi.org/10.1016/j.ecoser.2014.05.006

Hartel, T., Schweiger, O., Öllerer, K., Cog Iniceanu, D., & Arntzen, J. W. (2010). Amphibian distribution in a traditionally managed rural landscape of Eastern Europe: Probing the effect of landscape composition. *Biological Conservation*, *143*(5), 1118–1124. https://doi.org/10.1016/j.biocon.2010.02.006

Hassall, C., Thompson, D. J., & Harvey, I. F. (2010). The impact of climate-induced distributional changes on the validity of biological water quality metrics. *Environmental Monitoring and Assessment*, *160*, 451–456. https://doi.org/10.1007/s10661-008-0709-4

Hassel, K., & Soderstrom, L. (2005). The expansion of the alien mosses Orthodontium lineare and Campylopus introflexus in Britain and continental Europe. *Journal of the Hattori Botanical Laboratory, 97,* 183–193.

Hauck, M. (2009). Global warming and alternative causes of decline in arctic-alpine and boreal-montane lichens in north-western Central Europe. *Global Change Biology*, *15*(11), 2653–2661. https://doi.org/10.1111/j.1365-2486.2009.01968.x

Hauck, M. (2010). Ammonium and nitrate tolerance in lichens. *Environmental Pollution*, *158*(5), 1127–1133. https://doi.org/10.1016/j.envpol.2009.12.036

Hauck, M., Bruyn, U. de, & Leuschner, C. (2013). Dramatic diversity losses in epiphytic lichens in temperate broad-leaved forests during the last 150 years. *Biological Conservation*, *157*, 136–145. https://doi.org/10.1016/j.biocon.2012.06.015

Hauck, M., Zimmermann, J., Jacob, M., Dulamsuren, C., Bade, C., Ahrends, B., & Leuschner, C. (2012). Rapid recovery of stem increment in Norway spruce at reduced SO_2 levels in the Harz Mountains, Germany. *Environmental Pollution*, *164*, 132–141. https://doi.org/10.1016/j.envpol.2012.01.026

Hautier Y, Seabloom EW, Borer ET, Adler PB, Harpole WS, Hillebrand H, Lind EM, MacDougall, A. S., Stevens, C. J., Bakker, J. D., Buckley, Y. M., Chu, C., Collins, S. L., Daleo, P., Damschen, E. I., Davies, K. F., Fay, P. A., Firn, J., Gruner, D. S., Jin, V. L., Klein, J. A., Knops, J. M. H., La Pierre, K. J., Li, W., McCulley, R. L., Melbourne, B. A., Moore, J. L., O'Halloran, L. R., Prober, S. M., Risch, A. C., Sankaran, M., Schuetz, M., & Hector, A. (2014). Eutrophication weakens stabilizing effects of diversity in natural grasslands. *Nature, 508*(7497), 521-525. https://doi.org/10.1038/nature13014

Hawksworth, D. L., Kirk, P. M., Sutton, B. C., & Pegler, D. N. (1996). Ainsworth & Bisby's dictionary of the fungi. *Revista Do Instituto de Medicina Tropical de São Paulo*, *38*(4), 272–272. https://doi.org/10.1590/S0036-46651996000400018

Hector, A., Hautier, Y., Saner, P., Wacker, L., Bagchi, R., Joshi, J., Scherer-Lorenzen, M., Spehn, E. M., Bazeley-White, E., Weilenmann, M., Caldeira, M. C., Dimitrakopoulos, P. G., Finn, J. A., Huss-Danell, K., Jumpponen, A., Mulder, C. P. H., Palmborg, C., Pereira, J. S., Siamantziouras, A. -S. D., Terry, A. C., Troumbis, A. Y., Schmid, B., & Loreau, M. (2010). General stabilizing effects of plant diversity on grassland productivity through population asynchrony and overyielding. *Ecology, 91*(8), 2213–2220. https://doi.org/10.1890/09-1162.1

Hector, A. & Bagchi, R. (2007). Biodiversity and ecosystem multifunctionality. *Nature, 448*(7150), 188–190. https://doi.org/10.1038/nature05947

Hedberg, P., Kotowski, W., Saetre, P., Mälson, K., Rydin, H., & Sundberg, S. (2012). Vegetation recovery after multiple-site experimental fen restorations. *Biological Conservation*, *147*(1), 60–67. https://doi.org/10.1016/j.biocon.2012.01.039

Hédl, R., Petřík, P., & Boublík, K. (2011). Long-term patterns in soil acidification due to pollution in forests of the Eastern Sudetes Mountains. *Environmental Pollution*, *159*(10), 2586–2593. https://doi.org/10.1016/j.envpol.2011.06.014

Hedwall, P. O., & Brunet, J. (2016). Trait variations of ground flora species disentangle the effects of global change and altered land-use in Swedish forests during 20 years. *Global Change Biology*, *22*(12), 4038–4047. https://doi.org/10.1111/gcb.13329

Heilmann-Clausen, J., Barron, E. S., Boddy, L., Dahlberg, A., Griffith, G. W., Nordén, J., Ovaskainen, O., Perini, C., Senn-Irlet, B., & Halme, P. (2015). A fungal perspective on conservation biology. *Conservation Biology*, *29*(1), 61–68. https://doi.org/10.1111/cobi.12388

Heino, J. (2002). Concordance of species richness patterns among multiple freshwater taxa: a regional perspective. *Biodiversity and Conservation*, *11*, 137–147. https://doi.org/10.1023/A:1014075901605

Heino, J., Virkkala, R., & Toivonen, H. (2009). Climate change and freshwater biodiversity: Detected patterns, future trends and adaptations in northern regions. *Biological Reviews*, *84*(1), 39–54. https://doi.org/10.1111/j.1469-185X.2008.00060.x

Heino, J., Virtanen, R., Vuori, K. M., Saastamoinen, J., Ohtonen, A., & Muotka, T. (2005). Spring bryophytes in forested landscapes: Land use effects on bryophyte species richness, community structure and persistence. *Biological Conservation, 124*(4), 539-545. https://doi.org/10.1016/j.biocon.2005.03.004

Hejcman, M., Hejcmanová, P., Pavlů, V., & Beneš, J. (2013). Origin and history of grasslands in Central Europe - A review. *Grass and Forage Science*, *68*(3), 345–363. https://doi.org/10.1111/gfs.12066

Hejcman, M., Száková, J., Schellberg, J., Šrek, P., Tlustoš, P., & Balík, J. (2010). The Rengen grassland experiment: Bryophytes biomass and element concentrations after 65 years of fertilizer application. *Environmental Monitoring and Assessment*, *166*(1–4), 653–662. https://doi.org/10.1007/s10661-009-1030-6

HELCOM. (2009). *Biodiversity in the Baltic Sea*.

HELCOM. (2010). Ecosystem health of the Baltic sea 2003-2007: HELCOM initial holistic assessment. Helsinki, Finland: Helsinki Commission

HELCOM. (2013). HELCOM red list of Baltic Sea species in danger of becoming extinct. *Baltic Sea Environment Proceedings*, *140,* 110.

HELCOM. (2016). *Towards an assessment of ecological coherence of the marine protected areas network in the Baltic Sea region, (i), 141*. Retrieved from http://www.nature.com/doifinder/10.1038/nature13022

HELCOM. (2017a). *Abundance of coastal key fish species.* Retrieved July 6, 2017, from http://www.helcom.fi/baltic-sea-trends/indicators/abundance-of-key-coastal-fish-species

HELCOM. (2017b). *Abundance of waterbirds in the breeding season.* Retrieved July 26, 2017, from http://www.helcom.fi/baltic-sea-trends/indicators/abundance-of-waterbirds-in-the-breeding-season/

HELCOM. (2017c). *Abundance of waterbirds in the wintering season.* Retrieved July 26, 2017, from http://www.helcom.fi/baltic-sea-trends/indicators/abundance-of-waterbirds-in-the-wintering-season/

HELCOM. (2017d). *Distribution of Baltic seals.* HELCOM core indicator report. Retrieved July 26, 2017, from http://www.helcom.fi/baltic-sea-trends/indicators/distribution-of-baltic-seals/

HELCOM. (2017e). *First version of the "State of the Baltic Sea" report – June 2017 – to be updated in 2018.* Available at: http://stateofthebalticsea.helcom.fi

HELCOM. (2017f). *State of the soft-bottom macrofauna community*. Retrieved July 26, 2017, from http://www.helcom.fi/baltic-sea-trends/indicators/state-of-the-soft-bottom-macrofauna-community/

HELCOM. (2017g). *Trends in arrival of new non-indigenous species*. Retrieved July 26, 2017, from http://www.helcom.fi/baltic-sea-trends/indicators/trends-in-arrival-of-new-non-indigenous-species/

Henden, J.A., Ims, R.A., Yoccoz, N.G., Hellström, P., & Angerbjörn, A. (2010). Strength of asymmetric competition between predators in food webs ruled by fluctuating prey: The case of foxes in tundra. *Oikos,119*(1), 27-34. https://doi.org/10.1111/j.1600-0706.2009.17604.x

Henwood, W. D. (2010). Toward a strategy for the conservation and protection of the world's temperate grasslands. *Great Plains Research*, *20*(1), 121–134. Retrieved from http://digitalcommons.unl.edu/greatplainsresearch

Hering, D., Schmidt-Kloiber, A., Murphy, J., Lücke, S., Zamora-Muñoz, C., López-Rodríguez, M. J., Huber, T., & Graf, W. (2009). Potential impact of climate change on aquatic insects: A sensitivity analysis for European caddisflies (Trichoptera) based on distribution patterns and ecological preferences. *Aquatic Sciences*, *71*(1), 3–14. https://doi.org/10.1007/s00027-009-9159-5

Hermy, M., Honnay, O., Firbank, L., Grashof-Bokdam, C., & Lawesson, J. E. (1999). An ecological comparison between ancient and other forest plant species of Europe, and the implications for forest

conservation. *Biological Conservation*, *91*(1), 9–22. https://doi.org/10.1016/S0006-3207(99)00045-2

Herrera, P. M., García "Petu," J. A., & Balmori, A. (2015). Valladolid. In J. Kelcey (Ed.), *Vertebrates and invertebrates of European cities: Selected non-avian fauna* (pp. 207–253). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6

Hickling, R., Roy, D. B., Hill, J. K., Fox, R., & Thomas, C. D. (2006). The distributions of a wide range of taxonomic groups are expanding polewards. *Global Change Biology*, *12*(3), 450–455. https://doi.org/10.1111/j.1365-2486.2006.01116.x

Hiddink, J. G., Burrows, M. T., & García Molinos, J. (2015). Temperature tracking by North Sea benthic invertebrates in response to climate change. *Global Change Biology*, *21*(1), 117–129. https://doi.org/10.1111/gcb.12726

Hiddink, J. G., & ter Hofstede, R. (2008). Climate induced increases in species richness of marine fishes. *Global Change Biology*, *14*(3), 453–460. https://doi.org/10.1111/j.1365-2486.2007.01518.x

Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., & Troxler, T. G. (2014). *IPCC 2014, 2013 supplement to the 2006 IPCC guidelines for national greenhouse gas inventories: Wetlands*.

Hobohm, C., & Bruchmann, I. (2009). Endemische Gefäßpflanzen und ihre Habitate in Europa–Plädoyer für den Schutz der Grasland-Ökosysteme [Endemic vascular plants and their habitats in Europe – plea for the protection of grassland ecosystems]. *Berichte Der Reinhold-Tüxen-Gesellschaft*, *21*, 142–161.

Hochkirch, A., Nieto, A., Braud, Y., Buzzetti, F. M., Chobanov, D., Willemse, L., & Zuna-kratky, T. (2016). *European red list of grasshoppers, crickets and bush-crickets*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/60944

Hodd, R. L., Bourke, D., & Skeffington, M. S. (2014). Projected range contractions of European protected oceanic montane plant communities: Focus on climate change impacts is essential for their future conservation. *PLoS ONE, 9*(4), e95147. https://doi.org/10.1371/journal.pone.0095147

Hodgetts, N. (2015). *Checklist and country status of European bryophytes – towards a new red list for Europe*. *Irish Wildlife Manuals*. Ireland: National Parks and Wildlife Service. Retrieved from http://hdl.handle.net/2262/73373

Hodgetts, N. G. (1992). Measures to protect bryophytes in Great Britain. *Biological Conservation*, *59*(2–3), 259–264. https://doi.org/10.1016/0006-3207(92)90594-D

Hof, A. R. (2015). Alien species in a warming climate: a case study of the nutcracker and stone pines. *Biological Invasions*, *17*(5), 1533–1543. https://doi.org/10.1007/s10530-014-0813-z

Hofmeister, J., Hošek, J., Brabec, M., Dvořák, D., Beran, M., Deckerová, H., Burel, J., Kříž, M., Borovička, J., Běťák, J., Vašutová, M., Malíček, J., Palice, Z., Syrovátková, L., Steinová, J., Černajová, I., Holá, E., Novozámská, E., Čížek, L., Iarema, V., Baltaziuk, K., & Svoboda, T. (2015). Value of old forest attributes related to cryptogam species richness in temperate forests: A quantitative assessment. *Ecological Indicators*, *57*, 497–504. https://doi.org/10.1016/j.ecolind.2015.05.015

Holsinger, J. R. (1988). Troglobites: the evolution of cave-dwelling organisms. *American Scientist*, *76*(2), 146–153.

Hölzel, N., Haub, C., Ingelfinger, M. P., Otte, A., & Pilipenko, V. N. (2002). The return of the steppe large-scale restoration of degraded land in southern Russia during the post-Soviet era. *Journal for Nature Conservation*, *10*(2), 75–85. https://doi.org/10.1078/1617-1381-00009

Hooper, D. U., Adair, E. C., Cardinale, B. J., Byrnes, J. E. K., Hungate, B. A., Matulich, K. L., Gonzalez, A, Duffy, J. E., Gamfeldt, L., & O'Connor, M. I. (2012). A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature 486,* U105–U129. https://doi.org/10.1038/nature11118

Hudson, J. M. G., & Henry, G. H. R. (2009). Increased plant biomass in a high Arctic heath community from 1981 to 2008. *Ecology*, *90*(10), 2657–2663. http://doi.org/10.1890/09-0102.1

Hughes, A. R., Inouye, B. D., Johnson, M. T. J., Underwood, N., & Vellend, M. (2008). Ecological consequences of genetic diversity. *Ecology Letters 11,* 609–623. http://doi.org/10.1111/j.1461-0248.2008.01179.x

Hunkeler, P. (2007). Standing committee 27th meeting report for on-the-spot appraisal construction of the hydro-electric power station Lesce on the Dobra River (Croatia). In *Report of the on-the-spot appraisal* (*5-6 June 2007*).

Hunt, G. L., Drinkwater, K. F., Arrigo, K., Berge, J., Daly, K. L., Danielson, S., Daase, M., Hop, H., Isla, E., Karnovsky, N., Laidre, K., Mueter, F. J., Murphy, E. J., Renaud, P. E., Smith, W. O., Trathan, P., Turner, J., & Wolf-Gladrow, D. (2016). Advection in polar and sub-polar environments: Impacts on high latitude marine ecosystems. *Progress in Oceanography*, *149*, 40–81. https://doi.org/10.1016/j.pocean.2016.10.004

Hunt, G. L., Katō, H., & McKinnell, S. M. (2000). *Predation by marine birds and mammals in the subarctic North Pacific Ocean. PICES Scientific Report No. 14*.

Huntington, H., Weller, G., Bush, E., Callaghan, T. V., Kattsov, V. M., Nuttall M. (2005). Arctic climate impact assessment (ACIA). In C. Symon, L. Arris, & B. Heal (Eds.), *An introduction to the Arctic climate impact assessment* (pp. 10-16). Cambridge, UK: Cambridge University Press.

Hutchinson, W. F., van Oosterhout, C., Rogers, S. I., & Carvalho, G. R. (2003). Temporal analysis of archived samples indicates marked genetic changes in declining North Sea cod (*Gadus morhua*). *Proceedings of the Royal Society B: Biological Sciences*, *270*(1529), 2125–2132. https://doi.org/10.1098/rspb.2003.2493

Icelandic Natural History Institute. (1996). *Valisti 1 Plöntur [Red Data List (1) Plants]*.

Ignatov, M., Afonina, O., & Ignatova, E. (2006). Check-list of mosses of East Europe and North Asia. *Arctoa*, *15*, 1–130.

Ihlow, F., Courant, J., Secondi, J., Herrel, A., Rebelo, R., Measey, G. J., Lillo, F., De Villiers, F. A., Vogt, S., De Busschere, C., Backeljau, T., & Rodder, D. (2016). Impacts of climate change on the global invasion potential of the African clawed frog *Xenopus laevis*. *PLoS ONE, 11*(6), e0154869. https://doi.org/10.1371/journal.pone.0154869

Inger, R., Gregory, R., Duffy, J. P., Stott, I., Voříšek, P., & Gaston, K. J. (2015). Common European birds are declining rapidly while less abundant species' numbers are rising. *Ecology Letters*, *18*(1), 28–36. https://doi.org/10.1111/ele.12387

Innocenti, G., Stasolla, G., Goren, M., Stern, N., Levitt-Barmats, Y., Diamant, A., & Galil, B. S. (2017). Going down together: invasive host, *Charybdis longicollis* (Decapoda: Brachyura: Portunidae) and invasive parasite, *Heterosaccus dollfusi* (Cirripedia: Rhizocephala: Sacculinidae) on the upper slope off the Mediterranean coast of Israel. *Marine Biology Research*, *13*(2), 229–236. https://doi.org/10.1080/17451000.2016.1240873

Insarov, G., & Insarova, I. (2013). Lichens and plants in urban environment. In D. Malkinson, D. Czamanski, & I. Benenson (Eds.), *Modeling of Land-Use and Ecological Dynamics* (pp. 167–193). Berlin, Germany: Springer-Verlag.

Insarov, G., Moutchnik, E., & Insarova, I. [Инсаров, Г. Э., Мучник, Е. Э., & Инсарова, И. Д.]. (2010). Эпифитные лишайники в условиях загрязнения атмосферы Москвы: методология долговременного мониторинга [Epiphytic lichens under air pollution stress in Moscow: Methodology for long-term monitoring]. Проблемы экологического мониторинга и моделирования экосистем [*Problems of Ecological Monitoring and Ecosystem Modelling*], *23*, 277–296.

Insarov, G., & Schroeter, B. (2002). Lichen monitoring and climate change. In P. L. Nimis, C. Scheidegger, & P. A. Wolseley (Eds.), *Monitoring with Lichens - Monitoring Lichens* (pp. 183–201). The Hague, The Netherlands: Kluwer Academic Publishers.

Ionov R.N., & Lebedeva L.P. [Ионов, Р. Н., & Лебедева, Л. Р.]. (2004). Растительный Покров Западного Тянь-Шаня (Обзор современного состояния

флоры). Трансграничный проект по Центральной Азии по сохранению биоразнообразия Западного Тянь-Шаня [*Plant cover of western Tien Shan* (*review of the modern status on flora*). *Transbounda*. Bishkek, Kyrgyzstan: Global Environment Facility/World Bank.

IPBES. (2016a). *Assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production*. S. G. Potts, V. L. Imperatriz-Fonseca, H. T. Ngo, J. C. Biesmeijer, T. D. Breeze, L. V. Dicks, L. A. Garibaldi, R. Hill, J. Settele, A. J. Vanbergen, M. A. Aizen, S. A. Cunningham, C. Eardley, B. M. Freitas, N. Gallai, P. G. Kevan, A. Kovács-Hostyánszki, P. K. Kwapong, J. Li, X. Li, D. J. Martins, G. Nates-Parra, J. S. Pettis, R. Rader, & B. F. Viana (Eds.). Bonn, Germany: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IPBES. (2016b). *Summary for policymakers of the assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production*. S. G. Potts, V. L. Imperatriz-Fonseca, H. T. Ngo, J. C. Biesmeijer, T. D. Breeze, L. V. Dicks, L. A. Garibaldi, R. Hill, J. Settele, A. J. Vanbergen, M. A. Aizen, S. A. Cunningham, C. Eardley, B. M. Freitas, N. Gallai, P. G. Kevan, A. Kovács-Hostyánszki, P. K. Kwapong, J. Li, X. Li, D. J. Martins, G. Nates-Parra, J. S. Pettis, R. Rader, & B. F. Viana (Eds.). Bonn, Germany: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IPCC. (2013). *Climate change 2013: The physical science basis. Contribution of working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change*. T. F. Stocker, D. Qin, G. Plattner, M. M. B. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, & P. M. Midgley (Eds.). Cambridge, UK: Cambridge University Press.

IPCC. (2014a). *Climate change 2014: Impacts, adaptation, and vulnerability. Part A: Global and sectoral aspects. Contribution of working group II to the fifth assessment report of the Intergovernmental Panel on Climate Change*. C. B. Field, V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee,

M., K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White (Eds.). Cambridge, UK: Cambridge University Press.

IPCC. (2014b). *Climate change 2014: Impacts, adaptation, and vulnerability. Part B: Regional aspects. contribution of working group II to the fifth assessment report of the Intergovernmental Panel on Climate*. V. R. Barros, C. B. Field, D. J. Dokken, M. D. Mastrandrea, K. J. Mach, T. E. Billir, M. Chatterjee, M., K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea & L. L. White (Eds.). Cambridge, UK: Cambridge University Press.

Isbell, F., Calcagno, V., Hector, A., Connolly, J., Harpole, W. S., Reich, P. B., Scherer-Lorenzen, M., Schmid, B., Tilman, D., Van Ruijven, J., Weigelt, A., Wilsey, B. J., Zavaleta, E. S., & Loreau, M. (2011). High plant diversity is needed to maintain ecosystem services. *Nature*, *477*(7363), 199–202. http://doi.org/10.1038/nature10282

IUCN. (2016). *The IUCN Red List of Threatened Species. Version 2016-1*. Retrieved from http://www.iucnredlist.org

IUCN. (2017a). *How many species are threatened?* Retrieved from http://www.iucnredlist.org/about/summary-statistics

IUCN. (2017b). *The IUCN Red List of Threatened Species*. Retrieved January 31, 2017, from http://www.iucnredlist.org/

IUCN. (2017c). *The IUCN Red List of Threatened Species. Version 2017-1*. Retrieved from http://www.iucnredlist.org

Ivanov, O. A., & Sukhanov, V. V. (2015). Species structure of pelagic Ichthyocenes in Russian waters of far eastern seas and the Pacific Ocean in 1980–2009. *Journal of Ichthyology*, *55*(4), 497–526. https://doi.org/10.1134/S0032945215040037

Jähnig, S. C., Kuemmerlen, M., Kiesel, J., Domisch, S., Cai, Q., Schmalz, B., & Fohrer, N. (2012). Modelling of riverine ecosystems by integrated models: conceptual approach, a case study and research agenda. *Journal of Biogeography*, *39*, 2253–2263. https://doi.org/10.1111/jbi.12009

Janick, J. (Ed.). (2003). *Horticultural Reviews, Volume 29: Wild apple and fruit trees of Central Asia*. New York, USA: John Wiley & Sons.

Janssen, J. A. M., Rodwell, J. S., García Criado, M., Gubbay, S., Haynes, T., Nieto, A., Sanders, N., Landucci, F., Loidi, J., Ssymank, A.,Tahvanainen, T., Valderrabano, M., Acosta, A., Aronsson, M., Arts, G., Attorre, F., Bergmeier, E., Bijlsma, R.-J., Bioret, F., Biță-Nicolae, C., Biurrun, I., Calix, M., Capelo, J., Čarni, A., Chytrý, M., Dengler, J., Dimopoulos, P., Essl, F., Gard ell, H., Gigante, D., Giusso del Galdo, G., Hájek, G., Jansen, F., Jansen, J., Kapfer, J., Mickolajczak, A., Molina, J. A., Molnár, Z., Paternoster, D., Piernik, A., Poulin, B. Renaux, B., Schaminée, J. H. J., Šumberová, K., Toivonen, H., Tonteri, T., Tsiripidis, I., Tzonev, R., & Valachovič, M. (2016). *European red list of habitats. Part 2. Terrestrial and freshwater habitats*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/091372

Jansen, J., Rego, F., Goncalves, P., & Silveira, S. (1997). Fire, a strong land shaping element in the Sierra da Estrela (Portugal). *NNA Berichte*, *10*, 150–162.

Jelić, D., Kuljerić, M., Koren, T., Treer, D., Šalamon, D., Lončar, M., Podnar Lešić, M., Janev Hutinec, B., Bogdanović, T., Mekinić, S., & Jelić, K. (2012): Red book of amphibians and reptiles of Croatia. Ministry of Environmental and Nature protection, State Institute for Nature Protection. https://www.researchgate.net/publication/264938275_Red_book_of_Amphibians_and_Reptiles_of_Croatia

Jenkins, C., Pimm, S. L., & Joppa, L. N. (2013). *Global bird diversity*. Retrieved January 28, 2017, from www.BiodiversityMapping.org

Jeppesen, E., Brucet, S., Naselli-Flores, L., Papastergiadou, E., Stefanidis, K., Nõges, T., Attayde, J. L., Zohary, T., Coppens, J., Bucak, T., Menezes, R. F., Freitas, F. R. S., Kernan, M., Søndergaard, M., & Beklioğlu, M. (2015). Ecological impacts of global warming and water abstraction on lakes and reservoirs due to changes in water level and related changes in salinity. *Hydrobiologia*, *750*(1),

201–227. https://doi.org/10.1007/s10750-014-2169-x

Jeppesen, E., Kronvang, B., Meerhoff, M., Søndergaard, M., Hansen, K. M., Andersen, H. E., Lauridsen, T. L., Liboriussen, L., Beklioglu, M., Ozen, A., & Olesen, J. E. (2009). Climate change effects on runoff, catchment phosphorus loading and lake ecological state, and potential adaptations. *Journal of Environmental Quality*, *38*, 1930–1941. https://doi.org/10.2134/jeq2008.0113

Jeppesen, E., Meerhoff, M., Davidson, T. A., Trolle, D., Søndergaard, M., Lauridsen, T. L., Beklioglu, M., Brucet, S., Volta, P., Gonzalez-Bergonzoni, I., & Nielsen, A. (2014). Climate change impacts on lakes: An integrated ecological perspective based on a multi-faceted approach, with special focus on shallow lakes. *Journal of Limnology*, *73*(Suppl.), 88–111. https://doi.org/10.4081/jlimnol.2014.844

Jeppesen, E., Meerhoff, M., Holmgren, K., González-Bergonzoni, I., Teixeira-de Mello, F., Declerck, S. A. J., De Meester, L., Søndergaard, M., Lauridsen, T. L., Bjerring, R., Conde-Porcuna, J. M., Mazzeo, N., Iglesias, C., Reizenstein, M., Malmquist, H. J., Liu, Z., Balayla, D., & Lazzaro, X. (2010). Impacts of climate warming on lake fish community structure and potential effects on ecosystem function. *Hydrobiologia*, *646*(1), 73–90. https://doi.org/10.1007/s10750-010-0171-5

Jeppesen, E., Mehner, T., Winfield, I. J., Kangur, K., Sarvala, J., Gerdeaux, D., ask, M., Malmquist, H. J., Holmgren, K., Volta, P., Romo, S., Eckmann, R., Sandström, A., Blanco, S., Kangur, A., Ragnarsson Stabo, H., Tarvainen, M., Ventelä, A. M., Søndergaard, M., Lauridsen, T. L., & Meerhoff, M. (2012). Impacts of climate warming on the long-term dynamics of key fish species in 24 European lakes. *Hydrobiologia*, *694*(1), 1–39. https://doi.org/10.1007/s10750-012-1182-1

**Jeppesen, E., Sondergaard, M., Jensen, J. P., Havens, K. E., Anneville, O., Carvalho, L., Coveney, M. F., Deneke, R., Dokulil, M. T., Foy, B., Gerdeaux, D., Hampton, S. E., Hilt, S., Kangur, K., Kohler, J., Lammens, E. H. H. R., Lauridsen, T. L., Manca, M., Miracle,

M. R., Moss, B., Noges, P., Persson, G., Phillips, G., Portielje, R., Romo, S., Schelske, C. L., Straile, D., Tatrai, I., Willen, E., & Winder, M.** (2005). Lake responses to reduced nutrient loading - an analysis of contemporary long-term data from 35 case studies. *Freshwater Biology*, *50*(10), 1747–1771. https://doi.org/10.1111/j.1365-2427.2005.01415.x

Jeppesen, E., Søndergaard, M., & Liu, Z. (2017). Lake restoration and management in a climate change perspective: An introduction. *Water, 9*(2), 122. https://doi.org/10.3390/w9020122

Jeppesen, E., Winfield, I. J., Trochine, C., Brucet, S., Argillier, C., Arranz, I., Beklioglu, M., Benejam, L., Ferreira, T., Hesthagen, T, Kerstin Holmgren, K., Jeppesen, E., Kelly, F., Krause, T., Rask, M., Volta, P., Winfield, I. J., & Hesthagen, T. (2017). Non-native fish occurrence and biomass in 1943 western Palearctic lakes and reservoirs and their abiotic and biotic correlates. *Ecosystems, 21*(3), 395-409. https://doi.org/10.1007/s10021-017-0156-6

Jepson, P. D., Deaville, R., Barber, J. L., Aguilar, À., Borrell, A., Murphy, S., Barry, J., Brownlow, A., Barnett, J., Berrow, S., Cunningham, A. A., Davison, N. J., Ten Doeschate, M., Esteban, R., Ferreira, M., Foote, A. D., Genov, T., Giménez, J., Loveridge, J., Llavona, Á., Martin, V., Maxwell, D. L., Papachlimitzou, A., Penrose, R., Perkins, M. W., Smith, B., De Stephanis, R., Tregenza, N., Verborgh, P., Fernandez, A., & Law, R. J. (2016). PCB pollution continues to impact populations of orcas and other dolphins in European waters. *Scientific Reports*, *6,* 18573. https://doi.org/10.1038/srep18573

Ji, R., Jin, M., & Varpe, Ø. (2013). Sea ice phenology and timing of primary production pulses in the Arctic Ocean. *Global Change Biology*, *19*(3), 734–741. https://doi.org/10.1111/gcb.12074

Jiménez-Alfaro, B., Gavilán, R. G., Escudero, A., Iriondo, J. M., & Fernández-González, F. (2014). Decline of dry grassland specialists in Mediterranean high-mountain communities influenced by recent climate warming. *Journal of Vegetation Science*, *25*(6), 1394–1404. http://doi.org/10.1111/jvs.12198

Johannesen, E., Mørk, H. L., Korsbrekke, K., Wienerroither, R., Eriksen, E., Fossheim, M., Wenneck, T., Dolgov, A. V., Prokhorova, T., & Prozorkevich, D. (2017). *Arctic fishes in the Barents Sea 2004-2015: Changes in abundance and distribution*.

Johnson, M. S., Putwain, P. D., & Holliday, R. J. (1978). Wildlife conservation value of derelict metalliferous mine workings in Wales. *Biological Conservation*, *14*(2), 131–148. https://doi.org/10.1016/0006-3207(78)90029-0

Jonason, D., Andersson, G. K. S., Öckinger, E., Rundlöf, M., Smith, H. G., & Bengtsson, J. (2011). Assessing the effect of the time since transition to organic farming on plants and butterflies. *Journal of Applied Ecology*, *48*(3), 543–550. http://doi.org/10.1111/j.1365-2664.2011.01989.x

Jones, M. C., & Cheung, W. W. L. (2015). Multi-model ensemble projections of climate change effects on global marine biodiversity. *ICES Journal of Marine Science*, *72*(3), 741–752. https://doi.org/10.1093/icesjms/fsu172

Jones, M. C., Dye, S. R., Pinnegar, J. K., Warren, R., & Cheung, W. W. L. (2013). Applying distribution model projections for an uncertain future: The case of the Pacific oyster in UK waters. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *23*(5), 710–722. https://doi.org/10.1002/aqc.2364

Jönsson, M. T., Ruete, A., Kellner, O., Gunnarsson, U, & Snall, T. (2017). Will forest conservation areas protect functionally important diversity of fungi and lichens over time? *Biodiversity and Conservation 26*(11), 2547-2567. https://doi.org/10.1007/s10531-015-1035-0

Jönsson, M. T., & Thor, G. (2012). Estimating coextinction risks from epidemic tree death: Affiliate lichen communities among diseased host tree populations of *Fraxinus excelsior*. *PLoS ONE*, *7*(9), e45701. https://doi.org/10.1371/journal.pone.0045701

Joosten, H., Tanneberger, F., & Moen, A. (Eds.). (2017). *Mires and peatlands of Europe: Status, distribution and conservation*. Stuttgart, Germany: Schweizerbart Science Publishers.

Joosten, H., & Clarke, D. (2002). *Wise use of mires and peatlands – Background and principles including a framework for decision-making*. Devon, UK: International Mire Conservation Group and International Peat Society.

Joppa, L. N., O'Connor, B., Visconti, P., Smith, C., Geldmann, J., Hoffmann, M., Watson, J. E. M., Butchart, S. H. M., Virah-Swamy, M., Halpern, B. S., Ahmed, S. E., Balmford, A., Sutherland, W. J., Harfoot, M., Hilton-Taylor, C., Foden, W., Minin, E. Di, Pagad, S., Genovesi, P., Hutton, J., & Burgess, D. N. (2016). Filling in biodiversity threat gaps. *Science*, *352*(6284), 416–418. https://doi.org/10.1126/science.aaf3565

Jørgensen, L., Archambault, P., Armstrong, C., Dolgov, A., Edinger, E., Gaston, T., Hildebrand, J., Piepenburg, D., Smith, W., von Quillfeldt, C., Vecchione, M., & Rice, J. (2016a). Chapter 36G. Arctic Ocean. In *The first global integrated marine assessment - World ocean assessment I*. New York, USA: United Nations.

Jørgensen, P. S., Böhning-Gaese, K., Thorup, K., Tøttrup, A. P., Chylarecki, P., Jiguet, F., Lehikoinen, A., Noble, D. G., Reif, J., Schmid, H., van Turnhout, C., Burfield, I. J., Foppen, R., Voříšek, P., van Strien, A., Gregory, R. D., & Rahbek, C. (2016b). Continent-scale global change attribution in European birds - combining annual and decadal time scales. *Global Change Biology*, *22*(2), 530–543. https://doi.org/10.1111/gcb.13097

Jovanović Glavaš, O., Jalžić, B., & Bilandžija, H. (2017). Population density, habitat dynamic and aerial survival of relict cave bivalves from genus *Congeria* in the Dinaric karst. *International Journal of Speleology*, *46*(1), 13–22. https://doi.org/10.5038/1827-806X.46.1.2020

Juberthie, C. (2000). Conservation of subterranean habitats and species. In H. Wilkens, D. C. Culver, & W. Humphreys (Eds.), *Ecosystems of the world: Subterranean ecosystems* (pp. 691–700). Amsterdam, The Netherlands: Elsevier Science Publishers.

Jucker, T., Bouriaud, O., Avacaritei, D., & Coomes, D.A. (2014). Stabilizing effects of diversity on aboveground wood production in forest ecosystems: linking patterns and processes. *Ecology Letters, 17*(12), 1560–1569. https://doi.org/10.1111/ele.12382

Jucker, T., Avăcăriței, D., Bărnoaiea, I., Duduman, G., Bouriaud, O. and Coomes, D. A. (2016), Climate modulates the effects of tree diversity on forest productivity. *Journal of Ecology, 104*(2), 388–398. https://doi.org/10.1111/1365-2745.12522

Jungius, H. (2012). *Biodiversity preservation and integrated river basin development in the Syrdaria River valley of Kazakhstan. Final Evaluation Report*.

Kabisch, N., & Haase, D. (2013). Green spaces of European cities revisited for 1990-2006. *Landscape and Urban Planning, 110,* 113-122. https://doi.org/10.1016/j.landurbplan.2012.10.017

Kaiser, J., Clarke, K. R., Hinz, H., Austen, M. C. V, Somerfield, P. J., & Karakassis, I. (2006). Global analysis of response and recovery of benthic biota to fishing. *Marine Ecology Progress Series*, *311*, 1–14. https://doi.org/10.3354/meps311001

Kaland, P. E. (1986). The origin and management of Norwegian coastal heaths as reflected by pollen analysis. In K.-E. Behre (Ed.), *Anthropogenic indicators in pollen diagrams* (pp. 19–36). Rotterdam, The Netherlands: A.A. Balkema.

Kalkman, V. J., Boudot, J.-P., Bernard, R., Conze, K.-J., De Knijf, G., Dyatlova, E., Ferreira, S., Jović, M., Ott, J., Riservato, A., & Sahlén, G. (2010). *European red list of dragonflies*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/84650

Kamakhina, G. L. [Камахина, Г. Л.]. (2008). Ключевые инвазии чужеродных видов флоры и фауны Туркменистана [Key invasions of alien species of flora and fauna of Turkmenistan]. Проблемы Освоения Пустынь *[Problems of Desert Development]*, *4,* 34–38.

Kamenos, N. A. (2010). North Atlantic summers have warmed more than winters since 1353, and the response of marine zooplankton. *Proceedings of the National Academy of Sciences of the United States of America*, *107*(52), 22442–22447. https://doi.org/10.1073/pnas.1006141107

Kamilov, G. K. [Камилов Г. К.]. (1973). Рыбы и биологические основы рыбохозяйственного освоения водохранилищ Узбекистана [*Fishes and biological basis of fishery use of in Uzbekistan reservoirs]*. Tashkent, USSR: Fan.

Kamp, J., Koshkin, M. A., Bragina, T. M., Katzner, T. E., Milner-Gulland, E. J., Schreiber, D., Sheldon, R., Shmalenko, A., Smelansky, I., Terraube, J., & Urazaliev, R. (2016). Persistent and novel threats to the biodiversity of Kazakhstan's steppes and semi-deserts. *Biodiversity and Conservation, 25*(12), 2521-2541. https://doi.org/10.1007/s10531-016-1083-0

Kamp, J., Urazaliev, R., Balmford, A., Donald, P. F., Green, R. E., Lamb, A. J., & Phalan, B. (2015). Agricultural development and the conservation of avian biodiversity on the Eurasian steppes: A comparison of land-sparing and land-sharing approaches. *Journal of Applied Ecology*, *52*(6), 1578–1587. http://doi.org/10.1111/1365-2664.12527

Kamp, J., Urazaliev, R., Donald, P. F., & Hölzel, N. (2011). Post-Soviet agricultural change predicts future declines after recent recovery in Eurasian steppe bird populations. *Biological Conservation*, *144*(11), 2607–2614. https://doi.org/10.1016/j.biocon.2011.07.010

Kämpf, I., Mathar, W., Kuzmin, I., Hölzel, N., & Kiehl, K. (2016). Post-Soviet recovery of grassland vegetation on abandoned fields in the forest steppe zone of western Siberia. *Biodiversity and Conservation*, *25*(12), 2563-2580. https://doi.org/10.1007/s10531-016-1078-x

Karimov, B. (2011). An overview on desert aquaculture in Central Asia (Aral Sea Drainage Basin). In *Aquaculture in desert and arid lands* (pp. 61–84). Rome, Italy: FAO.

Karimov, B. K., Kamilov, B. G., & Matthies, M. (2014a). Unconventional water resources of agricultural origin and their re-utilization potential for development of desert land aquaculture in the Aral Sea basin. In *The global water system in the Anthropocene: Challenges for science and governance* (pp. 183–200). Cham, Switzerland: Springer International Publishing.

Karimov, B., Kamilov, B. G., Upare, M., Van Anrooy, R., Bueno, P., & Shohimardonov, D. R. (2009). *FAO Fisheries and Aquaculture Circular No. 1030/1. Inland capture fisheries and aquaculture in the republic of Uzbekistan: Current status and planning*.

Karimov B. K., Matthies, M., & Kamilov, B. G. (2014b). Unconventional water resources of agricultural origin and their re-utilization potential for development of desert land aquaculture in the Aral Sea basin. In A. Bhaduri, J. Bogardi, J. Leentvaar, & S. Marx (Eds.), *The global water system in the Anthropocene: Challenges for science and governance* (pp. 183–200). Cham, Switzerland: Springer International Publishing.

