CHAPTER 3

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

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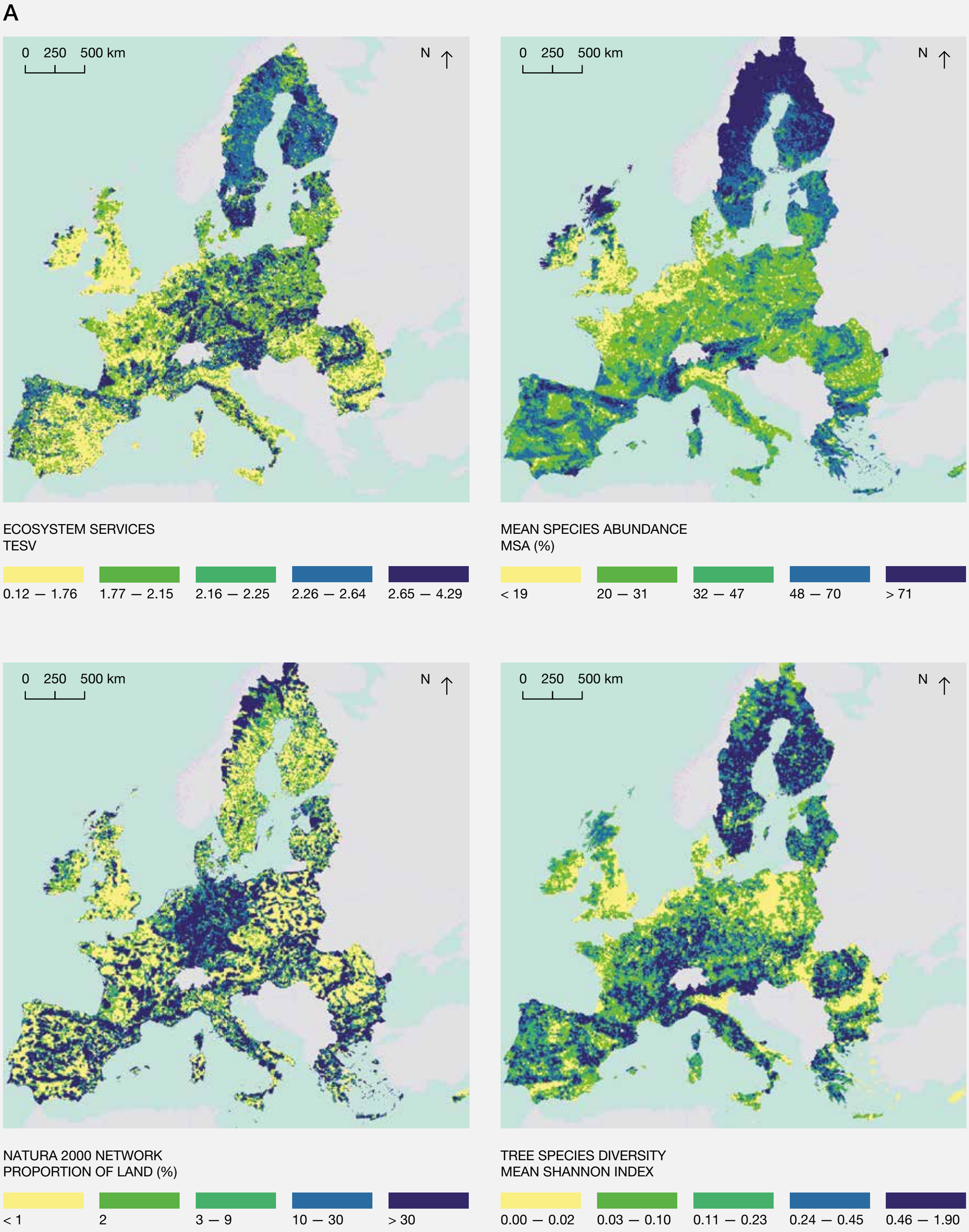
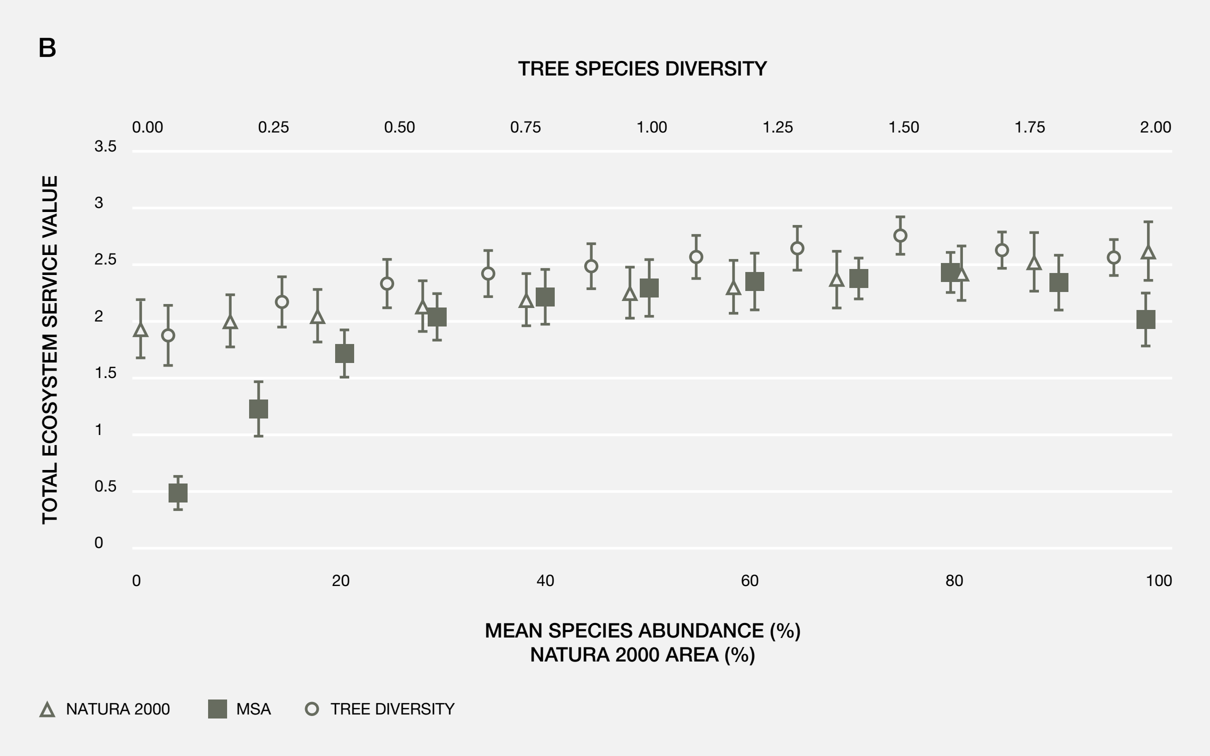
[1] Available at https://www.ipbes.net/sites/default/files/eca_ch_3_appendix_3.1_additional_references.pdf

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

Ⓐ 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. Ⓑ 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).





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:

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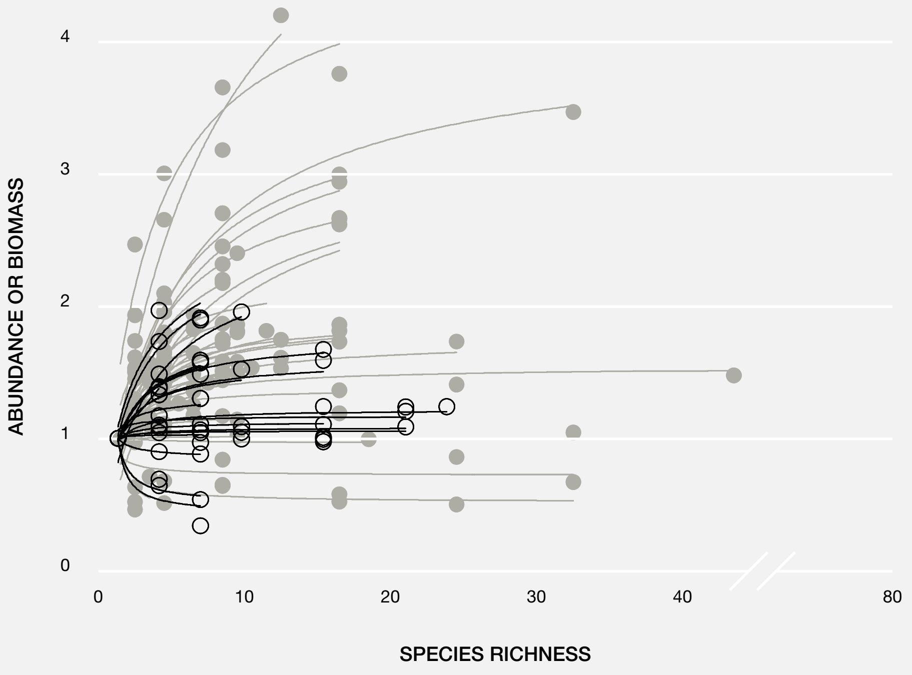
2. Available at https://www.ipbes.net/sites/default/files/eca_ch_3_appendix_3.1_additional_references.pdf

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.

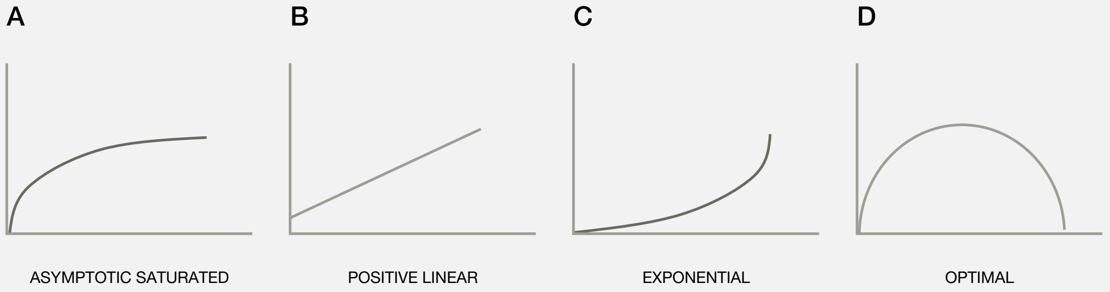


a) complementarity effects, i.e. functional complementarity in, for example, resource use of species or genotypes or phenotypes or due to positive (facilitative) species interactions; and b) selection effects, i.e. selection of particular functional traits of species or genotypes or phenotypes, with 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.**



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

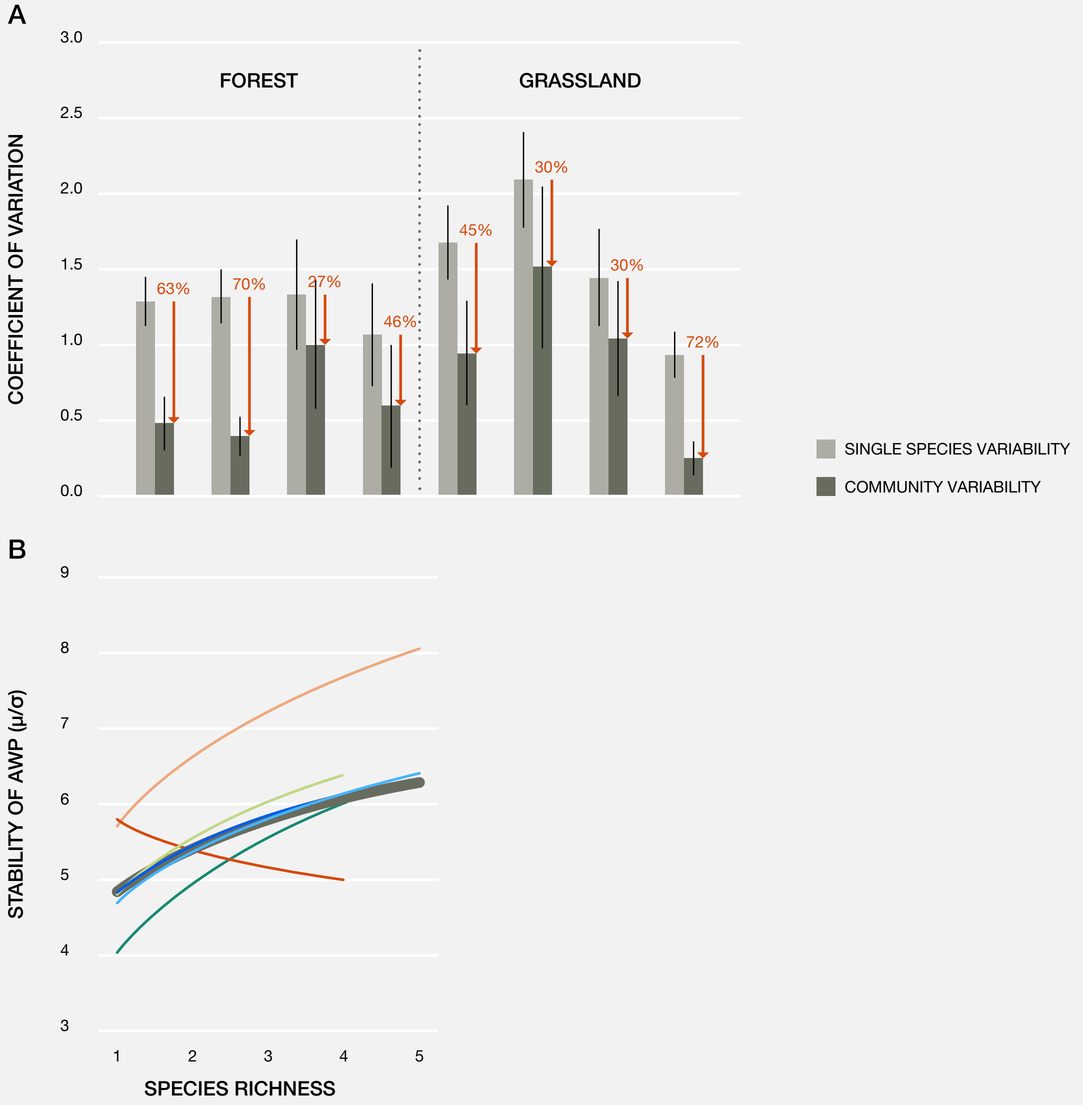
an increasing function of species richness, while population stability often decreases with species richness (Tilman *et al.*, 2014). Two main hypotheses known as the "portfolio effect" and the "insurance hypothesis" predict a stabilizing effect of species diversity. The "portfolio effect" posits higher 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 **Ⓐ 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).