Kašák, J., Mazalová, M., Šipoš, J., & Kuras, T. (2015). Dwarf pine: invasive plant threatens biodiversity of alpine beetles. *Biodiversity and Conservation*, *24*(10), 2399–2415. https://doi.org/10.1007/s10531-015-0929-1

Kasymov, A. G. [Касымов, А. Г.]. (1987). Каспийское море *[The Caspian Sea]*. Leningrad, USSR: Гидрометеоиздат [Gidrometeoizdat].

Kats, N. Y. [Кац, Н. Я.]. (1971). Болота Земного шара *[Swamps of the Earth]*. Moscow, USSR: Наука [Nauka].

Kauserud, H., Heegaard, E., Büntgen, U., Halvorsen, R., Egli, S., Senn-Irlet, B., Krisai-Greilhuber, I., Dämon, W., Sparks, T., Nordén, J., Høiland, K., Kirk, P., Semenov, M., Boddy, L., & Stenseth, N. C. (2012). Warming-induced shift in European mushroom fruiting phenology. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(36), 14488–14493. https://doi.org/10.1073/pnas.1200789109

Kazanci, N., Girgin, S., & Dügel, M. (2004). On the limnology of Salda Lake, a large and deep soda lake in southwestern Turkey: future management proposals. *Aquatic Conservation-Marine and Freshwater Ecosystems*, *14*(2), 151–162. https://doi.org/10.1002/aqc.609

Kazantcheev, E. N. [Казанчеев, Е. Н.]. (1981). Рыбы Каспийского моря *[Fishes of the Caspian Sea]*. Moscow, USSR: Легкая и пищевая промышленность [Legkaya I pischevaya promyshlennost].

Kazstat. (2005). Kazstat. Retrieved from www.stat.gov.kz

Kazstat. (2016). Kazstat. Retrieved from www.stat.gov.kz

Kearns, C. A., Inouye, D. W., & Waser, N. M. (1998). Endangered mutualisms: The conservation of plant-pollinator interactions. *Annual Review of Ecology and Systematics*, *29,* 83–112. https://doi.org/10.1146/annurev.ecolsys.29.1.83

Kędra, M., Moritz, C., Choy, E. S., David, C., Degen, R., Duerksen, S., Ellingsen, I., Górska, B., Grebmeier, J. M., Kirievskaya, D., van Oevelen, D., Piwosz, K., Samuelsen, A., & Węsławski, J. M. (2015). Status and trends in the structure of Arctic benthic food webs. *Polar Research*, *34,* 23775. https://doi.org/10.3402/polar.v34.23775

Keeley, J. E., Bond, W. J., Bradstock, R. A., Pausas, J. G., & Rundel, P. W. (2012). *Fire in Mediterranean ecosystems: Ecology, evolution and management*. Cambridge, UK: Cambridge University Press.

Keenleyside, C., & Tucker, G. (2010). *Farmland abandonment in the EU: an assessment of trends and prospects.* London, UK: Institute for European Environmental Policy.

Keith, S. A., Newton, A. C., Morecroft, M. D., Bealey, C. E., & Bullock, J. M. (2009). Taxonomic homogenization of woodland plant communities over 70 years. *Proceedings of the Royal Society B: Biological Sciences*, *276*(1672), 3539–3544. https://doi.org/10.1098/rspb.2009.0938

Kelcey, J. (Ed.). (2015). *Vertebrates and invertebrates of European cities: Selected non-avian fauna.* (1ˢᵗ *ed.*). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6

Kelcey, J., & Rheinwald, G. (2005). Conclusions. In J. Kelcey & G. Rheinwald (Eds.), *Birds in European cities* (pp. 417–422). St. Katharinen, Germany: Ginster Verlag.

Kell, S. P., Maxted, N., & Bilz, M. (2012). European crop wild relative threat assessment: knowledge gained and lessons learnt. In N. Maxted, M. E. Dulloo, B. Ford-Lloyd, L. Frese, J. Iriondo, & M. Pinheiro de Carvalho (Eds.), *Agrobiodiversity conservation: Securing the diversity of crop wild relatives and landraces* (pp. 218–242). Wallingford, UK: CAB International.

Kenis, M., & Branco, M. (2010). Impact of alien terrestrial arthropods in Europe. Chapter 5. *BioRisk*, *4*(1), 51–71. https://doi.org/10.3897/biorisk.4.42

Kestrup, Å. M., Smith, D. L., & Therriault, T. W. (Eds.). (2015). *PICES scientific report No. 48*. *Report of working group 21 on non-indigenous aquatic species.*

Kessler, A, & Smith, A. T. (2014). The status of the great bustard (*Otis tarda tarda*) in Central Asia: from the Caspian Sea to the Altai. *Aquila, 121,* 115–132.

Ketelaars, H. A. M., Lambregts-van De Clundert, F. E., Carpentier, C. J., Wagenvoort, A. J., & Hoogenboezem, W. (1999). Ecological effects of the mass occurrence of the Ponto-Caspian invader, *Hemimysis anomala* G.O. Sars, 1907 (Crustacea: Mysidacea), in a freshwater storage reservoir in the Netherlands, with notes on its autecology and new records. *Hydrobiologia*, *394*, 233–248. https://doi.org/10.1023/A:1003619631920

Ketner-Oostra, R., & Sýkora, K. V. (2004). Decline of lichen-diversity in calcium-poor coastal dune vegetation since the 1970s, related to grass and moss encroachment. *Phytocoenologia*, *34*(4), 521–549. https://doi.org/10.1127/0340-269X/2004/0034-0521

Kharin, N. G. (2002). *Vegetation degradation in Central Asia under the impact of human activities*. Dordrecht, The Netherlands: Kluwer Academic. http://doi.org/10.1007/978-94-010-0425-1

Kharuk, V. I., Ranson, K. J., Im, S. T., & Naurzbaev, M. M. (2006). Forest-tundra larch forests and climatic trends. *Russian Journal of Ecology*, *37*(5), 291–298. https://doi.org/10.1134/S1067413606050018

Khrustalev, Y. P., Ivlieva, O. V., Bespalova, A. A., & Pirumova, E. I., [Хрусталев, Ю. П., Ивлиева, О. В., Беспалова, А. Ф., & Пирумова, Е. И.]. (2001). Ландшафты Азовского моря и их преобразование под влиянием хозяйственной деятельности [Landscapes of the Azov sea and their transformation under the impact of economic activity]. In Экологические проблемы природопользования *[Ecological problems of nature management]* (pp. 110–123). Таганрог, Russian Federation: FGU "Azovmorvod".

Kideys, A. E. (2002). Fall and rise of the Black Sea ecosystem. *Science*, *297*(5586), 1482–1484. https://doi.org/10.1126/science.1073002

Killengreen, S.T., E. Strømseng, N.G. Yoccoz, & R.A. Ims. (2012). How ecological neighbourhoods influence the structure of the scavenger guild in Low Arctic tundra. *Diversity and Distributions* 18(6), 563–74. https://doi.org/10.1111/j.1472-4642.2011.00861.x

Kipriyanova, L. M., Yermolaeva, N. I., Bezmaternykh, D. M., Dvurechenskaya, S. Y., & Mitrofanova, E. Y. (2007). Changes in the biota of Chany Lake along a salinity gradient. *Hydrobiologia*, *576,* 83–93. https://doi.org/10.1007/s10750-006-0295-9

Kirby, R. R., & Beaugrand, G. (2009). Trophic amplification of climate warming. *Proceedings of the Royal Society B: Biological Sciences*, *276*(1676), 4095–103. https://doi.org/10.1098/rspb.2009.1320

Kirschbaum, U., Windisch, U., Vorbeck, A., & Hanewald, K. (2006). Mapping lichen diversity in Wetzlar and Giessen as an indicator of air quality: Comparison between the surveys of 1970, 1985, 1995 and 2005. *Gefahrstoffe Reinhaltung Der Luft*, *66*(6), 272–280.

Klanderud, K., & Birks, H. J. B. (2003). Recent increases in species richness and shifts in altitudinal distributions of Norwegian mountain plants. *The Holocene*, *13*(1), 1–6. https://doi.org/10.1191/0959683603hl589ft

Kleijn, D., Baquero, R. A., Clough, Y., Díaz, M., De Esteban, J., Fernández, F., Gabriel, D., Herzog, F., Holzschuh, A., Jöhl, R., Knop, E., Kruess, A., Marshall, E. J. P., Steffan-Dewenter, I., Tscharntke, T., Verhulst, J., West, T. M., & Yela, J. L. (2006). Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, *9*(3), 243–254. http://doi.org/10.1111/j.1461-0248.2005.00869.x

Kleijn, D., Rundlöf, M., Scheper, J., Smith, H. G., & Tscharntke, T. (2011). Does conservation on farmland contribute to halting the biodiversity decline? *Trends in Ecology and Evolution*, *26*(9), 474–481. https://doi.org/10.1016/j.tree.2011.05.009

Kleinbauer, I., Dullinger, S., Peterseil, J., & Essl, F. (2010). Climate change might drive the invasive tree *Robinia pseudacacia* into nature reserves and endangered habitats. *Biological Conservation*, *143*(2), 382–390. https://doi.org/10.1016/j.biocon.2009.10.024

Klimanov, V. A., & Sirin, A. A. (1997). The dynamics of peat accumulation by mires of northern Eurasia during the last three thousand years. In *Northern forested wetlands: Ecology and management* (pp. 313–324). Raton, USA: CRC Press.

Klonner, G., Dullinger, I., Wessely, J., Bossdorf, O., Carboni, M., Dawson, W., Essl, F., Gattringer, A., Haeuser, E.,

van Kleunen, M., Kreft, H., Moser, D., Pergl, J., Pyšek, P., Thuiller, W., Weigelt, P., Winter, M., & Dullinger, S. (2017). Will climate change increase hybridization risk between potential plant invaders and their congeners in Europe? *Diversity and Distributions*, *23*(8), 934–943. https://doi.org/10.1111/ddi.12578

Knop, E., Zoller, L., Ryser, R., Gerpe, C., Hörler, M., & Fontaine, C. (2017). Artificial light at night as a new threat to pollination. *Nature 548*, 206–209. https://doi.org/10.1038/nature23288

Kobelt, M., & Nentwig, W. (2008). Alien spider introductions to Europe supported by global trade. *Diversity and Distributions, 14*(2), 273-280. https://doi.org/10.1111/j.1472-4642.2007.00426.x

Koch, M., Bowes, G., Ross, C., & Zhang, X.-H. (2013). Climate change and ocean acidification effects on seagrasses and marine macroalgae. *Global Change Biology*, *19*(1), 103–132. https://doi.org/10.1111/j.1365-2486.2012.02791.x

Kochnev, A. A. [Кочнев, А. А.]. (2004). Белый медведь на Чукотке: тревоги и надежды [Polar bear in Chukotka: concerns and hopes]. Охрана Дикой Природы *[Wildlife Conservation]*, *3*(29), 7–14.

Kochnev, A., & Zdor, E. (2014). *Harvest and use of polar bears in Chukotka: Results of 1999-2012 studies*. Moscow, Russian Federation: Pi Kvadrat.

Koh, L. P. (2004). Species coextinctions and the biodiversity crisis. *Science*, *305*(5690), 1632–1634. https://doi.org/10.1126/science.1101101

Kokorin, A., Blyakharchuk, T. A., Gerasimchuk, I. V., Gruza, G. V., Kamennova, I. E., Parfenova, Y. I., Rankova, E. Y., Semenov, V. A., & Tchebakova, N. M. (2011). *Assessment report: Climate change and its impact on ecosystems, population and economy of the Russian portion of the Altai-Sayan ecoregion*. A. Kokorin (Ed.). Moscow, Russian Federation: WWF Russia. Retrieved from https://wwf.ru/upload/iblock/7c5/assessment_climate_altai_eng_.pdf

Kolman, R., Kapusta, A., & Duda, A. (2011). Re-establishing the Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus* Mitchill) in Poland. In P. Williot, E. Rochard, J. Gessner, & F. Kirschbaum (Eds.), *Biology and Conservation of the European Sturgeon Acipenser sturio L. 1758* (pp. 573–581). Berlin, Germany: Springer. https://doi.org/10.1007/978-3-642-20611-5

König, Cl., & Weick, F. (2008). *Owls of the world*. London, UK: Christopher Helm.

Kontorovich, A. E., Eder, L. V., Filimonova, I. V., Nemov, V. U., & Provornaya, I. V. [Конторович, А. Е., Эдер, Л. В., Филимонова, И. В., Немов, В. Ю., & Проворная, И. В.]. (2013). Нефтяная промышленность Дальнего Востока: современное состояние и перспективы развития [Oil industry of the Russian Far East: modern state and development perspectives]. Бурение и Нефть *[Drilling and Oil]*, *7–8*, 3–9.

Konvička, M., Fric, Z., & Beneš, J. (2006). Butterfly extinctions in European states: Do socioeconomic conditions matter more than physical geography? *Global Ecology and Biogeography*, *15*(1), 82–92. https://doi.org/10.1111/j.1466-822X.2006.00188.x

Kopecký, M., Hédl, R., & Szabó, P. (2013). Non-random extinctions dominate plant community changes in abandoned coppices. *Journal of Applied Ecology*, *50*(1), 79–87. https://doi.org/10.1111/1365-2664.12010

Korovin, V. A. (2015). Long-term changes in the community of birds of the agricultural landscape in the Middle Urals. *Contemporary Problems of Ecology*, *8*(2), 227–231. https://doi.org/10.1134/S1995425515020092

Körner, C. (2003). *Alpine plant life: functional plant ecology of high mountain ecosystems* (2nd *Edition*). Berlin, Germany: Springer.

Körner, C., Paulsen, J., & Spehn, E. M. A. (2011). A definition of mountains and their bioclimatic belts for global comparisons of biodiversity data. *Alpine Botany, 121*(2), 73-78. https://doi.org/10.1007/s00035-011-0094-4

Körner, C. (2012). *Alpine treelines: functional ecology of the global high elevation tree limits*. Basel, Switzerland: Springer.

Korotchenko, I., & Peregrym, M. (2012). Ukrainian steppes in the past, at present and in the future. In M. J. A. Werger & M. A. van Staalduinen (Eds.), *Eurasian steppes. Ecological problems and livelihoods in a changing world* (pp. 173–196). Dordrecht, The Netherlands: Springer. https://doi.org/10.1007/978-94-007-3886-7_5

Korotenko, V. A., & Domashov, I. A. [Коротенко, В. А., & Домашов, И. А.]. (2014). Экологическое изменение развития Кыргызстана *[Ecological dimension of Kyrgyzstan development]*. Bishkek, Kyrgyzstan: БИОМ [BIOM].

Korpinen, S., & Jormalainen, V. (2008). Grazing and nutrients reduce recruitment success of *Fucus vesiculosus* L. (Fucales: Phaeophyceae). *Estuarine, Coastal and Shelf Science*, *78*(2), 437–444. https://doi.org/10.1016/j.ecss.2008.01.005

Kortsch, S., Primicerio, R., Fossheim, M., Dolgov, A. V., & Aschan, M. (2015). Climate change alters the structure of arctic marine food webs due to poleward shifts of boreal generalists. *Proceedings of the Royal Society B: Biological Sciences*, *282*(1814), 31-39. Retrieved from http://rspb.royalsocietypublishing.org/content/282/1814/20151546

Koslow, J. A., Auster, P., Bergstad, O. A., Roberts, J. M., Rogers, A., Vecchione, M., Harris, P., Rice, J, & Bernal, P. (2016). Chapter 51. Biological communities on seamounts and other submarine features potentially threatened by disturbance. In *The first global integrated marine assessment - World ocean assessment I*. New York, USA: United Nations.

Kotlyakov, V. M. [Котляков, В. М.] (Ed.). (2000). *National atlas of Russia* [Национальный Атлас России]. Moscow, Russian Federation: Федеральное агентство геодезии и картографии [The Federal Agency of Geodesy and Cartography]. Retrieved from http://национальныйатлас.рф/cd2/index.html

Kotlyakov, V. M. [Котляков, В. М.]. (2004). Азовское море *[The Azov Sea]. In* Национальный атлас России *[The National Atlas of Russia]* (*pp. 254–257*). Moscow, Russian Federation: The Federal Agency of Geodesy and Cartography.

Kotlyakov, V. M. (2010). Криосфера и климат [The cryosphere and climate]. Экология и жизнь [Ecology and life], 11, 51-60.

Kotova, I., Kayukova, E., & Kotov, S. (2016). Peloids of Crimean salt lakes and the Dead Sea: controls on composition and formation. *Environmental Earth Sciences, 75*(16), 1207. https://doi.org/10.1007/s12665-016-5999-1

Kotta, J, Almroth-Rosell, E, Andersson, H., Eero, M., Eilola, M., Hinrichsen, H., Jänes, H., MacKenzie, B., Meier, H. E. M., Ojaveer, H., Pärnoja, M., Skov, H, & von Dewitz, B. (2016). *Report on the nature and types of driver interactions including their potential future. BIO-C3 Deliverable, D3.2*. http://doi.org/10.3289/BIO-C3_D3.2

Kottek, M., Grieser, J., Beck, C., Rudolf, B., & Rubel, F. (2006). World map of the Köppen-Geiger climate classification updated. *Meteorologische Zeitschrift, 15*(3), 259–263. http://doi.org/10.1127/0941-2948/2006/0130

Kotze, J., Venn, S., Niemelä, J., & Spence, J. (2014). Effects of urbanization on the ecology and evolution of arthropods. *Urban Ecology Patterns, Processes, and Applications*, *6*(38), 45–66. https://doi.org/10.1093/acprof

Kouba, Y., Martínez-García, F., De Frutos, Á., & Alados, C. L. (2015). Effects of previous land-use on plant species composition and diversity in Mediterranean forests. *PLoS ONE*, *10*(9), 1–15. http://doi.org/10.1371/journal.pone.0139031

Kovács-Hostyánszki, A., Li, J. L., Pettis, J., Settele, J., Aneni, T., Espíndola, A., Kahono, S., Hajnalka, S., Thompson, H., Vanbergen, A., & Vandame, R. (2016). Chapter 2: Drivers of change of pollinators, pollination networks and pollination. In S. G. Potts, V. L. Imperatriz-Fonseca, & H. T. Ngo, J. C. Biesmeijer, T. D. Breeze, L. V. Dicks, L. A. Garibaldi, R. Hill, J. Settele, A. J. Vanbergen, M. A. Aizen, S. A. Cunningham, C. Eardley, B. M. Freitas, N. Gallai, P. G. Kevan, A. Kovács-Hostyánszki, P. K. Kwapong, J. Li, X. Li, D. J. Martins, G. Nates-Parra, J. S. Pettis, R. Rader, & B. F. Viana (Eds.), *Assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem*

Services on pollinators, pollination and food production (pp. 27-149). Bonn, Germany: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

Kovalenko, A. E., Bondartseva, M. A., Karatygin, I. V., Melnik, V. A., Novozhilov, Y. K., Popov, E. S., & Pystina, K. А. [Коваленко, А. Е., Бондарцева, М. А., Каратыгин, И. В., Мельник, В. А., Новожилов, Ю. К., Попов, Е. С., & Пыстина, К.А.]. (2005). Состояние изученности и оценка видового разнообразия грибов и миксомицетов России [State of the study and estimation of the species diversity of fungi and myxomycetes of Russia - Fungi in natural and anthropogenic ecosystems]. In Труды международной конференции «Грибы в природных и антропогенных экосистемах», посвященной столетию со дня начала работы профессора А. С. Бондарцева в Ботаническом институте им. Комарова РАН [*Proceedings of international conference dedicated to the 100th anniversary of the beginning of the work of prof. A.S. Bondartsev in the Komarov Botanical Institute Russian Academy of Science]* (pp. 267–270). Saint-Petersburg, Russian Federation: Ботанический институт им. Комарова [Komarov Botanical Institute].

Kovalevskaya, S. S., Bondarenko, O. N., Nabiev, M. M., & Pakhomova, M. G. [Бондаренко, О. Н., Набиев, М. М., Пахомова, М. Г., & Цукерваник, Т. И.] (Eds.). (1968-1993). Определитель растений Средней Азии: критический конспект флоры Средней Азии. Т. 1-10 *[Guide to plants of Central Asia: a critical synopsis of the flora of Central Asia. V. 1-10].* Tashkent, Uzbekistan: Fan.

Krader, L. (1955). Ecology of Central Asian pastoralism. *Southwestern Journal of Anthropology*, *11*(4), 301–326. https://doi.org/10.1086/soutjanth.11.4.3628907

Kraemer, R., Prishchepov, A. V., Müller, D., Kuemmerle, T., Radeloff, V. C., Dara, A., Terekhov, A., & Frühauf, M. (2015). Long-term agricultural land-cover change and potential for cropland expansion in the former Virgin Lands area of Kazakhstan. *Environmental Research Letters*, *10*(5), 54012. http://doi.org/10.1088/1748-9326/10/5/054012

Krám, P., Hruška, J., & Shanley, J. B. (2012). Streamwater chemistry in three contrasting monolithologic Czech catchments. *Applied Geochemistry*, *27*(9), 1854–1863. http://doi.org/10.1016/j.apgeochem.2012.02.020

Kraus, D., & Krumm, F. (2013). *Integrative approaches as an opportunity for the conservation of forest biodiversity*. *International Journal of Environmental Studies*, *71*(2), 226-227. https://doi.org/10.1080/00207233.2014.889472

Kresin, V. S., Eremenko, E. V., Zakharchenko, M. A., & Yurchenko, A. I., [Кресин, В. С., Еременко, Е. В., & Захарченко, М. А. Юрченко, А. И.]. (2008). Динамика поступления соединений фосфора в украинские прибрежные воды Черного моря и комплекс водоохранных мероприятий [Dynamics of phosphorus compounds arrivals in the Ukrainian coastal Black Sea water and complex of water conservation measures]. Экология Окружающей Среды И Безопасность *[Ecology Environment and Safety], 5,* 28–33.

Kricsfalusy, V., Mróz, W., & Popov, S. (2008). Historical changes of the upper tree line in the Carpathian Mountains (Ukraine). *Mountain Forum Bulletin*, *8*, 15–18.

Kristiansen, T., Stock, C., Drinkwater, K. F., & Curchitser, E. N. (2014). Mechanistic insights into the effects of climate change on larval cod. *Global Change Biology*, *20*(5), 1559–1584. https://doi.org/10.1111/gcb.12489

Kristinsson, H., Zhurbenko, M., & Steen Hansen, E. (2010). Panarctic checklist of lichens and lichenicolous. *CAFF Technical Report No. 20.* Akureyri, Iceland: CAFF International Secretariat.

Kuemmerle, T., Levers, C., Erb, K., Estel, S., Jepsen, M. R., Müller, D., Plutzar, C., Sturck, J., Verkerk, P. J., Verburg, P. H., & Reenberg, A. (2016). Hotspots of land use change in Europe. *Environmental Research Letters*, *11*(6), 64020. https://doi.org/10.1088/1748-9326/11/6/064020

Kuijper, D. P. J., Jedrzejewska, B., Brzeziecki, B., Churski, M., Jedrzejewski, W., & Zybura, H. (2010). Fluctuating ungulate density shapes tree recruitment in natural stands of the

Białowieza Primeval Forest, Poland. *Journal of Vegetation Science*, *21*(6), 1082–1098. https://doi.org/10.1111/j.1654-1103.2010.01217.x

Kuksa, V. I. [Кукса, В. И.]. (1994). Южные моря (Аральское, Каспийское, Азовское и Черное) в условиях антропогенного стресса *[Southern Seas (*Aral*, *Caspian*, *Azov* and *Black*) in anthropogenic stress conditions]*. St. Peterburg, Russian Federation: Hydrometeoizdat.

Kulagin, V. M., Markov, P. A., & Tishkov, A. A. [Кулагин, В. М., Марков, П. А., & Тишков, А. А.]. (1990). Иссык-Кульский заповедник [Issyk-Kul strict natural reserve]. In Заповедники Средней Азии и Казахстана. Заповедники СССР *[Strict natural reserves of Central Asia and Kazakhstan. Strict natural reserves of the USSR]* (p. 399). Moscow, USSR: Мысль [Mysl].

Kull, K. & Zobel, M. (1991). High species richness in an Estonian wooded meadow. *Journal of Vegetation Science, 2,* 711-714. https://doi.org/10.2307/3236182

Kumpula, T., Forbes, B. C., Stammler, F., & Meschtyb, N. (2012). Dynamics of a coupled system: Multi-resolution remote sensing in assessing social-ecological responses during 25 years of gas field development in Arctic Russia. *Remote Sensing*, *4*(4), 1046–1068. https://doi.org/10.3390/rs40401046

Kumpula, T., Pajunen, A., Kaarlejärvi, E., Forbes, B. C., & Stammler, F. (2011). Land use and land cover change in Arctic Russia: Ecological and social implications of industrial development. *Global Environmental Change*, *21*(2), 550–562. https://doi.org/10.1016/j.gloenvcha.2010.12.010

Künzi, Y., Prati, D., Fischer, M., & Boch, S. (2015). Reduction of native diversity by invasive plants depends on habitat conditions. *American Journal of Plant Sciences*, *6,* 2718–2733. https://doi.org/10.4236/ajps.2015.617273

Kupianskaya, A. N., Proshchalykin, M. Y., & Lelej, A. S. (2014). Contribution to the fauna of bumble bees (Hymenoptera, Apidae: *Bombus* Latreille, 1802) of the Republic of Tyva, Eastern Siberia. *Euroasian Entomological Journal*, *13*(3), 290–294.

Kurtov, A. A. [Куртов, А. А.]. (2013). Центральная Азия: водные артерии как новые узлы противоречий [Central Asia: Waterways as New Contradictions]. In K. A. Kokarev, D. A. Aleksandrov, & I. Y. Frolova. [К.А. Кокарев, Д.А. Александров, & И.Ю. Фролова] (Eds.), Центральная Азия: проблемы и перспективы (взгляд из России иКитая) *[Central Asia: Problems and Perspectives (*A View from Russia and China*)]* (pp. 155–230). Moscow, Russian Federation: RISI. Retrieved from http://www.studfiles.ru/preview/5771470/

Kuz'mina, Z. V., & Treshkin, S. E. (1997). Soil salinization and dynamics of tugai vegetation in the southeastern Caspian Sea region and in the Aral Sea coastal region. *Eurasian Soil Science*, *30*(6), 642–649.

Kvile, K. O., Taranto, G. H., Pitcher, T. J., & Morato, T. (2014). A global assessment of seamount ecosystems knowledge using an ecosystem evaluation framework. *Biological Conservation*, *173*, 108–120. http://doi.org/10.1016/j.biocon.2013.10.002

Laidre, K. L., Stern, H., Kovacs, K. M., Lowry, L., Moore, S. E., Regehr, E. V., Ferguson, S. H., Wiig, Ø., Boveng, P., Angliss, R. P., Born, E. W., Litovka, D., Quakenbush, L., Lydersen, C., Vongraven, D., & Ugarte, F. (2015). Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. *Conservation Biology*, *29*(3), 724–737. https://doi.org/10.1111/cobi.12474

Lambin, E. F., & Meyfroidt, P. (2011). Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Scienes of the United States of America*, *108*(9), 3465–3472. https://doi.org/10.1073/pnas.1100480108

Landolt, J., Gross, A., Holdenrieder, O., & Pautasso, M. (2016). Ash dieback due to *Hymenoscyphus fraxineus*: what can be learnt from evolutionary ecology? *Plant Pathology*, *65*(7), 1056–1070. https://doi.org/10.1111/ppa.12539

Landsat-8. (2016). *Lake Chany*. Retrieved from https://earthexplorer.usgs.gov/

Lang, S. I., Cornelissen, J. H. C., Shaver, G. R., Ahrens, M., Callaghan, T. V.,

Molau, U., Ter Braak, T. J. F., Holzer, A., & Aerts, R. (2012). Arctic warming on two continents has consistent negative effects on lichen diversity and mixed effects on bryophyte diversity. *Global Change Biology*, *18*(3), 1096–1107. https://doi.org/10.1111/j.1365-2486.2011.02570.x

Larsen, J. N., Anisimov, O. A., Constable, A., Hollowed, A. B., Maynard, N. Prestrud, P. Prowse, T. D. & Stone, J. M. R. (2014). Polar regions. In V. R. Barros, C. B. Field, D. J. Dokken, M. D. Mastrandrea, K. J. Mach, T. E. Billir, M. Chatterjee, M., K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea & L. L. White (Eds.), *Climate change 2014: Impacts, adaptation, and vulnerability: Working group II contribution to the fifth assessment report of the Intergovernmental Panel on Climate Change* (pp. 1567–1612). Cambridge, United Kingdom: Cambridge University Press. Retrieved from http://www.citeulike.org/group/15400/article/13497155

Latvian Academy of Science. (1997). *Red Data Book of Latvia - Vascular Plants and Mosses*.

Lavergne, S., Thuiller, W., Molina, J., & Debussche, M. (2005). Environmental and human factors influencing rare plant local occurrence, extinction and persistence: A 115-year study in the Mediterranean region. *Journal of Biogeography*, *32*(5), 799–811. http://doi.org/10.1111/j.1365-2699.2005.01207.x

Lavergne, S., Mouquet, N., Ronce, O., & Thuiller, W. (2010). Biodiversity and climate change: Integrating evolutionary and ecological responses of species and communities. *Annual Review of Ecology, Evolution and Systematics, 41,* 321–350. http://doi.org/10.1146/annurev-ecolsys-102209-144628

Lavrinenko, O. V., & Lavrinenko, I. A., [Лавриненко О.В., & Лавриненко, И. А. (Eds.). (2006). Красная книга Ненецкого автономного округа *[Red Data Book of the Nenets Autonomus Okrug]*. Narian-Mar, Russian Federation: Ненецкий информационно-аналитический центр [Nenets Informational-Analitic Centre].

Le Roux, X, Barbault, R., Baudry, J., Burel, F., Doussan, I., Garnier, E., Herzog, F., Lavorel, S., Lifran, R.,

Roger-Esrade, L., Sarthou, J. P., & Trommetter, M. (2008). *Agriculture and biodiversity: Benefiting from synergies.*

Le Roux, X., Barbault, R. Baudry, J. Burel, F. Doussan, I. Garnier, E. Herzog, F. Lavorel, S. Lifran, R. Roger- Estrade, J. Sarthou, J. & Trommetter, M. (2009). *Agriculture and biodiversity: benefiting from synergies. Multidisciplinary Scientific Assessment, Synthesis Report, INRA.*

Le Viol, I., Jiguet, F., Brotons, L., Herrando, S., Lindström, A., Pearce-Higgins, J. W., Reif, J., Van Turnhout, C., & Devictor, V. (2012). More and more generalists: two decades of changes in the European avifauna. *Biology Letters*, *8*(5), 780–2. https://doi.org/10.1098/rsbl.2012.0496

Leeuwen, C. V., Emeljanenko, T., & Popova, L. (1994). *Nomads in Central Asia: animal husbandry and culture in transition (19th-20th century)*. Amsterdam, The Netherlands: Koninklijk Instituut voor de Tropen (KIT) (Royal Tropical Institute, RTI). Retrieved from https://www.cabdirect.org/cabdirect/abstract/19951800186

Lefcheck, J. S., Byrnes, J. E. K., Isbell, F., Gamfeldt, L., Griffin, J. N., Eisenhauer, N., Hensel, M. J. S., Hector, A., Cardinale, B. J., & Duffy, J. E. (2015). Biodiversity enhances ecosystem multifunctionality across trophic levels and habitats. *Nature Communications, 6,* 6936. https://doi.org/10.1038/ncomms7936

Lefèvre, F., & Fady, B. (2016). Introduction to Mediterranean forest systems: Mediterranean basin. In T. D. Paine & F. Lieitier (Eds.), *Insects and diseases of Mediterranean forest systems* (pp. 7–28). Switzerland: Springer.

Lehsten, V., Sykes, M. T., Scott, A. V., Tzanopoulos, J., Kallimanis, A., Mazaris, A., Verburg, P. H., Schulp, C. J. E., Potts, S. G., & Vogiatzakis, I. (2015). Disentangling the effects of land-use change, climate and CO_2 on projected future European habitat types. *Global Ecology and Biogeography*, *24*(6), 653-663. https://doi.org/10.1111/geb.12291

Lelej, A. S., & Storozhenko, S. Y. (2010). Insect taxonomic diversity in the Russian Far East. *Entomological Review*, *90*(2), 372–386. https://doi.org/10.1134/S001387381003005X

Lenoir, J., Gégout, J. C., Marquet, P. A., de Ruffray, P., & Brisse, H. (2008). A significant upward shift in plant species optimum elevation during the 20th century. *Science*, *320*(5884), 1768–1771. https://doi.org/10.1126/science.1156831

Leppäkoski, E., Gollasch, S., Gruszka, P., Ojaveer, H., Olenin, S., & Panov, V. (2002). The Baltic sea of invaders. *Canadian Journal of Fisheries and Aquatic Sciences*, *59*(7), 1175–1188. https://doi.org/10.1139/f02-089

Leppik, E., Jüriado, I., Suija, A., & Liira, J. (2013). The conservation of ground layer lichen communities in alvar grasslands and the relevance of substitution habitats. *Biodiversity and Conservation*, *22*(3), 591–614. https://doi.org/10.1007/s10531-012-0430-z

Lester, S.E., Halpern, B.S., Grorud-Colvert, K., Lubchenco, J. Ruttenberg, B.I., Gaines, S.D., Airamé, S., & Warner, R. R. (2009). Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series*, *348*, 33–46. https://doi.org/10.3354/meps08029

Levers, C., Müller, D., Erb, K., Haberl, H., Jepsen, M. R., Metzger, M. J., Meyfroidt, P., Plieninger, T., Plutzar, C., Stürck, J., Verburg, P. H., Verkerk, P. J., & Kuemmerle, T. (2015). Archetypical patterns and trajectories of land systems in Europe. *Regional Environmental Change, 18*(3), 715-732. https://doi.org/10.1007/s10113-015-0907-x

Levin, L. A., Mengerink, K., Gjerde, K. M., Rowden, A. A., Van Dover, C. L., Clark, M. R., Ramirez-Llodra, E., Currie, B., Smith, C. R., Sato, K. N., Gallo, N., Sweetman, A K., Iily, H., Armstrong, C. W., & Brider, J. (2016). Defining "serious harm" to the marine environment in the context of deep-seabed mining. *Marine Policy*, *74*(October), 245–259. https://doi.org/10.1016/j.marpol.2016.09.032

Lewandowska, A. M., Biermann, A., Borer, E. T., Cebrián-Piqueras, M. A., Declerck, S. A., De Meester, L., Van Donk, E., Gamfeldt, L., Gruner, D. S., Hagenah, N., Harpole, W. S., Kirkman, K. P., Klausmeier, C. A., Kleyer, M., Knops, J. M., Lemmens, P., Lind, E. M., Litchman, E., Mantilla-Contreras, J., Martens, K., Meier, S., Minden, V., Moore, J. L., Venterink, H. O., Seabloom, E. W., Sommer, U., Striebel, M., Trenkamp, A., Trinogga, J., Urabe, J., Vyverman, W., Van de Waal, D. B., Widdicombe, C. E., & Hillebrand, H.** (2016). The influence of balanced and imbalanced resource supply on biodiversity-functioning relationship across ecosystems. *Philosophical Transactions of the Royal Society B: Biological Sciences, 371,* 1694. https://doi.org/10.1098/rstb.2015.0283

Lilleleht, V. (1998). *Eesti Puname Raama. Ohustatud seened, taimed ja loomad [Red data book of Estonia. Threatened fungi, plants and animals]*. Tartu, Estonia: Eesti Teaduste Akadeemia, Looduskaitse Komisjon.

Lindgaard, A., & Henriksen, S. (Eds.). (2011). *Norsk rødliste for naturtyper [Norwegian Red List for Ecosystems and Habitat Types]*. Trondheim, Norway: Artsdatabanken.

Lindner, M. M., Maroschek, S., Netherer, A., Kremer, A., Barbati, J., Garcia-Gonzalo, R., Seidl, R., Delzon, S., Corona, P., Kolstrom, M., Lexer, M. J., & Marchetti, M. (2010). Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. *Forest Ecology and Management*, *259*(4), 698–709. https://doi.org/10.1016/j.foreco.2009.09.023

Lindsey, R. (2016). *Shrinking Aral Sea*. Retrieved from http://earthobservatory.nasa.gov/Features/WorldOfChange/aral_sea.php

Liška, J., Zdeněk, P., & Slavíková, Š. (2012). Lichen flora of the Czech Republic. *Preslia*, *84*(3), 851–862. Retrieved from http://www.scopus.com/inward/record.url?eid=2-s2.0-84865453402&partnerID=tZOtx3y1

Lisowska, M. (2011). Lichen recolonisation in an urban-industrial area of southern Poland as a result of air quality improvement. *Environmental Monitoring and Assessment*, *179*(1–4), 177–190. https://doi.org/10.1007/s10661-010-1727-6

Liu, Y., Webber, S., Bowgen, K., Schmaltz, L., Bradley, K., Halvarsson,

P., Abdelgadir, M., & Griesser, M. (2013). Environmental factors influence both abundance and genetic diversity in a widespread bird species. *Ecology and Evolution*, *3*(14), 4683–4695. https://doi. org/10.1002/ece3.856

Living Black Sea. (2016). *Evolution of the Black Sea ecosystem*. Retrieved from http:// blacksea-education.ru/e2-1.shtml

Lõhmus, A., Nellis, R., Pullerits, M., & Leivits, M. (2016). The potential for long-term sustainability in seminatural forestry: A broad perspective based on woodpecker populations. *Environmental Management*, *57*(3), 558–571. https://doi.org/10.1007/ s00267-015-0638-2

Lõhmus, A., & Runnel, K. (2014). Ash dieback can rapidly eradicate isolated epiphyte populations in production forests: A case study. *Biological Conservation*, *169*, 185–188. https://doi.org/10.1016/j. biocon.2013.11.031

Lomský, B., Šrámek, V., & Novotný, R. (2012). Changes in the air pollution load in the Jizera Mts.: Effects on the health status and mineral nutrition of the young Norway spruce stands. *European Journal of Forest Research*, *131*(3), 757–771. https://doi. org/10.1007/s10342-011-0549-6

Loreau, M. (2000) Biodiversity and ecosystem functioning: recent theoretical advances. *Oikos, 91,* 3–17. https://doi. org/10.1034/j.1600-0706.2000.910101.x

Loreau, M. (2008). Biodiversity and ecosystem functioning: The mystery of the deep sea. *Current Biology, 18,* R126–R128. https://doi.org/10.1016/j. cub.2007.11.060

Loreau, M. (2010). Linking biodiversity and ecosystems: towards a unifying ecological theory. *Philosophical Transactions of the Royal Society B: Biological Sciences, 365,* 49–60. https://doi.org/10.1098/ rstb.2009.0155

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J. P., Hector, A., Hooper, D. U., Huston, M. A., Raffaelli, D., Schmid, B., Tilman, D., & Wardle, D. A. (2001). Biodiversity and ecosystem functioning: Current knowledge and future challenges. *Science, 294*(5543),

804-808. https://doi.org/10.1126/ science.1064088

Lozano, F. D. (Ed.). (2000). *Lista roja de Flora vascular Española [Red list of Spanish vascular flora]*.

LPI. (2016). *Living Planet Index. 2016.* Retrieved from http://www. livingplanetindex.org/projects?main_page_ project=LivingPlanetReport&home_flag=1

Ludwig, G., & Schnittler, M. (1996). *Rote Liste gefährdeter Pflanzen Deutschlands [Red List of endangered plants in Germany]*.

Lukić-Bilela, L., Ozimec, R., Miculinić, K., & Basara, D. (2013). A comprehensive valorisation of Megara cave with a view to preservation and protection. *Natura Montenegrina*, *12*(3), 1–17.

Lukoyanov, V. A. [В.А. Лукоянов]. (2013). Доклад о состоянии природопользования и обохране окружающей среды Краснодарскогокрая в 2012 году [Report about the state of natural resources and environmental protection of the Krasnodar region in 2012]. Krasnodar: Administration and Ministry of Natural Resources of Krasnodar Region. Retrieved from http://www. mprkk.ru/media/main/attachment/attach/ df4e573261d9d541c06e3036616b6f5c.pdf

Luryeva, I. I. [Лурыева, И.И.]. (2014). Экологические аспекты разработки газовых месторождений [Ecological aspects of the development of gas fields]. Проблемы Освоения Пустынь [Problems of Desert Development], 3–4, 85–87.

Lynam, C. P., & Rossberg, A. G. (2017). *New univariate characterization of fish community size structure improves precision beyond the large fish indicator*. Retrieved from http://arxiv.org/abs/1707.06569

MacDonald, D., Crabtree, J. R., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Lazpita, J. G., & Gibon, A. (2000). Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. *Journal of Environmental Management*, *59*(1), 47–69. https://doi.org/10.1006/ jema.1999.0335

Mace, G. M., Gittleman, J. L. & Purvis, A. (2003). Preserving the tree of life. *Science,*

300(5626), 1707–1709. https://doi. org/10.1126/science.1085510

Mace, G. M., Norris, K. & Fitter, A. H. (2012). Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology and Evolution, 27,* 19–26. https:// doi.org/10.1016/j.tree.2011.08.006

Madre, F., Vergnes, A., Machon, N., & Clergeau, P. (2013). A comparison of 3 types of green roof as habitats for arthropods. *Ecological Engineering, 57,* 109-117. https://doi.org/10.1016/j. ecoleng.2013.04.029

Maes, J., Paracchini, M. L., & Zulian, G. (2011). *A European assessment of the provision of ecosystem services: Towards an atlas of ecosystem services*. Luxembourg: Publications Office of the European Union. https://doi. org/10.2788/63557

Maes, J., Paracchini, M. L., Zulian, G., Dunbar, M. B., & Alkemade, R. (2012). Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. *Biological Conservation, 155,* 1–12. https:// doi.org/10.1016/j.biocon.2012.06.016

Maestre, F. T., Eldridge, D. J., Soliveres, S., Kéfi, S., Delgado-Baquerizo, M., Bowker, M. A., García-Palacios, P., Gaitán, J., Gallardo, A., Lázaro, R., & Berdugo, M. (2016). Structure and functioning of dryland ecosystems in a changing world. *Annual Review of Ecology, Evolution, and Systematics, 47,* 215-237. https://doi.org/10.1146/annurev-ecolsys-121415-032311

Maestre, F. T., Quero, J. L., Gotelli, N. J., Escudero, A., Ochoa, V., Delgado-Baquerizo, M., García-Gómez, M., Bowker, M. A., Soliveres, S., Escolar, C., García-Palacios, P., Berdugo, M., Valencia, E., Gozalo, B., Gallardo, A., Aguilera, L., Arredondo, T., Blones, J., Boeken, B., Bran, D., Conceição, A. A., Cabrera, O., Chaieb, M., Derak, M., Eldridge, D. J., Espinosa, C. I., Florentino, A., Gaitán, J., Gabriel Gatica, M., Ghiloufi, W., Gómez-González, S., Gutiérrez, J. R., Hernández, R. M., Huang, X., Huber-Sannwald, E., Jankju, M., Miriti, M., Monerris, J., Mau, R. L., Morici, E., Naseri, K., Ospina, A., Polo, V., Prina, A., Pucheta, E., Ramírez-

Collantes, D. A., Romão, R., Tighe, M., Torres-Díaz, C., Val, J., Veiga, J. P., Wang, D., & Zaady, E. (2012). Plant species richness and ecosystem multifunctionality in global drylands. *Science, 335*(6065), 214-218. https://doi.org/10.1126/science.1215442

Magurran, A. E., Dornelas, M., Moyes, F., Gotelli, N. J., & McGill, B. (2015). Rapid biotic homogenization of marine fish assemblages. *Nature Communications, 6,* 8405. https://doi.org/10.1038/ncomms9405

Maharramova, E. H., Safarov, H. M., Kozlowski, G., Borsch, T., & Muller, L. A. (2015). Analysis of nuclear microsatellites reveals limited differentiation between colchic and hyrcanian populations of the wind-pollinated relict tree *Zelkova carpinifolia* (Ulmaceae). *American Journal of Botany, 102*(1), 119–128. http://doi.org/10.3732/ajb.1400370

Mair, L., Harrison, P. J., Räty, M., Bärring, L., Strandberg, G., & Snäll, T. (2017). Forest management could counteract distribution retractions forced by climate change: *Ecological Applications, 27*(5), 1485–1497. https://doi.org/10.1002/eap.1541

Mäkeläinen, S., De Knegt, H. J., Ovaskainen, O., & Hanski, I. K. (2016). Home-range use patterns and movements of the Siberian flying squirrel in urban forests: Effects of habitat composition and connectivity. *Movement Ecology, 4*(5), UNSP 13. https://doi.org/10.1186/s40462-016-0071-z

Makoedov, A. N., & Kozhemiako, O. H., [Макоедов, А. Н., & Кожемяко, О. Н.]. (2007). Основы рыбохозяйственной политики России *[The principles of fishery policy in Russian Federation]*. Moscow, Russia Federation: National Fish Resources.

Malaj, E., von der Ohe, P. C., Grote, M., Kühne, R., Mondy, C. P., Usseglio-Polatera, P., Brack, W., & Schäfer, R. B. (2014). Organic chemicals jeopardize the health of freshwater ecosystems on the continental scale. *Proceedings of the National Academy of Sciences of the United States of America, 111*(26), 9549–9554. https://doi.org/10.1073/pnas.1321082111

Malanson, G. P., Rose, J. P., Schroeder, P. J., & Fagre, D. B. (2011). Contexts for change in alpine tundra. *Physical Geography, 32*(2), 97–113. http://doi.org/10.2747/0272-3646.32.2.97

Máliš, F., Kopecký, M., Petřík, P., Vladovič, J., Merganič, J., & Vida, T. (2016). Life stage, not climate change, explains observed tree range shifts. *Global Change Biology, 22*(5), 1904–1914. https://doi.org/10.1111/gcb.13210

Malyshev, L., & Nimis, P. L. (1997). Climatic dependence of the ecotone between alpine and forest orobiomes in southern Siberia. *Flora, 192*, 109–120.

Mamaev, V. (2002). The Caspian Sea - enclosed and with many endemic species. In *Europe's biodiversity - biogeographical regions and seas. Seas around Europe. EEA Report No 1/2002*.

Manca, M., & DeMott, W. R. (2009). Response of the invertebrate predator *Bythotrephes* to a climate-linked increase in the duration of a refuge from fish predation. *Limnology and Oceanography, 54*(6), 2506–2512. https://doi.org/10.4319/lo.2009.54.6_part_2.2506

Mannerla, M., Andersson, M., Birzaks, J., Debowski, P., Degerman, E., Huhmarniemi, A., Häggström, H., Ikonen, E., Jokikokko, E., Jutila, E., Kesler, M., Kesminas, V., Kontautas, A., Pedersen, S, Persson, S., Romakkaniemi, A., Saura, A., Shibaev, S., Titov, S., Tuus, H., Tylik, K., & Yrjänä, T. (2011). Salmon and sea trout populations and rivers in the Baltic Sea. Helsinki, Finland: HELCOM.

Mansuroglu, S., Ortacesme, V., & Karaguzel, O. (2006). Biotope mapping in an urban environment and its implications for urban management in Turkey. *Journal of Environmental Management, 81*(3), 175-187. https://doi.org/10.1016/j.jenvman.2005.10.008

Manu, M., Szekely, L., Oromulu, L. V., Bărbuceanu, D., Honciuc, V., Maican, S., Fiera, C., Purice, D., & Ion, M. (2015). Bucharest. In J. Kelcey (Ed.), *Vertebrates and invertebrates of European cities: Selected non-avian fauna* (pp. 257–322). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6

Marhold, K. & Hindák, F. (1999). *Zoznam nizsích a vyssích rastlín Slovenska [Check List of Non-vascular and Vascular Plants of Slovakia]*. Bratislava, Slovakia: Veda.

Marian, S. (1903). *Insectele în limba: credințele, si obiceiurile Românilor [Insects in language: the beliefs and the customs of the Romanians]*.

Markensten, H., Moore, K., & Persson, I. (2010). Simulated lake phytoplankton composition shifts toward cyanobacteria dominance in a future warmer climate. *Ecological Applications, 20*(3), 752-767. https://doi.org/10.1890/08-2109.1

Markovic, D., Carrizo, S., Freyhof, J., Cid, N., Lengyel, S., Scholz, M., Kasperdius, H., & Darwall, W. (2014). Europe's freshwater biodiversity under climate change: Distribution shifts and conservation needs. *Diversity and Distributions, 20*(9), 1097-1107. https://doi.org/10.1111/ddi.12232

Marmor, L., Randlane, T., Jüriado, I., & Saag, A. (2017). Host tree preferences of red-listed epiphytic lichens in Estonia. *Baltic Forestry, 23*(2), 364–373.

Martin-Mehers, G. (2016). *Western gray whale advisory panel: Stories of influence*. IUCN, WWF, IFAW.

Martin, A. (2009). The Loch Ness monster and La Palma giant lizard *Gallotia auaritae*: are they really extant? *Oryx, 43*, 17-17. https://doi.org/10.1017/S0030605308431071

Masterman, E. W. G. (1921). Crocodiles in Palestine. *Palestine Exploration Fund: Quarterly Statement, 1920*, 19–21.

Mateo Miras, J. A., & Martínez-Solano, I. (2009). *Gallotia auaritae*. The IUCN Red List of Threatened Species. https://doi.org/10.2305/IUCN.UK.2009.RLTS.T61501A12492629.en

Mathar, W., Kleinebecker, T., & Hölzel, N. (2015). Environmental variation as a key process of co-existence in flood-meadows. *Journal of Vegetation Science, 26*(3), 480–491. https://doi.org/10.1111/jvs.12254

Mathar, W. P., Kämpf, I., Kleinebecker, T., Kuzmin, I., Tolstikov, A., Tupitsin, S., & Hölzel, N. (2015). Floristic diversity of meadow steppes in the western Siberian Plain: effects of abiotic site conditions, management and landscape structure. *Biodiversity and Conservation, 25*(12), 2361-2379. https://doi.org/10.1007/s10531-015-1023-4

Matveeva, N. V. (2015). *Plants and fungi of the polar deserts in the Northern hemisphere.* Saint-Petercburg, Russian Federation: Marafon Publlishing.

Mazzoleni, S., di Pasquale, G., Mulligan, M., di Martino, P., & Rego, F. (2004). *Recent dynamics of Mediterranean vegetation and landscape*. Chichester, UK: John Wiley & Sons, Ltd.

McCarthy, T., Mallon, D., Jackson, R., Zahler, P., & McCarthy, K. (2017). *Panthera uncia.* The IUCN Red List of Threatened Species. http://dx.doi.org/10.2305/IUCN.UK.2017-2.RLTS.T22732A50664030.en

Mccauley, B. D. J., Woods, P., Sullivan, B., Bergman, B., Jablonicky, C., Roan, A., Hirshfield, M., Boerder, K., & Worm, B. (2016). Ending hide and seek at sea. *Science*, *351*(6278), 1148–1150. https://doi.org/10.1126/science.aad5686

McCauley, D. J., Pinsky, M. L., Palumbi, S. R., Estes, J. A., Joyce, F. H., & Warner, R. R. (2015). Marine defaunation: Animal loss in the global ocean. *Science*, *347*(6219), 247–254. https://doi.org/10.1126/science.1255641

McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological Conservation*, *127*(3), 247–260. https://doi.org/10.1016/j.biocon.2005.09.005

McNeill, D. C. (2010). *Translocation of a population of great crested newts* (Triturus cristatus)*: a Scottish case study* (Doctoral dissertation). Retrieved from http://theses.gla.ac.uk/2184/

McQuatters-Gollop, A., Raitsos, D. E., Edwards, M., & Attrill, M. J. (2007). Spatial patterns of diatom and dinoflagellate seasonal cycles in the NE Atlantic Ocean. *Marine Ecology Progress Series*, *339*, 301–306. https://doi.org/10.3354/meps339301

McRae, L., Deinet, S., Gill, M. & Collen, B. (2012). *The Arctic species trend index: Tracking trends in Arctic marine populations. CAFF Assessment Series No. 7.* Akureyri, Iceland: CAFF International Secretariat. Retrieved from https://oaarchive.arctic-council.org/bitstream/handle/11374/219/ASTI_Tracking_Trends_Arctic_Marine_Populations_April_2012.pdf?sequence=1

MEA. (2005). *Ecosystems and human well-being: Synthesis*. Washington, DC, USA: Island Press.

Médail, F., & Diadema, K. (2009). Glacial refugia influence plant diversity patterns in the Mediterranean Basin. *Journal of Biogeography*, *36*(7), 1333–1345. http://doi.org/10.1111/j.1365-2699.2008.02051.x

MedPAN & RAC/SPA. (2016). *The 2016 status of marine protected areas in the Mediterranean. Main findings*. Retrieved from http://www.oceanactionhub.org/2016-status-marine-protected-areas-mediterranean-main-findings

Meerhoff, M., Teixeira-de Mello, F., Kruk, C., Alonso, C., González-Bergonzoni, I., Pacheco, J. P., Lacerot, G., Arim, M., Beklioglu, M., Brucet, S., Goyenola, G., Iglesias, C., Mazzeo, N., Kosten, S., & Jeppesen, E. (2012). Environmental warming in shallow lakes. A review of potential changes in community structure as evidenced from space-for-time substitution approaches. *Advances in Ecological Research, 46,* 259-349. https://doi.org/10.1016/B978-0-12-396992-7.00004-6

Meiri, S., Bauer, A., Allen, A., Castro-Herrera, F., Chirio, L., Colli, G., Das, I., Doan, T., Glaw, F., Grismer, L., Hoogmoed, M., Kraus, F., LeBreton, M., Meirte, D., Nagy, Z., Nogueira, C., Oliver, P., Pincheira-Donoso, D., Shea, G., Sindaco, R., Tallowin, O., Torres, S., Trape, J., Uetz, P., Wagner, P., Wang, Y. Z., Ziegler, T., & Roll, U. (2018). Extinct, obscure or imaginary: the lizard species with the smallest ranges. *Diversity & Distributions*, *in press.*

Meliadou, A., & Troumbis, A. Y. (1997). Aspects of heterogeneity in the distribution of diversity of the European herpetofauna. *Acta Oecologica, 18*(4), 393–412. https://doi.org/10.1016/S1146-609X(97)80031-8

Meltofte, H., Barry, T., Berteaux, D., Bültmann, H., Christiansen, J. S., Cook, J. A., Dahlberg, A., Daniëls, F, J. A., Ehrich, D.,Fjeldså, J., Friðriksson, F., Ganter, B., Gaston, A. J., Gillespie, L. J., Grenoble, L., Hoberg, E. P., Hodkinson, I. D., Huntington, H. P., Ims, R. A., Josefson, A. B., Kutz, S. J., Kuzmin, S. L., Laidre, K. L., Lassuy, D. R., Lewis, P. N., Lovejoy, C., Michel, C., Mokievsky, V., Mustonen, T., Payer, D. C.,Poulin, M., Reid, D. G., Reist, J. D., Tessler, D. F., Wrona, F. J. (2013). *Arctic Biodiversity Assessment. Synthesis*. Akureyri, Iceland: Conservation of Arctic Flora and Fauna (CAFF).

Mengerink, K. J., Dover, C. L. Van, Ardron, J., Baker, M., Escobar-briones, E., Gjerde, K., Koslow, J. A., Ramirez-Llodra, E., Lara-Lopez, A., Squires, D., Sutton, T., Sweetman, A. K., & Levin, L. A. (2014). A call for deep-ocean stewardship. *Science*, *344*(6185), 696-698. https://doi.org/10.1126/science.1251458

Merunková, K., Preislerová, Z., & Chytrý, M. (2014). Environmental drivers of species composition and richness in dry grasslands of northern and central Bohemia, Czech Republic. *Tuexenia*, *34*(1), 447–466. https://doi.org/10.14471/2014.34.017

Messager, M. L., Lehner, B., Grill, G., Nedeva, I., & Schmitt, O. (2016). Estimating the volume and age of water stored in global lakes using a geo-statistical approach. *Nature Communications*, *7*, 13603. https://doi.org/10.1038/ncomms13603

Meyer, S., Wesche, K., Krause, B., & Leuschner, C. (2013). Dramatic losses of specialist arable plants in central Germany since the 1950s/60s - a cross-regional analysis. *Diversity and Distributions*, *19*(9), 1175–1187. https://doi.org/10.1111/ddi.12102

Micheli, F., Halpern, B. S., Walbridge, S., Ciriaco, S., Ferretti, F., Fraschetti, S., Lewison, R., Nykjaer, L., & Rosenberg, A. A. (2013). Cumulative Human Impacts on Mediterranean and Black Sea Marine Ecosystems: Assessing Current Pressures and Opportunities. *Plos ONE*, *8*(12), e79889. https://doi.org/10.1371/journal.pone.0079889

Micklin, P. (2007). The Aral Sea disaster. *Annual Review of Earth and Planetary Sciences*, *35*(1), 47–72. https://doi.org/10.1146/annurev.earth.35.031306.140120

Mieszkowska, N., Kendall, M. A., Hawkins, S. J., Leaper, R., & Williamson, P. (2006). Changes in the range of some common rocky shore species in Britania response to climate change. *Hydrobiologia*, *555*, 241. https://doi.org/10.1007/s10750-005-1120-6

Mieszkowska, N., Sugden, H., Firth, L. B., & Hawkins. (2014). The role of sustained observations in tracking impacts of environmental change on marine biodiversity and ecosystems. *Philosophical Transactions of the Royal Society A: Mathematical, Physical, and Engineering Sciences*, *372*(2025), 20130339. https://doi.org/10.1098/rsta.2013.0339

Milkov, F. N. & Gvozdetskiy, N. A. [Мильков, Ф. Н., & Гвоздецкий, Н. А.]. (1969). Физическая география СССР. Общий обзор. Европейская часть СССР. Кавказ [The physical geography of the USSR. Overview. The European part of the USSR. The Caucasus]. Moscow, Russian Federation: Mysl.

Milkov, F. N. [Мильков, Ф. Н.]. (1977). Природные зоны СССР [Natural zones of the USSR]. Moscow, Russian Federation: Мысль [Mysl].

Millaku, F., Rexhepi, F., Krasniqi, E., Pajazitaj, Q., Mala, X., & Berisha, N. (2013). *Libri i Kuq i Florës Vaskulare të Republikës së Kosovës [The red book of vascular flora of the Republic of Kosovo]*. Retrieved from http://www.ammk-rks.net/repository/docs/Libri_i_Kuq_i_Flores_vaskulare_Shqip_(permbledhje).pdf

Millon, A., Petty, S. J., Little, B., Gimenez, O., Cornulier, T., & Lambin, X. (2014). Dampening prey cycle overrides the impact of climate change on predator population dynamics: A long-term demographic study on tawny owls. *Global Change Biology, 20*(6), 1770–81. https://doi.org/10.1111/gcb.12546

Milner-Gulland, E. J., Kreuzberg-Mukhina, E., Grebot, B., Ling, S., Bykova, E., Abdusalamov, I., Bekenov, A., Gärdenfors, U., Hilton-Taylor, C.,

Salnikov, V., & Stogova, L. (2006). Application of IUCN red listing criteria at the regional and national levels: a case study from Central Asia. *Biodiversity and Conservation, 15*(6), 1873-1886. https://doi.org/10.1007/s10531-005-4304-5

Milner-Gulland, E. J., & Singh, N. J. (2016). Two decades of saiga antelope research. In J. Bro-Jørgensen, & D. P. Mallon, *Antelope Conservation* (pp. 297–314). Chichester, UK: John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118409572.ch15

Minayeva, T. Y., Bragg, O. M., & Sirin, A. A. (2017a). Towards ecosystem-based restoration of peatland biodiversity. *Mires and Peat*, *19,* UNSP 01. https://doi.org/10.19189/MaP.2013.OMB.150

Minayeva, T., & Sirin, A. (2005). Use and conservation of mires in Russia. In G. M. Steiner (Ed.), *Mires – from Siberia to Tierra del Fuego* (pp. 275–292). Linz, Austria: Biologiezentrum Oberösterreichisches Landesmuseum.

Minayeva, T., & Sirin, A. (2009). Wetlands – Threatened Arctic ecosystems: Vulnerability to climate change and adaptation options. In *Climate change and Arctic sustainable development* (pp. 76–83). Paris, France: UNESCO.

Minayeva, T., & Sirin, A. (2010). Arctic peatlands. In *Arctic biodiversity trends 2010 – Selected indicators of change* (pp. 71–74). Akureyri, Iceland: CAFF International Secretariat.

Minayeva, T. Y., & Sirin, A. A. (2012). Peatland biodiversity and climate change. *Biology Bulletin Reviews*, *2*(2), 164–175. https://doi.org/10.1134/S207908641202003X

Minayeva, T., Sirin, A., Kershaw, P., & Bragg, O. (2017b). Arctic peatlands. In *The wetland book II: Distribution, description and conservation* (pp. 1–15). Dordrecht, The Netherlands: Springer. https://doi.org/10.1007/978-94-007-6173-5_109-2

Minayeva, T., Sirin, A., & Bragg, O. (2009). *A quick scan of peatlands in Central and Eastern Europe*. Wageningen, The Netherlands: Wetlands International.

Minayeva, T., Sirin, A., & Stracher, G. (2013). The peat fires of Russia. In G. Stracher (Ed.), *Coal and peat fires: A global perspective* (pp. 376–394). Amsterdam, The Netherlands: Elsevier B.V.

Mirzoyan, Z. A., Volovik, S. P., & Martyniuk, M. L. [Мирзоян, З. А., Воловик, С. П., & Мартынюк, М. Л.]. (2002). Развитие популяции Beroe ovata в Азово-Черноморском бассейне [Development of *Beroe ovata* populations in the Azov-Black Sea basin]. In Основные проблемы рыбного хозяйства и охраны рыбохозяйственных водоемов Азово-Черноморского басейна (2000-2001 гг.) *[Main problems of fisheries and protection of fishery water bodies of the Azov-Black Sea basin]* (pp. 180–192). Rostov-on-Don, Russian Federation: AzNIIRKh.

Mitrofanov, I. V., & Mamilov, N. S. (2015). Fish diversity and fisheries in the Caspian Sea and Aral–Syr Darya basin in the Republic of Kazakhstan at the beginning of the 21st Century. *Aquatic Ecosystem Health & Management*, *18*(2), 160–170. https://doi.org/10.1080/14634988.2015.1028870

Mittermeier, R. A., Robles Gil, P., Hoffmann, M., Pilgrom, J., Brooks, T., Mittermeier, C. G., Lamoreux, J., & da Fonseca, G. A. B. (2004). *Hotspots revisited. Earth's biologically richest and most endangered terrestrial ecoregions*. Chicago, USA: University of Chicago Press.

Moerland, W., De Baerdemaeker, A., Boesveld, A., Grutters, M. A. J., & Van de Poel, J. L. (2015). Rotterdam. In J. Kelcey (Ed.), *Vertebrates and Invertebrates of European Cities: Selected non-avian fauna* (pp. 453–494). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6

Möllmann, C., Müller-Karulis, B., Diekmann, R., Flinkman, J., & Gårdmark, A. (2007). *Ecosystem regime state in the Baltic proper, Gulf of Riga, Gulf of Finland, and the Bothnian Sea. HELCOM Indicator Fact Sheets 2007*.

Molnár, Z. (2014). Perception and management of spatio-temporal pasture heterogeneity by Hungarian herders. *Rangeland Ecology and Management 67,* 107–118. https://doi.org/10.2111/REM-D-13-00082.1

Molnár, Z., Biró, M., Bartha, S., & Fekete, G. (2012). Past trends, present state and future prospects of Hungarian forest-steppes. In M. J. A. Werger, & M. A. van Staalduinen, *Eurasian steppes. Ecological problems and livelihoods in a changing world* (pp. 209–252). Dordrecht, The Netherlands: Springer. https://doi.org/10.1007/978-94-007-3886-7_7

Möls, T., Vellak, K., Vellak, A., & Ingerpuu, N. (2013). Global gradients in moss and vascular plant diversity. *Biodiversity and Conservation*, *22*(6–7), 1537–1551. https://doi.org/10.1007/s10531-013-0492-6

Moning, C., & Müller, J. (2009). Critical forest age thresholds for the diversity of lichens, molluscs and birds in beech (Fagus sylvatica L.) dominated forests. *Ecological Indicators*, *9*(5), 922–932. https://doi.org/10.1016/j.ecolind.2008.11.002

Montero-Serra, I., Edwards, M., & Genner, M. J. (2015). Warming shelf seas drive the subtropicalization of European pelagic fish communities. *Global Change Biology*, *21*(1), 144–153. https://doi.org/10.1111/gcb.12747

Montes, C., & Martino, P. (1987). Las lagunas salinas españolas [Spanish salt lakes]. In *Bases Cientficas para la protección de los humedales en España* (pp. 95–145). Madrid, Spain: Real Academia de Ciencias Exactas, Físicas y Naturales de Madrid.

Mooij, W. M., Janse, J. H., De Senerpont Domis, L. N., Hülsmann, S., & Ibelings, B. W. (2007). Predicting the effect of climate change on temperate shallow lakes with the ecosystem model PCLake. *Hydrobiologia*, *584*(1), 443–454. https://doi.org/10.1007/s10750-007-0600-2

Moon, D. (2013). *The plough that broke the steppes: Agriculture and environment on Russia's grasslands, 1700-1914*. Oxford, UK: Oxford University Press. Retrieved from https://global.oup.com/academic/product/the-plough-that-broke-the-steppes-9780199556434?cc=cz&lang=en&

Moore, S. E., & Huntington, H. P. (2008). Arctic marine mammals and climate change: Impacts and reilience. *Ecological Applications*, *18*(2), S157–S165. https://doi.org/10.1890/06-0571.1

Mora, C., Aburto-Oropeza, O., Ayala-Bocos, A., Ayotte, P. M., Banks, S., Bauman, A. G., Beger, M., Bessudo, S., Booth, D. J., Brokovich, E., Brooks, A., Chabanet, P., Cinner, J. E., Cortés, J., Cruz-Motta, J. J., Cupul-Magaña, A., DeMartini, E. E., Edgar, G. J., Feary, D. A., Ferse, S. C. A., Friedlander, A. M., Gaston, K. J., Gough, C., Graham, N. A. J., Green, A., Guzman, H., Hardt, M., Kulbicki, M., Letourneur, Y., Ĺpez-Pérez, A., Loreau, M., Loya, Y., Martinez, C., Mascareñas-Osorio, I., Morove, T., Nadon, M. O., Nakamura, Y., Paredes, G., Polunin, N. V. C., Pratchett, M. S., Reyes Bonilla, H., Rivera, F., Sala, E., Sandin, S. A., Soler, G., Stuart-Smith, R., Tessier, E., Tittensor, D. P., Tupper, M., Usseglio, P., Vigliola, L., Wantiez, L., Williams, I., Wilson, S. K., & Zapata, F. A. (2011). Global human footprint on the linkage between biodiversity and ecosystem functioning in reef fishes. *PLoS Biology 9*(4), e1000606. https://doi.org/10.1371/journal.pbio.1000606

Mora, C., Danovaro R., Loreau M. (2014). Alternative hypotheses to explain why biodiversity-ecosystem functioning relationships are concave-up in some natural ecosystems but concave-down in manipulative experiments. *Scientific Reports 4*, 5427. https://doi.org/10.1038/srep05427

Morato, T., Watson, R., Pitcher, T. J., & Pauly, D. (2006). Fishing down the deep. *Fish and Fisheries*, *7*(1), 24–34. https://doi.org/10.1111/j.1467-2979.2006.00205.x

Moreira, F., Viedma, O., Arianoutsou, M., Curt, T., Koutsias, N., Rigolot, E., Barbati, A., Corona, P., Vaz, P., Xanthopoulos, G., Mouillot, F., & Bilgili, E. (2011). Landscape - wildfire interactions in southern Europe: Implications for landscape management. *Journal of Environmental Management*, *92*(10), 2389–2402. http://doi.org/10.1016/j.jenvman.2011.06.028

Moreira, F. & Russo, D. (2007). Modelling the impact of agricultural abandonment and wildfires on vertebrate diversity in Mediterranean Europe. *Landscape Ecology 22*(10), 1461-1476. https://doi.org/10.1007/s10980-007-9125-3

Morgunov, B. (2011). *The diagnostic analysis of the environment of the Arctic zone of the Russian Federation (Extended summary)*. Moscow, Russian Federation: Научный мир [Scientific World].

Morin, X., Fahse, L., de Mazancourt, C., Scherer-Lorenzen, M., & Bugmann, H. (2014). Temporal stability in forest productivity increases with tree diversity due to asynchrony in species dynamics. *Ecology Letters. 17*(12), 1526–1535. https://doi.org/10.1111/ele.12357

Moss, B. (2015). Biodiversity climate change impacts report card technical paper: Freshwaters, climate change and UK conservation. *Freshwater Ecology, Biodiversity Report Card Paper 17*, 1–63.

Mouchet M. A., Villeger S., Mason N. W. H., & Mouillot D. (2010). A functional guide to functional diversity measures. *Functional Ecology, 24*, 867–876. https://doi.org/10.1111/j.1365-2435.2010.01695.x

Mouillot, D., Albouy, C., Guilhaumon, F., Ben Rais Lasram, F., Coll, M., Devictor, V., Meynard, C. N., Pauly, D., Tomasini, J. A., Troussellier, M., Velez, L., Watson, R., Douzery, E. J. P., & Mouquet, N. (2011). Protected and threatened components of fish biodiversity in the Mediterranean Sea. *Current Biology*, *21*(12), 1044–1050. https://doi.org/10.1016/j.cub.2011.05.005

Mouillot, D., Bellwood, D. R., Baraloto, C., Chave, J., Galzin, R., Harmelin-Vivien, M., Kulbicki, M., Lavergne, S., Lavorel, S., Mouquet, N., Paine, C. E. T., Renaud, J., & Thuiller, W. (2013). Rare species support vulnerable functions in high-diversity ecosystems. *PLoS Biology, 11*(5), e1001569. https://doi.org/10.1371/journal.pbio.1001569

Moya, Ó., Contreras-Díaz, H., Oromí, P., & Juan, C. (2004). Genetic structure, phylogeography and demography of two ground-beetle species endemic to the Tenerife laurel forest (Canary Islands). *Molecular Ecology*, *13*(10), 3153–3167. http://doi.org/10.1111/j.1365-294X.2004.02316.x

Mulec, J., & Kosi, G. (2009). Lampenflora algae and methods of growth control. *Journal of Cave and Karst Studies*, *71*(2), 109–115.

Müller, J., Klaus, V. H., Kleinebecker, T., Prati, D., Hölzel, N., & Fischer, M.

(2012). Impact of land-use intensity and productivity on bryophyte diversity in agricultural grasslands. *PloS One*, 7(12), e51520. https://doi.org/10.1371/journal. pone.0051520

Müller, R., Heinicke, T., Juschus, O., & Zeitz, J. (2016). Genesis and abiotic characteristics of three high-altitude peatlands in the Tien Shan Mountains (Kyrgyzstan), with focus on silty peatland substrates. *Mires and Peat, 18*(24), 1–19. http://doi.org/10.19189/MaP.2015. OMB.217

Mullon, C., Steinmetz, F., Merino, G., Fernandes, J. A., Cheung, W. W. L., Butenschön, M., & Barange, M. (2016). Quantitative pathways for Northeast Atlantic fisheries based on climate, ecological–economic and governance modelling scenarios. *Ecological Modelling*, *320,* 273–291. https://doi.org/10.1016/j. ecolmodel.2015.09.027

Mumladze, L., Chaladze, G., Asanidze, Z., Saghinadze, S., & Khachidze, E. (2008). *Refugial forest from the western Lesser Caucasus. Final Report. Priject ID – 090107.*

Muñoz-Fuentes, V., Vilà, C., Green, A. J., Negro, J. J., & Sorenson, M. D. (2007). Hybridization between white-headed ducks and introduced ruddy ducks in Spain. *Molecular Ecology*, *16*(3), 629–638. https:// doi.org/10.1111/j.1365-294X.2006.03170.x

Murray, J. W., Top, Z., & Ozsoy, E. (1989). Hydrographic properties and ventilation of the Black Sea. *Deep Sea Resources*, *38*(Suppl.), S663–S689. https:// doi.org/10.1016/S0198-0149(10)80003-2

Mustonen, T., & Helander, E. (Eds.). (2004). *Snowscapes, dreamscapes - A snowchange community book of change*. Tampere, Finland: Tampere Polytechnic.

Nakhutsrishvili, G. (2003). High mountain vegetation of the Caucasus region. In L. Nagy, G. Grabherr, C. Körner, & D. B. A. Thompson (Eds.), *Alpine biodiversity in Europe* (pp. 93–103). Berlin, Germany: Springer.

Nakhutsrishvili, G., Zazanashvili, N., & Batsatsashvili, K. (2011). Regional profile: Colchic and Hyrcanic temperate rainforests of the western Eurasian Caucasus. In D.

A. DellaSala (Ed.), *Temperate and boreal rainforests of the world: Ecology and conservation* (pp. 214–222). Washington, DC, USA: Island Press.

Nakhutsrishvili, G., Zazanashvili, N., Batsatsashvili, K., & Montalvo Mancheno, C. S. (2015). Colchic and Hyrcanian forests of the Caucasus: similarities, differences and conservation status. *Flora Mediterranea, 25,* 185–192. http://doi.org/10.7320/ FlMedit25SI.185

Namsaraev, B. B., Abidueva, E. Y., & Lavrent'eva, E. V. [Намсараев, Б. Б., Абидуева, Е. Ю., & Лаврентьева, Е. В.]. (2008). Экология микроорганизмов экстремальных водных систем *[Ecology of microorganisms in extreme aquatic systems]*. Ulan-Ude, Russian Federation: Издательство Бурятского Государственного Университета [Publishing House of Buryat State University].

NASA. (2004). *Caspian Sea.* Retrieved from https://eoimages.gsfc.nasa.gov/ images/imagerecords/70000/70736/ CaspianSea.A2004100.0945.1km.jpg

NASA. (2014). Revelation of Aral Sea disaster. Retrieved from http://www. capitalwired.com/nasas-images-revelation- of-aral-sea-disaster/23259/

Nascimbene, J., Lazzaro, L., & Benesperi, R. (2015). Patterns of β-diversity and similarity reveal biotic homogenization of epiphytic lichen communities associated with the spread of black locust forests. *Fungal Ecology*, *14*, 1–7. https://doi.org/10.1016/j. funeco.2014.10.006

Nascimbene, J., Nimis, P. L., & Ravera, S. (2013a). Evaluating the conservation status of epiphytic lichens of Italy: A red list. *Plant Biosystems*, *147*(4), 898–904. https://doi.org/10.1080/1126350 4.2012.748101

Nascimbene, J., Thor, G., & Nimis, P. L. (2013b). Effects of forest management on epiphytic lichens in temperate deciduous forests of Europe - A review. *Forest Ecology and Management, 298*, 27-38. https://doi. org/10.1016/j.foreco.2013.03.008

Nash, T. H. (Ed.). (2008a). *Lichen Biology, 2nd Edition*. Cambridge, UK: Cambridge University Press.

Nash, T. H. (2008b). Lichen sensitivity to air pollution. In T. H. Nash (Ed.), *Lichen Biology, 2nd edition* (pp. 301–316). Cambridge UK: Cambridge University Press. https://doi. org/10.1017/CBO9780511790478.016

Natcheva, R., Ganeva, A., & Spiridonov, G. (2006). Red List of the bryophytes in Bulgaria. *Phytologia Balcanica*, *12*(1), 55–62.

Navarro, L.M. & Pereira H.M. (2012). Rewilding abandoned landscapes in Europe. *Ecosystems 15,* 900–912.

Navarro, L. M., & Pereira, H. M. (2015). Rewilding abandoned landscapes in Europe. In: L. M. Navarro & H. M. Pereira (Eds.), *Rewilding European landscapes* (pp. 3–23). Cham, Switzerland: Springer. http:// doi.org/10.1007/978-3-319-12039-3_1

Neaves, L. E., Whitlock, R., Piertney, S. B., Burke, T., Butlin, R. K., & Hollingsworth, P. M. (2015). *Implications of climate change for genetic diversity and evolvability in the UK. Biodiversity climate change report card technical paper 15*.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D. R., Chan, K. M. A., Daily, G. C., Goldstein, J., Kareiva, P. M., Lonsdorf, E., Naidoo, R., Ricketts, T. H., & Shaw, M. R. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, *7*(1), 4–11. http://doi. org/10.1890/080023

Nesterova, D. A., & Terenko, L. M., [Нестерова, Д. А., & Теренко, Л. М.]. (2009). Фитопланктон Каркинитского залива в сентябре 2008 г. [The Karkinit Bay phytoplankton in September 2008]. Экологическая Безопасность Прибрежной И Шельфовой Зон Моря *[Ecological Safety of Sea Coastal and Shelf Zones]*, *20*, 293–300.

Neumann, T. (2010). Climate change effects on the Baltic Sea ecosystem: A model study. *Journal of Marine Ecosystems, 81*(3), 213-224. https://doi.org/10.1016/j. jmarsys.2009.12.001

Newbold, T., Hudson, L. N., Hill, S. L. L., Contu, S., Lysenko, I., Senior, R. A., Börger, L., Bennett, D. J., Choimes, A., Collen, B., Day, J., De Palma, A., Díaz, S., Echeverria-Londoño, S., Edgar, M. J., Feldman, A., Garon, M., Harrison, M. L. K., Alhusseini, T., Ingram, D. J., Itescu, Y., Kattge, J., Kemp, V., Kirkpatrick, L., Kleyer, M., Correia, D. L. P., Martin, C. D., Meiri, S., Novosolov, M., Pan, Y., Phillips, H. R. P., Purves, D. W., Robinson, A., Simpson, J., Tuck, S. L., Weiher, E., White, H. J., Ewers, R. M., Mace, G. M., Scharlemann, J. P. W., & Purvis, A. (2015). Global effects of land use on local terrestrial biodiversity. *Nature*, *520*(7545), 45–50. https://doi.org/10.1038/nature14324

Nicastro, K. R., Zardi, G. I., Teixeira, S., Neiva, J., Serrao, E. A., & Pearson, G. A. (2013). Shift happens: trailing edge contraction associated with recent warming trends threatens a distinct genetic lineage in the marine macroalga *Fucus vesiculosus. BMC Biology*, *11*, 6. https://doi.org/10.1186/1741-7007-11-6

Nicolaev S., Alexandrov L., Boicenco L., Coatu V., Diaconeasa D., Dumitrache C., Dumitrescu O., Golumbeanu, L. Lazar, V. Malciu, R. Mateescu, V. Maximov, D. Micu, E. Mihailov, M. Nenciu M., Nita V., Oros A., Spinu A., Stoica E., Tabarcea C., Timofte F., Tiganus, D., & Zaharia, T. (2013). Report on the state of marine and coastal environment in 2013. *Recherches Marines*, *43*, 5–138.

Niemelä, J., & Kotze, D. J. (2009). Carabid beetle assemblages along urban to rural gradients: A review. *Landscape and Urban Planning*, *92*(2), 65–71. https://doi.org/10.1016/j.landurbplan.2009.05.016

Nieto, A., & Alexander, K. N. (2010). *European red list of saproxylic beetles*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/84561

Nieto, A., Ralph, G. M., Comeros-Raynal, M. T., Kemp, J., García Criado, M., Allen, D. J., Dulvy, N. K., Walls, R. H. L., Russell, B., Pollard, D., García, S., Craig, M., Collette, B. B., Pollom, R., Biscoito, M., Chao, N. L., Abella, A., Afonso, P., Álvarez, H., Carpenter, K. E., Clò, S., Cook, R., Costa, M. J., Delgado, J., Dureuil, M., Ellis, J. R., Farrell, E. D., Fernandes, P., Florin, A.-B. Fordham, S., Fowler, S., de Sola, L. G., Herrera, J. G., Goodpaster, A., Harvey,M., Heessen, H., Herler, J., Jung, A., Karmovskaya, E., Keskin, Ç., Knudsen,S. W., Kobyliansky, S., Kovačić, M., Lawson, J. M., Lorance, P., Phillips, S. M., Munroe, T., Nedreaas, K., Nielsen, J., Papaconstantinou, C., Polidoro, B., Pollock, C. M., Rijnsdorp, A. D., Sayer, C., Scott, J., Serena, F., Smith-Vaniz, W. F., Soldo, A., Stump, E., & Williams, J. T. (2015). *European red list of marine fishes*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/082723

Nieto, A., Roberts, S. P. M., Kemp, J., Rasmont, P., Kuhlmann, M., García Criado, M., Biesmeijer, J. C., Bogusch, P., Dathe, H. H., De la Rúa, P., De Meulemeester, T., Dehon, M., Dewulf, A., Ortiz-Sánchez, F. J., Lhomme, P., Pauly, A., Potts, S.G., Praz, C., Quaranta, M., Radchenko, V. G., Scheuchl, E., Smit, J., Straka, J., Terzo, M., Tomozii, B., Window, J., & Michez, D. (2014). *European red list of bees*. Luxembourg: Publication Office of the European Union. http://doi.org/10.2779/77003

Niklfeld, H. (1999). *Rote Listen gefährdeter Pflanzen Österreichs [Red list of endangered plants of Austria]*.