Ⓑ 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.

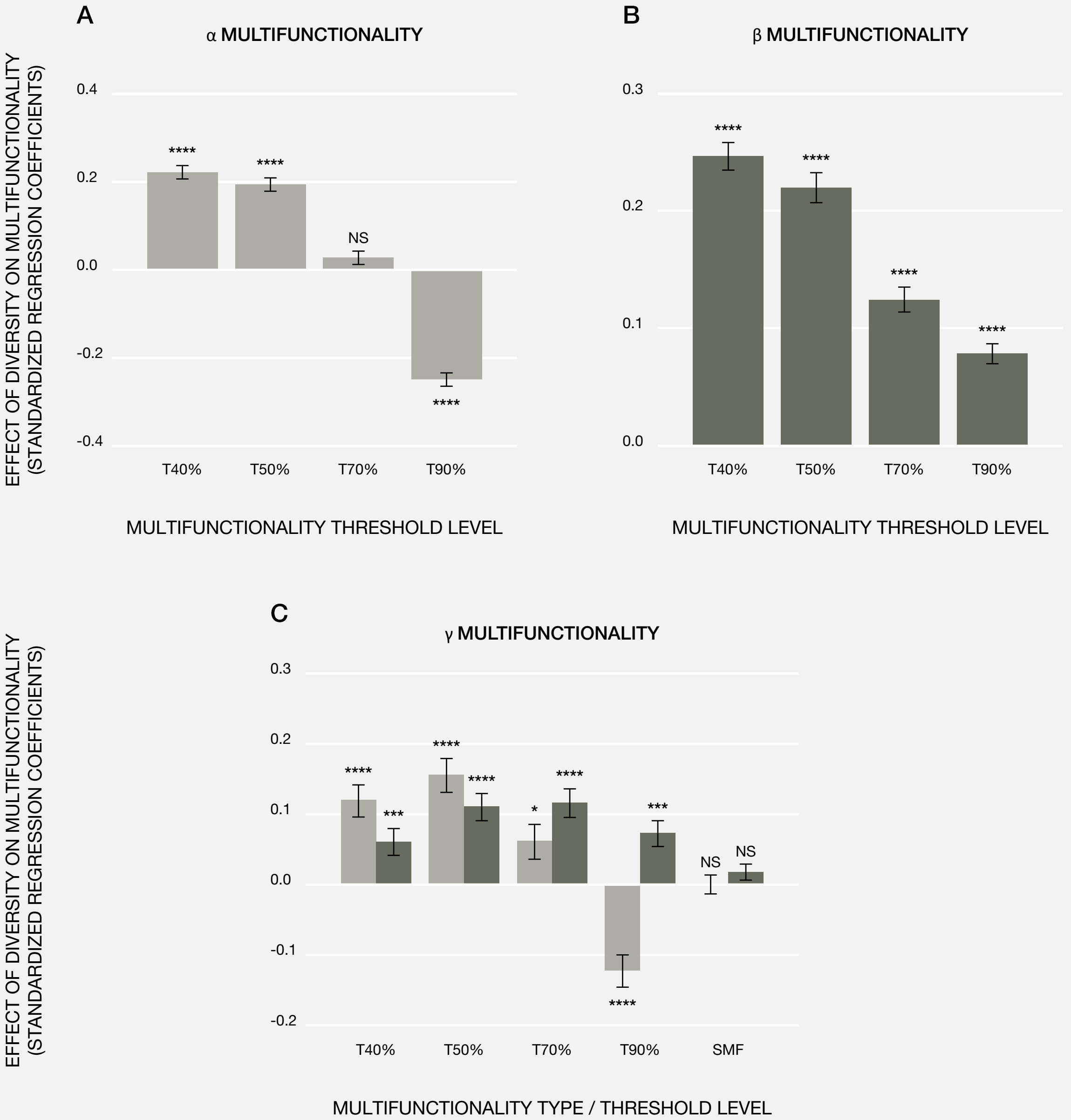
Inclusion of ecosystem multifunctionality (i.e. the provision of multiple ecosystem functions or services) in community models shows that multifunctional redundancy is generally lower than single-function redundancy. This means that a moderate loss of species can lead to a stronger loss of ecosystem 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- Ⓐ, β- Ⓑ, or γ- Ⓒ 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 km^2 in the Russian Arctic, 35,100 km^2 in Svalbard and 11,800 km^2 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 km^2 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).

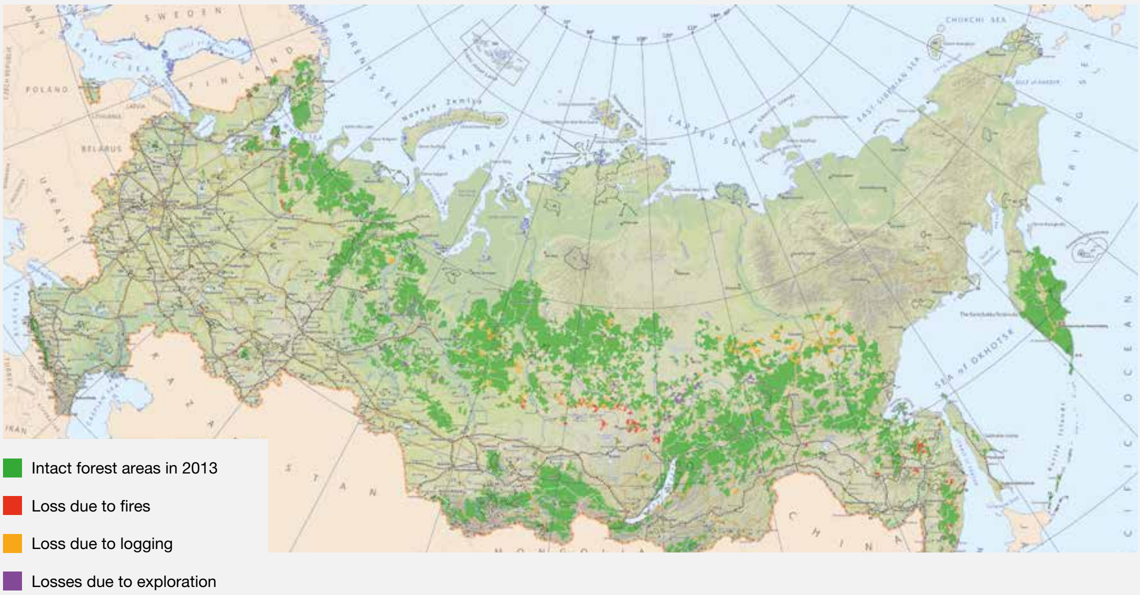
For people in remote forested areas, old-growth mountain or boreal forests are the only source of wood, and a source of food. Planted forests exclusively used for timber cover about 10% of the European Union (EEA, 2016). Throughout the forests of Europe and Central Asia, biodiversity is an important 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).**



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; Şekercioğlu *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 7 **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

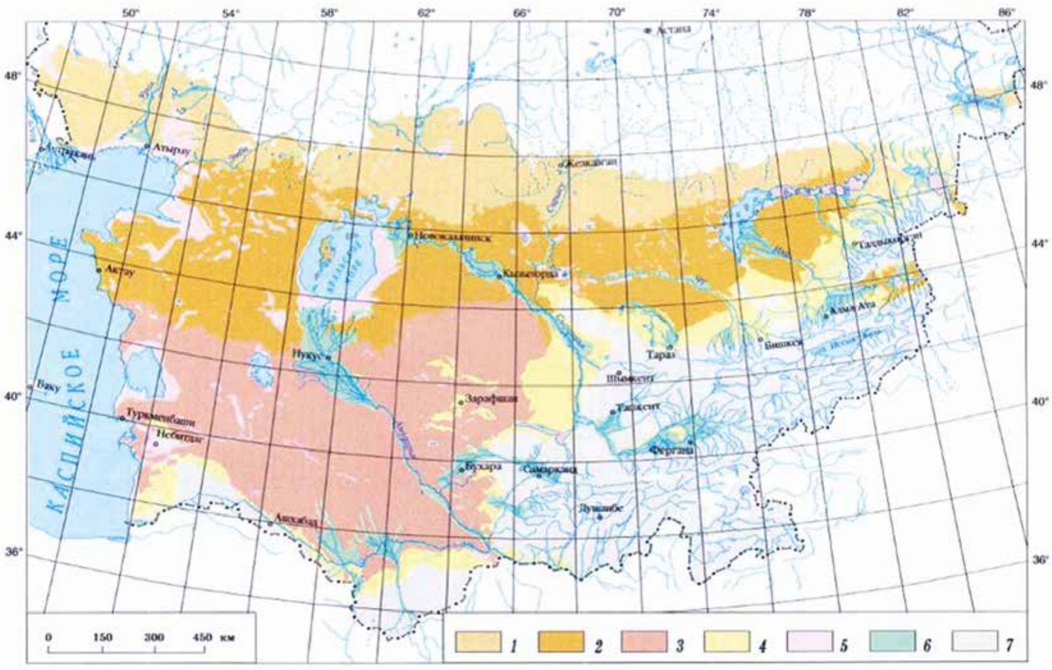
Deserts comprise low and high altitude plains with precipitation of no more than 100 mm/year (FAO, 1989) or no more than 250 mm/year (as per Koeppen-Geiger Classification, Kottek *et al.*, 2006), with rare or absent vegetation on desert soils (Kharin, 2002). While the largest extent of deserts is found 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 oasises. 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).

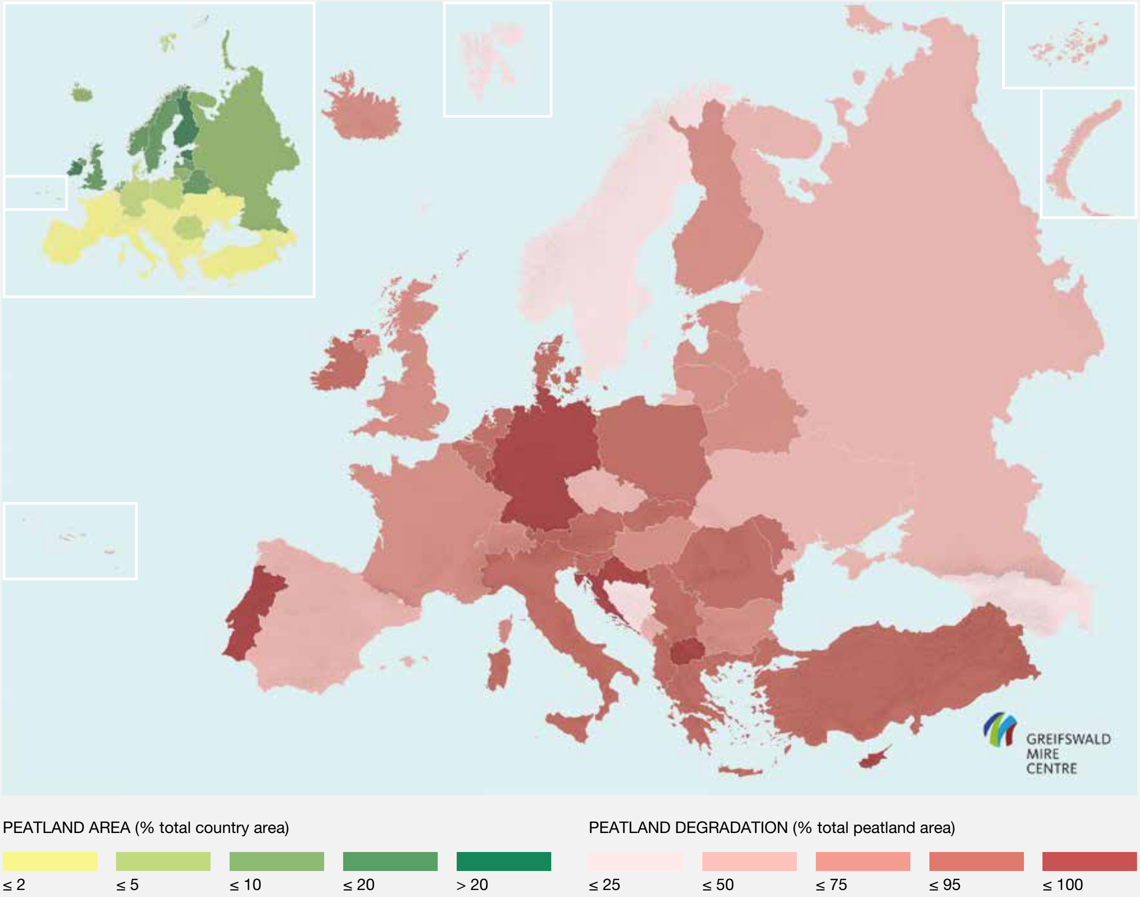
Peatlands often demonstrate a unique structural and functional integrity which has developed over centuries. Saturated peatland conditions select the plant species that may grow and form peat. The accumulated peat (which may consist to more than 90% of water) regulates the moisture balance and further 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 km^2, 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 km^2, 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 km^2 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 km^2 or 8.1% of the country and together with shallow peat lands (<30 cm) as much as 3,690,000 km^2 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.