Nikolsky, G. V. [Никольский, Г. В.]. (1938). Рыбы Таджикистана [*Fishes of Tajikistan*]. Moscow, USSR: Academy of Science of the USSR.

Nikolsky, G. V. [Никольский, Г. В.]. (1971). Частная ихтиология [*Special ichthyology*]. Moscow, USSR: Высшая школа [Higher School].

Nissenbaum, A. (1975). The microbiology and biogeochemistry of the Dead Sea. *Microbial Ecology*, *2*(2), 139–161. http://doi.org/10.1007/BF02010435

Noce, S., Collalti, A., Valentini, R., & Santini, M. (2016). Hot spot maps of forest presence in the Mediterranean basin. *iForest*, *9*(5), 766–774. http://doi.org/10.3832/ifor1802-009

Nogués-Bravo, D., Araújo, M. B., Errea, M. P., & Martínez-Rica, J. P. (2007). Exposure of global mountain systems to climate warming during the 21st Century. *Global Environmental Change*, *17*(3–4), 420–428. https://doi.org/10.1016/j.gloenvcha.2006.11.007

Nordén, B., Dahlberg, A., Brandrud, T. E., Fritz, Ö., Ejrnaes, R., & Ovaskainen, O. (2014). Effects of ecological continuity on species richness and composition in forests and woodlands: A review. *Écoscience*, *21*(1), 34–45. https://doi.org/10.2980/21-1-3667

Norkko, A., Laakkonen, T., & Laine, A. (2007). *Trends in soft-sediment macrozoobenthic communities in the open sea areas of the Baltic Sea*.

Norse, E. A., Brooke, S., Cheung, W. W. L., Clark, M. R., Ekeland, I., Froese, R., Gjerde, K. M., Haedrich, R. L., Heppell, S. S., Morato, T., Morgan, L. E., Pauly, D., Sumaila, R., & Watson, R. (2012). Sustainability of deep-sea fisheries. *Marine Policy*, *36*(2), 307–320. https://doi.org/10.1016/j.marpol.2011.06.008

Nowak, A., & Nowak, S. (2015). Distribution patterns of segetal weeds of cereal crops in Tajikistan. *Pakistan Journal of Botany*, *47*(4), 1415–1422.

Nowak, A., Nowak, S., & Nobis, M. (2011). Distribution patterns, ecological characteristic and conservation status of endemic plants of Tadzhikistan - A global hotspot of diversity. *Journal for Nature Conservation*, *19*(5), 296–305. https://doi.org/10.1016/j.jnc.2011.05.003

Nowak, A., Nowak, S., Nobis, M., & Nobis, A. (2014). A report on the conservation status of segetal weeds in Tajikistan. *Weed Research*, *54*(6), 635–648. https://doi.org/10.1111/wre.12103

Nybø, S., & Evju, M. (Eds.). (2017). *Fagsystem for fastsetting av god økologisk tilstand. Forslag fra et ekspertråd [System for the determination of good ecological condition. Suggestions from an expert advice]*. Retrieved from https://www.regjeringen.no/no/dokumenter/fagsystem-for-fastsetting-av-god-okologisk-tilstand/id2558481/

Odgaard, B. V. (1994). The Holocene vegetation history of northern West Jutland, Denmark. *Opera Botanica*, *123*, 1–171. https://doi.org/10.1111/j.1756-1051.1994.tb00625.x

Ogus, T. (Ed.). (2008). The state of marine living resources. In *State of the Environment of the Black Sea* (*2001-2006/7*). Istanbul, Turkey: Commission on the Protection of the Black Sea Against Pollution (BSC).

Ojaveer, H., Jaanus, A., Mackenzie, B. R., Martin, G., Olenin, S., Radziejewska, T., Telesh, I., Zettler, M. L., & Zaiko, A. (2010). Status of biodiversity in the Baltic sea. *PLoS ONE*, *5*(9), 1–19. https://doi.org/10.1371/journal.pone.0012467

Ojaveer, H., Olenin, S., Narščius, A., Florin, A.-B., Ezhova, E., Gollasch, S., Jensen, K. R., Lehtiniemi, M., Minchin, D., Normant-Saremba, M., & Strāke, S. (2016). Dynamics of biological invasions and pathways over time: a case study of a temperate coastal sea. *Biological Invasions*, *19*(3), 799-813. https://doi.org/10.1007/s10530-016-1316-x

Oliva, J., Boberg, J., & Stenlid, J. (2013). First report of *Sphaeropsis sapinea* on Scots pine (*Pinus sylvestris*) and Austrian pine (*P. nigra*) in Sweden. *New Disease Reports, 27,* 23. http://dx.doi.org/10.5197/j.2044-0588.2013.027.023

Oliver, T. H., Heard, M. S., Isaac, N. J. B., Roy, D. B., Procter, D., Eigenbrod, F., Freckleton, R., Hector, A., Orme, C. D. L., Petchey, O. L., Proença, V., Raffaelli, D., Suttle, K. B., Mace, G. M., Martín-López, B., Woodcock, B. A., & Bullock, J. M. (2015). Biodiversity and resilience of ecosystem functions. *Trends in Ecology and Evolution, 30*(11), 673-84. https://doi.org/10.1016/j.tree.2015.08.009

Olivier L., Galland, J-P., Maurin, H., & Roux, J-P. (1995). *Livre rouge de la flore menacée de France I espèces prioritaires [Red list of threatened flora of France – I priority species]*. Paris, France: MNHN.

Olson, K. A. (2013). *Saiga crossing options - Guidelines and recommendations to mitigate barrier effects of border fencing and railroad corridors on saiga antelope in Kazakhstan*.

Oltean, M., Negrean, G., Popescu, A, Roman, N., Dihoru, G., Sanda, V., & Mihailescu, S. (1994). *Lista rosie a plantelor superioare din România [Red list of higher plants of Romania]*. Bucharest, Romania: Academia Romana Institutul de Biologie.

Oren, A. (2006). Life at high salt concentrations. In E. Rosenberg, E. F. DeLong, S. Lory, E. Stackebrandt, & F. Thompson (Eds.), *The prokaryotes* (pp. 421–440). New York, USA: Springer.

Orgiazzi, A., Bardgett, R. D., Barrios, E., Behan-Pelletier, V., Briones, M. J. I., Chotte, J-L., De Deyn, G. B., Eggleton, P., Fierer, N., Fraser, T., Hedlund, K., Jeffrey, S., Johnson, N. C., Jones, A., Kandeler, E., Kaneko, N., Lavelle, P., Lemanceau, P., Miko, L., Montanarella, L., de Souza Moreira, F. M., Ramirez, K. S., Scheu, S., Singh, B.K., Six, J., van der Putten, W.H., & Wall, D. H. (Eds.). (2016). *Global soil biodiversity atlas*. Luxembourg: Publications Office of the European Union. https://doi.org/doi:10.2788/799182

Orlandi, S., Probo, M., Sitzia, T., Trentanovi, G., Garbarino, M., Lombardi, G., & Lonati, M. (2016). Environmental and land use determinants of grassland patch diversity in the western and eastern Alps under agro-pastoral abandonment. *Biodiversity and Conservation*, *25*(2), 275–293. https://doi.org/10.1007/s10531-016-1046-5

Orlova, T. Y., Konovalova, G. V, Stonik, I. V, Selina, M. S., Tatyana, V., & Shevchenko, O. G. (2002). Harmful algal blooms on the eastern coast of Russia. In *Harmful algal blooms in the PICES region of the North Pacific. PICES Report 23* (*August 2002*). PICES. Retrieved from https://www.pices.int/publications/scientific_reports/Report23/default.aspx

Orlov, A. A., Chechevishnikov, A. L., Alekseenkova, E. S., Borishpolets, K. P., Krylov, A. V., Kudeneeva, Yu. S., Mizin, V. I., Nikitin, A. I., Fedorchenko, A. V., & Chernyavskii, S. I. [Орлов, А. А., Чечевичников, А. Л., Алексеенкова, Е. С., Боришполец, К. П., Крылов, А. В., Куденеева, Ю. С., Мизин, В. И., Никитин, А. И., Федорченко, А. В., & Чернявский, С. И.]. (2011). Проблема пресной воды. Глобальный контекст политики России *[Problem of fresh water. Global context of Russian politics]*. Moscow, Russian Federation: MGIMO-University.

Örmeci, C., & Ekercin, S. (2005). Water quality monitoring using satellite image data: a case study around the Salt Lake in Turkey. In *Proc. SPIE 5977, Remote Sensing of the*

Ocean, Sea Ice, and Large Water Regions 2005, 59770K (*October 20, 2005*). https://doi.org/10.1117/12.628558

OSPAR. (2008). *Case reports for the OSPAR list of threatened and/or declining species and habitats*. *Biodiversity series*.

OSPAR. (2010). *Quality status report 2010*. Retrieved from https://qsr2010.ospar.org/en/index.html

OSPAR. (2017). *Intermediate assessment 2017.* Retrieved July 6, 2017, from https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/

Österblom, H., Hansson, S., Larsson, U., Hjerne, O., Wulff, F., Elmgren, R., & Folke, C. (2007). Human-induced trophic cascades and ecological regime shifts in the baltic sea. *Ecosystems*, *10*(6), 877–889. https://doi.org/10.1007/s10021-007-9069-0

Otero, M.M., Numa, C., Bo, M., Orejas, C., Garrabou, J., Cerrano, C., KružicÅL, P., Antoniadou, C., Aguilar, R., Kipson, S., Linares, C., Terr.n-Sigler, A., Brossard, J., Kersting, D., Casado-Amez.a, P., Garc.a, S., Goffredo, S., Oca.a, O., Caroselli, E., B. (2017). *Overview of the conservation status of Mediterranean anthozoans*. Gland, Switzerland: IUCN.

Oug, E., Cochrane, S. K. J., Sundet, J. H., Norling, K., & Nilsson, H. C. (2011). Effects of the invasive red king crab (*Paralithodes camtschaticus*) on soft-bottom fauna in Varangerfjorden, northern Norway. *Marine Biodiversity*, *41*(3), 467–479. https://doi.org/10.1007/s12526-010-0068-6

Overland, J. E., & Stabeno, P. J. (2004). Is the climate of the Bering Sea warming and affecting the ecosystem? *Eos, Transactions American Geophysical Union*, *85*(33), 309. https://doi.org/10.1029/2004EO330001

Pabi, S., van Dijken, G. L., & Arrigo, K. R. (2008). Primary production in the Arctic Ocean, 1998-2006. *Journal of Geophysical Research: Oceans*, *113*(8), C08005. https://doi.org/10.1029/2007JC004578

Pacifici, M., Foden, W. B., Visconti, P., Watson, J. E. M., Butchart, S. H. M., Kovacs, K. M., Scheffers, B. R., Hole,

D. G., Martin, T. G., Akçakaya, H. R., Corlett, R. T., Huntley, B., Bickford, D., Carr, J. a., Hoffmann, A. a., Midgley, G. F., P., P.-K., Pearson, R. G., Williams, S. E., Willis, S. G., Young, B., & Rondinini, C. (2015). Assessing species vulnerability to climate change. *Nature Climate Change*, *5*(February), 215–225. https://doi.org/10.1038/nclimate2448

Paillet, Y., Bergès, L., HjÄltén, J., Ódor, P., Avon, C., Bernhardt-Römermann, M., Bijlsma, R. J., De Bruyn, L., Fuhr, M., Grandin, U., Kanka, R., Lundin, L., Luque, S., Magura, T., Matesanz, S., Mészáros, I., SebastiÀ, M. T., Schmidt, W., Standovár, T., Tóthmérész, B., Uotila, A., Valladares, F., Vellak, K., & Virtanen, R. (2010). Biodiversity differences between managed and unmanaged forests: Meta-analysis of species richness in Europe. *Conservation Biology*, *24*(1), 101–112. https://doi.org/10.1111/j.1523-1739.2009.01399.x

Paine, T. D., & Lieutier, F. (Eds.). (2016). Insects and diseases of Mediterranean forest systems. Cham, Switzerland: Springer International Publishing. http://doi.org/10.1007/978-3-319-24744-1

Pajunen, A. M., Oksanen, J., & Virtanen, R. (2011). Impact of shrub canopies on understorey vegetation in western Eurasian tundra. *Journal of Vegetation Science*, *22*(5), 837–846. https://doi.org/10.1111/j.1654-1103.2011.01285.x

Pakeman, R. J., & Nolan, A. J. (2009). Setting sustainable grazing levels for heather moorland: a multi-site analysis. *Journal of Applied Ecology*, *46*(2), 363–368. http://doi.org/10.1111/j.1365-2664.2008.01603.x

Palmer, A. N. (1991). Origin and morphology of limestone caves. *Geological Society of America Bulletin, 103*(1), 1-21. https://doi.org/10.1130/0016-7606(1991)103<0001:OAMOLC>2.3.CO;2

Paltsyn, M. Y., Spitsyn, S. V., Kuksin, A.N. & Istomov, S. V. (2012). *Snow leopard conservation in Russia*. Krasnoyarsk, Russian Federation: WWF Russia.

Pape, T., Bickel, D., & Meier, R. (2009). *Diptera diversity: Status, challenges and tools*. Leiden, The Netherlands: Brill.

Parfenov, V. I. Kozlovskaya, N. V., & Vynaev, G. V. [Парфенов, В. И., Козловская, Н. В., & Вынаев, Г. В.]. (1987). *Rare and threatened plant species of Byelorussia and Lithuania*. Minsk, USSR: Наука и техника [Science and Technology].

Parish, F., Sirin, A., Charman, D., Joosten, H., Minayeva, T., & Silvius, M. (2008). *Assessment on peatlands, biodiversity and climate change: Main report*. Wageningen, The Netherlands: Global Environment Centre. Retrieved from http://www.imcg.net/media/download_gallery/books/assessment_peatland.pdf

Parmesan, C. (2006). Ecological and evolutionary responses to recent climate change. *Annual of Ecology, Evolution and Systematics*, *37*(1), 637–669. https://doi.org/10.2307/annurev.ecolsys.37.091305.30000024

Parnikoza, I., & Vasiluk, A. (2011). Ukrainian steppes: current state and perspectives for protection. *Annales Universitatis Mariae Curie-Skłodowska. Sectio C. Biologia*, *9*(1), 23–37. https://doi.org/10.2478/v10067-011-0018-0

Pärtel, M., Bruun, H. H., & Sammul, M. (2005). Biodiversity in temperate European grasslands: origin and conservation. In *Grassland Science in Europe, Volume 10* (pp. 1–14). Retrieved from http://lup.lub.lu.se/record/532202/file/625284.pdf

Passy, S. I., & Legendre, P. (2006). Are algal communities driven toward maximum biomass? *Proceedings of the Royal Society B: Biological Sciences, 273*, 2667–2674. https://doi.org/10.1098/rspb.2006.3632

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C. C., Alexander, J., Edwards, P. J., Arévalo, J. R., Cavieres, L. A., Guisan, A., Haider, S., Jakobs, G., McDougall, K., Millar, C. I., Naylor, B. J., Parks, C. G., Rew, L. J., & Seipel, T. (2009). Ain't no mountain high enough: Plant invasions reaching new elevations. *Frontiers in Ecology and the Environment*, *7*(9), 479–486. https://doi.org/10.1890/080072

Pauli, H., Gottfried, G., Dirnböck, T., Dullinger, S., & Grabherr, G. (2003). Assessing the long-term dynamics of endemic plants at summit habitats. In L. Nagy, G. Grabherr, C. Körner, & D. B. A. Thompson (Eds.), *Alpine biodiversity in Europe* (pp. 195–207). Berlin, Germany: Springer.

Pauli, H., Gottfried, M., Dullinger, S., Abdaladze, O., Akhalkatsi, M., Alonso, J. L. B., Coldea, G., Dick, J., Erschbamer, B., Calzado, R. F., Ghosn, D., Holten, J. I., Kanka, R., Kazakis, G., Kollar, J., Larsson, P., Moiseev, P., Moiseev, D., Molau, U., Mesa, J. M., Nagy, L., Pelino, G., Puscas, M., Rossi, G., Stanisci, A., Syverhuset, A. O., Theurillat, J. P., Tomaselli, M., Unterluggauer, P., Villar, L., Vittoz, P., & Grabherr, G. (2012). Recent plant diversity changes on Europe's mountain summits. *Science*, *336*(6079), 353–355. https://doi.org/DOI 10.1126/science.1219033

Pauls, S. U., Nowak, C., Bálint, M., & Pfenninger, M. (2013) The impact of global climate change on genetic diversity within populations and species. *Molecular Ecology, 22,* 925–946. https://doi.org/10.1111/mec.12152

Pausas, J. G., Llovet, J., Anselm, R., & Vallejo, R. (2008). Are wildfires a disaster in the Mediterranean basin? – A review. *International Journal of Wildland Fire*, *17*(6), 713–723. http://doi.org/10.1071/WF07151

Pavlov V. A., & Sundet, J. H. (2011). Snow crab. In T. Jakobsen & V. K. Ozhigin (Eds.), *The Barents Sea: ecosystem, resources, management: half a century of Russian-Norwegian cooperation* (pp. 168–172). Trondheim, Norway: Tapir Academic Press. Retrieved from https://brage.bibsys.no/xmlui/handle/11250/109444

Pavlovskaya, L. P. (1995). *Fishery in the lower Amu-Darya under the impact of irrigated agriculture*. Retrieved September 20, 2017, from http://www.fao.org/docrep/V9529E/v9529E04.htm

PBL. (2010). *Rethinking Global Biodiversity Strategies: Exploring structural changes in production and consumption to reduce biodiversity loss*. The Hague/Bilthoven: Netherlands Environmental Assessment Agency.

PBL. (2012). *Roads from Rio + 20*. Retrieved from https://roadsfromrio.pbl.nl.

PBL. (2014). *How sectors can contribute to sustainable use and conservation of*

biodiversity. *CBD technical series 78*. Montreal, Canada: Secretariat of the Convention on Biological Diversity. Retrieved from http://www.cbd.int/doc/publications/cbd-ts-79-en.pdf

Pe'er, G., Dicks, L. V., Visconti, P., Arlettaz, R., Báldi, A., Benton, T. G., Collins, S., Dieterich, M., Gregory, R. D., Hartig, F., Henle, K., Hobson, P. R., Kleijn, D., Neumann, R. K., Robijns, T., Schmidt, J., Shwartz, A., Sutherland, W. J., Turbé, A., Wulf, F., & Scott, A. V. (2014). EU agricultural reform fails on biodiversity. *Science*, *344*(6188), 1090–1092. https://doi.org/10.1126/science.1253425

Peay, K. G., Kennedy, P. G., & Talbot, J. M. (2016). Dimensions of biodiversity in the Earth mycobiome. *Nature Reviews Microbiology*, *14*(7), 434–447. https://doi.org/10.1038/nrmicro.2016.59

Peeler, E. J., Oidtmann, B. C., Midtlyng, P. J., Miossec, L., & Gozlan, R. E. (2011). Non-native aquatic animals introductions have driven disease emergence in Europe. *Biological Invasions, 13*(6), 1291-1303. https://doi.org/10.1007/s10530-010-9890-9

Pekel, J.-F., Cottam, A., Gorelick, N., & Belward, A. S. (2016). High-resolution mapping of global surface water and its long-term changes. *Nature*, (540), 418–422. https://doi.org/10.1038/nature20584

Pellissier, L., Anzini, M., Maiorano, L., Dubuis, A., Pottier, J., Vittoz, P., & Guisan, A. (2013). Spatial predictions of land-use transitions and associated threats to biodiversity: The case of forest regrowth in mountain grasslands. *Applied Vegetation Science*, *16*(2), 227–236. https://doi.org/10.1111/j.1654-109X.2012.01215.x

Peñuelas, J., Filella, I., & Comas, P. (2002). Changed plant and animal life cycles from 1952-2000 in the Mediterranean region. *Global Change Biology*, *8*(8), 531–544. http://doi.org/10.1046/j.1365-2486.2002.00489.x

Pereira, H. M., Leadley, P. W., Proença, V., Alkemade, R., Scharlemann, J. P. W., Fernandez-Manjarrés, J. F., Araújo, M. B., Balvanera, P., Biggs, R., Cheung, W. W. L., Chini, L., Cooper, H. D., Gilman, E. L., Guénette, S.,

Hurtt, G. C., Huntington, H. P., Mace, G. M., Oberdorff, T., Revenga, C., Rodrigues, P., Scholes, R. J., Sumaila, U. R., & Walpole, M. (2010). Scenarios for global biodiversity in the 21st century. *Science*, *330*(6010), 1496–501. https://doi.org/10.1126/science.1196624

Pérez-Ruzafa, A., García-Charton, J. A., & Marcos, C. (2017). North East Atlantic vs. Mediterranean marine protected areas as fisheries management tool. *Frontiers in Marine Science, 4,* 245. https://doi.org/10.3389/fmars.2017.00245

Pérez-Ruzafa, A., Gilabert, J., Gutiérrez, J.M., Fernández, A.I., Marcos, C. & Sabah, S. (2002). Evidence of a planktonic food web response to changes in nutrient input dynamics in the Mar Menor coastal lagoon, Spain. *Hydrobiologia, 475/476,* 359-369. https://doi.org/10.3389/10.1023/A:1020343510060

Pérez-Ruzafa, Á., González-Wangüemert, M., Lenfant, P., Marcos, C., & García-Charton, J. A. (2006). Effects of fishing protection on the genetic structure of fish populations. *Biological Conservation*, *129*(2), 244–255. https://doi.org/10.1016/j.biocon.2005.10.040

Pérez-Ruzafa, A., & Marcos, C. (2012). Fisheries in coastal lagoons: An assumed but poorly researched aspect of the ecology and functioning of coastal lagoons. *Estuarine, Coastal and Shelf Science*, *110*, 15–31. https://doi.org/10.1016/j.ecss.2012.05.025

Pérez-Ruzafa, A., Marcos, C., & Pérez-Ruzafa, I. M. (2011). Mediterranean coastal lagoons in an ecosystem and aquatic resources management context. *Physics and Chemistry of the Earth*, *36*(5–6), 160-166. https://doi.org/10.1016/j.pce.2010.04.013

Peringer, A., Siehoff, S., Chételat, J., Spiegelberger, T., Buttler, A., & Gillet, F. (2013). Past and future landscape dynamics in pasture-woodlands of the Swiss Jura Mountains under climate change. *Ecology and Society*, *18*(3), 11. https://doi.org/10.5751/ES-05600-180311

Perry, A. L., Low, P. J., Ellis, J. R., & Reynolds, J. D. (2005). Climate change and distribution shifts in marine fishes.

Science, *308*(5730), 1912–5. https://doi.org/10.1126/science.1111322

Peskova, T. J. [Пескова, Т. Ю.]. (2000). Половая структура популяций земноводных при обитании в чистых и загрязненных пестицидами водоемах [Sex-ratio structure of the amphibians inhabiting pure and pesticide-polluted reservoirs]. Современная герпетология (Сборник трудов) [*Modern Herpetology (Collected Proceedings)]*, *1*, 26–35.

Petchey O.L. (2000). Species diversity, species extinction, and ecosystem function. *American Naturalist, 155,* 696–702. https://doi.org/10.1086/303352

Petersen, S., Krätschell, A., Augustin, N., Jamieson, J., Hein, J. R., & Hannington, M. D. (2016). News from the seabed – Geological characteristics and resource potential of deep-sea mineral resources. *Marine Policy*, *70*, 175–187. https://doi.org/10.1016/j.marpol.2016.03.012

Petitpierre, B., MacDougall, K., Seipel, T., Broennimann, O., Guisan, A., & Kueffer, C. (2015). Will climate change increase the risk of plant invasions into mountains? *Ecological Applications*, *26*(2), 1507090237160008. https://doi.org/10.1890/14-1871.1

Petr, T., Ismukhanov, K., Kamilov, B., Pulakhton, D., & Umarov, P. D. (2004). Irrigation systems and their fisheries in the Aral Sea Basin, Central Asia. In T. Welcomme & R. Petr (Eds.), *Proceedings of the second international symposium on the management of large rivers for fisheries volume II*. Bangkok, Thailand: RAP Public. Retrieved from http://www.fao.org/docrep/007/ad526e/ad526e00.htm

Pettersson, R. B., Ball, J. P., Renhorn, K. E., Esseen, P. A., & Sjöberg, K. (1995). Invertebrate communities in boreal forest canopies as influenced by forestry and lichens with implications for passerine birds. *Biological Conservation*, *74*(1), 57–63. https://doi.org/10.1016/0006-3207(95)00015-V

Pham, C. K., Diogo, H., Menezes, G., Porteiro, F., Braga-Henriques, A., Vandeperre, F., & Morato, T. (2014a). Deep-water longline fishing has reduced impact on vulnerable marine ecosystems

Scientific Reports, 4, 4837. http://dx.doi.org/10.1038/srep04837

Pham, C. K., Ramirez-Llodra, E., Alt, C. H. S., Amaro, T., Bergmann, M., Canals, M., Company, J. B., Davies, J., Duineveld, G., Galgani, F., Howell, K. L., Huvenne, V. A. I., Isidro, E., Jones, D. O. B., Lastras, G., Morato, T., Gomes-Pereira, J. N., Purser, A., Stewart, H., Tojeira, I., Tubau, X., Van Rooij, D., & Tyler, P. A. (2014b). Marine litter distribution and density in European seas, from the shelves to deep basins. *PLoS ONE*, *9*(4), e95839. https://doi.org/10.1371/journal.pone.0095839

Phitos, D. (1995). *The red data book of rare and threatened plants of Greece*. Athens, Greece, WWF.

Phoenix, G. K., Emmett, B. A., Britton, A. J., Caporn, S. J. M., Dise, N. B., Helliwell, R., Jones, L., Leake, J. R., Leith, I. D., Sheppard, L. J., Sowerby, A., Pilkington, M. G., Rowe, E. C., Ashmore, M. R., & Power, S. A. (2012). Impacts of atmospheric nitrogen deposition: responses of multiple plant and soil parameters across contrasting ecosystems in long-term field experiments. *Global Change Biology*, *18*(4), 1197–1215. https://doi.org/10.1111/j.1365-2486.2011.02590.x

Piano, E., De Wolf, K., Bona, F., Bonte, D., Bowler, D. E., Isaia, M., Lens, L., Merckx, T., Mertens, D., Van Kerckvoorde, M., De Meester, L., & Hendrickx, F. (2017). Urbanization drives community shifts towards thermophilic and dispersive species at local and landscape scales. *Global Change Biology*, *23*(7), 2554–2564. https://doi.org/10.1111/gcb.13606

Plotnikov, I. S. [Плотников, И. С.]. (2016). Многолетние изменения фауны свободноживущих водных беспозвоночных Аральского моря *[Multiyear changes of fauna of aquatic invertebrates of the Aral Sea]*. Saint-Petersburg, Russian Federation: ЗИН РАН [ZIN RAN].

Pocock M. J. O. & Jennings, N. (2008). Testing biotic indicator taxa: the sensitivity of insectivorous mammals and their prey to the intensification of lowland agriculture. *Journal of Applied Ecology, 45*(1): 151–160. https://doi.org/10.1111/j.1365-2664.2007.01361.x

Poláková, J., Tucker, G., Hart, K., Dwyer, J., & Rayment, M. (2011). *Addressing biodiversity and habitat preservation through measures applied under the Common Agricultural Policy. Report Prepared for DG Agriculture and Rural Development, Contract No. 30-CE0388497/00-44.* London, UK: Institute for European Environmental Policy.

Poloczanska, E. S., Brown, C. J., Sydeman, W. J., Kiessling, W., Schoeman, D. S., Moore, P. J., Brander, K., Bruno, J. F., Buckley, L. B., Burrows, M. T., Duarte, C. M., Halpern, B. S., Holding, J., Kappel, C. V, O'Connor, M. I., Pandolfi, J. M., Parmesan, C., Schwing, F., Thompson, S. A., & Richardson, A. J. (2013). Global imprint of climate change on marine life. *Nature Climate Change*, *3*(10), 919–925. https://doi.org/Doi 10.1038/Nclimate1958

Poloczanska, E. S., Burrows, M. T., Brown, C. J., Molinos, J. G., Halpern, B. S., Hoegh-Guldberg, O., Kappel, C. V., Moore, P. J., Richardson, A. J., Schoeman, D. S., & Sydeman, W. J. (2016). Responses of marine organisms to climate change across oceans. *Frontiers in Marine Science*, *3,* 62. https://doi.org/10.3389/fmars.2016.00062

Ponge, J.-F., Salmon, S., Benoist, A., & Geoffroy, J.-J. (2015). Soil macrofaunal communities are heterogeneous in heathlands with different grazing intensity. *Pedosphere*, *25*(4), 524–533. http://doi.org/10.1016/S1002-0160(15)30033-3

Popova, E. N., & Semenov, S. M. (2013). Current and expected changes in Colorado beetle climatic habitat in Russia and neighboring countries. *Russian Meteorology and Hydrology*, *38*(7), 509–514. https://doi.org/10.3103/S1068373913070108

Portnov, B. A., & Safriel, U. N. (2004). Combating desertification in the Negev: dryland agriculture vs. dryland urbanization. *Journal of Arid Environments*, *56*(4), 659–680. https://doi.org/10.1016/S0140-1963(03)00087-9

Prada, S., Sequeira, M. M. D., Figueira, C., & Oliveira Da Silva, M. (2009). Fog precipitation and rainfall interception in the natural forests of Madeira Island (Portugal). *Agricultural and Forest Meteorology*, (149),

1179–1187. https://doi.org/10.1016/j.agrformet.2009.02.010

Prather, C. M., Pelini, S. L., Laws, A., Rivest, E., Woltz, M., Bloch, C. P., Del Toro, I., Ho, C. K., Kominoski, J., Scott Newbold, T. A., Parsons, S., & Joern, A. (2013). Invertebrates, ecosystem services and climate change. *Biological Reviews*, *88*(2), 327–348. https://doi.org/10.1111/brv.12002

Prilipko, L. I. [Прилипко, Л. И.]. (1970). Растительный покров Азербайджана *[Plant cover of Azerbaijan]*. Baku, Azerbaijan: Elm.

Prishchepov, A. A., Müller, D., Dubinin, M., Baumann, M., & Radeloff, V. C. (2013). Determinants of agricultural land abandonment in post-Soviet European Russia. *Land Use Policy*, *30*(1), 873–884. https://doi.org/10.1016/j.landusepol.2012.06.011

Procházka F. (2000). *Cerný a cervený seznam cévnatých rostlin Ceské republiky [Black and Red lists of Vascular Plants of the Czech Republic]*. Praha, Czech Republic: Príroda.

Program and Action Plan [Программа и план действий]. (2015). Программа и план действий по адаптации к изменению климата сектора "Лес и биоразнообразие" на 2015-2017 гг. *[Program and Action Plan on adaptation to climate change in sector "Forest and Biodiversity" on 2015-2017]*. Bishkek, The Kyrgyz Republic: State Agency on Nature Protection and Forestry under Government of the Kyrgyz Republic.

Prop, J., Aars, J., Bårdsen, B.-J., Hanssen, S. A., Bech, C., Bourgeon, S., de Fouw, J., Gabrielsen, G. W., Lang, J., Noreen, E., Oudman, T., Sittler, B., Stempniewicz, L., Tombre, I., Wolters, E. & Moe, B. (2015). Climate change and the increasing impact of polar bears on bird populations. *Frontiers in Ecology and Evolution*, *3*, 33. https://doi.org/10.3389/fevo.2015.00033

Puig, P., Canals, M., Company, J. B., Martín, J., Amblas, D., Lastras, G., Palanques, A., & Calafat, A. M. (2012). Ploughing the deep sea floor. *Nature, 489*, 286-289. https://doi.org/10.1038/nature11410

Purvis, O. W. (2015). Lichens on Betula in the Ural Mountains; relationships with bark acidity and element concentrations as indicators of geology and anthropogenic influences. *British Lichen Society Bulletin*, *117*, 15–28.

Purvis, O. W., Tittley, I., Chimonides, P. D. J., Bamber, R., Hayes, P. A., James, P. W., Rumsey, F. J., & Read, H. (2010). Long-term biomonitoring of lichen and bryophyte biodiversity at Burnham Beeches SAC and global environmental change. *Systematics and Biodiversity*, *8*(2), 193–208. https://doi.org/10.1080/14772001003782088

Pykälä, J. (2003) Effects of restoration with cattle grazing on plant species composition and richness of semi-natural grasslands. *Biodiversity & Conservation 12,* 2211-2226. https://doi.org/10.1023/A:1024558617080

Pyšek, P., Křivánek, M., & Jarošík, V. (2009). Planting intensity, residence time, and species traits determine invasion success of alien woody species. *Ecology*, *90*(10), 2734–2744. https://doi.org/10.1890/08-0857.1

Quaas, M. F., Reusch, T. B. H., Schmidt, J. O., Tahvonen, O., & Voss, R. (2016). It is the economy, stupid! Projecting the fate of fish populations using ecological-economic modeling. *Global Change Biology*, *22*(1), 264–270. https://doi.org/10.1111/gcb.13060

Quézel, P., & Médail, F. (2003). *Ecologie et biogéographie des forêts du Bassin Méditerranéen [Ecology and biogeography of the forests of the Mediterranean basin].* Paris, France: Elsevier.

Rabalais, N. N., Díaz, R. J., Levin, L. A., Turner, R. E., Gilbert, D., & Zhang, J. (2010). Dynamics and distribution of natural and human-caused hypoxia. *Biogeosciences*, *7*, 585–619. https://doi.org/10.5194/bg-7-585-2010

Rachkovskaya, E. I., & Bragina, T. M. (2012). Steppes of Kazakhstan: Diversity and present state. In M. J. A., Werger & M. A. van Staalduinen (Eds.), *Eurasian steppes. Ecological problems and livelihoods in a changing world* (pp. 103–148). Dordrecht, The Netherlands: Springer. https://doi.org/10.1007/978-94-007-3886-7_3

Radu, G., & Anton, E. (2014). Impact of turbot fishery on cetaceans in the Romanian Black Sea area. *Scientia Marina*, *78*(S1), 103–109. https://doi.org/10.3989/scimar.04029.27A

Rakonczay, Z. (1989). *Red data book on mosses, vascular plants and animals (invertebrates and vertebrates).* Budapest, Hungary: Vörös Könyv. Akadémiai Kiado.

Ram, D., Axelsson, A.-L., Green, M., Smith, H. G., & Lindström, Å. (2017). What drives current population trends in forest birds – forest quantity, quality or climate? A large-scale analysis from northern Europe. *Forest Ecology and Management*, *385*, 177–188. https://doi.org/10.1016/j.foreco.2016.11.013

Ramirez-Llodra, E., Tyler, P. A., Baker, M. C., Bergstad, O. A., Clark, M. R., Escobar, E., Levin, L. A., Menot, L., Rowden, A. A., Smith, C. R., & Van Dover, C. L. (2011). Man and the last great wilderness: Human impact on the deep sea. *PLoS ONE*, *6*(8), e22588. https://doi.org/10.1371/journal.pone.0022588

Ramsar. (2015a). *COP 12 Doc. 11: Regional overview of the implementation of the Convention and its Strategic Plan in Europe.*

Ramsar. (2015b). *COP 12 Doc. 12: Regional overview of the implementation of the Convention and its Strategic Plan in Asia.*

Ramsar. (n.d.). *Ramsar Sites Information Service*. Retrieved from https://rsis.ramsar.org/

Randi, E. (2008). Detecting hybridization between wild species and their domesticated relatives. *Molecular Ecology*, *17*(1), 285–293. https://doi.org/10.1111/j.1365-294X.2007.03417.x

Randin, C. F., Engler, R., Normand, S., Zappa, M., Zimmermann, N. E., Pearman, P. B., Vittoz, P., Thuiller, W., & Guisan, A. (2009). Climate change and plant distribution: Local models predict high-elevation persistence. *Global Change Biology*, *15*(6), 1557–1569. https://doi.org/10.1111/j.1365-2486.2008.01766.x

Randlane, T., Juriado, I., Suija, A., Lohmus, P., & Leppik, E. (2008).

Lichens in the new red list of Estonia. *Folia Cryptogamica Estonica, 44,* 113–120.

Rangwala, I., Sinsky, E., & Miller, J. R. (2013). Amplified warming projections for high altitude regions of the northern hemisphere mid-latitudes from CMIP5 models. *Environmental Research Letters*, *8*(2), 24040. https://doi.org/10.1088/1748-9326/8/2/024040

Rassi, P., Hyvärinen, E., Juslén, A., & Mannerkoski, I. (2010). *The 2010 red list of Finnish species.* Helsinki, Finland: Ympäristöministeriö & Suomen ympäristökeskus.

Raybaud, V., Beaugrand, G., Dewarumez, J. M., & Luczak, C. (2014). Climate-induced range shifts of the American jackknife clam *Ensis directus* in Europe. *Biological Invasions*, *17*(2), 725–741. https://doi.org/10.1007/s10530-014-0764-4

Raynolds, M. K., Walker, D. A., & Maier, H. A. (2006). NDVI patterns and phytomass distribution in the circumpolar Arctic. *Remote Sensing of Environment*, *102*(3–4), 271–281. http://doi.org/10.1016/j.rse.2006.02.016

Reading, C. J., Luiselli, L.M., Akani, G.C., Bonnet, X., Amori, G., Ballouard, J.M., Filippi, E., Naulleau, G., Pearson, D., & Rugiero, L. (2010). Are snake populations in widespread decline? *Biology Letters*, *6*, 777–780. http://doi.org/10.1098/rsbl.2010.0373

Reif, J., Voříšek, P., Šťastný, K., Bejček, V., & Petr, J. (2008). Agricultural intensification and farmland birds: New insights from a Central European country. *Ibis*, *150*(3), 596–605. https://doi.org/10.1111/j.1474-919X.2008.00829.x

Reimers, E., Miller, F. L., Eftestøl, S., Colman, J. E., & Dahle, B. (2006). Flight by feral reindeer *Rangifer tarandus tarandus* in response to a directly approaching human on foot or on skis. *Wildlife Biology, 12*(4), 403-413. https://doi.org/10.2981/0909-6396(2006)12[403:FBFRRT]2.0.CO;2

Reinecke, J., Klemm, G., & Heinken, T. (2014). Vegetation change and homogenization of species composition in temperate nutrient deficient Scots pine forests after 45 yr. *Journal of Vegetation*

Science, *25*(1), 113–121. https://doi.
org/10.1111/jvs.12069

**Renaud, P. E., Sejr, M. K., Bluhm, B.
A., Sirenko, B., & Ellingsen, I. H.** (2015).
The future of Arctic benthos: Expansion,
invasion, and biodiversity. *Progress in
Oceanography*, *139*, 244–257. https://doi.
org/10.1016/j.pocean.2015.07.007

Rhymer, J. M., & Simberloff, D. (1996).
Extinction by hybridization and introgression.
Annual Review of Ecology and Systematics,
27(1), 83–109. https://doi.org/10.1146/
annurev.ecolsys.27.1.83

**Rice, J., Arvanitidis, C., Boicenco,
L., Kasapidis, P., Mahon, R., Malone,
T., Montevecchi, W., Coll Monton,
M., Moretzsohn, F., Ouellet, P.,
Oxenford, H., Smith, T., Tunnell, J.
W., Vanaverbeke, J., & Van Gaever, S.**
(2016). Chapter 36A. North Atlantic
Ocean. In *The first global integrated marine
assessment - World ocean assessment I*.
New York, USA: United Nations.

Richardson, D. M., & Pysek, P.
(2006). Plant invasions: merging the
concepts of species invasiveness and
community invasibility. *Progress in Physical
Geography*, *30*(3), 409–431. https://doi.
org/10.1191/0309133306pp490pr

**Richman, N. I., Böhm, M., Adams, S. B.,
Alvarez, F., Bergey, E. A., Bunn, J. J. S.,
Burnham, Q., Cordeiro, J., Coughran,
J., Crandall, K. A., Dawkins, K. L.,
DiStefano, R. J., Doran, N. E., Edsman,
L., Eversole, A. G., Füreder, L, Furse, J.
M., Gherardi, F., Hamr, P., Holdich, D.
M., Horwitz, P., Johnston, K., Jones, C.
M., Jones, J. P. G., Jones, R. L., Jones,
T. G., Kawai, T., Lawler, S., López-Mejıa,
M., Miller, R. M., Pedraza-Lara, C.,
Reynolds, J. D., Richardson, A. M. M.,
Schultz, M. B., Schuster, G. A., Sibley,
P. J., Souty-Grosset, C., Taylor, C. A,
Thoma, R. F., Walls, J., Walsh, T. S., &
Collen, B.** (2015). Multiple drivers of decline
in the global status of freshwater crayfish
(Decapoda: Astacidea). *Philosophical
Transactions of the Royal Society B:
Biological Sciences, 370,* 20140060. http://
dx.doi.org/10.1098/rstb.2014.0060

**Richner, N., Holderegger, R., Linder,
H. P., & Walter, T.** (2015). Reviewing
change in the arable flora of Europe: A

meta-analysis. *Weed Research*, *55*(1),
1–13. https://doi.org/10.1111/wre.12123

**Rigling, D., Hilfiker, S., Schöbel,
C., Meier, F., Engesser, R.,
Scheidegger, C., … Queloz, V.**
(2016). Das Eschentriebsterben
Biologie, Krankheitssymptome und
Handlungsempfehlungen [Ash die-
back biology, disease symptoms and
recommendations for action]. *Merkblatt Für
Die Praxis [Fact Sheet for Practice]*, *57*, 1–8.
Retrieved from www.wsl.ch/publikationen

Rintelen, T., & Van Damme, D. (2011).
Dreissena caspia. The IUCN Red List
of Threatened Species. http://dx.doi.
org/10.2305/IUCN.UK.2011-2.RLTS.
T188971A8669278.en

**Roberge, J. M., Lämås, T., Lundmark,
T., Ranius, T., Felton, A., & Nordin, A.**
(2015). Relative contributions of set-
asides and tree retention to the long-
term availability of key forest biodiversity
structures at the landscape scale. *Journal
of Environmental Management*, *154*,
284–292. https://doi.org/10.1016/j.
jenvman.2015.02.040

Robinson, R., & Sutherland, W. (2002).
Post-war changes in arable farming and
biodiversity in Great Britain. *Journal of
Applied Ecology*, *39,* 157–176. https://doi.
org/10.1046/j.1365-2664.2002.00695.x

**Rodrigues, A. S. L., Brooks, T. M.,
Butchart, S. H. M., Chanson, J., Cox,
N., Hoffmann, M., & Stuart, S. N.**
(2014). Spatially explicit trends in the
global conservation status of vertebrates.
PLoS ONE 9(11), e113934. https://doi.
org/10.1371/journal.pone.0113934

**Rogers, A. D., Brierley, A., Croot, P.,
Cunha, M. R., Danovaro, R., Devey, C.,
Hoel, A. H., Ruhl., H. A., Sarradin, P-M.,
Trevisanut, S., van den Hove, S., Vieira,
H., & Visbeck, M.** (2015) *Delving deeper:
critical challenges for 21st century deep-sea
research.* K. E. Larkin, K. Donaldson, &
N. McDonough. (Eds.). Ostend, Belgium:
European Marine Board.

**Rogers, A. D., Tyler, P. A., Connelly,
D. P., Copley, J. T., James, R., Larter,
R. D., Linse, K., Mills, R. A., Garabato,
A. N., Pancost, R. D., Pearce, D. A.,
Polunin, N. V. C., German, C. R., Shank,
T., Boersch-Supan, P. H., Alker, B. J.,**

**Aquilina, A., Bennett, S. A., Clarke, A.,
Dinley, R. J. J., Graham, A. G. C., Green,
D. R. H., Hawkes, J. A., Hepburn, L.,
Hilario, A., Huvenne, V. A. I., Marsh,
L., Ramirez-Llodra, E., Reid, W. D. K.,
Roterman, C. N., Sweeting, C. J., Thatje,
Sven, & Zwirglmaier, K.** (2012). The
discovery of new deep-sea hydrothermal
vent communities in the Southern Ocean
and implications for biogeography. *PLoS
Biology*, *10*(1), e1001234. https://doi.
org/10.1371/journal.pbio.1001234

**Roll, U., Feldman, A., Novosolov, M.,
Allison, A., Bauer, A.M., Bernard, R.,
Böhm, M., Castro-Herrera, F., Chirio, L.,
Collen, B., Colli, G.R., Dabool, L., Das, I.,
Doan, T.M., Grismer, L.L., Hoogmoed,
M., Itescu, Y., Kraus, F., LeBreton, M.,
Lewin, A., Martins, M., Maza, E., Meirte,
D., Nagy, Z.T., Nogueira, C.C., Pauwels,
O.S.G., Pincheira-Donoso, D., Powney,
G., Sindaco, R., Tallowin, O., Torres-
Carvajal, O., Trape, J.-F., Vidan, E., Uetz,
P., Wagner, P., Wang, Y., Orme, C.D.L.,
Grenyer, R., & Meiri, S.** (2017). The global
distribution of tetrapods reveals a need for
targeted reptile conservation. *Nature Ecology
& Evolution 1,* 1677-1682. https://doi.
org/10.1038/s41559-017-0332-2

Rondinini, C., & Visconti, P. (2015).
Scenarios of large mammal loss in Europe
for the 21st century. *Conservation Biology*,
29(4), 1028-1036. https://doi.org/10.1111/
cobi.12532

**Rönkkönen, S., Ojaveer, E., Raid, T., &
Viitasalo, M.** (2004). Long-term changes in
Baltic herring (*Clupea harengus membras*)
growth in the Gulf of Finland. *Canadian
Journal of Fisheries and Aquatic Sciences*,
61(2), 219–229. https://doi.org/10.1139/
f03-167

**Roques, A., Rabitsch, W., Rasplus,
J., Lopez-Vaamonde, C., Nentwig,
W., & Kenis, M.** (2009). Alien
terrestrial invertebrates of Europe. In
DAISIE, *Handbook of alien species in
Europe* (pp. 63–79). Dordrecht, The
Netherlands: Springer.

**Rosa García, R., Fraser, M. D.,
Celaya, R., Ferreira, L. M. M., García,
U., & Osoro, K.** (2013). Grazing land
management and biodiversity in the
Atlantic European heathlands: a review.
Agroforestry Systems, *87*(1), 19–43. http://
doi.org/10.1007/s10457-012-9519-3

Rosen Michel, T., & Röttger, C. (2014). *Central Asian mammals initiative: Saving the last migrations*. D. Mallon, S. Michel, R. Vagg, & P. Zahler (Eds.). Bonn, Germany: UNEP / CMS Secretariat.

Roshydromet [Росгидромет]. (2014). *Second assessment report on climate change and its consequences in Russian Federation.* [Второй оценочный доклад Росгидромета об изменениях климата и их последствиях на территории Российской Федерации]. Retrieved from http://downloads.igce.ru/publications/OD_2_2014/v2014/htm/1.htm

Rosstat [Росстат]. (2015). Сельское хозяйство, охота и охотничье хозяйство, лесоводство в России. Стат.сб./Росстат. Москва. 201 с. [*Agriculture, hunting and wildlife management, and forestry in Russia. Statistical Compendium, Moscow. 201 pp.*] Retrieved from http://www.gks.ru/free_doc/doc_2015/selhoz15.pdf

Roué, M., & Molnár, Z. (Eds.). (2017). *Knowing our lands and resources: Indigenous and local knowledge of biodiversity and ecosystem services in Europe and Central Asia*. Paris, France: UNESCO.

Rundlöf, M., & Smith, H. G. (2006). The effect of organic farming on butterfly diversity depends on landscape context. *Journal of Applied Ecology*, *43*(6), 1121–1127. http://doi.org/10.1111/j.1365-2664.2006.01233.x

Ruiz-Labourdette, D., Nogués-Bravo, D., Ollero, H. S., Schmitz, M. F., & Pineda, F. D. (2012). Forest composition in Mediterra nean mountains is projected to shift along the entire elevational gradient under climate change. *Journal of Biogeography*, *39*(1), 162–176. https://doi.org/10.1111/j.1365-2699.2011.02592.x

Russell, D. J. F., Wanless, S., Collingham, Y. C., Huntley, B., & Hamer, K. C. (2015). Predicting future European breeding distributions of British seabird species under climate change and unlimited/no dispersal scenarios. *Diversity*, *7*(4), 342–359. https://doi.org/10.3390/d7040342

Russian Academy of Sciences. (2008). Фундаментальные и прикладные проблемы ботаники в начале XXI века: Материалы всероссийской конференции (Петрозаводск, 22–27 сентября 2008 г.). Часть 2: Альгология. Микология. Лихенология. Бриология. [Fundamental and applied problems of botany at the beginning of the XXI century: Materials of all-Russian conference (Petrozavodsk, 22-27 September 2008) Part 2: Algology. Mycology. Lichenology. Bryology]. Petrozavodsk, Russian Federation: Karelian research centre of the Russian Academy of Sciences.

Russian Forest Protection Centre [Российского центра защиты леса]. (n.d.). Огнёвка самшитовая - 3 года на Российском Кавказе [The box tree moth (*Cydalima perspectalis*) - 3 years in the Russian Caucasus]. Retrieved December 8, 2016, from http://rcfh.ru/23_09_2014_de743.html

Russo, D. (2007). *The effects of land abandonment on animal species in Europe: conservation and management implications.*

Rydin, H., & Jeglum, J. K. (2013). *The biology of peatlands* (2nd edition). Oxford, UK: Oxford University Press.

Sabovljevit, M., Ganeva, A., Tsakiri, E., & Stefanut, S. (2001). Bryology and bryophyte protection in south-eastern Europe. *Biological Conservation*, *101*(1), 73–84. https://doi.org/10.1016/S0006-3207(01)00043-X

Sadygov, T. N. [Садыгов, Т. Н.]. (2012). Концепция возобновления Прикуринских тугайных лесов [The concept of Pikulinski tugai forests restoration]. Научные Труды АГАУ [*Scientific Proceedings of ASAU]*, *2*, 56–59.

Safarov, I. S. [Сафаров, И. С.]. (1979). Субтропические леса Талыша [*Subtropical forests of Tallish]*. Baku, Azerbaijan: Elm.

Safarov, H. (2009). Rare and endangered plant species in Hirkan National Park and its environs. In N. Zazanashvili & D. Mallon (Eds.), *Status and protection of globally threatened species in the Caucasus* (pp. 193-198). Tbilisi, Georgia: CEPF, WWF.

Sahlén, G., Bernard, R., A., C. R., Ketelaar, R., & Suhling, F. (2004). Critical species of Odonata in Europe. *International Journal of Odonatology*, *7*, 385–398. https://doi.org/10.1080/13887890.2004.9748223

Sala, E., Kizilkaya, Z., Yildirim, D., & Ballesteros, E. (2011). Alien marine fishes deplete algal biomass in the Eastern Mediterranean. *PLoS ONE*, *6*(2), 1–5. https://doi.org/10.1371/journal.pone.0017356

Saltankin, V. P. (2012). Russian water reservoirs. In: L. Bengtsson, R. W. Herschy, & R. W. Fairbridge (Eds.), *Encyclopedia of lakes and reservoirs. Encyclopedia of earth sciences series* (pp. 691–697). Dordrecht, The Netherlands: Springer. https://doi.org/10.1007/978-1-4020-4410-6_250

Saltré, F., Duputié, A., Gaucherel, C., & Chuine, I. (2015). How climate, migration ability and habitat fragmentation affect the projected future distribution of European beech. *Global Change Biology*, *21*(2), 897–910. https://doi.org/10.1111/gcb.12771

Sandrea, R., & Sandrea, I. (2010). Deepwater crude oil output: How large will the uptick be? *Oil and Gas Journal*, *108*(41), 48–53. Retrieved from https://www.ogj.com/articles/print/volume-108/issue-41/exploration-development/deepwater-crude-oil-output-how-large.html

Santini, A., Ghelardini, L., De Pace, C., Desprez-Loustau, M. L., Capretti, P., Chandelier, A., Cech, T., Chira, D., Diamandis, S., Gaitniekis, T., Hantula, J., Holdenrieder, O., Jankovsky, L., Jung, T., Jurc, D., Kirisits, T., Kunca, A., Lygis, V., Malecka, M., Marcais, B., Schmitz, S., Schumacher, J., Solheim, H., Solla, A., Szabò, I., Tsopelas, P., Vannini, A., Vettraino, A. M., Webber, J., Woodward, S., & Stenlid, J. (2013). Biogeographical patterns and determinants of invasion by forest pathogens in Europe. *New Phytologist*, *197*(1), 238–250. https://doi.org/10.1111/j.1469-8137.2012.04364.x

Sauer, J., Domisch, S., Nowak, C., & Haase, P. (2011). Low mountain ranges: summit traps for montane freshwater species under climate change. *Biodiversity and Conservation*, *20*, 3133–3146. https://doi.org/10.1007/s10531-011-0140-y

Savvaitova, K. A., & Petr, T. (1999). Fish and fisheries in Lake Issyk-Kul (Tien Shan), River Chu and Pamir Lakes. In T. Petr (Ed.), *Fish and Fisheries at Higher Altitudes: Asia* (pp. 168–187). Rome, Italy: FAO.

Schatz, B. & Dounias, E. (2016). Pollination: Threats and opportunities in European beekeeping. In S. Thiébault & J.-P. Moatti (Eds.), *The Mediterranean region under climate change: a scientific update* (*COP 22 UN FCCC, Marrakech* (*MAR*), *2016/11/7-18*)*, pp. 551-558*. Marseille, France: IRD

Scheidegger, C., Bilovitz, P. O., Werth, S., Widmer, I., & Mayrhofer, H. (2012). Hitchhiking with forests: Population genetics of the epiphytic lichen *Lobaria pulmonaria* in primeval and managed forests in southeastern Europe. *Ecology and Evolution*, 2(9), 2223–2240. https://doi.org/10.1002/ece3.341

Scheidegger, C., & Clerc, P. (2002). *Rote Liste der gefährdeten Arten der Schweiz: Baum- und erdbewohnende Flechten [Red list of endangered species in Switzerland: Tree and Erdbewohnende lichen]*. Bern, Switzerland: Bundesamt für Umwelt, Wald und Landschaft BUWAL.

Scheidegger, C., & Werth, S. (2009). Conservation strategies for lichens: insights from population biology. *Fungal Biology Reviews*, 23(3), 55–66. https://doi.org/10.1016/j.fbr.2009.10.003

Scherrer, D., & Körner, C. (2011). Topographically controlled thermal-habitat differentiation buffers alpine plant diversity against climate warming. *Journal of Biogeography*, 38(2), 406–416. https://doi.org/10.1111/j.1365-2699.2010.02407.x

Schmid B. (2002). The species richness-productivity controversy. *Trends in Ecology and Evolution, 17*(3), 113–114. http://doi.org/10.1016/S0169-5347(01)02422-3

Schmidt, N. M., Ims, R. A., Høye, T. T., Gilg, O., Hansen, L. H., Hansen, J., Lund, M., Fuglei, E., Forchhammer, M. C., & Sittler, B. (2012). Response of arctic predator guilds to collapsing lemming cycles. *Proceedings of the Royal Society B: Biological Sciences*, 279(1746), 4417–4422. https://doi.org/10.1098/rspb.2012.1490

Schneider-Binder, E. (2007). Xerophilous and xero-mesophilous grasslands on slumping hills around the Saxon villages Apold and Saschiz (Transylvania, Romania). *Transylvanian Review of Systematical and Ecological Research*, 4, 55–64. Retrieved from https://search.proquest.com/openview/00559c262e577f36a9987e28a286449e/1?pq-origsite=gscholar&cbl=54813

Schneider, M. K., Lüscher, G., Jeanneret, P., Arndorfer, M., Ammari, Y., Bailey, D., Balázs, K., Báldi, A., Choisis, J.-P., Dennis, P., Eiter, S., Fjellstad, W., Fraser, M. D., Frank, T., Friedel, J. K., Garchi, S., Geijzendorffer, I. R., Gomiero, T., Gonzalez-Bornay, G., Hector, A., Jerkovich, G., Jongman, R. H. G., Kakudidi, E., Kainz, M., Kovács-Hostyánszki, A., Moreno, G., Nkwiine, C., Opio, J., Oschatz, M.-L., Paoletti, M. G., Pointereau, P., Pulido, F. J., Sarthou, J., Siebrecht, N., Sommaggio, D., Turnbull, L. A, Wolfrum, S., & Herzog, F. (2014). Gains to species diversity in organically farmed fields are not propagated at the farm level. *Nature Communications, 5,* 4151. http://doi.org/10.1038/ncomms5151

Scholefield, P., Firbank, L., Butler, S., Norris, K., Jones, L. M., & Petit, S. (2011). Modelling the European farmland bird indicator in response to forecast land-use change in Europe. *Ecological Indicators*, 11(1), 46–51. https://doi.org/10.1016/j.ecolind.2009.09.008

Schwarz, U. (2012). *Hydropower projects in protected areas on the Balkans.*

Schwörer, C., Henne, P. D., & Tinner, W. (2014). A model-data comparison of Holocene timberline changes in the Swiss Alps reveals past and future drivers of mountain forest dynamics. *Global Change Biology*, 20(5), 1512–1526. https://doi.org/10.1111/gcb.12456

Science for Environment Policy. (2015). *Ecosystem services and biodiversity. In-depth report 11.* Bristol, UK: UWE. http://ec.europa.eu/science-environment-policy

Sciberras, M., Jenkins, S. R., Mant, R., Kaiser, M. J., Hawkins, S. J., & Pullin, A. S. (2015). Evaluating the relative conservation value of fully and partially protected marine areas. *Fish and Fisheries*, 16(1), 58–77. https://doi.org/10.1111/faf.12044

Seaward, M. (2008). Environmental role of lichens. In T. H. Nash (Ed.), *Lichen Biology, 2nd edition* (pp. 274–298). Cambridge, UK: Cambridge University Press.

Šebesta, J., Šamonil, P., Lacina, J., Oulehle, F., Houška, J., & Buček, A. (2011). Acidification of primeval forests in the Ukraine Carpathians: Vegetation and soil changes over six decades. *Forest Ecology and Management*, 262(7), 1265–1279. https://doi.org/10.1016/j.foreco.2011.06.024

Sedelnikova, N. V. (1988). Lichens. In *Red Book of Novosibirsk Oblast: Plants* (pp. 123–129). Novosibirsk, USSR: Nauka.

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., Kartesz, J., Kenis, M., Kreft, H., Kühn, I., Lenzner, B., Liebhold, A., Mosena, A., Moser, D., Nishino, M., Pearman, D., Pergl, J., Rabitsch, W., Rojas-Sandoval, J., Roques, A., Rorke, S., Rossinelli, S., Roy, H. E., Scalera, R., Schindler, S., Štajerová, K., Tokarska-Guzik, B., van Kleunen, M., Walker, K., Weigelt, P., Yamanaka, T., & Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications*, 8, 14435. https://doi.org/10.1038/ncomms14435

Šefferová Stanová, V., Šeffer, J., & Janák, M. (2008). *Management of Natura 2000 habitats. Alkaline fens.*

Segerstråle, S. G. (1957). Baltic sea. *Geological Society of America Memoirs*, 67, 751–800.

Şekercioğlu, C. H., Anderson, S., Akçay, E., Bilgin, R., EmreCanf, Ö., Semiz, G., Tavşanoğlu, Ç., BakiYokeş, M., Soyumert, A., İpekdal, K., Sağlam, I. K., Yücel, M., & Nüzhet Dalfes, J. (2011). Turkey's globally important biodiversity in crisis. *Biological Conservation*, 144(12), 2752–2769. https://doi.org/10.1016/j.biocon.2011.06.025

Selvi, F., Carrari, E., & Coppi, A. (2016). Impact of pine invasion on the taxonomic and phylogenetic diversity of a relict Mediterranean forest ecosystem. *Forest Ecology and Management*, 367, 1–11. http://doi.org/10.1016/j.foreco.2016.02.013

Senn-irlet, B., Heilmann-Clausen, J., Genney, D., & Dahlberg, A. (2007). Guidance for Conservation of Macrofungi in Europe. Retrieved from https://www.wsl.ch/eccf/Guidance_Fungi.pdf

Serusiaux, E. (1989). *Liste rouge des Macrilichens dans la communaute Europeenne [Red list of Macrilichens in the European Community]*. Liège, Belgium: Université de Liège. Retrieved from http://hdl.handle.net/2268/138066

Sheldon, R. D., Kamp, J., Koshkin, M., Urazaliev, R., Iskakov, T., Field, R., Salemgareev, A. R., Khrokov, V. V., Zhuly, V. A. Skylarenko, S. L., & Donald, P. F. (2013). Breeding ecology of the globally threatened sociable lapwing *Vanellus gregarius* and the demographic drivers of recent declines. *Journal of Ornithology*, *154*, 501–516. https://doi.org/10.1007/s10336-012-0921-4

Shelyak-Sosonka, Y. R. (1996). *Chervona Kniga Ukrainy - Roslinniy Svit [Red Data Book of Ukraine - Plant Kingdom]*. Kiev, Ukraine: Ukrainskaya Enciklopedia.

Shevchenko, V. V., & Datsky, A. V. (2014). Биоэкономика использования промысловых ресурсов минтая Северной Пацифики *[Bioeconomics of Alaska Pollack fishery resources using in North Pacific]*. Moscow, Russian Federation: Publishing House VNIRO.

Shiganova, T. A. [Шиганова, Т. А.]. (2000). Некоторые итоги изучения биологии вселенца M. leidyi (A. Agassiz) в Черном море [Some results of studying the biology of the invader *M. leidyi* (A. Agassiz) in the Black Sea]. In S. P. Volovik (Ed.), Гребневик *Mnemiopsis leidyi* (*A. Agassiz*) в Азовском и Черном морях и последствия его вселения *[Ctenophore Mnemiopsis leidyi* (*A. Agassiz*) *in the Azov and Black seas and the consequences of its invasion]* (pp. 33–75). Rostov-on-Don, Russian Federation: БКИ [BKI].

Shiganova, T. A., Bulgakova, Y. V., Sorokin, P. Y., & Lukashev, Y. F. [Шиганова, Т. А., Булгакова, Ю. В., Сорокин, П. Ю., & Лукашев, Ю. Ф.]. (2000). Результаты исследований нового вселенца Beroe ovata в Черном море [Investigation of a new settler *Beroe ovata* in the Black Sea]. Известия Академии Наук, Серия Биологическая [Proceedings of the Academy of Sciences, Biological Series], 2, 247–256.

Shore, R. F., Fletcher, M. R., & Walker, L. A. (2003). Agricultural pesticides and mammals in Britain. In: F. Tattersall & W. Manley (Eds.), *Conservation and conflict. Mammals and farming in* Britain (pp. 37–50). Otley, UK: Westbury Publishing.

Shukurov, E. Dj. [Шукуров, Э. Дж.]. (2007). Сочинения *[Compositions]*. Bishkek, Kyrgyzstan: БИОМ [BIOM]. Retrieved from https://s3.eu-central-1.amazonaws.com/biom/lib/book/shukurov_compositions.pdf

Shukurov, E. Dj. [Шукуров, Э. Дж.]. (2009). Природные основы устойчивого развития Кыргызстана *[Natural basis of sustainable development in Kyrgyzstan]*. Bishkek, Kyrgyzstan: Regional Expert Centre on education for sustainable development.

Shukurov, E. Dj, Mitropolsky, O. V., Talskykh, V. N., Zhodubaeva, L. Y., & Shevchenko, V. V. [Шукуров, Э. Дж., Митропольский, О. В., Тальских, В. Н., Жолдубаева, Л. Ы., & Шевченко, В. В.]. (2005). Атлас биологического разнообразия Западного-Тянь-Шаня *[Atlas of biological diversity of western Tien Shan]*. Bishkek, Kyrgyzstan: ОсОО «Бисмарк» [LLC "Bismark"]. Retrieved from https://s3.eu-central-1.amazonaws.com/biom/lib/book/atlas_biodiv_west_tian_shan.pdf

Shukurov, E. Dj., Korotenko, V. A., Kirilenko, V. A., Vashneva, N. S., & Domashov, I. A. [Шукуров, Э. Дж., Коротенко, В. А., Кириленко, В. А., Вашнева, Н. С., & Домашов, И. А.]. (2015). Экологическая безопасность в контексте устойчивого развития Кыргызстана [Environmental safety of Kyrgyzstan in the context of sustainable development]. Bishkek, Kyrgyzstan: БИОМ [BIOM].

Shukurov, E. Dj. [Шукуров, Э. Дж.]. (2016). Зоогеография Кыргызстана *[Zoogeography of Kyrgyzstan]*. Bishkek, Kyrgyzstan: БИОМ [BIOM]. Retrieved from https://s3.eu-central-1.amazonaws.com/biom/work/pub/zoogeo.pdf

Shuman, J. K., & Shugart, H. H. (2009). Evaluating the sensitivity of Eurasian forest biomass to climate change using a dynamic vegetation model. *Environmental Research Letters*, *4*(4), 45024. https://doi.org/10.1088/1748-9326/4/4/045024

Shuntov, V. P., & Temnykh, O. S. [Шунтов, В. П., & Темных, О. С.]. (2013). Иллюзии и реалии экосистемного подхода к изучению и управлению морскими и океаническими биологическими ресурсами [Illusions and realities of the ecosystem approach to the study and management of marine and oceanic biological resources]. Известия ТИНРО *[Izvestiya TINRO]*, *173*, 3–29.

Shurin, J. B., Winder, M., Adrian, R., Keller, W. B., Matthews, B., Paterson, A. M., Paterson, M. J., Pinel-Alloul, B., Rusak, J. A., & Yan, N. D. (2010). Environmental stability and lake zooplankton diversity - contrasting effects of chemical and thermal variability. *Ecology Letters*, *13*(4), 453–463. https://doi.org/10.1111/j.1461-0248.2009.01438.x

Shustov, M. V. [Шустов, М. В.]. (2015). Лишайники в Красных книгах Ульяновской и Самарской областей [Red List of lichens in the Uliyanovsk and the Samara region]. Известия Самарского научного центра Российской академии наук *[News of Samara Scientific Center of the Russian Academy of Sciences]. 17*(6), 322-325.

Sieber, A., Uvarov N. V., Baskin, L. M., Radeloff, V. C., Bateman, B. L., Pankov, A. B., Kuemmerle, T. (2015). Post-Soviet land-use change effects on large mammals' habitat in European Russia. *Biological Conservation 191*, 567–576. http://dx.doi.org/10.1016/j.biocon.2015.07.041

Siedentop, S., & Fina, S. (2012). Who sprawls most? Exploring the patterns of urban growth across 26 European countries. *Environment and Planning A: Economy and Space*, *44*(11), 2765–2784. https://doi.org/10.1068/a4580

Silic, C. (1996). *Spisak biljnih vrsta (Pteridohpyta i Spermatophyta) za crvenu knjigu Bosne i Hercegovine [List of plant species (Pteridohpyta and Spermatophyta) for the red book of Bosnia and Herzegovina]*.

Sillero, N., Campos, J., Bonardi, A., Corti, C., Creemers, R., Crochet, P.-A., Crnobrnja-Isailovic, J., Denoël, M., Ficetola, G.F., Gonçalves, J., Kuzmin, S., Lymerakis, P., de Pous, P., Rodríguez, A., Sindaco, R., Speybroeck, J., Toxopeus, B., Vieites, D.R., & Vences, M. (2014). Updated distribution

and biogeography of amphibians and reptiles of Europe. *Amphibia-Reptilia 35,* 1-31. https://doi.org/10.1163/15685381-00002935

Sirami, C., Nespoulous, A., Cheylan, J. P., Marty, P., Hvenegaard, G. T., Geniez, P., Schatz, B., & Martin, J. L. (2010). Long-term anthropogenic and ecological dynamics of a Mediterranean landscape: Impacts on multiple taxa. *Landscape and Urban Planning*, *96*(4), 214–223. http://doi.org/10.1016/j.landurbplan.2010.03.007

Silva, J., Toland, J., Jones, W., Elridge, J., Thorpe, E., Campbell, M., & O'Hara, E. (2008). *LIFE and endangered plants. Conserving Europe's threatened flora.* Luxembourg: Office for Official Publications of the European Union. https://doi.org/10.2779/99297

Simpson, M., & Prots, B. (2013). Predicting the distribution of invasive plants in the Ukrainian Carpathians under climatic change and intensification of anthropogenic disturbances: implications for biodiversity conservation. *Environmental Conservation*, *40*(2), 167–181. https://doi.org/10.1017/s037689291200032x

Sirenko, B. I., & Gagaev, S. Y. (2007). Unusual abundance of macrobenthos and biological invasions in the Chukchi Sea. *Russian Journal of Marine Biology*, *33*(6), 355–364. https://doi.org/10.1134/S1063074007060016

Sirenko, B. I. [Сиренко, Б. И.]. (2009). The present state of investigations of the Chukchi Sea fauna. In B. I. Sirenko [Б. И. Сиренко] (Ed.), Экосистемы и биоресурсы Чукотского моря и сопредельных акваторий. Исследования фауны морей *[Ecosystems and biological resources of the Chukchi Sea and adjacent areas. Explorations of the Fauna of the Seas]* (pp. 8-31). Saint-Petersburg, Russian Federation: Зоологический институт Российской Академии Наук [Zoological Institute of RAS].

Sirin, A., Minayeva, T., Ilyasov, D., Suvorov, G., Martynenko, V., Fedotov, Yu., Glukhova, T., Valyaeva, N., Tsuganova, O., Maslov, A., Muldashev, A., Shirokikh, P., & Kuznetsov, E. (2016). Peatlands in sub humid regions under changing climate and human activities. In *Proceedings 15th International Peat*

Congress 2016 "Peatlands in Harmony". Kuching, Sarawak, Malaysia, 14-16 August 2016 (pp. 409–413).

Sirin, A., Minayeva, T., Vozbrannaya, A., & Bartalev, S. (2011). How to avoid peat fires? *Science in Russia*, (2), 13–21.

Sirkiä, S., Lindén, A., Helle, P., Nikula, A., Knape, J., & Lindén, H. (2010). Are the declining trends in forest grouse populations due to changes in the forest age structure? A case study of Capercaillie in Finland. *Biological Conservation*, *143*(6), 1540–1548. https://doi.org/10.1016/j.biocon.2010.03.038

Sitzia, T., Semenzato, P., & Trentanovi, G. (2010). Natural reforestation is changing spatial patterns of rural mountain and hill landscapes: A global overview. *Forest Ecology and Management, 259*(8), 1354-1362. https://doi.org/10.1016/j.foreco.2010.01.048

Sket, B. (1999). The nature of biodiversity in hypogean waters and how it is endangered. *Biodiversity and Conservation*, *8*(10), 1319–1338. https://doi.org/doi:10.1023/A:1008916601121

Sket, B. (2012a). Dinaric karst, diversity in. In W. B. White. & D. C. Culver (Eds.), *Encyclopedia of caves. Second edition* (pp. 158–165). Amsterdam, The Netherlands: Elsevier Academic Press.

Sket, B. (2012b). Diversity patterns in Dinaric karst. In W. B. White & D. C. Culver (Eds.), *Encyclopedia of caves. Second edition* (pp. 228–238). Amsterdam, The Netherlands: Elsevier Academic Press.

Sket, B., Paragamian, K., & Trontelj, P. (2004). A census of obligate subterranean fauna of the Balkan Penninsula. In I. Griffiths, B. Kryštufek, & J. M. Reed (Eds.), *Balkan biodiversity: pattern and process in the European hotspot* (pp. 309–323). Dordrecht, The Netherlands: Kluwer Academic.

Skorka, P., Lenda, M., & Tryjanowski, P. (2010). Invasive alien goldenrods negatively affect grassland bird communities in Eastern Europe. *Biological Conservation*, *143*(4), 856–861. https://doi.org/10.1016/j.biocon.2009.12.030

Slingenberg, A., Braat, L., Windt, H. Van Der, Rademaekers, K., Eichler, L., &

Turner, K. (2009). *Study on understanding the causes of biodiversity loss and the policy assessment framework.*

Smale, D. A., Burrows, M. T., Moore, P., O'Connor, N., & Hawkins, S. J. (2013). Threats and knowledge gaps for ecosystem services provided by kelp forests: A northeast Atlantic perspective. *Ecology and Evolution*, *3*(11), 4016–4038. https://doi.org/10.1002/ece3.774

Smaliychuk, A., Müller, D., Prishchepov, A. V., Levers, C., Kruhlov, I., & Kuemmerle, T. (2016). Recultivation of abandoned agricultural lands in Ukraine: Patterns and drivers. *Global Environmental Change*, *38*, 70–81. https://doi.org/10.1016/j.gloenvcha.2016.02.009

Smelansky, I. E. (2003). *Biodiversity of agricultural lands in Russia: current state and trends*. Moscow, Russian Federation: IUCN.

Smelansky, I. E., & Tishkov, A. A. (2012). The steppe biome in Russia: Ecosystem services, conservation status, and actual challenges. In M. J. A. Werger & M. van Staalduinen (Eds.), *Eurasian steppes: ecological problems and livelihoods in a changing world* (pp. 45–101). Dordrecht, The Netherlands: Springer. https://doi.org/10.1007/978-94-007-3886-7

Smelansky, I., & Simonov, E. (2008). Temperate grassland region: Russian steppes. In B. Peart (Ed.), *Compendium of regional templates on the status of temperate grasslands conservation and protection* (pp. 87–99). Vancouver, Canada: IUCN World Commission on Protected Areas.

Smelyansky, I., Elizarov, A., Sobolev, N., & Blagovidov, A. [Смелянский, И., Елизаров, А., Соболев, Н., & Благовидов, А.]. (2006). Стратегия сохранения степей России: позиция неправительственных организаций *[Russian Steppe Conservation Strategy: NGOs position.* Moscow, Russian Federation: Издательство Центра охраны дикой природы [Publishing House of Biodiversity Conservation Center]. Previous "author" was the strategy

Smelansky, I. E., Buivolov, J. A., Bazhenov, Y. A., Bakirova, R. T., Borovik, L. P., Borodin, A. P., Bykova, E. P., Vlasov, A. A., Gavrilenko, V.

S., Goroshko, O. A., Gribkov, A.V., Kirilyuk, V. E., Korsun, O. V., Kreindlin, M. L., Kuksin, G. V., Lysenko, G. N., Polchaninova, N. Yu, Pulyaev, A. I., Ryzhkov, O. V., Ryabinina, Z. N., & Tkachuk T. E. [Смелянский, И. Э., Буйволов, Ю. А., Баженов, Ю. А., Бакирова, Р. Т., Боровик, Л. П., Бородин, А. П., Быкова, Е. П., Власов, А. А., Гавриленко, В. С., Горошко, О. А., Грибков, А. В., Кирилюк, В. Е., Корсун, О. В., Крейндлин, М. Л., Куксин, Г. В., Лысенко, Г. Н., Полчанинова, Н. Ю., Пуляев, А. И., Рыжков, О. В., Рябинина, З. Н., & Ткачук, Т. Е.]. (2015) Степные пожары и управление пожарной ситуацией в степных ООПТ: экологические и природоохранные аспекты. Аналитический обзор *[Steppe fires and fire management in steppe protected areas: Environmental and conservation aspects. Analytical survey]*. Moscow, Russian Federation: Издательство Центра охраны дикой природы [Biodiversity Conservation Center Press].

Smith, G. F., Gittings, T., Wilson, M., French, L., Oxbrough, A., O'Donoghue, S., O'Halloran, S., Kelly, D. L., Mitchell, S. J. G., Kelly, T., Iremonger, S., Mckee, A. M., & Giller, P. (2008). Identifying practical indicators of biodiversity for stand-level management of plantation forests. *Biodiversity and Conservation*, *17*(5), 991–1015. https://doi.org/10.1007/s10531-007-9274-3

Snickars, M., Weigel, B., & Bonsdorff, E. (2015). Impact of eutrophication and climate change on fish and zoobenthos in coastal waters of the Baltic Sea. *Marine Biology*, *162*(1), 141–151. https://doi.org/10.1007/s00227-014-2579-3

Sokolov, V. E. [Соколов, В. Е.]. (1986). Редкие и исчезающие животные. Млекопитающие *[Rare and endangered animals. Mammals]*. Moscow, USSR: Высшая школа [Higher School].

Sokos, C. K., Birtsas, P. K., Connelly, J. W., & Papaspyropoulos, K. G. (2013). Hunting of migratory birds: disturbance intolerant or harvest tolerant? *Wildlife Biology*, *19*(2), 113–125. https://doi.org/10.2981/12-032

Sokratov, S. A., Seliverstov, Y. G., & Shnyparkov, A. L. (2014). Assessment of the economic risk for the ski resorts of

changes in snow cover duration. *Ice and Snow, 54*(3), 100-106.

Soliveres, S., Manning, P., Prati, D., Gossner, M. M., Alt, F., Arndt, H., Baumgartner, V., Binkenstein, J., Birkhofer, K., Blaser, S., Blüthgen, N., Boch, S., Böhm, S., Börschig, C., Buscot, F., Diekötter, T., Heinze, J., Hölzel, N., Jung, K., Klaus, V. H., Klein, A.-M., Kleinebecker, T., Klemmer, S., Krauss, J., Lange, M., Morris, E. K., Müller, J., Oelmann, Y., Overmann, J., Pašalić, E., Renner, S. C., Rillig, M. C., Schaefer, H. M., Schloter, M., Schmitt, B., Schöning, I., Schrumpf, M., Sikorski, J., Socher, S. A., Solly, E. F., Sonnemann, I., Sorkau, E., Steckel, J., Steffan-Dewenter, I., Stempfhuber, B., Tschapka, M., Türke, M., Venter, P., Weiner, C. N., Weisser, W. W., Werner, M., Westphal, C., Wilcke, W., Wolters, V., Wubet, T., Wurst, S., Fischer, M., & Allan, E. (2016a). Locally rare species influence grassland ecosystem multifunctionality. *Philosophical Transactions of the Royal Society B: Biological Sciences, 371*(1694), 20150269. https://doi.org/10.1098/rstb.2015.0269

Soliveres, S., Van der Plas, F., Manning, P., Prati, D., Gossner, M., Renner, S., Alt, F., Arndt, H., Baumgartner, V., Binkenstein, J., Birkhofer, K., Blaser, S., Blüthgen, N., Boch, S., Böhm, S., Börschig, C., Buscot, F., Diekötter, T., Heinze, J., Hölzel, N., Jung, K., Klaus, V.H., Kleinebecker, T., Klemmer, S., Krauss, J., Lange, M., Morris, E.K., Müller, J., Oelmann, Y., Overmann, J., Pašalić, E., Rillig, M.C., Schaefer, H.M., Schloter, M., Schmitt, B., Schöning, I., Schrumpf, M., Sikorski, J., Socher, S.A., Solly, E.F., Sonnemann, I., Sorkau, E., Steckel, J., Steffan-Dewenter, I., Stempfhuber, B., Tschapka, M., Türke, M., Venter, P.C., Weiner, C.N., Weisser, W.W., Werner, M., Westphal, C., Wilcke, W., Wolters, V., Wubet, T., Wurst, S., Fischer, M., Allan, E. (2016b). Biodiversity at multiple trophic levels is needed for ecosystem multifunctionality. *Nature, 536*, 456-459. https://doi.org/10.1038/nature19092

Solomon, J., Shulkina, T., & Schatz, G. E. (2014). *Red list of endemic plants of the Caucasus: Armenia, Azerbaidjan, Georgia, Iran, Russia, and Turkey*. St. Louis, USA: Missouri Botanical Garden Press.