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).

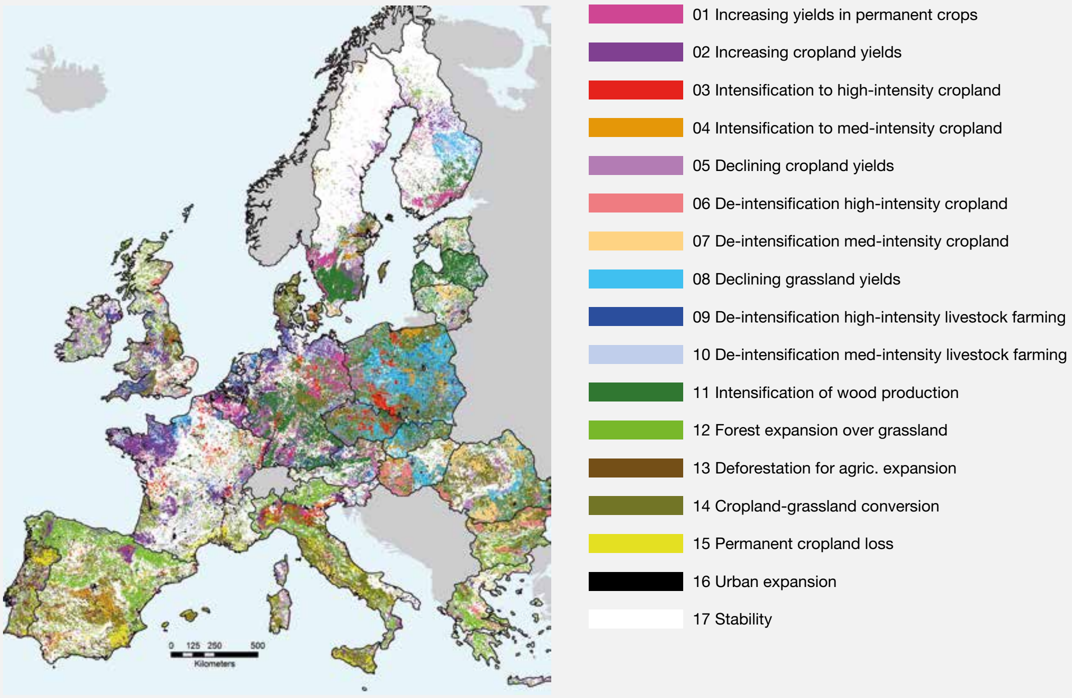
ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

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.

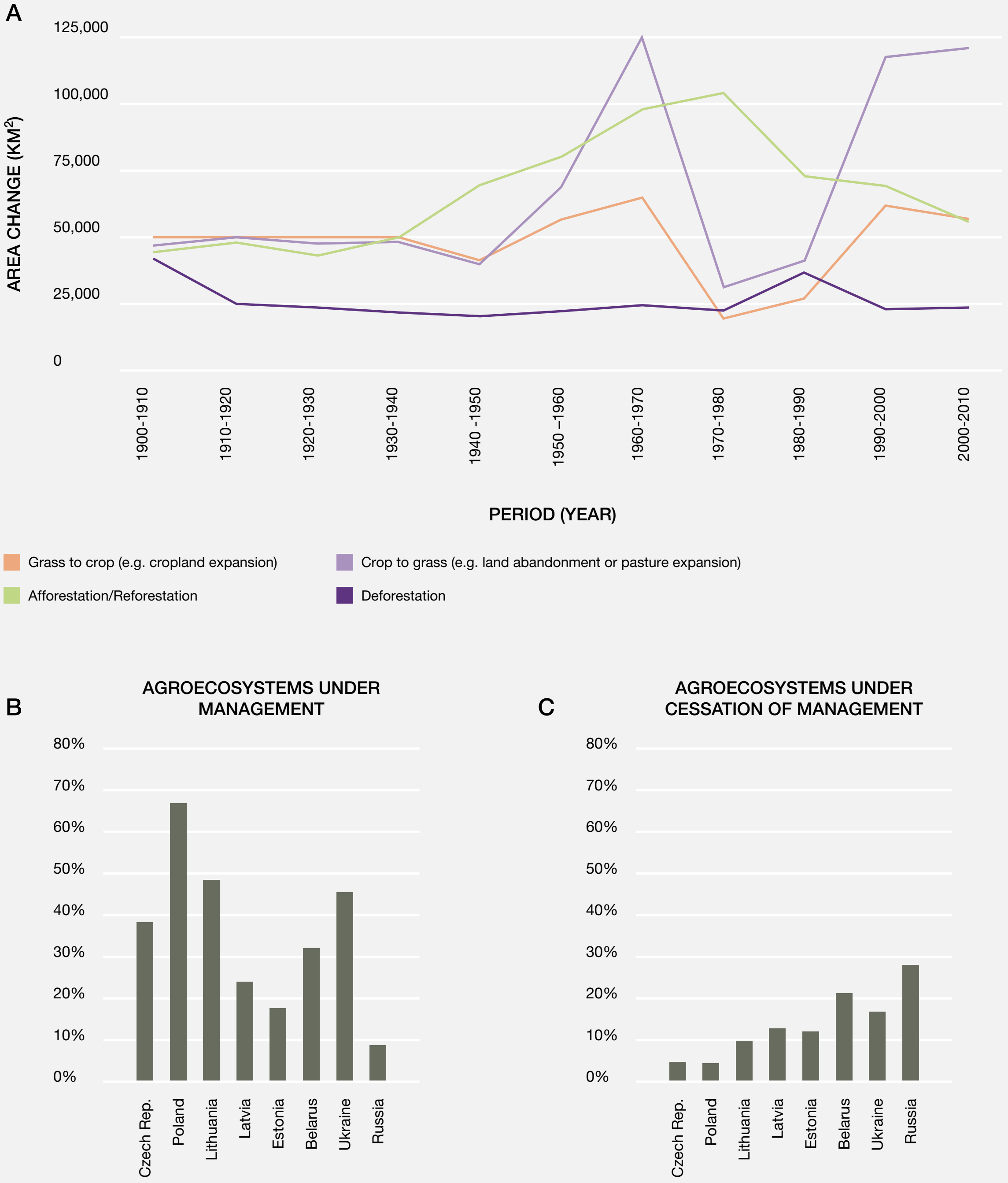
The resilience of natural peatlands to climate change is based on their self-regulation, but this capacity is not unlimited (Minayeva & Sirin, 2012). Substantial changes in peatland hydrology (by drainage), soil hydraulic properties (by long-term drainage), and peatland relief (by oxidation, subsidence 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.**



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).**



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										Importance of direct drivers									
		Past					Current					Climate		Land use*		Pollution*		Exploitation		Invasives	
		ECA	WE	CE	EE	CA	ECA	WE	CE	EE	CA	P	C	P	C	P	C	P	C	P	C
Farmland birds	Species richness	↘	↘	↘	↘	↘	↘	↘	↘	↔	↘	1	1	2	2	1	1	0	0	0	0
	Species abundance	↓	↓	↘	↘	↘	↘	↘	↘	↗	↔	1	1	2	2	1	1	0	0	0	0
	Functional diversity	↘					↘					1	1	2	2	1	1	0	0	0	0
Avian breeds	Breed richness	↘	↓	↘	↘	↘	↘	↘	↘	↘	↘	0	0	2	2	0	0				
	Local breed abundance	↘	↓	↘	↘	↘	↘	↘	↘	↘	↘	0	0	2	2	0	0				
	Genetic diversity	↘	↓	↘	↘	↘	↘	↘	↘	↘	↘	0	0	2	2	0	0				
Mammal breeds	Breed richness	↘	↓	↘	↘	↘	↘	↘	↘	↘	↘	0	0	2	2	0	0				
	Local breed abundance	↘	↓	↘	↘	↘	↘	↘	↘	↘	↘	0	0	2	2	0	0				
	Genetic diversity	↘	↓	↘	↘	↘	↘	↘	↘	↘	↘	0	0	2	2	0	0				
Butterflies	Species richness	↘					↘							2	2	2	2	0	0		
	Species abundance	↘					↘							2	2	2	2	0	0		
	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
	Species abundance	↘					↘					0	0	2	2	2	2	0	0	1	1
	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				
	Ex situ genetic diversity	↕	↕	↕	↕	↕	↕	↕	↕	↕	↕	0	0	2	2	2	2				
	Ex situ functional diversity	↘					↘					0	0	2	2	2	2				
	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)**.

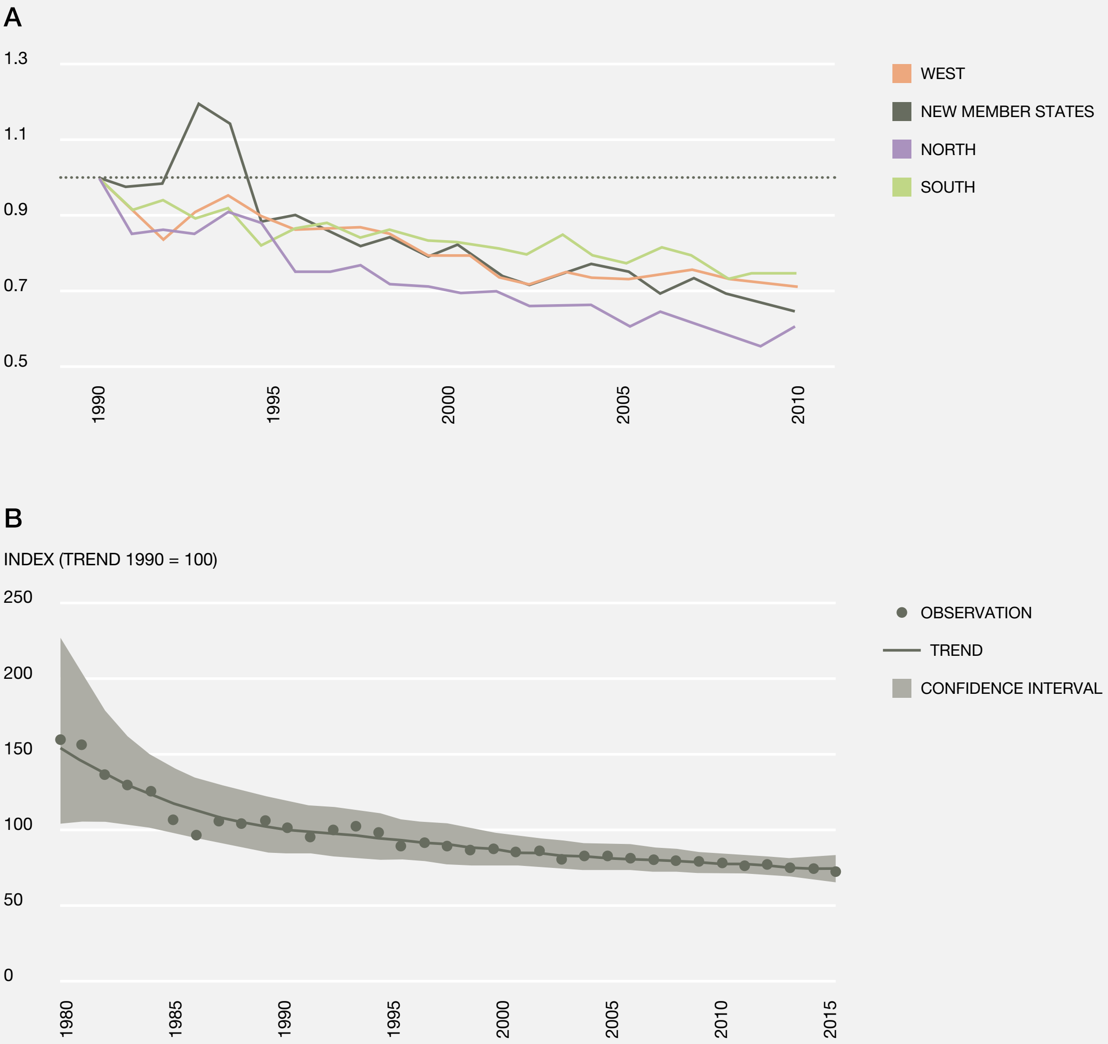
Agriculture-detrimental and beneficial insects – Temporal trends in the abundance or distribution of insects, which can cause major changes for agriculture have been reported. For instance, important changes in the distribution of crop pests, in particular due to climate change in northern areas of Europe 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 **Ⓐ Temporal variations in the abundance of common farmland birds between four European regions between 1990 and 2011. Source: Pe'er *et al.* (2014). Ⓑ 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).**

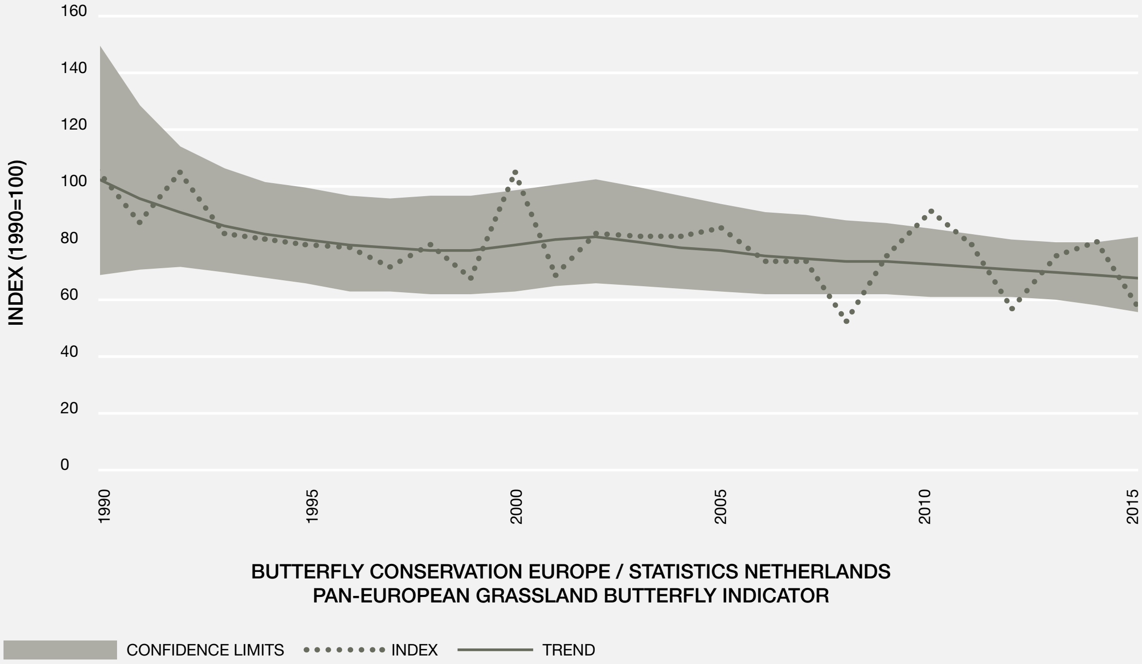


Figure 3 14 **Ⓐ Change in the climatic range of the Colorado beetle from 1991 to 2010 compared with 1951 to 1970. 1: unsuitable areas for the beetle; 2: range increment; 3: suitable in both periods. Source: Popova & Semenov (2013). Ⓑ Annual growth rate (%/yr) in the number of honey bee colonies for countries reporting those data to FAO between 1961 and 2012. Source: FAO (2013a).**

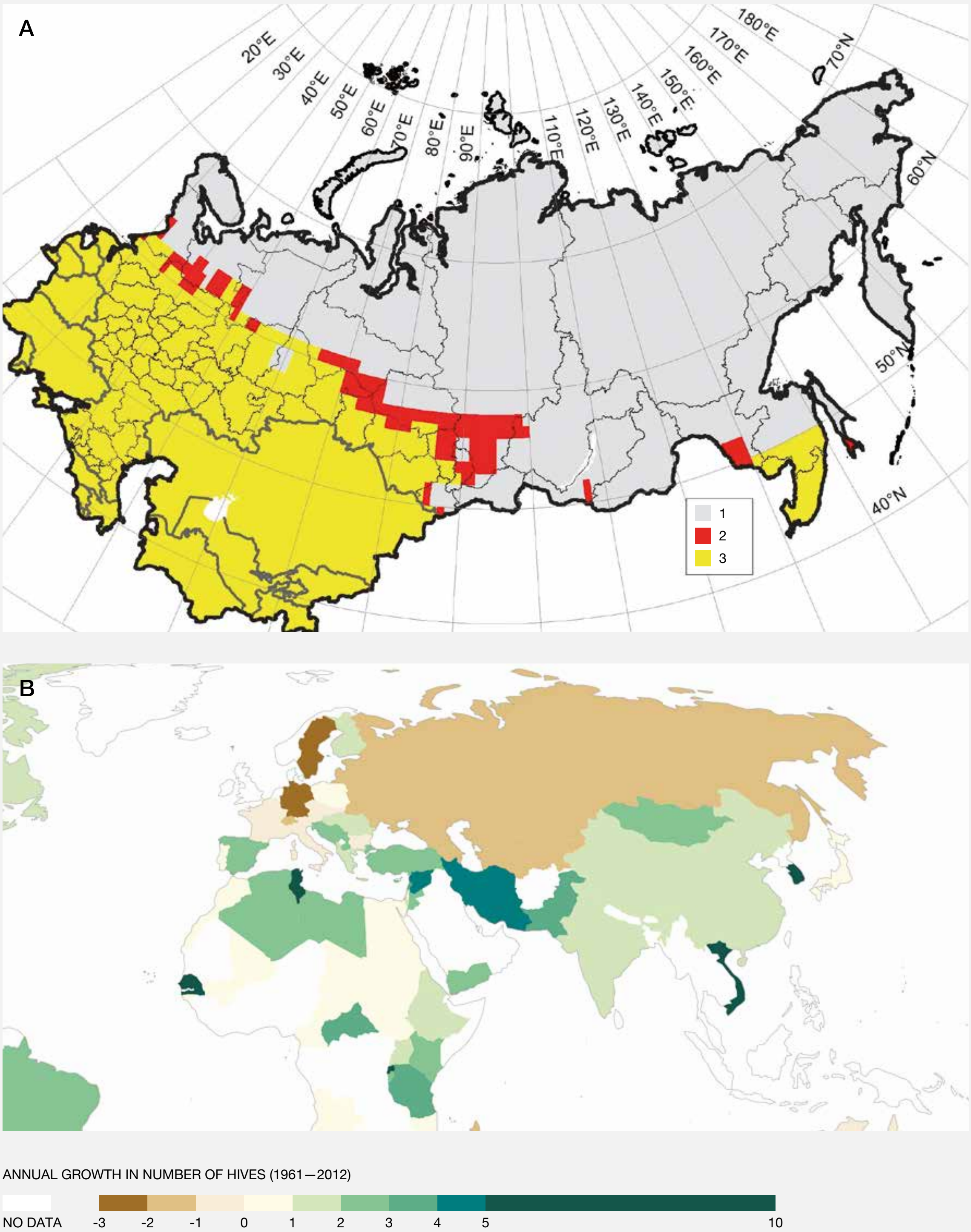
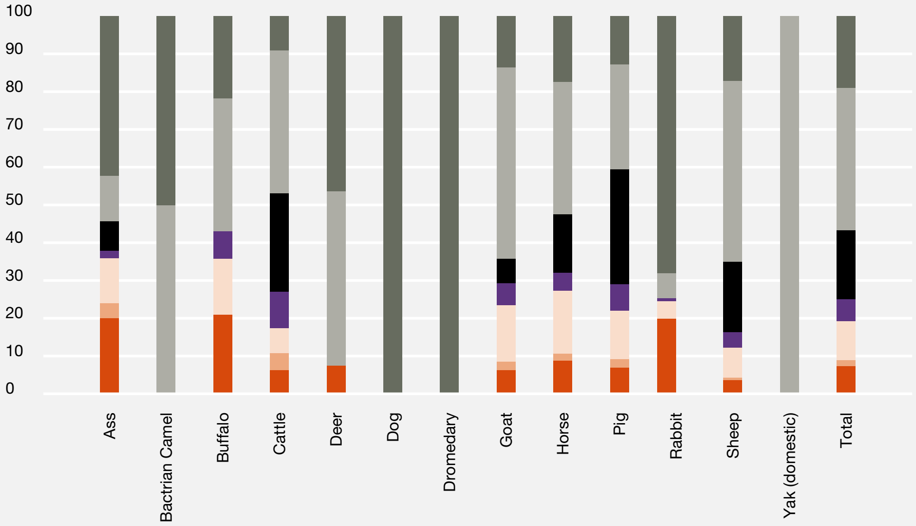


Figure ③ ⑮ **Risk status of mammalian breeds in geographical Europe and the Caucasus up to December 2005. Relative (figures) and absolute (tables) numbers are presented. Source: FAO (2007).**



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).

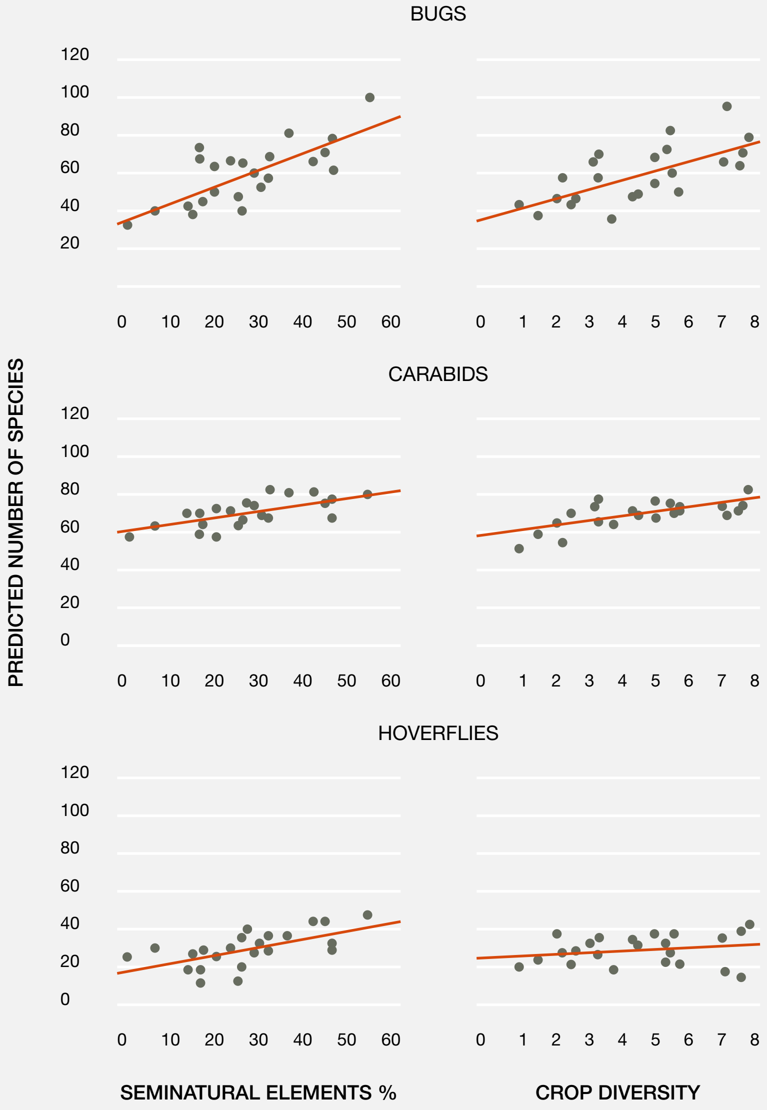
ATTRIBUTION OF BIODIVERSITY TRENDS TO DIRECT DRIVERS

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-

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).

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.