Soós, Á., Papp, L., & Oosterbroek, P. (1992). *Catalogue of palaearctic diptera vol 1.* Budapest, Hungary: Hungarian Natural History Museum.

Sorokin, Y. I. (2002). *Black Sea ecology and oceanography*. Leiden, The Netherlands: Backhuys Publishers.

Sorokovikova, L. M., Popovskaya, G. I., Tomberg, I. V., Sinyukovich, V. N., Kravchenko, O. S., Marinaite, I. I., Bashenkhaeva, N. V., & Khodzher, T. V. (2013). The Selenga River water quality on the border with Mongolia at the beginning of the 21st century. *Russian Meteorology and Hydrology*, *38*(2), 126–133. https://doi.org/10.3103/S1068373913020106

Soto, C. G. (2001). The potential impacts of global climate change on marine protected areas. *Reviews in Fish Biology and Fisheries*, *11*(3), 181–195. https://doi.org/10.1023/a:1020364409616

Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N., Ferdaña, Z. A., Finlayson, M., Halpern, B. S., Jorge, M. A., Lombana, A., Lourie, S. A., Martin, K. D., Mcmanus, E., Molnar, J., Recchia, C. A., & Robertson, J. (2007). Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. *BioScience*, *57*(7), 573. http://doi.org/10.1641/B570707

Sparrius, L. B., & Kooijman, A. M. (2011). Invasiveness of *Campylopus introflexus* in drift sands depends on nitrogen deposition and soil organic matter. *Applied Vegetation Science*, *14*(2), 221–229. https://doi.org/10.1111/j.1654-109X.2010.01120.x

Stanners, D., & Bourdeau, P. (Eds.). (1995). *Europe's Environment: The Dobris Assessment*. Copenhagen, Denmark: European Environment Agency.

Šťastný, K., Červený, J., Řezáč, M., Kurka, A., Veselý, P., Kadlec, T., Konvička, M., Juřičková, L., Harabiš, F., & Marhoul, P. (2015). Prague. In J. Kelcey (Ed.), *Vertebrates and invertebrates of European cities: Selected non-avian fauna* (pp. 379–451). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6

Šťastný, K., Červený, J., Rom, J., Solský, M., Hanel, L., Andreska, J., Vojar, J., & Kerouš, K. (2015). Prague.

In J. Kelcey (Ed.), *Vertebrates and invertebrates of European cities: Selected non-avian fauna* (pp. 119–153). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6

Stelmakh, L. V., & Mansurova, I. M., [Стельмах, Л. В., & Мансурова, И. М.]. (2012). Эколого-физиологические основы биоразнообразия фитопланктона Черного моря [Ecological and physiological basis of phytoplankton biodiversity in the Black Sea]. Экосистемы *[Ecosystems], 7,* 149–158.

Stendera, S., Adrian, R., Bonada, N., Cañedo-Argüelles, M., Hugueny, B., Januschke, K., Pletterbauer, F., & Hering, D. (2012). Drivers and stressors of freshwater biodiversity patterns across different ecosystems and scales: A review. *Hydrobiologia*, *696*(1), 1–28. https://doi.org/10.1007/s10750-012-1183-0

Stenger-Kovács, C., Lengyel, E., Buczkó, K., Tóth, F., Crossetti, L., Pellinger, A., Doma, Z. Z., & Padisák, J. (2014). Vanishing world: alkaline, saline lakes in Central Europe and their diatom assemblages. *Inland Waters*, (4(4)), 383–396. https://doi.org/10.5268/IW-4.4.722

STOA. (2013). *Technology options for feeding 10 billion people - Interactions between climate change & agriculture and between biodiversity & agriculture*. https://doi.org/10.2861/43440

Stoate, C., Báldi, A., Beja, P., Boatman, N. D., Herzon, I., van Doorn, A., de Snoo, G. R., Rakosy, L., & Ramwell, C. (2009). Ecological impacts of early 21st century agricultural change in Europe – A review. *Journal of Environmental Management 91,* 22–46. https://doi.org/10.1016/j.jenvman.2009.07.005

Stoate, C., Boatman, N. D., Borralho, R. J., Carvalho, C. R., de Snoo, G. R., & Eden, P. (2001). Ecological impacts of arable intensification in Europe. *Journal of Environmental Management*, *63*(4), 337–365. https://doi.org/10.1006/jema.2001.0473

Stöcklin, J., Bosshard, A., Klaus, G., Rudmann-Maurer, K., & Fischer, M. (2007). *Landnutzung und biologische Vielfalt in den Alpen – Fakten, Perspektiven, Empfehlungen [Land use and biodiversity in the Alps – facts, perspectives, recommendations]*. Zürich, Switzerland: VDF.

Stofer, S., Bergamini, A., Aragón, G., Carvalho, P., Coppins, B. J., Davey, S., Dietrich, M., Farkas, E., Kärkkäinen, K., Keller, C., Lökös, L., Lommi, S., Máguas, C., Mitchell, R., Pinho, P., Rico, V. J., Truscott, A.-M., Wolseley, P. A., Watt, A., & Scheidegger, C. (2006). Species richness of lichen functional groups in relation to land use intensity. *The Lichenologist*, *38*(4), 331. https://doi.org/10.1017/S0024282906006207

Stokland, J. N., Siitonen, J., & Jonsson, B. G. (2012). *Biodiversity in dead wood*. Cambridge, UK: Cambridge University Press. https://doi.org/10.1017/CBO9781139025843

Stone, R. (2008). A new great lake - or dead sea? *Science*, *320*(5879), 1002–1005. https://doi.org/10.1126/science.320.5879.1002

Strayer, D. L., & Dudgeon, D. (2010). Freshwater biodiversity conservation: recent progress and future challenges. *Journal of the North American Benthological Society*, *29*(1), 344–358. https://doi.org/10.1899/08-171.1

Stuart-Smith, S. J., & Jepson, P. D. (2017). Persistent threats need persistent counteraction: Responding to PCB pollution in marine mammals. *Marine Policy*, *84,* 69–75. https://doi.org/10.1016/j.marpol.2017.06.033

Succow, M., & Uppenbrink, M. (Eds.). (2009). *Potential analysis for further nature conservation in Azerbaijan: A spatial and political investment strategy*. Greifswald, Germany: Michael Succow Foundation. Retrieved from https://books.google.ru/books?id=qR3dlbuBkYgC&printsec=%20frontcover&hl=ru&source=gbs_ge_summary_r&cad=0#v=onepage&q&f=false

Sugar, I. (1994). *Crvena knjiga biljnih vrsta Republike Hrvatske [Red list of plant species of Croatia]*. Zagreb, Croatia: Ministarstvo graditeljstva i zastite okolisa, Zavod za zastitu prirode.

Sutton, M. A., Mason, K. E., Sheppard, L. J., Sverdrup, H., Heuber, R., & Hicks, W. K. (2014). *Nitrogen Deposition, Critical Loads and Biodiversity*. Dordrecht, The Netherlands: Springer.

Svendsen, L. M., Pyhälä, M., Gustafsson, B., Sonesten, L., & Knuuttila, S. (2015). *Inputs of nitrogen and phosphorus to the Baltic Sea. HELCOM core indicator report*.

Svetasheva, T. (2017). Fungal conservation in Russia: an update. In *Fungal Conservation in a Changing Europe*.

Svetasheva, T. 2015. (2015). *Armillaria ectypa*. The IUCN Red List of Threatened Species. http://www.iucnredlist.org/details/75097245/0

Synnes, M. (2007). Bioprospecting of organisms from the deep sea: scientific and environmental aspects. *Clean Technologies and Environmental Policy*, *9*(1), 53–59. https://doi.org/10.1007/s10098-006-0062-7

Szabó, P. (2013). The end of common uses and traditional management in a Central European wood. In I. D. Rotherham (Ed.), *Cultural severance and the environment: The ending of traditional and customary practice on commons and landscapes managed in common* (pp. 205–213). Dordrecht, The Netherlands: Springer. https://doi.org/10.1007/978-94-007-6159-9

Takala, T., Kouki, J., & Tahvanainen, T. (2014). Bryophytes and their microhabitats in coniferous forest pastures: should they be considered in the pasture management? *Biodiversity and Conservation*, *23*(12), 3127–3142. https://doi.org/10.1007/s10531-014-0769-4

Takala, T., Tahvanainen, T., & Kouki, J. (2012). Can re-establishment of cattle grazing restore bryophyte diversity in abandoned mesic semi-natural grasslands? *Biodiversity and Conservation*, *21*(4), 981–992. https://doi.org/10.1007/s10531-012-0234-1

Takhtadzhyan, A. L. [Тахтаджян, А. Л.]. (1978). Флористические области земли *[The floristic regions of the world]*. Leningrad, USSR: Nauka.

Taskavak, E., Atatür, M.K., Ghaffari, H., Meylan, P.A. (2016). *Rafetus euphraticus* (Daudin 1801) – Euphrates softshell turtle.

In A. G. J. Rhodin, J. B. Iverson, P. P. van Dijk, R A. Saumure, K A. Buhlmann, P. C. H. Pritchard, & R. A. Mittermeier (Eds.), *Conservation biology of freshwater turtles and tortoises: A compilation project of the IUCN/SSC tortoise and freshwater turtle specialist group.*

Taubmann, J., Theissinger, K., Feldheim, K. A., Laube, I., Graf, W., Haase, P., Johannesen, J., & Pauls, S. U. (2011). Modelling range shifts and assessing genetic diversity distribution of the montane aquatic mayfly *Ameletus inopinatus* in Europe under climate change scenarios. *Conservation Genetics, 12,* 503–515. https://doi.org/10.1007/s10592-010-0157-x

Taylor, M. L., & Roterman, C. N. (2017). Invertebrate population genetics across Earth's largest habitat: The deep-sea floor. *Molecular Ecology, 26*(19),4872-4896. https://doi.org/10.1111/mec.14237

Temple, H. J., & Cox, N. A. (2009). *European red list of amphibians*. Luxembourg: Office for Official Publications of the European Communities. https://doi.org/10.2779/73661

Terraube, J., Arroyo, B.E., Madders, M. & Mougeot, F. (2011). Diet specialisation and foraging efficiency under fluctuating food abundance in sympatric avian predators. *Oikos, 120,* 234–44. https://doi.org/10.1111/j.1600-0706.2010.18554.x

Terraube, J., Arroyo, B. E., Bragin, A., Bragin, E. & Mougeot, F. (2012). Ecological factors influencing the breeding distribution and success of a nomadic, specialist predator. *Biodiversity and Conservation, 21,* 1835–52. https://doi.org/10.1007/s10531-012-0282-6

Terlizzi, A., Felline, S., Lionetto, M. G., Caricato, R., Perfetti, V., Cutignano, A., & Mollo, E. (2011). Detrimental physiological effects of the invasive alga *Caulerpa racemosa* on the Mediterranean white seabream *Diplodus sargus*. *Aquatic Biology, 12*(2), 109–117. https://doi.org/10.3354/ab00330

Thackeray, S. J., Sparks, T. H., Frederiksen, M., Burthe, S., Bacon, P. J., Bell, J. R., Botham, M. S., Brereton, T. M., Bright, P. W., Carvalho, L., Clutton-Brock, T., Dawson, A.,

Edwards, M., Elliott, J. M., Harrington, R., Johns, D., Jones, I. D., Jones, J. T., Leech, D. I., Roy, D. B., Scott, W. A., Smith, M., Smithers, R. J., Winfield, I. J., & Wanless, S. (2010). Trophic level asynchrony in rates of phenological change for marine, freshwater and terrestrial environments. *Global Change Biology, 16*(12), 3304–3313. https://doi.org/10.1111/j.1365-2486.2010.02165.x

The Russian Academy of Sciences. (2014). *The second international conference "Lichenology in Russia: Problems and perspectives", dedicated to the 300th anniversary of the Komarov Botanical Institute RAS and the 100th anniversary of the Institute of Cryptogamic Plants, Saint Petersburg, November 5–8, 2014. Programme and proceedings.* M. P. Andreev, D. E. Himelbrant, E. S. Kuznetsova, & I. S. Stepanchikova (Eds.). Saint Petersburg, Russian Federation: Publishing house of the StPtbSETU "LETI". Retrieved from https://www.binran.ru/files/publications/Proceedings/Proceedings_Lichenology_in_Russia_2014/Lichenology_in_Russia_2014_Proceedings.pdf

Theurillat, J. P., & Guisan, A. (2001). Potential impact of climate change on vegetation in the European alps: A review. *Climatic Change, 50*(1–2), 77–109. https://doi.org/10.1023/A:1010632015572

Thomas, J. A., Simcox, D. J., & Clarke, R. T. (2008). Successful conservation of a threatened *Maculinea* Butterfly. *Science, 325*(5936), 80-83. https://doi.org/10.1126/science.1175726

Thomas, Y., Pouvreau, S., Alunno-Bruscia, M., Barillé, L., Gohin, F., Bryère, P., & Gernez, P. (2016). Global change and climate-driven invasion of the Pacific oyster (*Crassostrea gigas*) along European coasts: A bioenergetics modelling approach. *Journal of Biogeography, 43*(3), 568–579. https://doi.org/10.1111/jbi.12665

Thompson, I.D., Ferreira, J., Gardner, T., Guariguata, M., Pin Koh, L., Okabe, K., Pan, Y., Schmitt, C.B., Tylianakis, J., Barlow, J., Kapos, V., Kurz, W.A., Parrotta, J.A., Spalding, M.D. & van Vliet, N. (2012). Forest biodiversity, carbon and other ecosystem services: relationships and impacts of deforestation and forest degradation. In A. John, C. W. Parrotta, & S. Mansourian (Eds.), *Understanding*

relationships between biodiversity, carbon, forests and people: the key to achieving REDD+ objectives (pp. 21–50). Vienna, Austria: IUFRO.

Thorpe, A., Whitmarsh, D., Drakeford, B., Reid, C., Karimov, B., Timirkhanov, S., Satybekov, K., & Van Anrooy, R. (2011). *Feasibility of stocking and culture-based fisheries in Central Asia*. Ankara, Turkey: FAO.

Thuiller, W., Guéguen, M., Georges, D., Bonet, R., Chalmandrier, L., Garraud, L., Renaud, J., Roquet, C., Van Es, J., Zimmermann, N. E., & Lavergne, S. (2014a). Are different facets of plant diversity well protected against climate and land cover changes? A test study in the French Alps. *Ecography, 37*(12), 1254–1266. https://doi.org/10.1111/ecog.00670

Thuiller, W., Pironon, S., Psomas, A., Barbet-Massin, M., Jiguet, F., Lavergne, S., Pearman, P. B., Renaud, J., Zupan, L., & Zimmermann, N. E. (2014b). The European functional tree of bird life in the face of global change. *Nature Communications, 5,* 3118. https://doi.org/10.1038/ncomms4118

Thuiller, W., Lavergne, S., Roquet, C., Boulangeat, I., Lafourcade, B., & Araujo, M. (2011). Consequences of climate change on the tree of life in Europe. *Nature, 470*(7335), 531–534. https://doi.org/10.1038/nature09705

Thuiller, W., Lavorel, S., Araujo, M. B., Sykes, M. T., & Prentice, I. C. (2005). Climate change threats to plant diversity in Europe. *Proceedings of the National Academy of Sciences of the United States of America, 102*(23), 8245–8250. https://doi.org/10.1073/pnas.0409902102

Thurber, A. R., Sweetman, A. K., Narayanaswamy, B. E., Jones, D. O. B., Ingels, J., & Hansman, R. L. (2014). Ecosystem function and services provided by the deep sea, *11*(14), 3941–3963. https://doi.org/10.5194/bg-11-3941-2014

Tilman, D., Isbell, F., & Cowles, J. M. (2014). Biodiversity and ecosystem functioning. *Annual Review of Ecology, Evolution, and Systematics, 45,* 471–493. https://www.

annualreviews.org/doi/10.1146/annurev-ecolsys-120213-091917

Tilman, D., Reich, P. B., & Isbell, F. (2012) Biodiversity impacts ecosystem productivity as much as resources, disturbance, or herbivory. *Proceedings of the National Academy of Sciences of the United States of America, 109,* 10394–10397. https://doi.org/10.1073/pnas.1208240109

Timdal, E. (2015). *Norsk rødliste for arter 2015 [Norwegian red list of species 2015].* Retrieved January 17, 2017, from http://www.artsdatabanken.no/Rodliste/Artsgruppene/Lav

Tingstad, L., Gjerde, I., Dahlberg, A., & Grytnes, J. A. (2017). The influence of spatial scales on red list composition: Forest species in Fennoscandia. *Global Ecology and Conservation*, *11*, 247–297. https://doi.org/10.1016/j.gecco.2017.07.005

Tishkov, A. A. (2005). Managing conservation of the biota and ecosystems in the steppe zone. *Issues of Steppe Science, 6,* 47–58.

Tishkov, A.A. [Тишков, А. А.]. (2009). "Четвертый национальный доклад «Сохранение биоразнообразия в Российской Федерации» [Fourth national report "Biodiversity Conservation in the Russian Federation]. Moscow, Russian Federation: Ministry of Natural Resources of Russian Federation.

Tockner, K., Robinson, C. & Uehlinger, U. (2008). *Rivers of Europe.* London, UK: Elsevier.

Tockner, K., Pusch, M., Gessner, J., & Wolter, C. (2011). Domesticated ecosystems and novel communities: challenges for the management of large rivers. *Ecohydrology & Hydrobiology*, *11*(3–4), 167–174. https://doi.org/10.2478/v10104-011-0045-0

Tokarev, Y., & Shulman, G. (2007). Biodiversity in the Black Sea: effects of climate and anthropogenic factors. *Hydrobiologia*, *580*(1), 23–33. https://doi.org/10.1007/s10750-006-0468-6

Török, P., Ambarlı, D., Kamp, J., Wesche, K., & Dengler, J. (2016). Step(pe) up! Raising the profile of the Palaearctic natural grasslands. *Biodiversity and*

Conservation, *25*(12), 2187–2195. https://doi.org/10.1007/s10531-016-1187-6

Tosun, M. S. (2011). *Demographic divide and labor migration in the Euro-Mediterranean region*.

Tóth-Ronkay, M., Bajor, Z., Bárány, A., Földvári, G., Görföl, T., Halpern, B., Leél-Őssy, S., Mészáros, R., LPéntek, A., L. Tóth, B., Tóth, Z., & Vörös, J. (2015). Budapest. In J. Kelcey (Ed.), *Vertebrates and invertebrates of European cities: Selected non-avian fauna* (pp. 27–73). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6

Treshkin, S. Y., Kamalov, S. K., Bachiev, A., Mamutov, N., Gladishevand, A. I., & Aimbetov, I. (1998). Present status of the Tugai Forests in the lower Amu-Dar'ya basin and problems of their protection and restoration. In *Ecological research and monitoring of the Aral Sea deltas* (pp. 43–55). Paris, France: UNESCO.

Treshkin, S. (2001). The Tugai forests of floodplain of the Amudarya River: Ecology, dynamics and their conservation. In *Sustainable land use in deserts* (pp. 95–102). Berlin, Germany: Springer. http://doi.org/10.1007/978-3-642-59560-8_9

Trewick, S. (2017). Plate tectonics in biogeography. *International encyclopedia of geography: People, the earth, environment and technology*. https://doi.org/10.1002/9781118786352.wbieg0638

Trivedi, M. R., Berry, P. M., Morecroft, M. D., & Dawson, T. P. (2008). Spatial scale affects bioclimate model projections of climate change impacts on mountain plants. *Global Change Biology*, *14*(5), 1089–1103. https://doi.org/10.1111/j.1365-2486.2008.01553.x

Trontelj, P., Douady, C. J., Fišer, C., Gibert, J., Goričke, Š., Lefébure, T., Sket, B., & Zakšek, V. (2009). A molecular test for cryptic diversity in ground water: How large are the ranges of macro-stygobionts? *Freshwater Biology*, *54*(4), 727–744. https://doi.org/10.1111/j.1365-2427.2007.01877.x

Trontelj, P., & Zakšek, V. (2016). Genetic monitoring of iProteus/i populations. *Natura Sloveniae*, *18*(1), 53–54.

Tsikliras, A. C., Dinouli, A., Tsiros, V.-Z., & Tsalkou, E. (2015). The Mediterranean and Black Sea fisheries at risk from overexploitation. *PloS One*, *10*(3), e0121188. https://doi.org/10.1371/journal.pone.0121188

Tuck, S. L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L. A., & Bengtsson, J. (2014). Land-use intensity and the effects of organic farming on biodiversity: A hierarchical meta-analysis. *Journal of Applied Ecology*, *51*(3), 746–755. http://doi.org/10.1111/1365-2664.12219

Tuniyev, B. S. (1990). On the independence of the Colchis center of amphibian and reptile speciation. *Asiatic Herpetological Research*, *3*, 67–84.

Tuniyev, B. S. (1995). On the Mediterranean influence on the formation of herpetofauna of the Caucasian Isthmus and its main xerophylous refugia. *Russian Journal of Herpetology*, *2*, 95–119.

Tuniyev, B. S. (1997). About exact borders of the Colchis biogeographical province. *Russian Journal of Herpetology*, *4*, 182–185.

Tuniyev, B. S. (2012). First consequences of climate variation and use of natural resources in the biota of the West Caucasus In V. V. Snakin (Ed.), *Global environmental processes. Proceedings of International Scientific Conference* (*Moscow, October 2–4, 2012*). Retrieved from http://knigi.konflib.ru/8mehanika/146003-1-globalnie-ekologicheskie-processi-materiali-mezhdunarodnoy-nauchnoy-konferencii-moskva-2-4-oktyabrya-2012-g-mosk.php

Tuniyev, B. S. (2016). Rare species of shield-head vipers in the Caucasus. *Nature Conservation Research*, *1*, 11–25.

Tuniyev, B., Ananjeva, N.B., Agasyan, A., Orlov, N.L., & Tuniyev, S., B. (2009). *Eremias pleskei*. (errata version published in 2017). *The IUCN Red List of Threatened Species*. https://doi.org/10.2305/IUCN.UK.2009.RLTS.T164583A5910262.en

Turdakov, F. A. [Тудаков, Ф. А.]. (1963). *Fishes of Kirghizia* [Рыбы Киргизии]. Frunze, USSR: Publishing House of KirgizSSR Academy of Science.

Turdiboeva, M. U. [Турдибоева, М. У.]. (2015). О дергадации земель в Центральной Азии [About Land Degradation in Central Asia]. Молодой Ученый *[Young Scientist]*, *9*, 780–783.

Turdieva, M., Aleksandrovskiy, E., Kayimov, A., Djumabaeva, S., Mukanov, B., Saparmyradov, A., & Akmadov, K. (2007). Forests in Central Asia: current status and constraints. In R. Lal, M. Suleimenov, B. A. Stewart, D. O. Hansen, & P. Doraiswamy (Eds.), *Climate change and terrestrial carbon sequestration in Central Asia* (pp. 25–32). London, UK: Taylor & Francis.

Türk, R., & Hafellner, J. (1999). Rote Liste gefährdeter Flechten (Lichenes) Österreichs [Red list of endangered lichens of Austria]. In H. Niklfeld (Ed.), *Rote Listen gefährdeter Pflanzen Österreichs [Red list of endangered plants of Austria]* (pp. 187–228).

Tutayuk, V. Kh. (1975). Древесные реликты Талыша [Tree relicts of Talish]. Baku, Azerbaijan: Elm.

Uetz, P. (2017). *The Reptile Database*. Retrieved from http://www.reptile-database.org

Ulicsni, V., Svanberg, I., & Molnár, Z. (2016). Folk knowledge of invertebrates in Central Europe - folk taxonomy, nomenclature, medicinal and other uses, folklore, and nature conservation. *Journal of Ethnobiology and Ethnomedicine*, *12*(1), 47. https://doi.org/10.1186/s13002-016-0118-7

Urbanavichus, G. P. [Урбанавичюс, Г. П.]. (2010). *A checklist of the lichen flora of Russia* [Список лихенофлоры России]. M. P. Andreev (Ed.). St. Petersburg, Russian Federation: Nauka.

UNDP. (2015). Work for human development. (n.d.). Retrieved from www.undp.org

UNECE, & CAREC. (2011). *Development of regional cooperation to ensure water quality in Central Asia. Diagnostic Report and Cooperation Development Plan*.

UNEP. (2007). Deep-sea biodiversity and ecosystems. Retrieved from https://www.biodiversitylibrary.org/bibliography/57961#/summary

UNEP, FAO, & UNFF. (2009). *Vital Forest graphics*.

UNEP-WGMS. (2008). *Global Glacier Changes: facts and figures.*

UNESCO. (2009). Global open oceans and deep seabed (GOODS) - biogeographic classification. *IOC technical series*, *84*. Retrieved from http://www.citeulike.org/user/LNCScatalogo/article/10352567?updated=1336273165&rejected=

United Nations. (2016). *The first global integrated marine assessment - World ocean assessment I*. New York, USA.: United Nations.

Väinölä, R., Witt, J. D. S., Grabowski, M., Bradbury, J. H., Jazdzewski, K., & Sket, B. (2008). Global diversity of amphipods (Amphipoda; Crustacea) in freshwater. *Hydrobiologia*, *595*(1), 241–255. https://doi.org/10.1007/s10750-007-9020-6

Valkó, O., Deák, B., Török, P., Kelemen, A., Miglécz, T., Tóth, K., & Tóthmérész, B. (2016). Abandonment of croplands: problem or chance for grassland restoration? Case studies from Hungary. *Ecosystem Health and Sustainability*, *2*(2), e0128. https://doi.org/10.1002/ehs2.1208

Van Cauwenberghe, L., Vanreusel, A., Mees, J., & Janssen, C. R. (2013). Microplastic pollution in deep-sea sediments. *Environmental Pollution*, *182*, 495–499. https://doi.org/10.1016/j.envpol.2013.08.013

Van de Poel, J. L., De Baerdmaeker, A., Bakker, G., Moerland, W., & De Zwarte, N. (2015). Rotterdam. In J. Kelcey (Ed.), *Vertebrates and invertebrates of European cities: Selected non-avian fauna* (pp. 155–178). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6

van de Wouw, M., van Hintum, T., Kik, C., van Treuren, R., & Visser, B. (2010). Genetic diversity trends in twentieth century crop cultivars: A meta analysis. *Theoretical and Applied Genetics*, *120*(6), 1241–1252. https://doi.org/10.1007/s00122-009-1252-6

van der Plas, F., Manning, P., Soliveres, S., Allan, E., Scherer-Lorenzen, M.,

Verheyen, K., Wirth, C., Zavala, M. A., Ampoorter, E., Baeten, L., Barbaro, L., Bauhus, J., Benavides, R., Benneter, A., Bonal, D., Bouriaud, O., Bruelheide, H., Bussotti, F., Carnol, M., Castagneyrol, B., Charbonnier, Y., Coomes, D. A., Coppi, A., Bastias, C. C., Dawud, S. M., De Wandeler, H., Domisch, T., Finér, L., Gessler, A., Granier, A., Grossiord, C., Guyot, V., Hättenschwiler, S., Jactel, H., Jaroszewicz, B., Joly, F., Jucker, T., Koricheva, J., Milligan, H., Mueller, S., Muys, B., Nguyen, D., Pollastrini, M., Ratcliffe, S., Raulund-Rasmussen, L., Selvi, F., Stenlid, J., Valladares, F., Vesterdal, L., Zielínski, D., Fischer, M. (2016a). Biotic homogenization can decrease landscape-scale forest multifunctionality. *Proceedings of the National Academy of Sciences of the United States of America, 113*(13), 3557-3562. https://doi.org/10.1073/pnas.1517903113

van der Plas, F., Manning, P., Soliveres, S., Allan, E., Scherer-Lorenzen, M., Verheyen, K., Wirth, C., Zavala, M. A., Hector, A., Ampoorter, E., Baeten, L., Barbaro, L., Bauhus, J., Benavides, R., Benneter, A., Berthold, F., Bonal, D., Bouriaud, O., Bruelheide, H., Bussotti, F., Carnol, M., Castagneyrol, B., Charbonnier, Y., Coomes, D. A., Coppi, A., Bastias, C. C., Dawud, S. M., De Wandeler, H., Domisch, T., Finér, L., Gessler, A., Granier, A., Grossiord, C., Guyot, V., Hättenschwiler, S., Jactel, H., Jaroszewicz, B., Joly, F., Jucker, T., Koricheva, J., Milligan, H., Mueller, S., Muys, B., Nguyen, D., Pollastrini, M., Raulund-Rasmussen, L., Selvi, F., Stenlid, J., Valladares, F., Vesterdal, L., Zielínski, D., Fischer, M. (2016b). Jack-of-all-trades effects drive biodiversity–ecosystem multifunctionality relationships in European forests. *Nature Communications, 7*, 11109. https://doi.org/10.1038/ncomms11109

Van Der Wal, R., Pearce, I. S. K., & Brooker, R. W. (2005). Mosses and the struggle for light in a nitrogen-polluted world. *Oecologia*, *142*(2), 159–168. https://doi.org/10.1007/s00442-004-1706-0

Van Dover, C. L., Ardron, J. A., Escobar, E., Gianni, M., Gjerde, K. M., Jaeckel, A., Jones, D. O. B., Levin, L. A., Niner, H. J., Pendleton, L., Smith, C. R., Thiele, T., Turner, P. J., Watling, L., &

Weaver, P. P. E. (2017). Biodiversity loss from deep-sea mining. *Nature Geoscience*, *10*(7), 464–465. https://doi.org/10.1038/ngeo2983

van Herk, C. M. (2001). Bark pH and susceptibility to toxic air pollutants as independent causes of changes in epiphytic lichen composition in space and time. *Lichenologist*, *33*(5), 419–441. https://doi.org/10.1006/lich.2001.0337

Van Swaay , C. A. M., Van Strien, A. J., Aghababyan, K., Åström, S., Botham, M., Brereton, T. Carlisle, B., Chambers, P., Collins, S., Dopagne, C., Escobés, R., Feldmann, R., Fernández-García, J. M., Fontaine, B., Goloshchapova, S., Gracianteparaluceta, A., Harpke, A., Heliölä, J., Khanamirian, G., Komac, B., Kühn, E., Lang, A., Leopold, P., Maes, D., Mestdagh, X., Monasterio, Y., Munguira, M. L., Murray, T., Musche, M., Õunap, E., Pettersson, L. B., Piqueray, J., Popoff, S., Prokofev, I., Roth, T., Roy, D. B., Schmucki, R., Settele, J., Stefanescu, C., Švitra, G., Teixeira, S. M., Tiitsaar, A., Verovnik, R., & Warren, M. S. (2017). *The European butterfly indicator for grassland species 1990-2015.*

Van Swaay, C., Cuttelod, A., Collins, S., Maes, D., Munguira, M. L., Šašić, M., Settele, J., Verovnik, R., Verstrael, T., Warren, M., Wiemers, M., & Wynhoff, I. (2010). *European red list of butterflies*. Luxembourg: Publications Office of the European Union. https://doi.org/doi:10.2779/83897

van Swaay, C., van Strien, A., Harpke, A., Fontaine, B., Stefanescu, C., Roy, D., Maes, D., Kühn, E., Õunap, E., Regan, E., Švitra, G., Prokofev, I., Heliölä, J., Settele, J., Pettersson, L., Botham, M., Musche, M., Titeux, N., Cornish, N., Leopold, P., & Julliard, R. (2015). *The European Butterfly Indicator for Grassland species: 1990-2013*. Copenhagen, Denmark: European Environment Agency.

Vanderpoorten, A., Engels, P., & Sotiaux, A. (2004). Trends in diversity and abundance of obligate epiphytic bryophytes in a highly managed landscape. *Ecography*, *27*(5), 567–576. https://doi.org/10.1111/j.0906-7590.2004.03890.x

Vandvik, V., Topper, J. P., Cook, Z., Daws, M. I., Heegaard, E., Maren, I. E., **& Velle, L. G.** (2014). Management-driven evolution in a domesticated ecosystem. *Biology Letters*, *10*(2), 20131082–20131082. http://doi.org/10.1098/rsbl.2013.1082

Vandvik, V., Heegaard, E., Måren, I. E., & Aarrestad, P. A. (2005). Managing heterogeneity: the importance of grazing and environmental variation on post-fire succession in heathlands. *Journal of Applied Ecology*, *42*(1), 139–149. http://doi.org/10.1111/j.1365-2664.2005.00982.x

Vangjeli, J., Ruci, B., & Mullaj, A. (1995). *Libri i kuq.* (*bimët e kërcënuara dhe të rralla të shqipërisë*) *[Red book of threatened and rare species of flora and fauna of Albania]*. Tirana, Albania: Institute of Biological Research, Academy of Science.

Vangjeli, J., Ruci, P., & Hoda, P. (1997). *Libri i Kuq* (*bimë, shoqerimë bimore dhe kafshë të rrëzikuara*). *[Red book of threatened and rare species of flora and fauna of Albania]*. Tirana, Albania: Regional Environmental Center for Central and Eastern Europe.

Vanhatalo, J., Vetemaa, M., Herrero, A., Aho, T., & Tiilikainen, R. (2014). By-catch of grey seals (*Halichoerus grypus*) in Baltic fisheries - a Bayesian analysis of interview survey. *PloS One*, *9*(11), e113836. https://doi.org/10.1371/journal.pone.0113836

Vanreusel, A., Hilario, A., Ribeiro, P. A., Menot, L., & Arbizu, P. M. (2016). Threatened by mining, polymetallic nodules are required to preserve abyssal epifauna. *Scientific Reports*, *6*(1), 26808. https://doi.org/10.1038/srep26808

Väre, H., Lampinen, R., Humphries, C., & Williams, P. (2003). Taxonomic diversity of vascular plants in the European alpine areas. In L. Nagy, G. Grabherr, C. Körner, & D. B. A. Thompson (Eds.), *Alpine biodiversity in Europe* (pp. 133–148). Berlin, Germany: Springer.

Varga, A. & Molnár, Z. (2014). The role of traditional ecological knowledge in managing wood- pastures. In T. Hartel & T. Plieninger (Eds.), *European wood-pastures in transition* (pp. 187-202). London, UK: Routledge.

Vasilakopoulos, P., Maravelias, C. D., & Tserpes, G. (2014). The alarming decline of Mediterranean fish stocks. *Current Biology*, *24*(14), 1643–1648. https://doi.org/10.1016/j.cub.2014.05.070

Vasiliev, Ya. Ya., Gorodkov, B. N., Ilinckiy, A. Ya., Lavrenko, E. M., Leskov, A. I., & Maleev, V. Ya. [Васильев, Я. Я., Городков, Б. Н., Ильинский, А. Я., Лавренко, Е. М., Лесков, А. И., & Малеев, В. Я.]. (1941). Пояснительный текст к карте растительности СССР в масштабе *1:5 000 000 [The explanatory text to the vegetation map of the USSR in scale 1:5 000 000]*.

Vasiluk, A. V., Parnikoza, I. Y., & Shevchenko, M. S. [Василюк, А. В., Парникоза, И. Ю., & Шевченко, М. С.]. (2010). Биоразнообразие степей под охраной Красной и Зеленой книг Украины [Biodiversity of steppes under protection of red book and green book of Ukraine]. Степной бюллетень *[Steppe Bulletin]*, *29*, 33–36.

Veen, P., Jefferson, R., Smidt, J., & Straaten, J. (2009). *Grasslands in Europe of high nature value*. Zeist, The Netherlands: KNNV Publishing.

Veisov, S.K., Khamraev, G.O., & Akyniyazov, A.D. [Вейсов, С. К., Хамраев, Г. О., & Акыниязов, А. Д.]. (2008). Динамика барханного рельефа Западного Туркменистана [The dynamics of Barkhan relief of the western Turkmenistan]. Проблемы Освоения Пустынь *[Problems of Desert Development], 4,* 16–19.

Velchev V. (Ed.). (1984). *Red data book of the People's Republic of Bulgaria. Volume 1. Plants*. Sofia, Bulgaria: Bulgarian Academy of Sciences.

Velle, L. G., Nilsen, L. S., Norderhaug, A., & Vandvik, V. (2014). Does prescribed burning result in biotic homogenization of coastal heathlands? *Global Change Biology*, *20*(5), 1429–1440. http://doi.org/10.1111/gcb.12448

Vellend, M., Baeten, L., Myers-Smith, I. H., Elmendorf, S. C., Beauséjour, R., Brown, C. D., De Frenne, P., Verheyen, K., & Wipf, S. (2013). Global meta-analysis reveals no net change in local-scale plant biodiversity over time. *Proceedings of the National Academy of Sciences of the United States of America, 110*(48), 19456-19459. http://doi.org/10.1073/pnas.1312779110

Venglovsky, B. I. [Венгловский, Б. И.]. (2006). Биоэкологические особенности восстановления и развития ореховых лесов Кыргызстана *[Bioecological features of the restoration and growing of walnut forests of Kyrgyzstan]*. Bishkek, Kyrgyzstan: The National Academy of Sciences of the Kyrgyz Republic.

Venn, S., Schulman, H., Törrönen, S., Salla, A., Pajunen, T., Kerppola, S., Paukkunen, J., Nieminen, M., Vilisics, F., & Karjalainen, S. (2015). Helsinki. In J. Kelcey (Ed.), *Vertebrates and invertebrates of European cities: Selected non-avian fauna* (pp. 323-377). New York, USA: Springer Science+Business Media. https://doi.org/10.1007/978-1-4939-1698-6_10

Ventosa, A., & Arahal, D. R. (2009). Physico-chemical characteristics of hypersaline environments and their biodiversity. In C. Gerday (Ed.), *Extremophiles* (pp. 247–262). Oxford, UK: EOLSS Publications.

Verboom, J., Alkemade, R., Klijn, J., Metzger, M. J., & Reijnen, R. (2007). Combining biodiversity modeling with political and economic development scenarios for 25 EU countries. *Ecological Economics, 62*(2), 267-276. https://doi.org/10.1016/j.ecolecon.2006.04.009

Vergés, A., Tomas, F., Cebrian, E., Ballesteros, E., Kizilkaya, Z., Dendrinos, P., Karamanlidis, A. A., Spiegel, D., & Sala, E. (2014). Tropical rabbitfish and the deforestation of a warming temperate sea. *Journal of Ecology*, *102*(6), 1518–1527. https://doi.org/10.1111/1365-2745.12324

Verheye, W. H. (Ed.). (2009). Volume V: Dry lands and desertification. In *Encyclopedia of Life Support Systems* (pp. 1–40). Paris, France: UNESCO.

Verheyen, K., Baeten, L., De Frenne, P., Bernhardt-Römermann, M., Brunet, J., Cornelis, J., Decocq, G., Dierschke, H., Eriksson, O., Hédl, R., Heinken, T., Hermy, M., Hommel, P., Kirby, K., Naaf, T., Peterken, G., Petřík, P., Pfadenhauer, J., Van Calster, H., Walther, G. R., Wulf, M., & Verstraeten, G. (2012). Driving factors behind the eutrophication signal in understorey plant communities of deciduous temperate forests. *Journal of Ecology*,

100(2), 352–365. https://doi.org/10.1111/j.1365-2745.2011.01928.x

Vershinin, A. [Вершинин А.]. (2003). Жизнь Черного Моря *[The life of the Black Sea]*. Moscow, Rusian Federation: MacCentr.

Vershinin, A. [Вершинин А.]. (2016). Живое Черное Море *[The living Black Sea]*. Moscow, Russian Federation: Kovcheg.

Vershinin, V. L., Vershinina, S. D., Berzin, D. L., & Zmeeva, D. V. (2015). Long-term observation of amphibian populations inhabiting urban and forested areas in Yekatarinburg, Russia. *Scientific Data*, *2*, 1–11. https://doi.org/10.1038/sdata.2015.18

Vetrov, A. A., & Romankevich, E. A. (2011). Primary production and fluxes of organic carbon to the seabed in the Russian Arctic seas as a response to the recent warming. *Oceanology*, *51*(2), 255–266. https://doi.org/10.1134/S0001437011020196

Vězda, A. (1983). Foliicole Flechten aus der Kolchis (West-Transkaukasien, UdSSR) [Foliicole lichen from the Colchis (West Transcaucasia, USSR)]. *Folia Geobotanica & Phytotaxonomica*, *18*(1), 45–70.

Vickery, J. A., Ewing, S. R., Smith, K. W., Pain, D. J., Bairlein, F., Škorpilová, J., & Gregory, R. D. (2014). The decline of Afro-Palaearctic migrants and an assessment of potential causes. *Ibis*, *156*(1), 1–22. https://doi.org/10.1111/ibi.12118

Vilà, M., Carrillo-Gavilán, A., Vayreda, J., Bugmann, H., Fridman, J., Grodzki, W., Haase, J., Kunstler, G., Schelhaas, M., & Trasobares, A. (2013). Disentangling biodiversity and climatic determinants of wood production. *PLoS ONE*, *8*(2), e53530. https://doi.org/10.1371/journal.pone.0053530

Vilà, M., & Hulme, P. E. (Eds.). (2017). *Impact of Biological Invasions on Ecosystem Services*. Cham, Switzerland: Springer International Publishing

Vilkov, E. V. (2013). Population trends in regular migrants as the basis for a prediction model for conservation of the birds of Eurasia. *Russian Journal of Ecology*,

44(2), 142–157. https://doi.org/10.1134/S106741361301013X

Villéger, S., Grenouillet, G., & Brosse, S. (2014). Functional homogenization exceeds taxonomic homogenization among European fish assemblages. *Global Ecology and Biogeography, 23*(12), 1450-1460. https://doi.org/10.1111/geb.12226

Villnäs, A., Norkko, J., Lukkari, K., Hewitt, J., & Norkko, A. (2012). Consequences of increasing hypoxic disturbance on benthic communities and ecosystem functioning. *PLoS ONE*, *7*(10), e44920. https://doi.org/10.1371/journal.pone.0044920

Vinogradov, K. A. [Виноградов, К. А.]. (1958). Очерки по истории отечественных гидробиологических исследований на Черном море *[Essays on the history of the National hydrobiological studies in the Black Sea]*. Kiev, USSR: Publishing house of the Academy of Sciences of the Ukrainian SSR.

Vinogradov, M. E., Shiganova, T. A., & Khoroshilov, V. S. (1995). The status of the main components of the zooplankton community in the Black Sea in 1993. *Oceanology*, *35*(3), 418–421.

Vinogradov, V. N. [Виноградов, В. Н.]. (1977). Лес и проблемы пустынь [The forest and problems of deserts]. Лесное Хозяйство *[Forestry], 9,* 55–60.

Virkkala, R., Heikkinen, R. K., Leikola, N., & Luoto, M. (2008). Projected large-scale range reductions of northern-boreal land bird species due to climate change. *Biological Conservation*, *141*(5), 1343–1353. https://doi.org/10.1016/j.biocon.2008.03.007

Virtanen, R., Grytnes, J. A., Lenoir, J., Luoto, M., Oksanen, J., Oksanen, L., & Svenning, J. C. (2013). Productivity-diversity patterns in arctic tundra vegetation. *Ecography*, *36*(3), 331–341. https://doi.org/10.1111/j.1600-0587.2012.07903.x

Virtanen, R., Johnston, A. E., Crawley, M. J., & Edwards, G. R. (2000). Bryophyte biomass and species richness on the park grass experiment, Rothamsted, UK. *Plant Ecology*, *151*(2), 129–141. https://doi.org/10.1023/A:1026533418357

Virtanen, T., Mikkola, K., Patova, E., & Nikula, A. (2002). Satellite image analysis of human caused changes in the tundra vegetation around the city of Vorkuta, north-European Russia. *Environmental Pollution*, *120*(3), 647–658. https://doi.org/10.1016/S0269-7491(02)00186-0

Visconti, P., Bakkenes, M., Baisero, D., Brooks, T., Butchart, S. H. M., Joppa, L., Alkemade, R., Marco, M. Di, Santini, L., Hoffmann, M., Maiorano, L., Pressey, R. L., Arponen, A., Boitani, L., Reside, A. E., Vuuren, D. van, & Rondinini, C. (2016). Projecting global biodiversity indicators under future development scenarios. *Conservation Letters*, *9*(1), 5–13. https://doi.org/10.1111/conl.12159

Vitasse, Y., Hoch, G., Randin, C. F., Lenz, A., Kollas, C., & Körner, C. (2012). Tree recruitment of European tree species at their current upper elevational limits in the Swiss Alps. *Journal of Biogeography*, *39*(8), 1439–1449. https://doi.org/10.1111/j.1365-2699.2012.02697.x

Voigt, C. C., Popa-Lisseanu, A. G., Niermann, I., & Kramer-Schadt, S. (2012). The catchment area of wind farms for European bats: A plea for international regulations. *Biological Conservation*, *153*, 80–86. https://doi.org/10.1016/j.biocon.2012.04.027

Vompersky, S. E., Ivanov, A. I., Tsyganova, O. P., Valyaeva, N. A., Glukhova, T. V., Dubinin, A. I., Glukhov, A. I., & Markelova, L. G. (1996). Bog organic soils and bogs of Russia and the carbon pool of their peat. *Eurasian Soil Science*, (28), 91–105.

Vompersky, S. E., Sirin, A. A., Sal'nikov, A. A., Tsyganova, O. P., & Valyaeva, N. A. (2011). Estimation of forest cover extent over peatlands and paludified shallow-peat lands in Russia. *Contemporary Problems of Ecology*, *4*(7), 734–741. http://doi.org/10.1134/S1995425511070058

Vompersky, S. E., Sirin, A. A., Tsyganova, O. P., Valyaeva, N. A., & Maykov, D. A. (2005). Mires and paludified lands of Russia: an attempt to analyse the spatial distribution and diversity. *Izvestiya RAN, Seriya Geografi Cheskaya, 5,* 21–33.

Volvenko, I. V. (2014). The new large database of the Russian bottom trawl surveys in the far eastern seas and the North Pacific Ocean in 1977-2010. *International Journal of Environmental Monitoring and Analysis*, *2*(6), 302–312. https://doi.org/10.11648/j.ijema.20140206.12

von Hertzen L, Hanski I, & Haahtela T. 2011. Natural immunity: Biodiversity loss and inflammatory diseases are two global megatrends that might be related. *EMBO Reports, 12,* 1089–1093. https://doi.org/10.1038/embor.2011.195

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A, Liermann, C. R., & Davies, P. M. (2010). Global threats to human water security and river biodiversity. *Nature*, *467*(7315), 555–561. https://doi.org/10.1038/nature09549

Vrahnakis, M., Janišová, M., Rūsiņa, S., Török, P., Venn, S., & Dengler, J. (2013). The European dry grassland group (EDGG): stewarding Europe's most diverse habitat type. In *Steppenlebensräume Europas– Gefährdung, Erhaltungsmaßnahmen und Schutz [Europe's steppe habitats – threats, conservation measures and protection]* (pp. 417–434).

Vyaznikova, K. S. [Вязникова, К.С.]. (2014). Влияние хозяйств марикультуры приморского гребешка на химико-экологическое состояние прибрежных акваторий [The impact of mariculture scallop farming on the chemical and ecological status of coastal waters]. In G. N. Kim, I. N. Kim, N. V. Dementyev, V. V. Barinov, E. N. Baklanov, & S. G. Volodina [Г. Н. Ким, И. Н. Ким, Н. В. Дементьева, В. В. Баринов, Е. Н. Бакланов, & С. Г. Володина] (Eds.), Актуальные проблемы освоения биологических ресурсов Мирового океана. Материалы III Международной научно-технической конференции (Владивосток, 27-29 мая 2014 года) *[Actual problems of the world ocean biological resource use. Proceedings of the 3rd International Scientific and Technical Conference (Vladivostok, 27-29 May, 2014)]* (pp. 65–67). Vladivostok, Russian Federation: Far Eastern State Technical Fisheries University.

Wagner, C., & Adrian, R. (2011). Consequences of changes in thermal regime for plankton diversity and trait composition in a polymictic lake: A matter of temporal scale. *Freshwater Biology*, *56*(10), 1949–1961. https://doi.org/10.1111/j.1365-2427.2011.02623.x

Wagner, N., Mingo, V., Schulte, U., & Lötters, S. (2015). Risk evaluation of pesticide use to protected European reptile species. *Biological Conservation*, *191*, 667–673. https://doi.org/10.1016/j.biocon.2015.08.002

Walker, D. A., Raynolds, M. K., Daniëls, F. J. A., Einarsson, E., Elvebakk, A., Gould, W., Katenin, A. E., Kholod, S. S., Markon, C. J., Melnikov, E. S., Moskalenko, N. G., Talbot, S. S., & Yurtsev, B. A. (2005). The Circumpolar Arctic vegetation map. *Journal of Vegetation Science, 16*(3), 267-282. https://doi.org/10.1111/j.1654-1103.2005.tb02365.x

Walker, M. D., Wahren, H. C., Hollister, R. D., Henry, G. H. R., Ahlquist, L. E., Alatalo, J. M., Bret-Harte, M. S., Calef, M. P., Callaghan, T. V., Carroll, A. B., Epstein, H. E., Jonsdottir, I. S., Klein, J. A., Manusson, B., Molau, U., Oberbaur, S. F., Rewa, S. P., Robinson, C. H., Shaver, G. R., Suding, K. N., Thompson, C. C., Tolvanen, A., Totland, O., Turner, P. L., Tweedie, C. E., Webber, P. J., & Wookey, P. A. (2006). Plant community responses to experimental warming across the tundra biome. *Proceedings of the National Academy of Sciences of the United States of America*, *103*(5), 1342–1346. https://doi.org/10.1073/pnas.0503198103

WallisDeVries, M. F., Noordijk, J., Colijn, E. O., Smit, J. T., & Veling, K. (2016). Contrasting responses of insect communities to grazing intensity in lowland heathlands. *Agriculture, Ecosystems & Environment*, *234*, 72–80. http://doi.org/10.1016/j.agee.2016.04.012

Wasmund, N., Göbel, J., & Bodungen, B. v. (2008). 100-years-changes in the phytoplankton community of Kiel Bight (Baltic Sea). *Journal of Marine Systems*, *73*(3–4), 300–322. https://doi.org/10.1016/j.jmarsys.2006.09.009

Wassmann, P., Duarte, C. M., Agusti, S., & Sejr, M. K. (2011). Footprints of climate change in the Arctic marine ecosystem. *Global Change Biology*, *17*(2),

1235–1249. https://doi.org/10.1111/j.1365-2486.2010.02311.x

Wassmann, P., Kosobokova, K. N., Slagstad, D., Drinkwater, K. F., Hopcroft, R. R., Moore, S. E., Ellingsen, I., Nelson, R. J., Carmack, E., Popova, E., & Berge, J. (2015). The contiguous domains of Arctic Ocean advection: Trails of life and death. *Progress in Oceanography*, *139*, 42–65. https://doi.org/10.1016/j.pocean.2015.06.011

Watson, M., Wilson, J. M., Koshkin, M., Sherbakov, B., Karpov, F., & Gavrilov, A. (2006). Nest survival and productivity of the critically endangered sociable lapwing *Vanellus gregarious*. *Ibis, 148*(3), 489–502. https://doi.org/10.1111/j.1474-919X.2006.00555.x

Watson, R. A., & Morato, T. (2013). Fishing down the deep: Accounting for within-species changes in depth of fishing. *Fisheries Research*, *140*, 63–65. https://doi.org/10.1016/j.fishres.2012.12.004

Webb, N. R. (1986). *Heathlands*. London, UK: Collins.

Webb, N. R. (1998). History and ecology of European heathlands. *Transactions of the Suffolk Naturalists Society, 34.*

Webb, J. R., Drewitt, A. L., & Measures, G. H. (2010). *Managing for species: Integrating the needs of England's priority species into habitat management. Part 1 Report. Natural England Research Reports, Number 024.* Sheffield, UK: Natural England. Retrieved from http://publications.naturalengland.org.uk/file/61078

Wedding, L. M., Reiter, S. M., Smith, C. R., Gjerde, K. M., Kittinger, J. N., Friedlander, A. M., Gaines, S. D., Clark, M. R., Thurnherr, A. M., Hardy, S. M., & Crowder, L. B. (2015). Managing mining of the deep seabed. *Science*, *349*(6244), 144–145. https://doi.org/10.1126/science.aac6647

Weeda E. J., Van Der Meijden, R., & Bakker, P. A. (1990). *Floron red data list 1990. Red data list of extinct, endangered and vunerable plants in Netherlands in the period 1980 -1990. Gorteria: Tijdschrift voor Onderzoek aan de Wilde Flora. 16,* 1-26.

Wegner, P., Kleinstäuber, G., Baum, F., & Schilling, F. (2005). Long-term investigation of the degree of exposure of German peregrine falcons (*Falco peregrinus*) to damaging chemicals from the environment. *Journal of Ornithology*, *146*, 34–54. https://doi.org/10.1007/s10336-004-0053-6

Wehn, S., Lundemo, S., & Holten, J. I. (2014). Alpine vegetation along multiple environmental gradients and possible consequences of climate change. *Alpine Botany*, *124*(2), 155–164. https://doi.org/10.1007/s00035-014-0136-9

Weinert, M., Mathis, M., Kröncke, I., Neumann, H., Pohlmann, T., & Reiss, H. (2016). Modelling climate change effects on benthos: Distributional shifts in the North Sea from 2001 to 2099. *Estuarine, Coastal and Shelf Science*, *175*, 157–168. https://doi.org/10.1016/j.ecss.2016.03.024

Wennersten, L., & Forsman, A. (2012). Population-level consequences of polymorphism, plasticity and randomized phenotype switching: a review of predictions. *Biological Reviews, 87,* 756–767. https://doi.org/10.1111/j.1469-185X.2012.00231.x

Werger, M. J. A., & van Staalduinen, M. A. (Eds.). (2012). *Eurasian Steppes: ecological problems and livelihoods in a changing world*. Dordrecht, The Netherlands: Springer.

Werner, Y., Disi, M., & Mousa Disi, A. M. (2006). *Acanthodactylus beershebensis*. The IUCN Red List of Threatened Species. https://doi.org/10.2305/IUCN.UK.2006.RLTS.T61454A12488658.en

Wesche, K., Ambarlı, D., Kamp, J., Török, P., Treiber, J., & Dengler, J. (2016). The Palaearctic steppe biome: a new synthesis. *Biodiversity and Conservation*, *25*(12), 2197–2231. https://doi.org/10.1007/s10531-016-1214-7

Westerbom, M., Kilpi, M., & Mustonen, O. (2002). Blue mussels, *Mytilus edulis* at the edge of the range: population structure, growth and biomass along a salinity gradient in the north-eastern Baltic Sea. *Marine Biology*, *140*(5), 991–999. https://doi.org/10.1007/s00227-001-0765-6

Westling, A. (2015). *Rödlistade arter i Sverige 2015 [Red list species of Sweden 2015].*

Wielgolaski, F E. (1972). Vegetation types and plant biomass in tundra. *Arctic and Alpine Research, 4*(4), 291-305.

Wiedmann, M., Aschan, M., Certain, G., Dolgov, A., Greenacre, M., Johannesen, E., Planque, B., & Primicerio, R. (2014). Functional diversity of the Barents Sea fish community. *Marine Ecology Progress Series*, *495*, 205–218. https://doi.org/10.3354/meps10558

Wiens, J. J. (2016). Climate-related local extinctions are already widespread among plant and animal species. *PLoS Biology*, *14*(12), e2001104. https://doi.org/10.1371/journal.pbio.2001104

Wiig, Ø., Amstrup, S., Atwood, T., Laidre, K., Lunn, N., Obbard, M., Regehr, E., & Thiemann, G. (2015), *Ursus maritimus*. The IUCN Red List of Threatened Species. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en

Williams, W. (1981). Inland salt lakes: An introduction. *Hydrobiologia, 81–82,* 1–14. https://doi.org/10.1007/BF00048701

Wilson, J. B., Peet, R. K., Dengler, J., & Pärtel, M. (2012). Plant species richness: The world records. *Journal of Vegetation Science*, *23*(4), 796–802. https://doi.org/10.1111/j.1654-1103.2012.01400.x

Wingfield, M. J., Brockerhoff, E. G., Wingfield, B. D., & Slippers, B. (2015). Planted forest health: The need for a global strategy. *Science*, *349*(6250), 832–836. https://doi.org/10.1126/science.aac6674

Wipf, S., Stöckli, V., Herz, K., & Rixen, C. (2013). The oldest monitoring site of the Alps revisited: accelerated increase in plant species richness on Piz Linard summit since 1835. *Plant Ecology and Diversity*, *6*(3–4), 447–455. https://doi.org/10.1080/17550874.2013.764943

Wirth, V, Hauck M, S. M. (2013). *Die Flechten Deutschlands (Vol. 1 and 2) [The lichens of Germany (Vol. 1 and 2)]*. Stuttgart, Germany: Eugen Ulmer.

Wirth, V., Hauck, M., Von Brackel, W., Cezanne, R., De Bruyn, U., Dürhammer, O., … Heinrich, D. (2011). *Rote Liste und Artenverzeichnis der Flechten und flechtenbewohnenden Pilze Deutschlands* (*Vol. 70*) *[Red List and species index of lichen and lichen-inhabited mushrooms of Germany* (*Vol. 70*)].* Naturschutz und Biologische Vielfalt.

Wolf, A., Callaghan, T. V., & Larson, K. (2008). Future changes in vegetation and ecosystem function of the Barents Region. *Climatic Change*, *87*(1–2), 51–73. http://doi.org/10.1007/s10584-007-9342-4

Wolf, A., Lazzarotto, P., & Bugmann, H. (2012). The relative importance of land use and climatic change in Alpine catchments. *Climatic Change*, *111*(2), 279–300. https://doi.org/10.1007/s10584-011-0209-3

Wolseley, P. A. (1995). A global perspective on the status of lichens and their conservation. *Mitteilungen Der Eidgenössischen Forschungsanstalt Für Wald, Schnee Und Landschaft*, *70*, 11–27.

Woods, R. G., & Coppins, B. J. (2012). *A conservation evaluation of British lichens and lichenicolous fungi. Species status 13*. Peterborough, UK: Joint Nature Conservation Committee.

World Ocean Review. (2014). *Mineral resources.* http://worldoceanreview.com/en/wor-3/mineral-resources/

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., Jackson, J. B. C., Lotze, H. K., Micheli, F., Palumbi, S. R., Sala, E., Selkoe, K. A., Stachowicz, J. J., & Watson, R. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science*, *314*(5800), 787–90. https://doi.org/10.1126/science.1132294

Wraber T., Skoberne, P., & Watton, I.. (1989). *Rdeci seznam ogrozenih praprotnic in semenk SR Slovenije [The Red Data List of Threatened Vascular Plants in Socialist Republic of Slovenia]*.

Wright, H. L., Lake, I. R., & Dolman, P. M. (2012). Agriculture-a key element for conservation in the developing world. *Conservation Letters*, *5*(1), 11–19. https://doi.org/10.1111/j.1755-263X.2011.00208.x

WWF. (2006). *An ecoregional conservation action plan for the Caucasus. (Second edition).*

WWF. (2015). *Intact forest landscapes in Russia: current condition and losses over the last 13 years*.

WWF & TNC. (2017). *Freshwater ecoregions of the World*. Retrieved from www.feow.org

Xu, L., Myneni, R. B., Chapin III, F. S., Callaghan, T. V., Pinzon, J. E., Tucker, C. J., Zhu, Z., Bi, J., Ciais, P., Tømmervik, H., Euskirchen, E. S., Forbes, B. C., Piao, S. L., Anderson, B. T., Ganguly, S., Nemani, R. R., Goetz, S. J., Beck, P. S. a., Bunn, a. G., Cao, C., & Stroeve, J. C. (2013). Temperature and vegetation seasonality diminishment over northern lands. *Nature Climate Change*, *3*(3), 581–586. http://doi.org/10.1038/nclimate1836

Yablokov, A. V., Belkovich, V. M., & Borisov, V. I., [Яблоков, А. В., Белькович, В. М., & Борисов, В. И.]. (1972). Киты и дельфины *[Whales and dolphins]*. Moscow, USSR: Наука [Science].

Yablokov, A. V., Koltsov, N. K., Levchenko, V. F., Sechenov, I. M., & Kerzhentsev, A. S. [Яблоков, А. В. Кольцов, Н. К., Левченко В. Ф., Сеченов И. М., & Керженцев, А. С.]. (2014). Переход к управляемой эволюции биосферы - выход из глобального экологического кризиса [The transition to a managed evolution of biosphere - the way out of the global environmental crisis]. Астраханский Вестник Экологического Образования *[Astrakhan Herald of Ecological Education]*, (3(29)), 28–37.

Yakubov, Kh.E., Yakubov, M.A., & Yakubov, S.Kh. [Якубов, Х. Э., Якубов, М. А., & Якубов, С. Х.]. (2011). Коллекторно-дренажный сток Центральной Азии и оценка его использования на орошение *[Collector-drainage waters in Central Asia and estimation of their use for irrigation]*.

Yakushev, E. V. (1999). An approach to modelling anoxic conditions in the Black Sea. In S. T. Besiktepe, Ü. Ünlüata, & A. S. Bologa (Eds.), *Environmental degradation of the Black Sea: Challenges and remedies: Proceedings of the NATO advanced research workshop* (*Nato science partnership subseries: 2* (pp. 93–108).

Yankova, M., Pavlov, D., Ivanova, P., Karpova, E., Boltachev, A., Öztürk, B., Bat, L., Oral, M., & Mgeladze, M. (2014). Marine fishes in the Black Sea: Recent conservation status. *Mediterranean Marine Science*, *15*(2), 366–379. https://doi.org/10.12681/mms.700

Yoccoz, N. G., Delestrade, A., & Loison, A. (2010). Impact of climatic change on alpine ecosystems: inference and prediction. *Revue de Géographie Alpine*, *98*(4), 355-366. Retrieved from: https://journals.openedition.org/rga/1293

Yokes, A., & Baki, M. (2012). *Alien opisthobranchs from Turkish coasts: first record of* Plocamopherus tilesii *Bergh, 1877 from the Mediterranean.*

Yom-Tov, Y. (2003). Poaching of Israeli wildlife by guest workers. *Biological Conservation*, *110*, 11–20. https://doi.org/10.1016/S0006-3207(02)00169-6

Yusifov, E. F., & Hajiyev, V. J. (2004). *Hyrkan Biospher reservation*. Baku, Azerbaijan: Elm.

Zacharias, I., Dimitriou, E., Dekker, A., & Dorsman, E. (2007). Overview of temporary ponds in the Mediterranean region: Threats, management and conservation issues *Journal of Environmental Biology, 28*(1), 1–9.

Zagmajster, M., Culver, D. C., & Sket, B. (2008). Species richness patterns of obligate subterranean beetles (Insecta: Coleoptera) in a global biodiversity hotspot - Effect of scale and sampling intensity. *Diversity and Distributions, 14*(1), 95-105. https://doi.org/10.1111/j.1472-4642.2007.00423.x

Zaharia, T., Maximov, V., Radu, G., Anton, E., Spinu, A., & Nenciu, M. (2014). Reconciling fisheries and habitat protection in Romanian coastal marine protected areas. *Scientia Marina*, *78*(S1), 95–101. https://doi.org/10.3989/scimar.04028.25B

Zaitsev, Y., & Mamaev, V. (1997). *Marine biological diversity in the Black Sea: a study of change and decline*. New York, USA: GEF BSEP United Nations Publications.

Zalota, A. K., & Spiridonov, V. A. (2015). Understanding and forecasting invasive marine decapods distribution in the waters of Northern Eurasia. In *Abstracts of the 50ᵗʰ European Marine Biological Symposium 21-25 September* (pp. 23-23).

Zamin, T. J., Baillie, J. E. M., Miller, R. M., Rodríguez, J. P., Ardid, A., & Collen, B. (2010). National red listing beyond the 2010 target. *Conservation Biology*, 24(4), 1012–1020. https://doi.org/10.1111/j.1523-1739.2010.01492.x

Zarfl, C., Lumsdon, A. E., & Tockner, K. (2015). A global boom in hydropower dam construction. *Aquatic Sciences*, 77, 161–170. https://doi.org/10.1007/s00027-014-0377-0

Zarzycki, K., & Kaźmierczakowa, R. (Eds.). (2001). *Polska czerwona księga roślin. Paprotniki i rośliny kwiatowe [Polish red book of plants. Pteridophytes and flowering plants]*. Cracow, Poland: W. Szafer Institute of Botany.

Zavaleta, E. S., Pasari, J. R., Hulvey, K. B., & Tilman, G. D. (2010). Sustaining multiple ecosystem functions in grassland communities requires higher biodiversity. *Proceedings of the National Academy of Sciences of the United States of America, 107*(4), 1443–1446. https://doi.org/10.1073/pnas.0906829107

Zavialov, P. O. (2005). *Physical oceanography of the dying Aral Sea*. Berlin, Germany: Springer. https://doi.org/10.1007/3-540-27234-8_1

Zazanashvili, N., & Mallon, D. (2009). *Status and protection of globally threatened species in the Caucasus*. Tbilisi, Georgia: CEPF, WWF.

Zechmeister, H. G., & Moser, D. (2001). The influence of agricultural land-use intensity on bryophyte species richness. *Biodiversity and Conservation*, 10(10), 1609–1625. https://doi.org/10.1023/a:1012008828522

Zektser, I. S. (2000). *Groundwater and the environment: Applications for the global community*. Boca Raton, USA: CRC Press. Retrieved from https://books.google.ru/books?hl=ru&lr=&id=V1g5SWyevXUC&oi=%20fnd&pg=PP1&dq=Zektser+I.S.,+

Everett+L.G.+Groundwater+and+the+Environment:+Applications+for+the+Global+Community&ots=ZlsiUrsneX&sig=SzFGIJ7AdJjO-NzpTZsPeTaumuM&redir_esc=y#v=onepage&q=Zektser%20I.S.%25&f=false

Zemp, M., Haeberli, W., Hoelzle, M., & Paul, F. (2006). Alpine glaciers to disappear within decades? *Geophysical Research Letters, 33*(13), L13504. https://doi.org/10.1029/2006GL026319

Zhakova, L. V. [Жакова, Л. В.]. (2013). О влиянии многолетних изменений солености Аральского моря на динамику сообществ макрофитов [Effect of long-term changes of the salinity on the water flora composition and distribution of macrophytes in Aral]. Труды Зоологического Института РАН *[Proceedings of the Zoological Institute of the Russian Academy of Sciences]*, (Annex 3), 113–119. Retrieved from https://www.zin.ru/journals/trudyzin/doc/vol_317_s2/TZ_317_2_Supplement_Zhakova.pdf

Zimina, O. L., Lyubin, P. A., Jørgensen, L. L., Zakharov, D. V., & Lyubina, O. S. (2015). Decapod crustaceans of the Barents Sea and adjacent waters: species composition and peculiarities of distribution. *Arthropoda Selecta, 24*(3), 417–428.

Zimnitskiy, A. V., Efremov, Yu. V., & Ilyichev, Yu. G. [Зимницкий, А. В., Ефремов, Ю. В., & Ильичев, Ю. Г.]. (2015). Современное оледенение Передней Азии (в границах Турции) [Present-day glaciation of Western Asia (on the Turkey territory)]. Лед и снег [Ice and snow], 55(4), 50-60.

Ziv, B., Saaroni, H., Pargament, R., Harpaz, T., & Alpert, P. (2014). Trends in rainfall regime over Israel, 1975-2010, and their relationship to large-scale variability. *Regional Environmental Change, 14*(5), 1751-1764. https://doi.org/10.1007/s10113-013-0414-x

Zoï. (2009). *Climate change in Central Asia by Zoi environment - issuu*. Retrieved September 20, 2017, from https://issuu.com/zoienvironment/docs/climate_change

Zoï. (2011). *Biodiversity in Central Asia. A visual synthesis*. Retrieved from http://www.zoinet.org/web/sites/default/files/publications/Biodiversity-CA-EN.pdf

Zoï. (2012). *Vital Caspian graphics 2 - Opportunities, aspirations and challenges*. Retrieved from http://www.grida.no/publications/vg/caspian2/

Zonn, I. S., Glantz, M. H., Kostianoy, A. G., & Kosarev, A. N. (2009). *The Aral Sea encyclopedia*. Berlin, Germany: Springer-Verlag. https://doi.org/10.1007/978-3-540-85088-5_10

Zotz, G., & Bader, M. Y. (2009). Epiphytic plants in a changing world-global: Change effects on vascular and non-vascular epiphytes. In U. Lüttge, W. Beyschlag, B. Büdel, & D. Francis (Eds.), *Progress in Botany 70* (pp. 147–170). Berlin, Germany: Springer. https://doi.org/10.1007/978-3-540-68421-3

Zvereva, E. L., & Kozlov, M. V. (2011). Impacts of industrial polluters on bryophytes: A meta-analysis of observational studies. *Water, Air, and Soil Pollution*, 218(1–4), 573–586. https://doi.org/10.1007/s11270-010-0669-5

Zwick, P. (2004). Key to the West Palearctic genera of stoneflies (Plecoptera) in the larval stage. *Limnologica - Ecology and Management of Inland Waters, 34*, 315–348.

ANNEX I

Glossary

A

Abundance (ecological)
The size of a population of a particular life form in a given area.

Acceptance
Acceptance of IPBES outputs at a session of its Plenary signifies that the material has not been subjected to line-by-line discussion and agreement, but nevertheless presents a comprehensive and balanced view of the subject matter.

Acidification
Ongoing decrease in pH away from neutral value of 7. Often used in reference to oceans, freshwater or soils, as a result of uptake of carbon dioxide from the atmosphere.

Actor
Individual person or group representative that is involved in a specific decision-making context.

Adaptation
Adjustment in natural or human systems to a new or changing environment, whether through genetic or behavioural change.

Adaptive capacity
The general ability of institutions, systems, and individuals to adjust to potential damage, to take advantage of opportunities, or to cope with the consequences.

Adaptive management
A systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices. In active adaptive management, management is treated as a deliberate experiment for purposes of learning.

Afforestation
Converting grasslands or shrublands into tree plantations. Afforestation is sometimes suggested as a tool to sequester carbon, but it can have negative impacts on biodiversity and ecosystem function, for example by reducing runoff and so decreasing water production.

Agenda setting
One of four phases in the policy cycle. Agenda setting motivates and sets the direction for policy design and implementation.

Agri-environmental schemes
Schemes that provide funding to farmers and land managers to farm in ways that support biodiversity, enhance the landscape, and improve the quality of water, air and soil (see also agroecology as integral to such schemes).

Agricultural intensification
An increase in agricultural production per unit of input (which may be labour, land, time, fertilizer, seed, feed or cash).

Agrobiodiversity
Agrobiodiversity or agricultural biodiversity is the biological diversity that sustains key functions, structures and processes of agricultural ecosystems. It includes the variety and variability of animals, plants and micro-organisms, at the genetic, species and ecosystem levels.

Agroecology
The science and practice of applying ecological concepts, principles and knowledge (i.e., the interactions of, and explanations for, the diversity, abundance and activities of organisms) to the study, design and management of sustainable agroecosystems. It includes the roles of human beings as a central organism by way of social and economic processes in farming systems. Agroecology examines the roles and interactions among all relevant biophysical, technical and socioeconomic components of farming systems and their surrounding landscapes.

Agroecosystem
An ecosystem, dominated by agriculture, containing assets and functions such as biodiversity, ecological succession and food webs. An agroecosystem is not restricted to the immediate site of agricultural activity (e.g. the farm), but rather includes the region that is impacted by this activity, usually by changes to the complexity of species assemblages and energy flows, as well as to the net nutrient balance.

Agroforestry
A collective name for land-use systems and technologies where woody perennials (trees, shrubs, palms, bamboos, etc.) are deliberately used on the same land-management units as agricultural crops and/or animals, in some form of spatial arrangement or temporal sequence.

Aichi Biodiversity Targets
The 20 targets set by the Conference of the Parties to the Convention for Biological Diversity (CBD) at its tenth meeting, under the Strategic Plan for Biodiversity 2011-2020.

Alien species
See "invasive alien species".

Annual
In botany, refers to plants that grow from seed to maturity, reproduction and death in one year. Related terms are biennial (plants that take two years to complete their life cycles), and perennial (plants that take several many years to complete their life cycles).

Anthropocentric value
See "values".

Anthropogenic assets
Built-up infrastructure, health facilities, or knowledge - including indigenous and local knowledge systems and technical or scientific knowledge - as well as formal and non-formal education, work, technology (both physical objects and procedures), and financial assets. Anthropogenic assets have been highlighted to emphasize that a good quality of life is achieved by a co-production of benefits between nature and people.

Approval
Approval of IPBES outputs signifies that the material has been subject to detailed, line-by-line discussion and agreement by consensus at a session of the Plenary.

Aquaculture
The farming of aquatic organisms, including fish, molluscs, crustaceans and aquatic plants, involving interventions such as regular stocking, feeding, protection from predators, to enhance production. (In contrast, aquatic organisms which are exploitable by the public as a common property resource, are classed as fisheries, not aquaculture).

Archetypes
In the context of scenarios, an over-arching scenario that embodies common characteristics of a number of more specific scenarios.

Aridification
A chronic reduction in soil moisture caused by an increase of mean annual temperature or a decrease in yearly precipitation.

Assessment reports
Assessment reports are published outputs of scientific, technical and socioeconomic issues that take into account different approaches, visions and knowledge systems, including global assessments of biodiversity and ecosystem services with a defined geographical scope, and thematic or methodological assessments based on the standard or the fast-track approach. They are composed of two or more sections including a summary for policymakers, an optional technical summary, and individual chapters and their executive summaries. Assessments are the major output of IPBES, and they contain syntheses of findings on topics that have been selected by the IPBES Plenary.

B

Baseline
A minimum or starting point to which to compare other information (e.g. for comparisons between past and present or before and after an intervention).

Beneficiary
Different social actors and groups who may be benefiting from nature and its contributions to people in different ways and to different degrees, including individual, household or collective levels.

Benefit sharing
Distribution of benefits between stakeholders.

Benefits
Advantage that contribute to well-being from the fulfilment of needs and wants. In the context of nature's contributions to people.

Benthic
Occurring at the bottom of a body of water; related to benthos.

Benthos
A group of organisms, including invertebrates, that live in or on the bottom in aquatic habitats.

Biocapacity
The capacity of a country, a region, or the world, to produce useful biological materials for its human population and to absorb waste materials.

Biocentric perspectives
Recognizing the importance of non-human life.

Biocultural diversity
The diversity exhibited collectively by natural and cultural systems. It incorporates three concepts: firstly, that the diversity of life includes human cultures and languages; secondly, that links exist between biodiversity and human cultural diversity; and finally, that these links have developed over time through mutual adaptation and possibly co-evolution between humans, plants and animals.

Biodiversity
The variability among living organisms from all sources including terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part. This includes variation in genetic, phenotypic, phylogenetic, and functional attributes, as well as changes in abundance and distribution over time and space within and among species, biological communities and ecosystems.

Biodiversity hotspot
A generic term for an area high in such biodiversity attributes as species richness or endemism. It may also be used in assessments as a precise term applied to geographic areas defined according to two criteria: (i) containing at least 1,500 species of the world's 300,000 vascular plant species as endemics, and (ii) being under threat, in having lost 70% of its primary vegetation.

Biodiversity loss
The reduction of any aspect of biological diversity (i.e. diversity at the genetic, species and ecosystem levels) is lost in a particular area through death (including extinction), destruction or manual removal; it can refer to many scales, from global extinctions to population extinctions, resulting in decreased total diversity at the same scale.

Biodiversity offset
A biodiversity offset is a tool proposed by developers and planners for compensating for the loss of biodiversity in one place by biodiversity gains in another.

Biofuel
Fuel made from biomass.

Biological diversity
See "biodiversity".

Biomass
The mass of non-fossilized and biodegradable organic material originating from plants, animals and micro-organisms in a given area or volume.

Biome
Biomes are global-scale zones, generally defined by the type of plant life that they support in response to average rainfall and temperature patterns. For example, tundra, coral reefs or savannas.

Biosphere
The sum of all the ecosystems of the world. It is both the collection of organisms living on the Earth and the space that they occupy on part of the Earth's crust (the lithosphere), in the oceans (the hydrosphere) and in the atmosphere. The biosphere is all the planet's ecosystems.

Biota
All living organisms of an area; the flora and fauna considered as a unit.

Biotic homogenization
See "homogenization".

Bureau
The IPBES Bureau is a subsidiary body established by the Plenary which carries out the governance functions of IPBES. It is made up of representatives nominated from each of the United Nations regions, and is chaired by the Chair of IPBES.

Bushmeat
Meat for human consumption derived from wild animals.

Bycatch
The commercially undesirable species caught during a fishing process.

C

Cap-and-trade
An economic policy instrument in which the State sets an overall environmental target (the cap) and assigns environmental impact allowances (or quotas) to actors that they can trade among each other.

Capacity-building (or capacity development)

Defined by the United Nations Development Programme as "the process through which individuals, organisations and societies obtain, strengthen and maintain their capabilities to set and achieve their own development objectives over time". IPBES promotes and facilitates capacity-building, to improve the capacity of countries to make informed policy decisions on biodiversity and ecosystem services.

Carbon cycle

The carbon cycle is the process by which carbon is exchanged among the ecosystems of the Earth.

Carbon sequestration

The long-term storage of carbon in plants, soils, geologic formations, and the ocean. Carbon sequestration occurs both naturally and as a result of anthropogenic activities and typically refers to the storage of carbon that has the immediate potential to become carbon dioxide gas.

Carbon storage

The technological process of capturing waste carbon dioxide from industry or power generation, and storing it so that it will not enter the atmosphere.

Carrying capacity

In ecology, the carrying capacity of a species in an environment is the maximum population size of the species that the environment can sustain indefinitely. The term is also used more generally to refer to the upper limit of habitats, ecosystems, landscapes, waterscapes or seascapes to provide tangible and intangible goods and services (including aesthetic and spiritual services) in a sustainable way.

Catalogue of policy support tools and methodologies

The IPBES catalogue of policy support tools and methodologies is an evolving online resource with two main goals. The first goal is to enable decision-makers to gain easy access to information on policy support tools and methodologies to better inform and assist the different phases of policymaking and implementation. The second goal is to allow a range of users to provide input to the catalogue and assess the usability of tools and methodologies in their specific contexts, including resources required and types of outputs that can be obtained, thus helping to identify and bridge gaps with respect to available tools and methodologies.

Certainty

In the context of IPBES, the summary terms to describe the state of knowledge are the following:
- Well established (certainty term): comprehensive meta-analysis or other synthesis or multiple independent studies that agree.
- Established but incomplete (certainty term): general agreement although only a limited number of studies exist but no comprehensive synthesis and, or the studies that exist imprecisely address the question.
- Unresolved (certainty term): multiple independent studies exist but conclusions do not agree.
- Inconclusive (certainty term): limited evidence, recognising major knowledge gaps.

Citizens/laypeople

Actors living in the area / context of interest that are directly or indirectly impacted by decisions / recommendations and hold their own (subjective) interest.

Climate change

As defined in Article 1 of the UNFCCC, "a change of climate which is attributed directly or indirectly to human activity that alters the composition of the global atmosphere and which is in addition to natural climate variability observed over comparable time periods".

Co-management

Process of management in which government shares power with resource users, with each given specific rights and responsibilities relating to information and decision-making.

Co-production

In the context of the IPBES conceptual framework, this is the joint contribution by nature and anthropogenic assets in generating nature's contributions to people.

Community based monitoring and information systems (CBMIS)

Initiatives by indigenous peoples and local community organizations to monitor their community's well-being and the state of their territories and natural resources, applying a mix of traditional knowledge and innovative tools and approaches. It is a system that promotes evidence-based policymaking while empowering communities to participate in the process.

Community-based natural resource management

Community-based natural resource management: an approach to natural resource management that involves the full participation of indigenous peoples' and local communities and resource users in decision-making activities, and the incorporation of local institutions, customary practices, and knowledge systems in management, regulatory, and enforcement processes. Under this approach, community-based monitoring and information systems are initiatives by indigenous peoples and local community organizations to monitor their community's well-being and the state of their territories and natural resources, applying a mix of traditional knowledge and innovative tools and approaches.