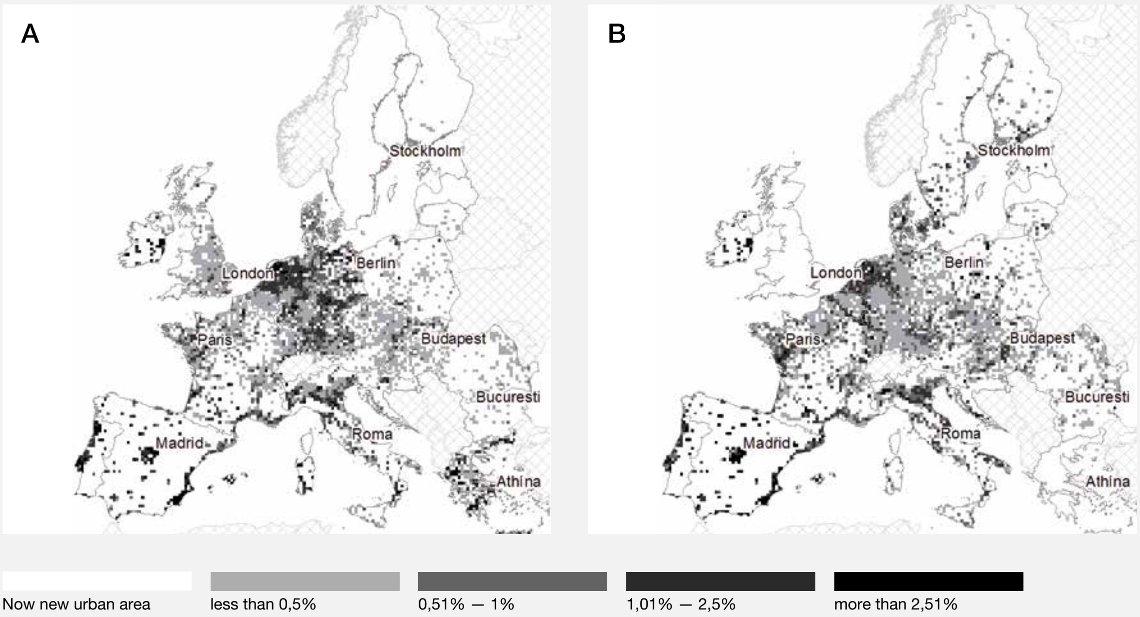
Relict natural habitats such as steppe grasslands and limestone caves in Budapest (Tóth-Ronkay *et al.*, 2015) and calcareous sand dunes in Rotterdam (Van de Poel *et al.*, 2015), support communities of specialized species, though fragmentation often leads to species losses and reduces the potential 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 Ⓐ 1990–2000 and Ⓑ 2000–2006. Source: Siedentop & Fina (2012) based on CORINE land-cover data.**



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.

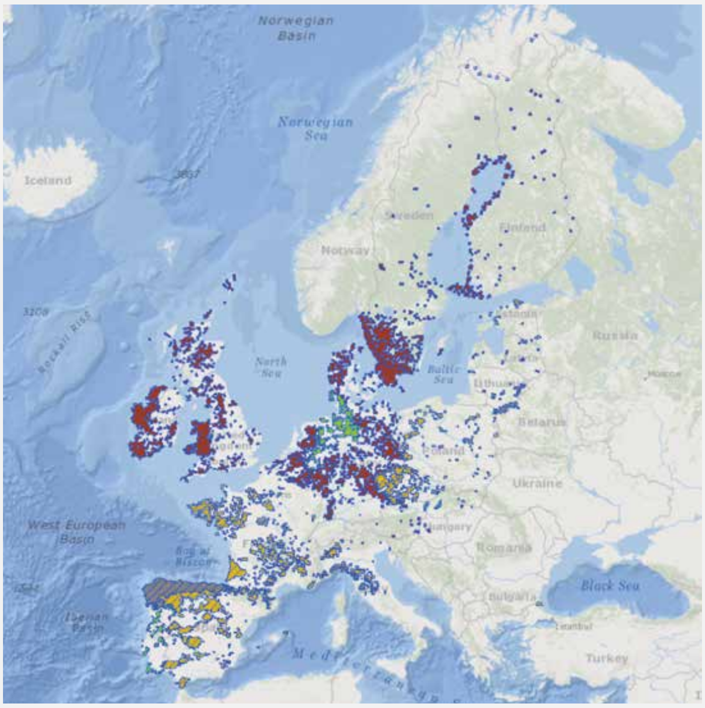
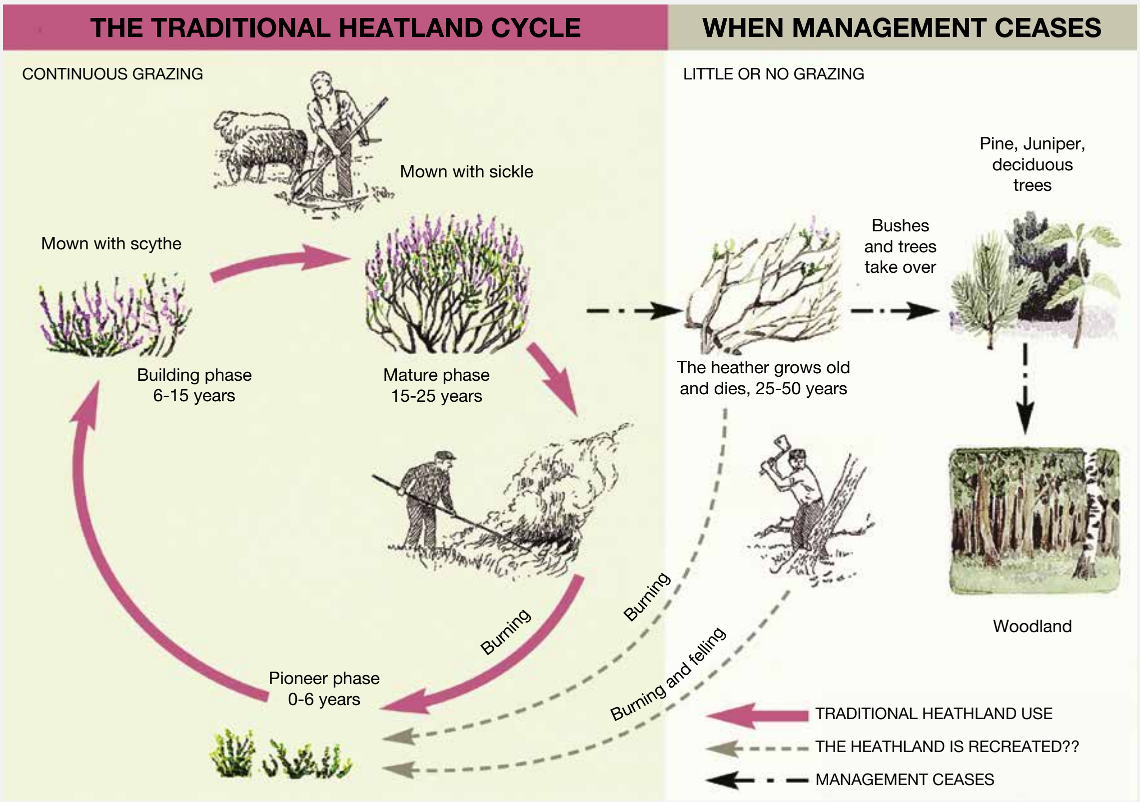


Figure 3 21 **The traditional management cycle with prescribed burning, grazing, and mowing, and the successional dynamics occurring after abandonment of management in heathlands. Source: Developed from Gimingham (1972).**



"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] 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.

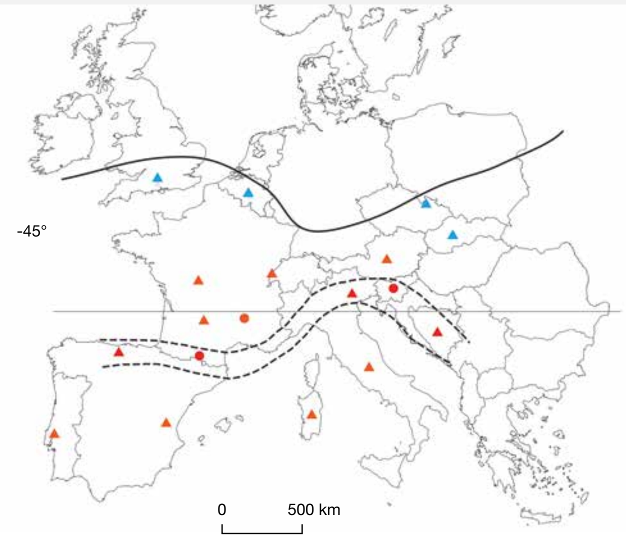
We distinguish to two types of subterranean systems, subterranean terrestrial systems (dry caves, epikarst, MSS (*milieu souterrain superficiel*)) and subterranean aquatic systems (flooded caves, groundwater, interstitial).

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 km^2 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).

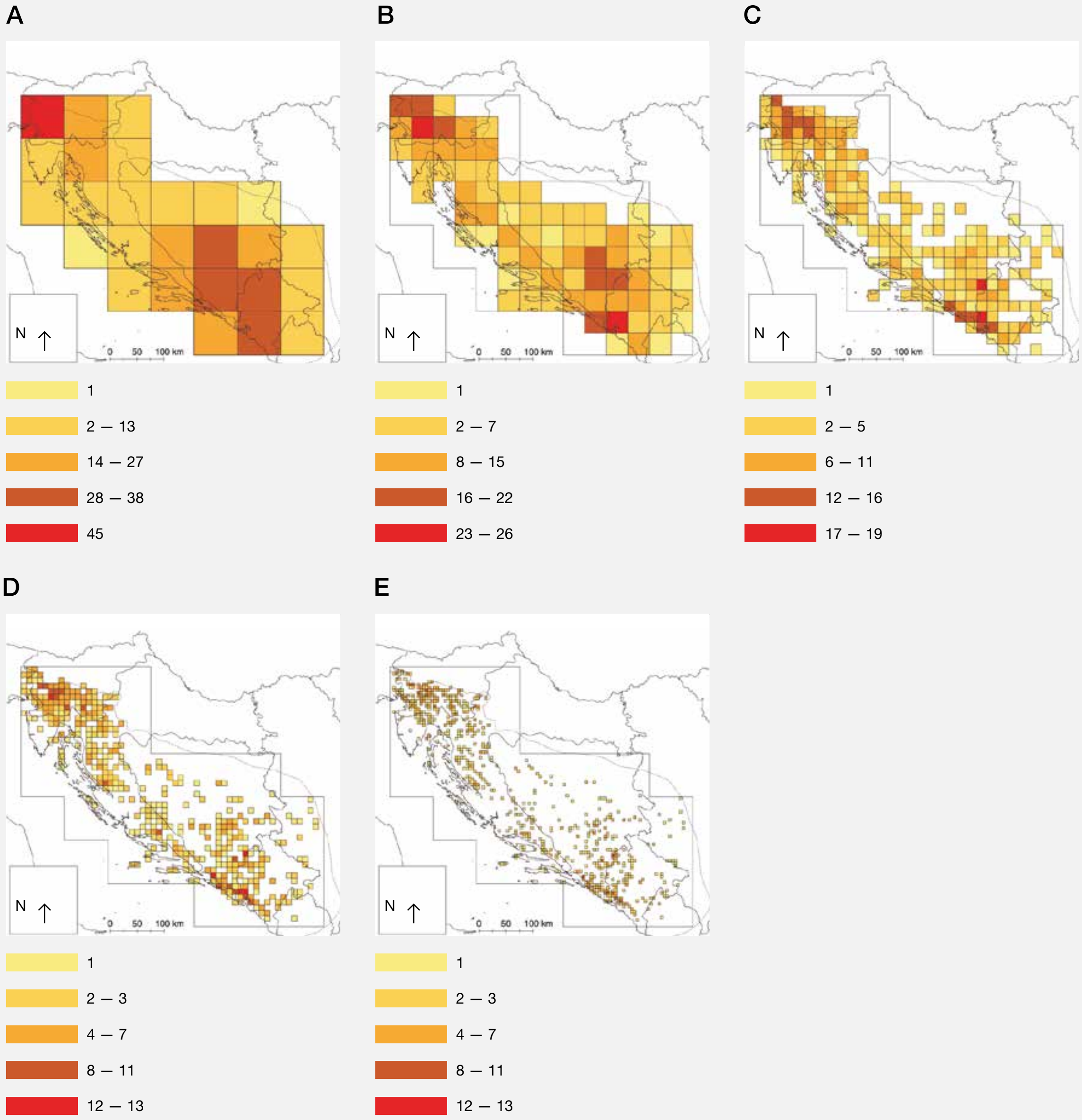


Figure 3 24 **Map of obligately subterranean aquatic (stygobiotic) species numbers in 0.2 x 0.2 ° grid cells distributed across six Western and Central European countries. Source: Deharveng *et al.* (2009).**

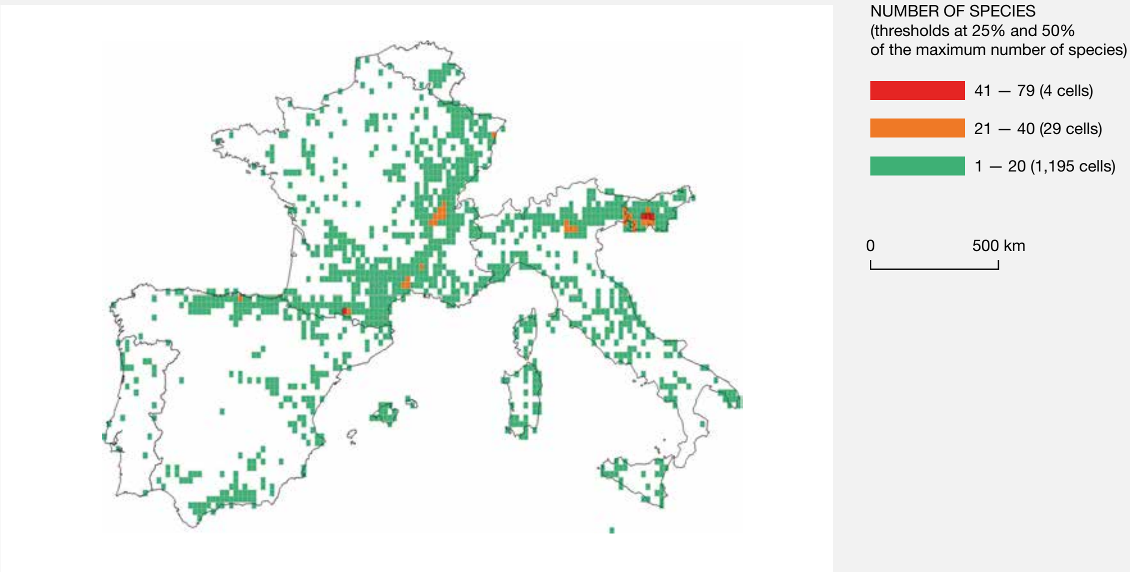


Figure 3 25 **Olm *Proteus anguinus*, an endemic species of Dinaric karst (Rupećica Cave in Ogulinsko Zagorje, Ogulin, Croatia, 2014). Photo: Dušan Jelić.**

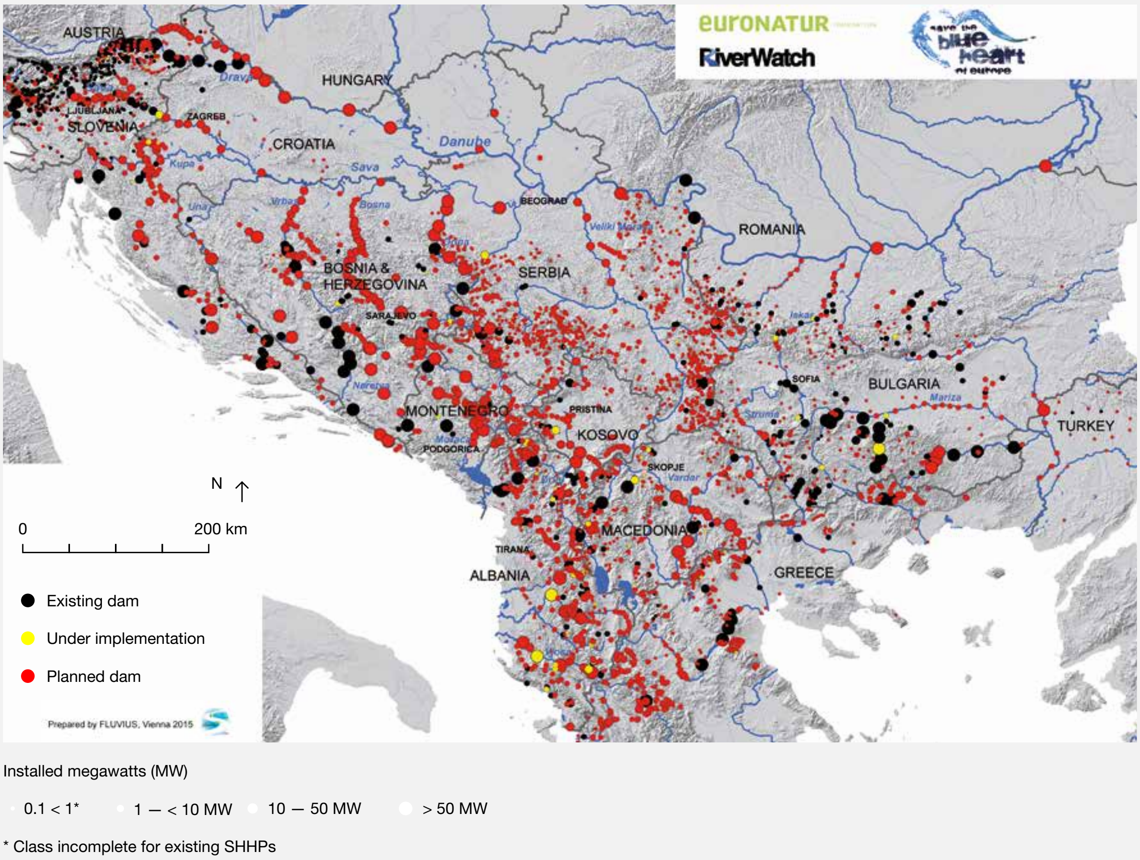
There are also visible geographic patterns within Western and Central Europe. The first one is a gradient in species richness with diversity decreasing from south to north and highest biodiversity within the mid-European high subterranean diversity ridge **(Figure 3.22)**. For details see Culver & Pipan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

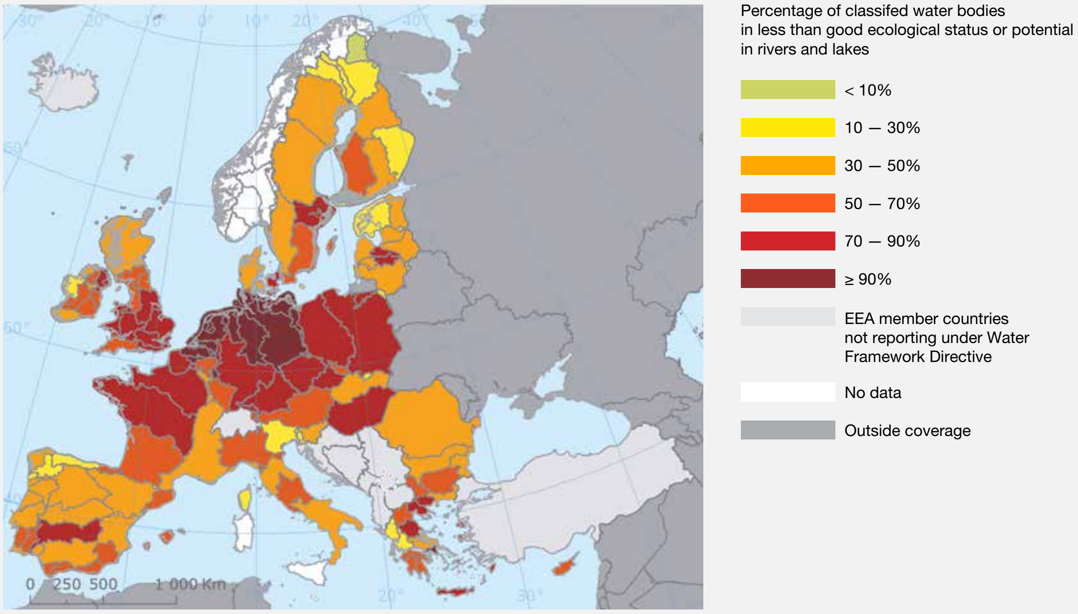
The extent of wetlands in Western, Central and Eastern Europe has declined by 50% from 1970 to 2008 (Dixon *et al.*, 2016). According to the State of the Environment Report review of the state of freshwater systems, only 53% of geographical Europe's rivers and lakes have a good ecological status in 2015 (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).**

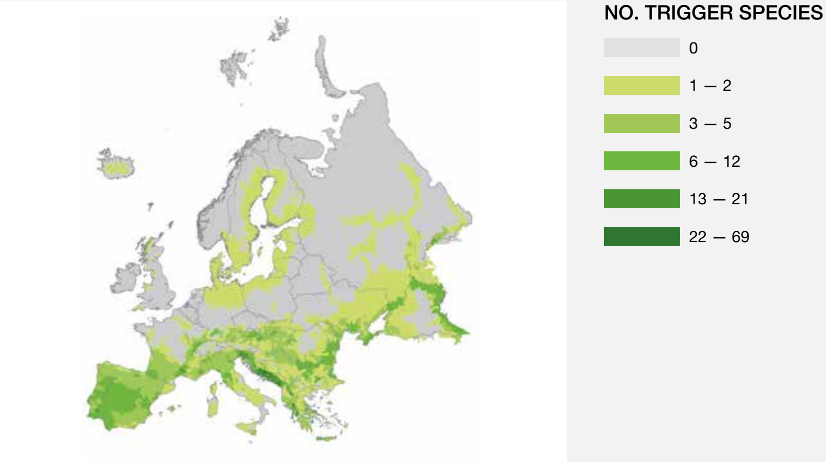


et al., 2004). Concerning species, 30% assessments have an "unfavourable to bad" conservation status and 45% assessments were classified as "unfavourable to inadequate" (EEA, 2015a). For Eastern Europe, fresh water quality remains poor, with variation from contaminated to extremely polluted for the majority 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).**



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).

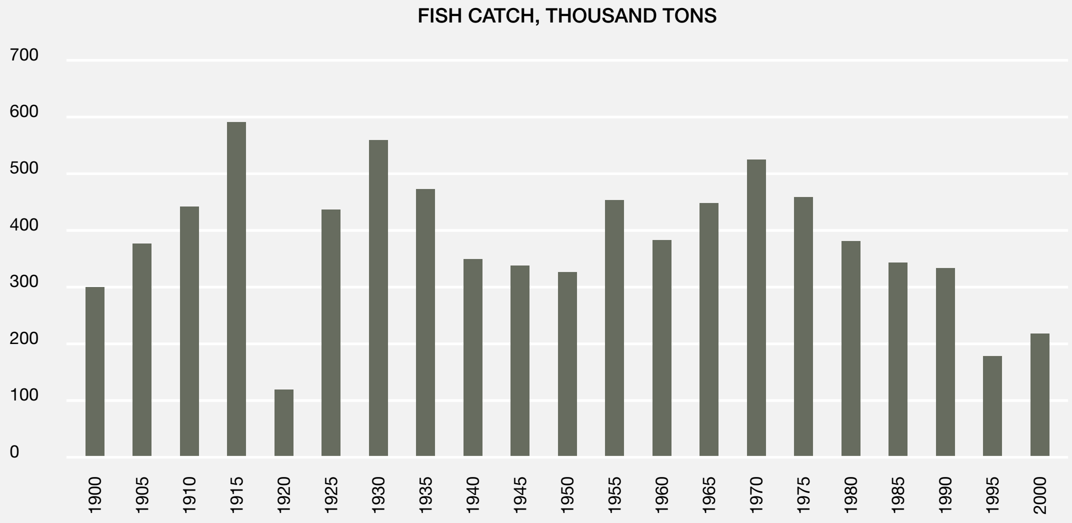
The Caspian Sea is threatened by pollution from untreated wastewater from industry and agriculture along the Volga River (an estimated 80% of the total load) (Glantz & Zonn, 1997) offshore oil and gas production, processing, extraction and transportation, and shipping. Industrial pollution impacts biological 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).**



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**.