Confidence

See "certainty".

Conservation agriculture

Approach to managing agro-ecosystems for improved and sustained productivity, increased profits and food security while preserving and enhancing the resource base and the environment. It is characterized by three linked principles, namely: 1) continuous minimum mechanical soil disturbance; 2) permanent organic soil cover; and 3) diversification of crop species grown in sequences and/or associations. This covers a wide range of approaches from minimum till to permaculture/"mimicking nature".

Corridor

A geographically defined area which allows species to move between landscapes, ecosystems and habitats, natural or modified, and ensures the maintenance of biodiversity and ecological and evolutionary processes.

Cost-benefit analysis

A technique designed to determine the feasibility of a project or plan by quantifying its costs and benefits.

Cropland

A land cover/use category that includes areas used for the production of crops for harvest.

Cross-scale analysis

Cross-scale effects are the result of spatial

and/or temporal processes interacting with other processes at another scale. These interactions create emergent effects that can be difficult to predict.

Cross-sectoral
Relating to interactions between sectors (that is, the distinct parts of society, or of a nation's economy), such as how one sector affects another sector, or how a factor affects two or more sectors.

Customary law
Law consisting of commonly repeated customs, practices and beliefs that are accepted as legal requirements or obligatory rules of conduct.

D

Decomposition
Breakdown of complex organic substances into simpler molecules or ions by physical, chemical and/or biological processes.

Deforestation
Human-induced conversion of forested land to non-forested land. Deforestation can be permanent, when this change is definitive, or temporary when this change is part of a cycle that includes natural or assisted regeneration.

Degraded land
Land in a state that results from persistent decline or loss of biodiversity and ecosystem functions and services that cannot fully recover unaided.

Denitrification
Reduction of nitrates and nitrites to nitrogen by microorganisms.

Desertification
Land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities. Desertification does not refer to the natural expansion of existing deserts.

Direct driver
See "driver".

Distributional equity/justice
Allocation of costs, benefits, risks and responsibilities as well as of the products of nature requiring the disaggregation of values to highlight who benefits and who loses, and to demonstrate the consequences for those affected.

Double counting of services
Erroneously including the same ecosystem service more than once in an economic analysis.

Downscaling
The transformation of information from coarser to finer spatial scales through statistical modelling or spatially nested linkage of structural models.

Driver
In the context of IPBES, drivers of change are all the factors that, directly or indirectly, cause changes in nature, anthropogenic assets, nature's contributions to people and a good quality of life.

Drylands
Arid, semi-arid and dry sub-humid areas. The term excludes hyper-arid areas, also known as deserts. Drylands are characterized by water scarcity and cover approximately 40% of the world's terrestrial surface.

E

Ecoregion
A large area of land or water that contains a geographically distinct assemblage of natural communities that: (a) Share a large majority of their species and ecological dynamics; (b) Share similar environmental conditions, and; (c) Interact ecologically in ways that are critical for their long-term persistence (source: WWF). In contrast to biomes, an ecoregion is generally geographically specific, at a much finer scale. For example, the "East African Montane Forest" ecoregion of Kenya (WWF ecoregion classification) is a geographically specific and coherent example of the globally occurring "tropical and subtropical forest" biome.

Ecological community
An assemblage or association of populations of two or more different species occupying the same geographical area and in a particular time.

Ecological footprint
A measure of the amount of biologically productive land and water required to support the demands of a population or productive activity. Ecological footprints can be calculated at any scale: for an activity, a person, a community, a city, a region, a nation or humanity as a whole.

Ecological infrastructure
Ecological infrastructure refers to the natural or semi-natural structural elements of ecosystems and landscapes that are important in delivering ecosystem services. It is similar to "green infrastructure", a term sometimes applied in a more urban context. The ecological infrastructure needed to support pollinators and improve pollination services includes patches of semi-natural habitats, including hedgerows, grassland and forest, distributed throughout productive agricultural landscapes, providing nesting and floral resources. Larger areas of natural habitat are also ecological infrastructure, although these do not directly support agricultural pollination in areas more than a few kilometres away from pollinator-dependent crops.

Economic and financial instruments
Economic and financial instruments can be used to change people's behaviour towards desired policy objectives. Instruments typically encompass a wide range of designs and implementation approaches. They include traditional fiscal instruments, including for example subsidies, taxes, charges and fiscal transfers. Additionally, instruments such as tradable pollution permits or tradable land development rights rely on the creation of new markets. Further instruments represent conditional and voluntary incentive schemes such as payments for ecosystem services. All these can in principle be used to correct for policy or/and market failures and reinstate full-cost pricing. They aim at reflecting social costs or benefits of the conservation and use of biodiversity and ecosystem services of a public good nature ("getting the price right"). Financial instruments, in contrast, are often extra-budgetary and can be financed from domestic sources or foreign aid, external borrowing, debt for nature swaps, etc. Economic instruments do not necessarily imply that commodification of environmental functions is promoted. Generally, they are meant to change behaviour of individuals (e.g., consumers and producers) and public actors (e.g., local and regional governments).

Economic valuation
See "values".

Ecosystem
A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit.

Ecosystem accounting
The process of constructing formal accounts for ecosystems.

Ecosystem degradation
A persistent (long-time) reduction in the capacity to provide ecosystem services.

Ecosystem function
The flow of energy and materials through the biotic and abiotic components of an ecosystem. It includes many processes such as biomass production, trophic transfer through plants and animals, nutrient cycling, water dynamics and heat transfer.

Ecosystem health
Ecosystem health is a metaphor used to describe the condition of an ecosystem, by analogy with human health. Note that there is no universally accepted benchmark for a healthy ecosystem. Rather, the apparent health status of an ecosystem can vary, depending upon which metrics are employed in judging it, and which societal aspirations are driving the assessment.

Ecosystem management
An approach to maintaining or restoring the composition, structure, function, and delivery of services of natural and modified ecosystems for the goal of achieving sustainability. It is based on an adaptive, collaboratively developed vision of desired future conditions that integrates ecological, socioeconomic, and institutional perspectives, applied within a geographic framework, and defined primarily by natural ecological boundaries.

Ecosystem services
The benefits people obtain from ecosystems. In the Millennium Ecosystem Assessment, ecosystem services can be divided into supporting, regulating, provisioning and cultural. This classification, however, is superseded in IPBES assessments by the system used under "nature's contributions to people." This is because IPBES recognises that many services fit into more than one of the four categories. For example, food is both a provisioning service and also, emphatically, a cultural service, in many cultures.

Ecotourism
Sustainable travel undertaken to access sites or regions of unique natural or ecological quality, promoting their conservation, low visitor impact, and socio-economic involvement of local populations.

Endangered species
A species at risk of extinction in the wild.

Endemic species
Plants and animals that exist only in one geographic region.

Endemism
The ecological state of a species being unique to a defined geographic location, such as an island, nation, country or other defined zone, or habitat type; organisms that are indigenous to a place are not endemic to it if they are also found elsewhere.

Energy security
Access to clean, reliable and affordable energy services for cooking and heating, lighting, communications and productive uses.

Equity
Fairness of rights, distribution, and access. Depending on context, this can refer to resources, services, or power.

Eutrophic
A condition of an aquatic system in which increased nutrient loading leads to progressively increasing amounts of algal growth and biomass accumulation. When the algae die off and decompose, the amount of dissolved oxygen in the water becomes reduced. The term is sometimes applied more broadly than just to aquatic systems.

Eutrophication
Nutrient enrichment of an ecosystem, generally resulting in increased primary production and reduced biodiversity. In lakes, eutrophication leads to seasonal algal blooms, reduced water clarity, and, often, periodic fish mortality as a consequence of oxygen depletion. The term is most closely associated with aquatic ecosystems but is sometimes applied more broadly.

Exclusive economic zone
An exclusive economic zone (EEZ) is a concept adopted at the Third United Nations Conference on the Law of the Sea (1982), whereby a coastal State assumes jurisdiction over the exploration and exploitation of marine resources in its adjacent section of the continental shelf, taken to be a band extending 200 miles from the shore. The exclusive economic zone comprises an area which extends either from the coast, or in federal systems

from the seaward boundaries of the constituent states (3 to 12 nautical miles, in most cases) to 200 nautical miles (370 kilometres) off the coast. Within this area, nations claim and exercise sovereign rights and exclusive fishery management authority over all fish and all continental shelf fishery resources.

Externality
A positive or negative consequence (benefit or cost) of an action that affects someone other than the agent undertaking that action and for which the agent is neither compensated nor penalized through the markets.

Extinction debt
The future extinction of species due to events in the past, owing to a time lag between an effect such as habitat destruction or climate change, and the subsequent disappearance of species.

F

Feedback
The modification or control of a process or system by its results or effects.

Food security
The World Food Summit of 1996 defined food security as existing "when all people at all times have access to sufficient, safe, nutritious food to maintain a healthy and active life".

Formal institutions
Include law and policies e.g., regulations and directives, and fiscal, agricultural or planning policies, to name just a few examples. These are typically based on legal instruments, treaties and customary laws. Informal institutions in turn include social norms and rules, such as those related to collective action.

Functional diversity
The range, actual values, relative abundance and distribution of functional trait attributes in a given community.

Functional traits
Any feature of an organism, expressed in the phenotype and measurable at the individual level, which has demonstrable links to the organism's function. As such, a functional trait determines the organism's response to external abiotic or biotic factors (response trait), and/or its effects on ecosystem properties or benefits or

detriments derived from such properties (effect trait). In plants, functional traits include morphological, ecophysiological, biochemical and regeneration traits. In animals, these traits include e.g. body size, litter size, age of sexual maturity, nesting habitat, time of activity.

G

Generalist species
A species able to thrive in a wide variety of environmental conditions and that can make use of a variety of different resources (for example, a flower-visiting insect that lives on the floral resources provided by several to many different plants).

Good quality of life
Within the context of the IPBES conceptual framework – the achievement of a fulfilled human life, a notion which may vary strongly across different societies and groups within societies. It is a context-dependent state of individuals and human groups, comprising aspects such as access to food, water, energy and livelihood security, and also health, good social relationships and equity, security, cultural identity, and freedom of choice and action. "Living in harmony with nature", "living-well in balance and harmony with Mother Earth" and "human well-being" are examples of different perspectives on a "good quality of life".

Governance
The way the rules, norms and actions in a given organization are structured, sustained, and regulated.

Grassland
Type of ecosystem characterized by a more or less closed herbaceous (non-woody) vegetation layer, sometimes with a shrub layer, but – in contrast to savannas – without, or with very few, trees. Different types of grasslands are found under a broad range of climatic conditions.

H

Habitat
The place or type of site where an organism or population naturally occurs. Also used to mean the environmental attributes required by a particular species or its ecological niche.

Habitat connectivity
The degree to which the landscape facilitates the movement of organisms (animals, plant reproductive structures, pollen, pollinators, spores, etc.) and other environmentally important resources (e.g., nutrients and moisture) between similar habitats. Connectivity is hampered by fragmentation.

Habitat degradation
A general term describing the set of processes by which habitat quality is reduced. Habitat degradation may occur through natural processes (e.g. drought, heat, cold) and through human activities (forestry, agriculture, urbanization).

Habitat fragmentation
A general term describing the set of processes by which habitat loss results in the division of continuous habitats into a greater number of smaller patches of lesser total and isolated from each other by a matrix of dissimilar habitats. Habitat fragmentation may occur through natural processes (e.g., forest and grassland fires, flooding) and through human activities (forestry, agriculture, urbanization).

Habitat service
The importance of ecosystems to provide living space for resident and migratory species (thus maintaining the gene pool and nursery service).

Harmonization
The process of bringing something together, and comparing (e.g., models or scenarios) to facilitate compatibility or consistency.

Hedgerow
A row of shrubs or trees that forms the boundary of an area such as a garden, field, farm, road or right-of-way.

Hedonic pricing
An economic valuation approach that utilizes information about the implicit demand for an environmental attribute of marketed commodities.

Homogenization
When used in the ecological sense "homogenization" means a decrease in the extent to which communities differ in species composition.

Human appropriation of net primary production (HANPP)
The aggregate impact of land use on biomass available each year in ecosystems.

I

Impact assessment
A formal, evidence-based procedure that assesses the economic, social, and environmental effects of public policy or of any human activity.

Important Bird & Biodiversity Areas
A Key Biodiversity Area identified using an internationally agreed set of criteria as being globally important for bird populations.

Indicators
A quantitative or qualitative factor or variable that provides a simple, measurable and quantifiable characteristic or attribute responding in a known and communicable way to a changing environmental condition, to a changing ecological process or function, or to a changing element of biodiversity.

Indigenous and community conserved areas
Natural and modified ecosystems including significant biodiversity, ecological services and cultural values voluntarily conserved by indigenous and local communities through customary laws or other effective means.

Indigenous and local knowledge systems
Indigenous and local knowledge systems are social and ecological knowledge practices and beliefs pertaining to the relationship of living beings, including people, with one another and with their environments. Such knowledge can provide information, methods, theory and practice for sustainable ecosystem management.

Indigenous peoples and local communities
Indigenous peoples and local communities (IPLCs) are, typically, ethnic groups who are descended from and identify with the original inhabitants of a given region, in contrast to groups that have settled, occupied or colonized the area more recently. IPBES does not intend to create or develop new definitions of what constitutes "indigenous peoples and local communities.

Indirect driver
See "driver".

Indirect use value
See "values".

Institutional failure

These are often catalogued as (i) law and policy failures (e.g., perverse subsidies), (ii) market failures (externalities in the use of public goods and services), (iii) organizational failure (e.g., lack of transparency and political legitimacy in decision-making) and (iv) informal institutional failures (e.g., break of collective action norms due to erosion of trust).

Institutions

Encompasses all formal and informal interactions among stakeholders and social structures that determine how decisions are taken and implemented, how power is exercised, and how responsibilities are distributed.

Instrumental value

See "values".

Integrated assessment models

Interdisciplinary models that aim to describe the complex relationships between environmental, social, and economic drivers that determine current and future state of the ecosystem and the effects of global change, in order to derive policy-relevant insights. One of the essential characteristics of integrated assessments is the simultaneous consideration of the multiple dimensions of environmental problems.

Integrated pest management

Also known as integrated pest control. It is a broadly-based approach that integrates various practices for economic control of pests. Integrated pest management (or IPM) aims to suppress pest populations below the economic injury level (i.e., to below the level that the costs of further control outweigh the benefits derived). It involves careful consideration of all available pest control techniques and then integration of appropriate measures to discourage development of pest populations while keeping pesticides and other interventions to economically justifiable levels with minimal risks to human health and the environment. IPM emphasizes the growth of a healthy crop with the least possible disruption to agro-ecosystems and encourages natural pest control mechanisms.

Integrated valuation

See "values".

Intrinsic value

See "values".

Invasive alien species

Species whose introduction and/or spread by human action outside their natural distribution threatens biological diversity, food security, and human health and well-being. "Alien" refers to the species having been introduced outside its natural distribution ("exotic", "non-native" and "non-indigenous" are synonyms for "alien"). "Invasive" means "tending to expand into and modify ecosystems to which it has been introduced". Thus, a species may be alien without being invasive, or, in the case of a species native to a region, it may increase and become invasive, without actually being an alien species.

Invasive species

See "invasive alien species".

IPBES conceptual framework

The IPBES conceptual framework has been designed to build shared understanding across disciplines, knowledge systems and stakeholders of the interplay between biodiversity and ecosystem drivers, and of the role they play in building a good quality of life.

IUCN protected area category

IUCN protected area management categories classify protected areas according to their management objectives.

K

Key biodiversity areas

Sites contributing significantly to the global persistence of biodiversity. They represent the most important sites for biodiversity worldwide, and are identified nationally using globally standardized criteria and thresholds.

Knowledge systems

A body of propositions that are adhered to, whether formally or informally, and are routinely used to claim truth. They are organized structures and dynamic processes (a) generating and representing content, components, classes, or types of knowledge, that are (b) domain-specific or characterized by domain-relevant features as defined by the user or consumer, (c) reinforced by a set of logical relationships that connect the content of knowledge to its value (utility), (d) enhanced by a set of iterative processes that enable the evolution, revision, adaptation, and advances, and (e) subject to criteria of relevance, reliability, and quality.

L

Land degradation

Refers to the many processes that drive the decline or loss in biodiversity, ecosystem functions or services and includes the degradation of all terrestrial ecosystems.

Land sharing

A situation where low-yield farming enables biodiversity to be maintained within agricultural landscapes.

Land sparing

Land sparing, also called "land separation" involves restoring or creating non-farmland habitat in agricultural landscapes at the expense of field-level agricultural production - for example, woodland, natural grassland, wetland, and meadow on arable land. This approach does not necessarily imply high-yield farming of the non-restored, remaining agricultural land.

Land use

The human use of a specific area for a certain purpose (such as residential, agriculture, recreation, industrial, etc.). Influenced by, but not synonymous with, land cover. Land-use change refers to a change in the use or management of land by humans, which may lead to a change in land cover.

Land-use change

See "land use".

Landscape

An area of land that contains a mosaic of ecosystems, including human-dominated ecosystems.

Landscape configuration

The distribution, size and abundances of patch types represented within a landscape. Configuration is spatially explicit because it refers not only to the variety and abundance of patch types, but also to their placement or location (dispersion) in the landscape.

Leaching

The dissolution and movement of dissolved substances by water.

Living in harmony with nature

Within the context of the IPBES conceptual framework – a perspective on good quality of life based on the interdependence that exists among human beings, other living species and elements of nature. It implies that we should live peacefully alongside all

other organisms even though we may need to exploit other organisms to some degree.

M

Mainstreaming biodiversity
Mainstreaming, in the context of biodiversity, means integrating actions or policies related to biodiversity into broader development processes or policies such as those aimed at poverty reduction, or tackling climate change.

Mangrove
Group of trees and shrubs that live in the coastal intertidal zone. Mangrove forests only grow at tropical and subtropical latitudes near the equator because they cannot withstand freezing temperatures.

Meta-analysis
A quantitative statistical analysis of several separate but similar experiments or studies in order to test the pooled data for statistical significance.

Millennium Ecosystem Assessment
The Millennium Ecosystem Assessment is a major assessment of the human impact on the environment published in 2005.

Mitigation
In the context of IPBES, an intervention to reduce negative or unsustainable uses of biodiversity and ecosystems.

Models
Qualitative or quantitative representations of key components of a system and of relationships between these components. Benchmarking (of models) is the process of systematically comparing sets of model predictions against measured data in order to evaluate model performance. Validation (of models) typically refers to checking model outputs for consistency with observations. However, since models cannot be validated in the formal sense of the term (i.e. proven to be true), some scientists prefer to use the words "benchmarking" or "evaluation". A dynamic model is a model that describes changes through time of a specific process. A process-based model (also known as "mechanistic model") is a model in which relationships are described in terms of explicitly stated processes or mechanisms based on established scientific understanding, and model parameters therefore have clear ecological interpretation, defined beforehand.

Hybrid models are models that combine correlative and process-based modelling approaches. A correlative model (also known as "statistical model") is a model in which available empirical data are used to estimate values for parameters that do not have predefined ecological meaning, and for which processes are implicit rather than explicit. Integrated assessment models are interdisciplinary models that aim to describe the complex relationships between environmental, social, and economic drivers that determine current and future state of the ecosystem and the effects of global change, in order to derive policy-relevant insights. One of the essential characteristics of integrated assessments is the simultaneous consideration of the multiple dimensions of environmental problems.

Monitoring
The repeated observation of a system in order to detect signs of change.

Monoculture
The agricultural practice of producing or growing a single crop, plant, or livestock species, variety, or breed in a field or farming system at a time.

Mosaic landscape
A pattern of landscapes with multiple patches and corridors.

Mother Earth
An expression used in a number of countries and regions to refer to the planet Earth and the entity that sustains all living things found in nature with which humans have an indivisible, interdependent physical and spiritual relationship (see "nature").

Multidisciplinary expert panel
The IPBES multidisciplinary expert panel is a subsidiary body established by the IPBES Plenary which oversees the scientific and technical functions of the Platform, a key role being to select experts to carry out assessments.

N

Native species
Indigenous species of animals or plants that naturally occur in a given region or ecosystem.

Natural capital
An economic metaphor for the limited stocks of physical and biological resources found on Earth.

Nature
In the context of IPBES, nature refers to the natural world with an emphasis on its living components. Within the context of Western science, it includes categories such as biodiversity, ecosystems (both structure and functioning), evolution, the biosphere, humankind's shared evolutionary heritage, and biocultural diversity. Within the context of other knowledge systems, it includes categories such as Mother Earth and systems of life, and it is often viewed as inextricably linked to humans, not as a separate entity (see "Mother Earth").

Nature's contributions to people (NCP)
Nature's contributions to people (NCP) are all the contributions, both positive and negative, of living nature (i.e. diversity of organisms, ecosystems, and their associated ecological and evolutionary processes) to the quality of life of people. Beneficial contributions from nature include such things as food provision, water purification, flood control, and artistic inspiration, whereas detrimental contributions include disease transmission and predation that damages people or their assets. Many NCP may be perceived as benefits or detriments depending on the cultural, temporal or spatial context.

Non-anthropocentric
A non-anthropocentric value is a value centred on something other than human beings. These values can be non-instrumental (e.g. a value ascribed to the existence of specific species for their own sake) or instrumental to non-human ends (e.g. the instrumental value a habitat has for the existence of a specific species).

Non-indigenous or non-native or alien species
See "invasive alien species".

O

Ocean acidification
See "acidification".

Organic agriculture
Any system that emphasises the use of techniques such as crop rotation, compost or manure application, and biological pest control in preference to synthetic inputs. Most certified organic farming schemes prohibit all genetically modified organisms and almost all synthetic inputs. Its origins are in a holistic management system that avoids off-farm inputs, but some organic

agriculture now uses relatively high levels of off-farm inputs.

Overexploitation

Harvesting species from the wild at rates faster than natural populations can recover. Includes overfishing, and overgrazing.

P

Participatory mapping

A key method that many indigenous communities apply in order to collect data, information and monitoring and to use it in science- policy- society interface processes.

Participatory scenario development (and planning)

Approaches characterized by more interactive, and inclusive, involvement of stakeholders in the formulation and evaluation of scenarios. Aimed at improving the transparency and relevance of decision-making, by incorporating demands and information of each stakeholder, and negotiating outcomes between stakeholders.

Particulate and gaseous pollutants

Air pollutants such as ozone, nitrogen oxides and ammonia.

Particulate matter

A mixture of solid particles (dust, dirt, soot, or smoke) and liquid droplets.

Pastoralism

Extensive livestock production in rangelands.

Peatlands

Wetlands which accumulate organic plant matter in situ because waterlogging prevents aerobic decomposition and the much slower rate of the resulting anaerobic decay is exceeded by the rate of accumulation.

Pelagic

Organisms that live in the water column.

Perennial

See "annual".

Permafrost

Perennially frozen ground that occurs wherever the temperature remains below 0°C for several years.

Phytophilia

The positive effect of green vegetation in landscapes on human beings.

Plankton

Typically microscopic aquatic organisms that drift or swim weakly. Phytoplankton are the plant forms of plankton (e.g., diatoms), and are the dominant plants in the sea. Zooplankton are the animal forms of plankton.

Plenary

Within the context of IPBES – the decision-making body comprising all of the members of IPBES.

Policy instrument

Set of means or mechanisms to achieve a policy goal.

Policy mix

A combination of policy instruments which has evolved to influence the quantity and quality of biodiversity conservation and ecosystem service provision in public and private sectors.

Policy support tools

Approaches and techniques based on science and other knowledge systems that can inform, assist and enhance relevant decisions, policymaking and implementation at local, national, regional and global levels to protect nature, thereby promoting nature's benefits to people and a good quality of life.

Policy/policy tools

Instruments used by governance bodies at all scales to implement their policies. Environmental policies, for example, could be implemented through tools such as legislation, economic incentives or dis-incentives, including taxes and tax exemptions, or tradable permits and fees.

Polycentric governance

An organizational structure where multiple independent actors mutually order their relationships with one another under a general system of rules.

Poverty

A state of economic deprivation. Its manifestations include hunger and malnutrition, limited access to education and other basic services. Other corollaries of poverty are social discrimination and exclusion as well as the lack of participation in decision-making.

Precautionary principle

Pertains to risk management and states that if an action or policy has a suspected risk of causing harm to the public or to the environment, in the absence of scientific consensus that the action or policy is not harmful, the burden of proof that it is not harmful falls on those taking an action. The principle is used to justify discretionary decisions when the possibility of harm from making a certain decision (e.g., taking a particular course of action) is not, or has not been, established through extensive scientific knowledge. The principle implies that there is a social responsibility to protect the public from exposure to harm, when scientific investigation has found a plausible risk or if a potential plausible risk has been identified.

Process-based model

See "models".

Protected area

A protected area is a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values.

R

Ramsar site(s)

A Ramsar site is a wetland site designated of international importance especially as Waterfowl Habitat under the Ramsar Convention, an intergovernmental environment treaty established in 1975 by UNESCO, coming into force in 1975. Ramsar site refers to wetland of international significance in terms of ecology, botany, zoology, limnology or hydrology. Such a site meets at least one of the criteria of identifying wetlands of international importance set by Ramsar Convention and is designated by appropriate national authority to be added to Ramsar list.

Rangeland

Natural grasslands used for livestock grazing.

REDD+

Reducing emissions from deforestation and forest degradation (REDD+) is a mechanism developed by Parties to the United Nations Framework Convention on Climate Change (UNFCCC). It creates a financial value for the carbon stored in forests by offering incentives for developing countries to reduce emissions from forested lands and invest in low-carbon paths to sustainable development. Developing countries would

receive results-based payments for results-based actions. REDD+ goes beyond simply deforestation and forest degradation, and includes the role of conservation, sustainable management of forests and enhancement of forest carbon stocks.

Regime shift(s)
Substantial reorganization in system structure, functions and feedbacks that often occurs abruptly and persists over time.

Rehabilitation
Restoration activities that move a site towards a natural state baseline in a limited number of components (i.e. soil, water, and/or biodiversity), including natural regeneration, conservation agriculture, and emergent ecosystems.

Relational value
See "values".

Remediation
Any action taken to rehabilitate ecosystems.

Resilience
The level of disturbance that an ecosystem or society can undergo without crossing a threshold to a situation with different structure or outputs. Resilience depends on factors such as ecological dynamics as well as the organizational and institutional capacity to understand, manage, and respond to these dynamics.

Resolution (spatial or temporal)
See "scale".

Restoration
Any intentional activities that initiates or accelerates the recovery of an ecosystem from a degraded state.

Richness
The number of biological entities (species, genotypes, etc.) within a given sample. Sometimes used as synonym of species diversity.

Rights-based instruments and customary norms
Synergizing rights and norms for the conservation and protection of systems of Mother Earth can foster complementarity with human well-being. International and national human rights instruments whether binding or non-binding can be creatively interpreted to fit socio-ecological systems and foster resilience. Strengthening of

collective rights, customary norms and institutions of indigenous peoples and local communities, can promote adaptive governance including the equitable and fair management of natural resources.

S

Salinization
The process of increasing the salt content in soil is known as salinization. Salinization can be caused by natural processes such as mineral weathering or by the gradual withdrawal of an ocean. It can also come about through artificial processes such as irrigation.

Savanna
Ecosystem characterized by a continuous layer of herbaceous plants, mostly grasses, and a discontinuous upper layer of trees that may vary in density.

Scale
The spatial, temporal, quantitative and analytical dimensions used to measure and study any phenomenon. The temporal scale is comprised of two properties: 1) temporal extent – the total length of the time period of interest for a particular study (e.g. 10 years, 50 years, or 100 years); and 2) temporal grain (or resolution) – the temporal frequency with which data are observed or projected within this total period (e.g. at 1-year, 5-year or 10-year intervals). The spatial scale is comprised of two properties: 1) spatial extent – the size of the total area of interest for a particular study (e.g. a watershed, a country, the entire planet); and 2) spatial grain (or resolution) – the size of the spatial units within this total area for which data are observed or predicted (e.g. fine-grained or coarse-grained grid cells).

Scenarios
Representations of possible futures for one or more components of a system, particularly for drivers of change in nature and nature's contributions, including alternative policy or management options.

Seascape(s)
Seascape can be defined as a spatially heterogeneous area of coastal environment (i.e. intertidal, brackish) that can be perceived as a mosaic of patches, a spatial gradient, or some other geometric patterning. The tropical coastal "seascape" often includes a patchwork of mangroves, seagrass beds, and coral reefs that

produces a variety of natural resources and ecosystem services.

Sector
A distinct part of society, or of a nation's economy.

Semi-natural habitat(s)
An ecosystem with most of its processes and biodiversity intact, though altered by human activity in strength or abundance relative to the natural state.

Socioecological system
An ecosystem, the management of this ecosystem by actors and organizations, and the rules, social norms, and conventions underlying this management.

Soil compaction
Defined as an increase in density and a decline of porosity in a soil that impedes root penetration and movements of water and gases.

Soil degradation
The diminishing capacity of the soil to provide ecosystem goods and services as desired by its stakeholders.

Soil organic matter
Matter consisting of plant and/or animal organic materials, and the conversion products of those materials in soils.

Soil quality
Soil quality is a measure of the soil's ability to provide ecosystem and social services through its capacities to perform its functions under changing conditions. Soil quality reflects how well a soil performs the functions of maintaining biodiversity and productivity, partitioning water and solute flow, filtering and buffering, nutrient cycling, and providing support for plants and other structures.

Species
An interbreeding group of organisms that is reproductively isolated from all other organisms, although there are many partial exceptions to this rule in particular taxa. Operationally, the term species is a generally agreed fundamental taxonomic unit, based on morphological or genetic similarity, that once described and accepted is associated with a unique scientific name.

Species composition
The array of species in a specific sample, community, or area.

Species distribution mode

Species distribution models relate field observations of the presence/absence of a species to environmental predictor variables, based on statistically or theoretically derived response surfaces, for prediction and inference. The predictor variables are often climatic but can include other environmental variables.

Species richness

The number of species within a given sample, community, or area.

Stakeholders

Any individuals, groups or organizations who affect, or could be affected (whether positively or negatively) by a particular issue and its associated policies, decisions and action.

Storylines (or scenario storylines)

Qualitative narratives which provide the descriptive framework from which quantitative exploratory scenarios can be formulated.

Summary for policymakers

Is a component of any report, providing a policy-relevant but not policy prescriptive summary of that report.

Sustainability

A characteristic or state whereby the needs of the present and local population can be met without compromising the ability of future generations or populations in other locations to meet their needs.

Sustainable Development Goals (SDGs)

A set of goals adopted by the United Nations in 2015 to end poverty, protect the planet, and ensure prosperity for all, as part of the 2030 Agenda for Sustainable Development.

Sustainable use (of biodiversity and its components)

The use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations.

Synergies

See "trade-off".

T

Taxon

A category applied to a group in a formal system of nomenclature, e.g., species, genus, family etc. (plural: taxa).

Telecoupling

Refers to socioeconomic and environmental interactions over distances. It involves distant exchanges of information, energy and matter (e.g., people, goods, products, capital) at multiple spatial, temporal and organizational scales.

Threatened species

In the IUCN Red List terminology, a threatened species is any species listed in the Red List categories, critically endangered, endangered, or vulnerable.

Tipping point

A set of conditions of an ecological or social system where further perturbation will cause rapid change and prevent the system from returning to its former state.

Trade-off

A trade-off is a situation where an improvement in the status of one aspect of the environment or of human well-being is necessarily associated with a decline in or loss of a different aspect. Trade-offs characterize most complex systems, and are important to consider when making decisions that aim to improve environmental and/or socio-economic outcomes. Trade-offs are distinct from synergies (the latter are also referred to as "win-win" scenarios): synergies arise when the enhancement of one desirable outcome leads to enhancement of another.

Transformation

A change in the fundamental attributes of natural and human systems that reflect strengthened, altered, or aligned paradigms, goals, or values towards promoting adaptation that supports sustainable development, including poverty reduction.

Transhumance

Form of pastoralism or nomadism organized around the migration of livestock between mountain pastures in warm seasons and lower altitudes the rest of the year. The seasonal migration may also occur between lower and upper latitudes. A traditional farming practice based on indigenous and local knowledge.

Transitional pathways

A course of actions and strategies that aim to achieve the vision. They are closely related to "policy or target-seeking scenarios".

Trophic cascades

The chain of knock - on extinctions observed or predicted to occur following the loss of one or a few species that play a critical role (e.g. as a pollinator) in ecosystem functioning.

Trophic level

The level in the food chain in which one group of organisms serves as a source of nutrition for another group of organisms (e.g. primary producers, primary or secondary consumers, decomposers).

U

Units of analysis

Units of analysis result from subdividing the Earth's surface into units solely for the purposes of analysis. The terrestrial and aquatic units of analysis serve as a framework for comparison within and across IPBES assessments and represent a pragmatic solution. The terrestrial and aquatic units of analysis used by IPBES are not intended to be prescriptive for purposes other than those of IPBES assessments. They are likely to evolve as the work of IPBES develops.

V

Values

- Value systems: Set of values according to which people, societies and organizations regulate their behaviour. Value systems can be identified in both individuals and social groups.
- Value (as principle): A value can be a principle or core belief underpinning rules and moral judgments. Values as principles vary from one culture to another and also between individuals and groups.
- Value (as preference): A value can be the preference someone has for something or for a particular state of the world. Preference involves the act of making comparisons, either explicitly or implicitly. Preference refers to the importance attributed to one entity relative to another one.
- Value (as importance): A value can be the importance of something for itself or for others, now or in the future, close by or at a distance. This importance can be considered in three broad classes.
  1. The importance that something has

subjectively, and may be based on experience. 2. The importance that something has in meeting objective needs. 3. The intrinsic value of something.

- Value (as measure): A value can be a measure. In the biophysical sciences, any quantified measure can be seen as a value.
- Non-anthropocentric value: A non-anthropocentric value is a value centred on something other than human beings. These values can be non-instrumental or instrumental to non-human ends.
- Intrinsic value: The value inherent to nature, independent of human experience and evaluation, and therefore beyond the scope of anthropocentric valuation approaches.
- Anthropocentric value: Human-centred, the value that something has for human beings and human purposes.
- Instrumental value: The direct and indirect contribution of nature's benefits to the achievement of a good quality of life. Within the specific framework of the total economic value, instrumental values can be classified into use (direct and indirect use values) on the one hand, and non-use values (option, bequest and existence values) on the other. Sometimes option values are considered as use values as well.
- Non-instrumental value: The value attributed to something as an end in itself, regardless of its utility for other ends.
- Relational value: The values that contribute to desirable relationships, such as those among people and between people and nature, as in "living in harmony with nature".
- Integrated valuation: The process of collecting, synthesizing, and communicating knowledge about the ways in which people ascribe importance and meaning of nature's contributions, to facilitate deliberation and agreement for decision-making and planning.

Vision
A desirable future (an endpoint in time) which we want to achieve. Visions usually consist of statements depicting the explicit desires, assumptions, beliefs and paradigms that underlie the desired future.

W

Water security
The capacity of a population to safeguard sustainable access to adequate quantities of and acceptable quality water for sustaining livelihoods, human well-being, and socio-economic development, for ensuring protection against water-borne pollution and water-related disasters, and for preserving ecosystems in a climate of peace and political stability.

Water stress
Water stress occurs in an organism when the demand for water exceeds the available amount during a certain period or when poor quality restricts its use.

Well-being
A perspective on a good life that comprises access to basic resources, freedom and choice, health and physical well-being, good social relationships, security, peace of mind and spiritual experience. Well-being is achieved when individuals and communities can act meaningfully to pursue their goals and can enjoy a good quality of life. The concept of human well-being is used in many Western societies and its variants, together with living in harmony with nature, and living well in balance and harmony with Mother Earth. All these are different perspectives on a good quality of life.

Western science
Also called modern science, Western scientific knowledge or international science, and used in the context of the IPBES conceptual framework as a broad term to refer to knowledge typically generated in universities, research institutions and private firms following paradigms and methods typically associated with the "scientific method" consolidated in Post-Renaissance Europe on the basis of wider and more ancient roots. It is typically transmitted through scientific journals and scholarly books. Some of its central tenets are observer independence, replicable findings, systematic scepticism, and transparent research methodologies with standard units and categories.

Wetlands
Areas that are subject to inundation or soil saturation at a frequency and duration, such that the plant communities present are dominated by species adapted to growing in saturated soil conditions, and/or that the soils of the area are chemically and physically modified due to saturation and indicate a lack of oxygen; such areas are frequently termed peatlands, marshes, swamps, sloughs, fens, bogs, wet meadows, etc.

Worldviews
Defined by the connections between networks of concepts and systems of knowledge, values, norms and beliefs.

Individual person's worldviews are moulded by the community the person belongs to. Practices are embedded in worldviews and are intrinsically part of them (e.g. through rituals, institutional regimes, social organization, but also in environmental policies, in development choices, etc.).

ANNEX II

Acronyms

ABTs	Aichi Biodiversity Targets		**NASA**	National Aeronautics and Space Administration
AR	Assessment Report (specifically in the context of the Intergovernmental Panel on Climate Change)		**NBSAP**	National Biodiversity Strategy and Action Plan
			NCP	Nature's Contributions to People
BPA	Bisphenol A; Biodiversity Promotion Areas		**NEC Directive**	National Emissions Ceiling Directive
CAP	Common Agricultural Policy		**NGO**	Non-Governmental Organization
CBD	Convention on Biological Diversity		**OECD**	Organisation for Economic Co-operation and Development
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora		**OPERAs**	OPerationalising Ecosystem Research Applications
CMIP	Coupled Model Intercomparison Project Phase		**OSPAR**	Oslo/Paris convention (for the Protection of the Marine Environment of the North-East Atlantic)
CMS	Convention on the Conservation of Migratory Species of Wild Animals			
COP	Conference of the Parties / Conference of the Contracting Parties		**PAHs**	Polycyclic Aromatic Hydrocarbons
			PBT	Polybutylene Terephthalate
DDD	Dichlorodiphenyldichloroethane		**PCB**	Polychrorinated Byphenil
DDE	Dichlorodiphenyldichloroethylene		**POP**	Persistent Organic Pollutant
DDT	Dichlorodiphenyltrichloroethane		**RCP**	Representative Concentration Pathway
ECA	Europe and Central Asia		**RED**	Renewable Energy Directive
ECOLEX	A partnership of three organizations compiling environmental, (ECO), law (LEX) related information		**REDD+**	Reducing Emissions from Deforestation and Degradation
			SDG	Sustainable Development Goal
EEA	European Environment Agency		**SRES**	Special Report on Emissions Scenarios
EU	European Union		**SSP**	Shared Socioeconomic Pathways
FAO	Food and Agriculture Organization (of the United Nations)		**TEEB**	The Economics of Ecosystems and Biodiversity
			TgC	Teragrams of Carbon
FSC	Forest Stewardship Council		**UNDP**	United Nations Development Programme
GDP	Gross Domestic Product		**UNECE**	United Nations Economic Commission for Europe
GHG	Greenhouse Gas		**UNEP**	United Nations Environment Programme (now also known as UN Environment)
Ha	Hectare(s)			
HELCOM	Helsinki Commission		**UNESCO**	United Nations Educational, Scientific, and Cultural Organization
HWSD	Harmonized World Soil Database			
IGBP	International Geosphere-Biosphere Programme		**UNFCCC**	United Nations Framework Convention on Climate Change
IHDP	International Human Dimensions Programme			
ILK	Indigenous and Local Knowledge		**USLE**	Universal Soil Loss Equation
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services		**USSR**	Union of Soviet Socialist Republics
			WFD	Water Framework Directive
IPCC	Intergovernmental Panel on Climate Change		**WG**	Working Group
IUCN	International Union for Conservation of Nature		**WHO**	World Health Organization
LRTAP	Long-range Transboundary Air Pollution (Convention)		**WTO**	World Trade Organization
			WWF	World Wide Fund for Nature
MEA	Millennium Ecosystem Assessment			
MPA	Marine Protected Area			

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The mission of IPBES is to strengthen the science-policy interface for biodiversity and ecosystem services for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development.

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