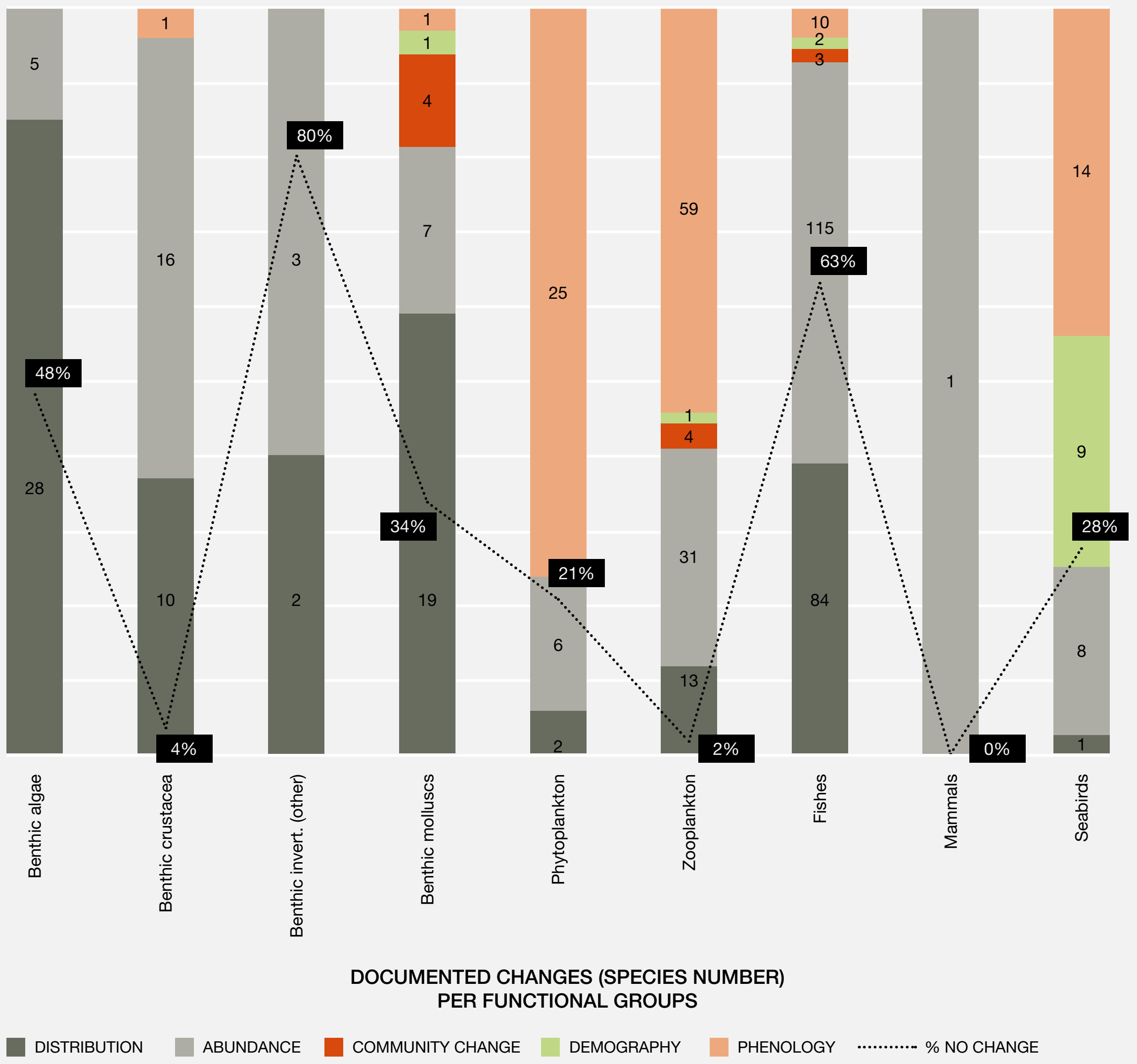
Shifts in range, in particular northward expansion of more than 140 km per decade on average across taxa (Poloczanska *et al.* 2013), have been shown **(Figure 3.34)**. This is exemplified by the subtropicalization of European pelagic fish communities (Montero-Serra *et al.*, 2015), by movements of calanoid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

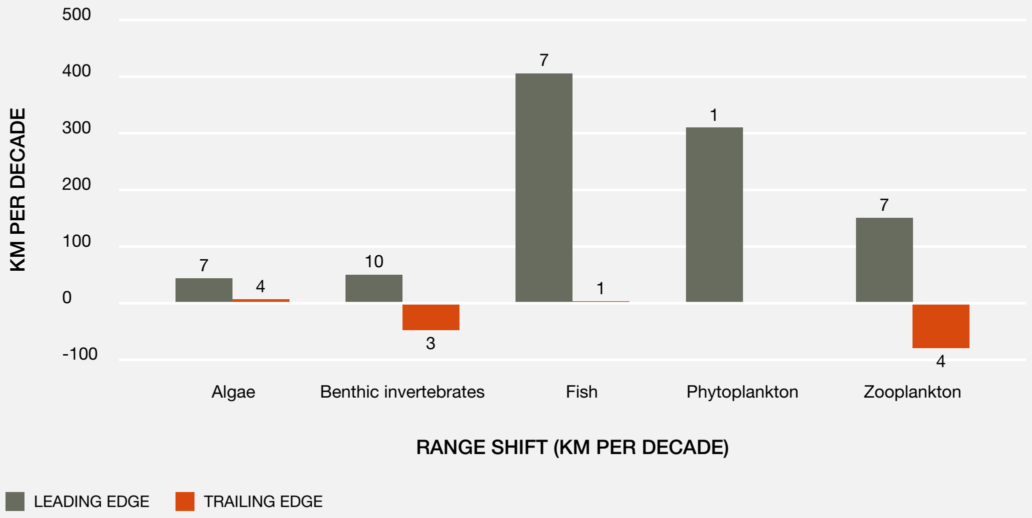


and current trend over a large range of taxa (Seebens *et al.*, 2017).

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).



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)**.

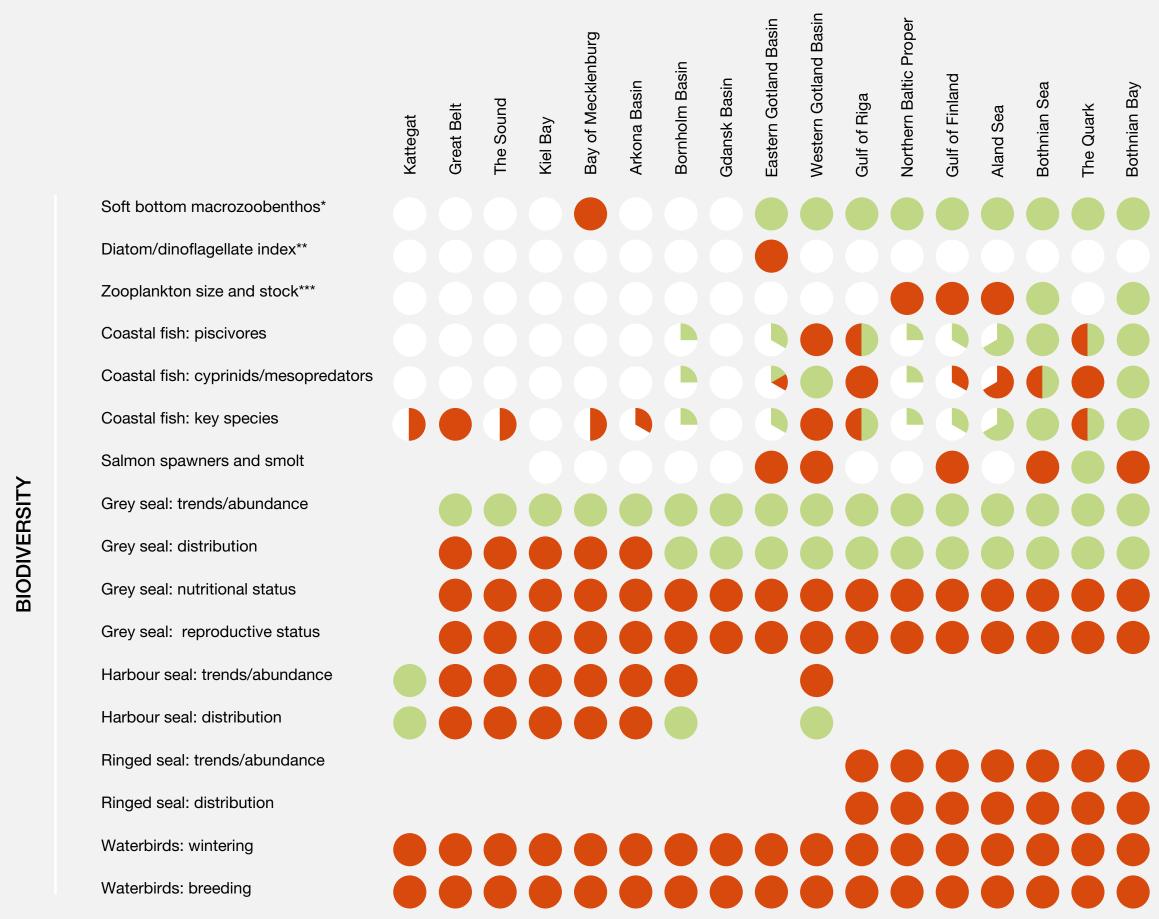
Regime shifts and fish trends

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 km^2, 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 $\mu g\ l^{-1}$); 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).

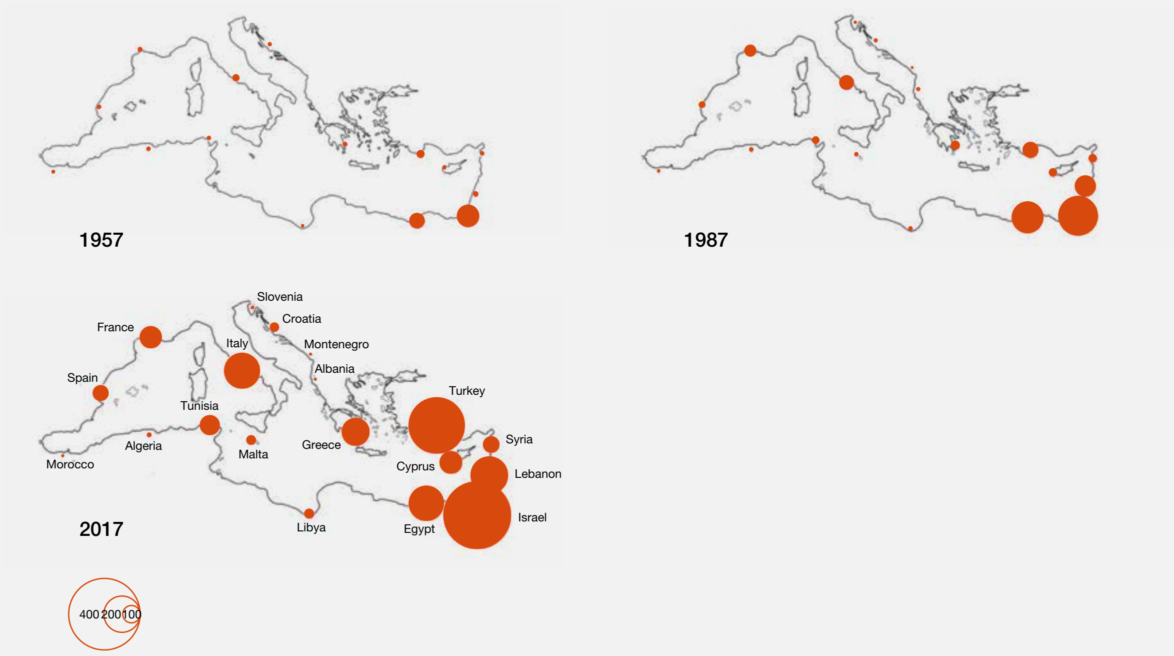
3.3.4.4 The Black and Azov Seas

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).

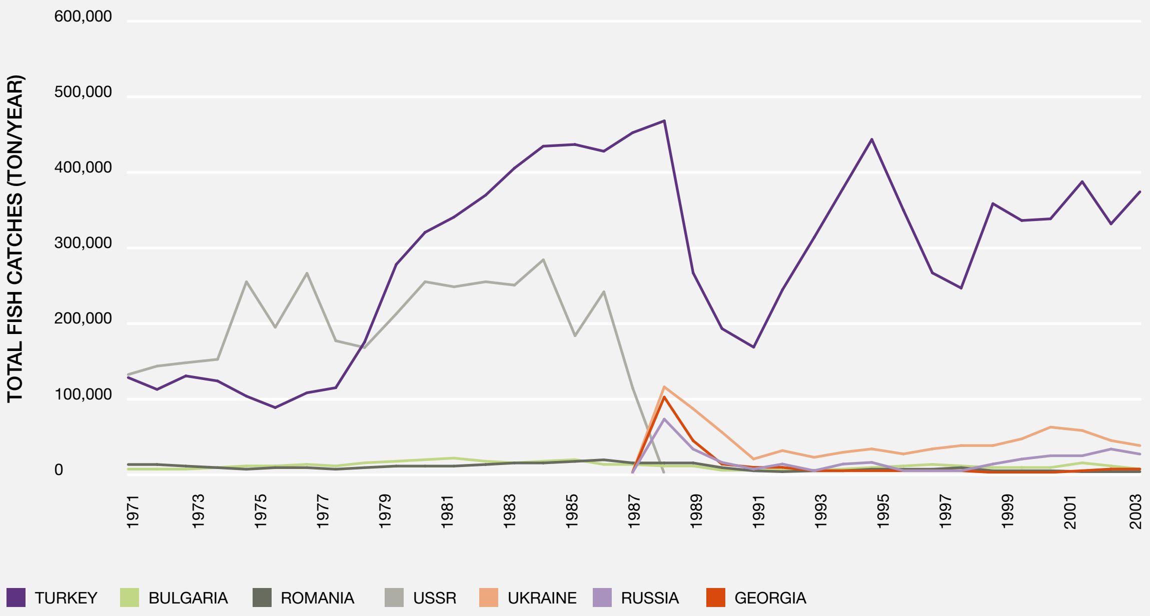
Two major rivers flow into the Sea of Azov: the Don and Kuban, and salinity is at its lowest (about 1‰) near the mouth of the Don (Kotlyakov, 2004). Flora and fauna are composed of different biogeographic groups with a predominance of eurythermic and euryhaline species. Only for the last 6,000-7,000 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).

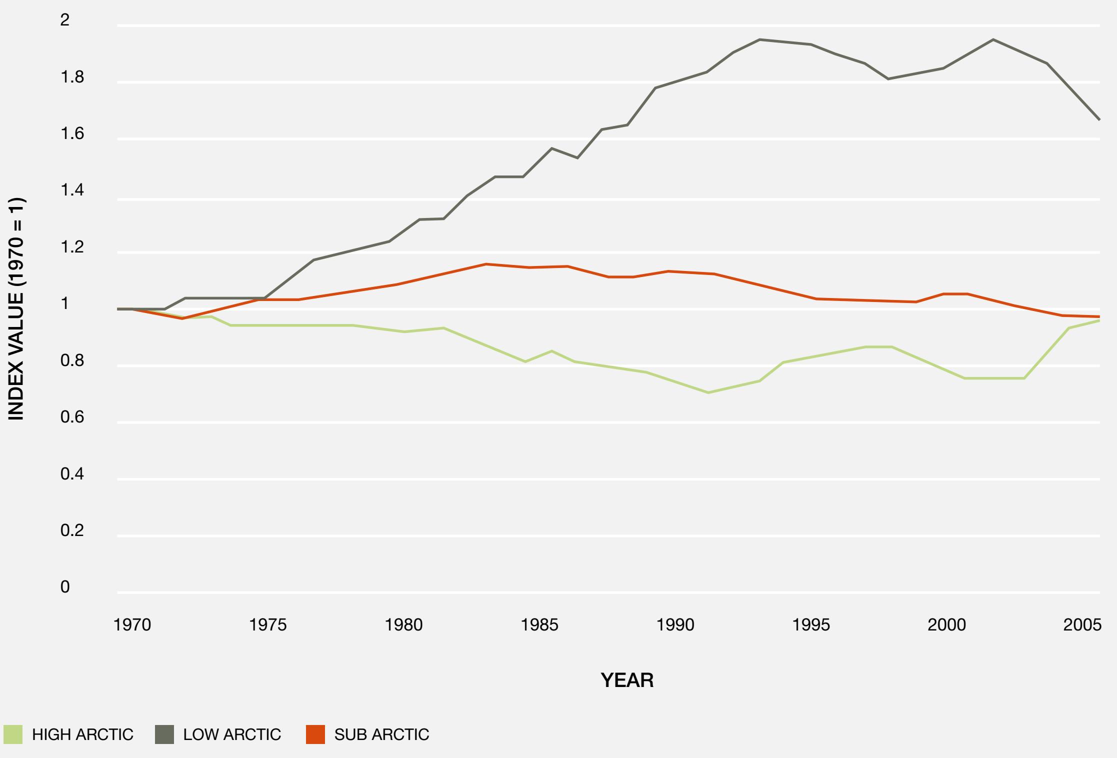
There are also different trends in species and abundance of Arctic fish in the northern Barents Sea (Johannesen *et al.*, 2017). Overall there was a negative trend in the number of Arctic fish species from 2004-2015 but, while some species declined across the area, others declined only in the southern 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).**



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.).

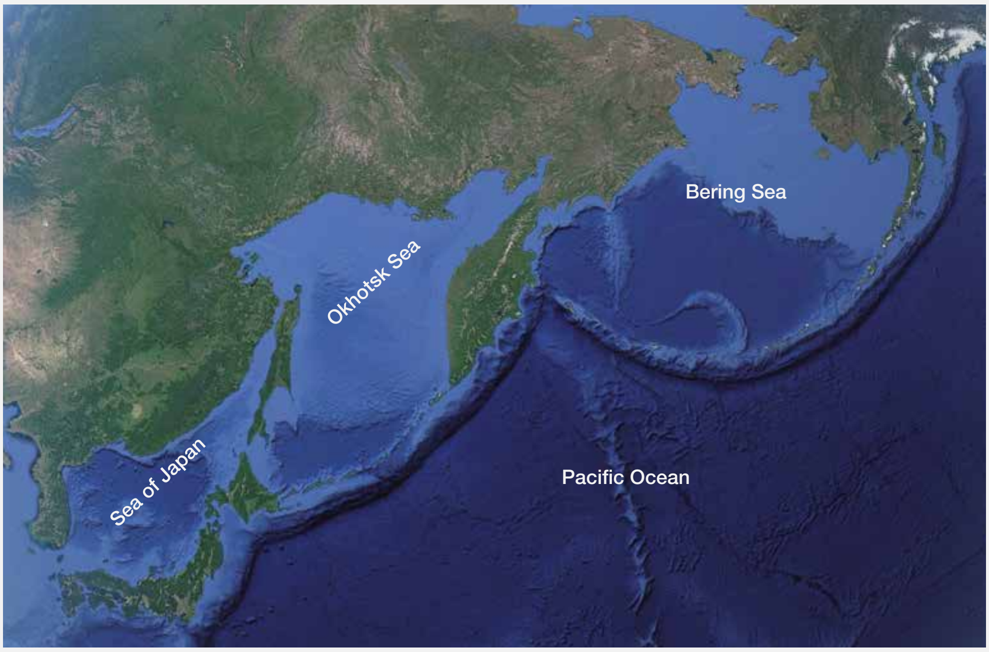
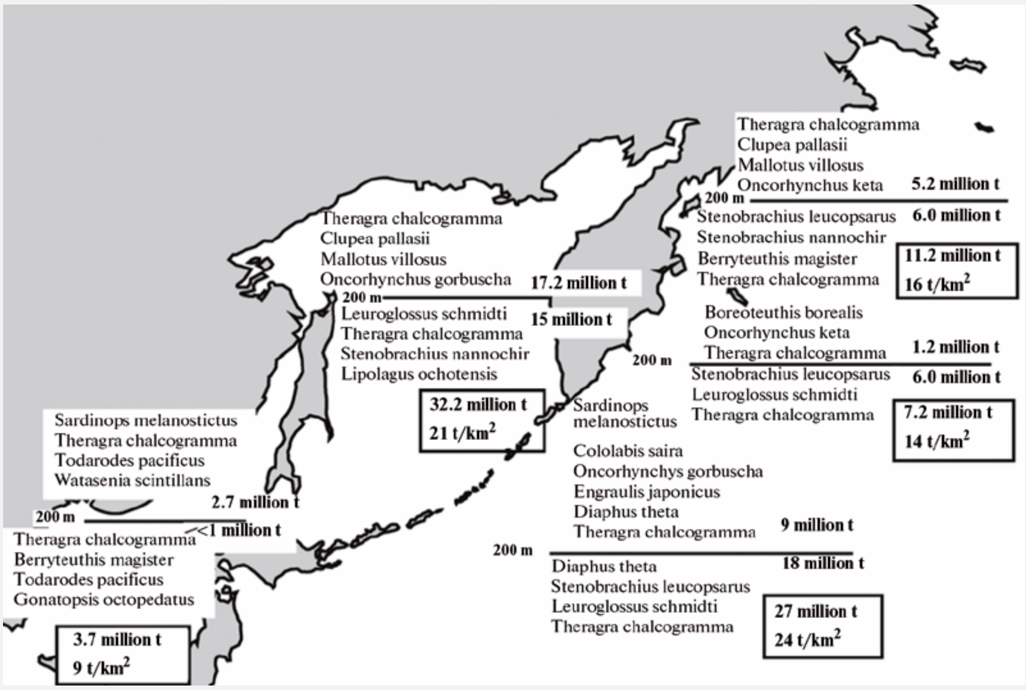


Figure ③ ㊵ Mean annual composition of the most abundant nekton species and total nekton biomass in different regions of the Far Eastern Seas: numerator — in the epipelagic layer (0–200 m), denominator — in the mesopelagic layer (200–1000 m), in frame — total biomass (106 t) and concentrations (t/km²) 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).**

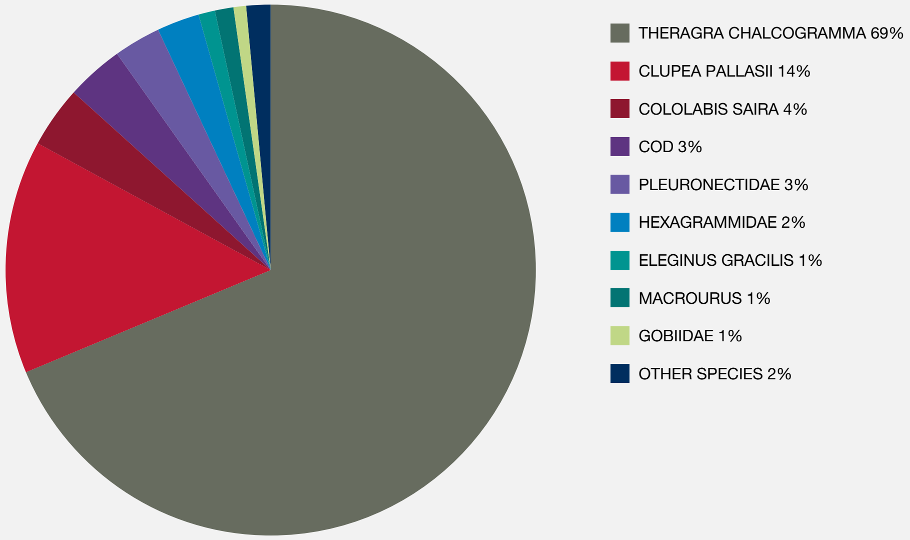
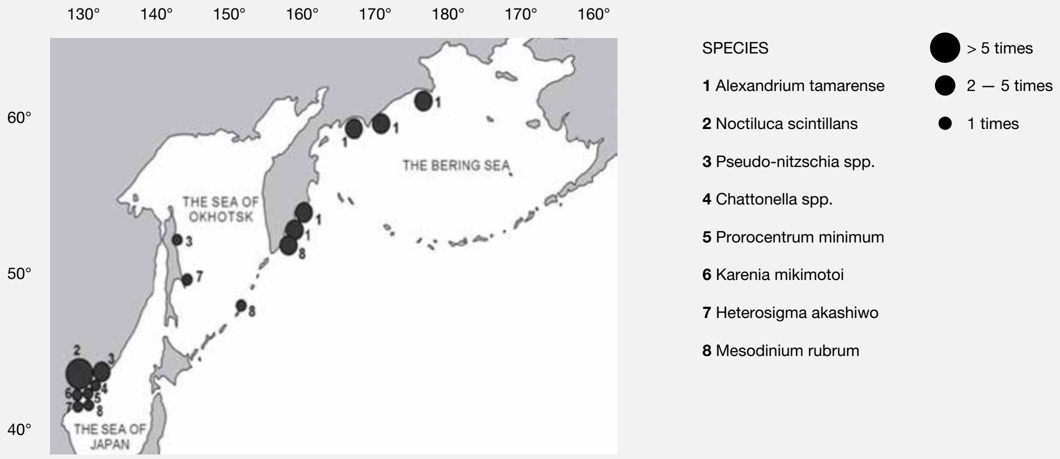


Figure ③ ㊷ **Harmful algal blooms in the Far East seas of Russia in 1980–2005. Source: Orlova *et al.* (2002).**



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 1 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

		GENERAL TREND								CLIMATE CHANGE							
		Past				Present				Past				Present			
		ECA				ECA				ECA				ECA			
Unit of analysis	Indicator	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	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
🟡				🟡				🟡				🟡				🟢				🟢				🟢				🟢			
🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
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🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴
🟡				🟡				🟡				🟡				🟢				🟢				🟡				🟡			
🟡		🟡		🟡	•	🟡	•	🟡	•	🟡	•	🟡	•	🟡	•	🟢	•	🟢	•	🟢	•	🟢	•	🟡	•	🟡	•	🟡	•	🟡	•
🔴				🔴				🟡				🟡				🔴				🔴				🟡				🔴			
🔴	•	🔴	•	🔴	•	🔴	•	🟡	•	🟡	•	🟡	•	🟡	•	🔴	•	🔴	•	🔴	•	🔴	•	🟡	•	🟡	•	🔴	•	🔴	•
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🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
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🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟡	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟡	🟡	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
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🟡	🟡	🟡	🟡	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🟡	🔴	🔴	🔴	🟡
🔴				🔴				🟡				🟡				🔴				🔴				🟡				🟡			
🔴	🔴	🔴	🟡	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟢	🟡	🟡	🟡	🟢
🔴				🔴				🟡				🟡				🔴				🔴				🔴				🔴			
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟡	🔴	🔴	🔴	🟡
🔴				🟡				🟢				🟢				🔴				🟡				🟡				🟡			
🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🔴				🟡				🟡				🔴				🔴				🟡				🟡				🔴			
🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴	🟡	🟡	🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴
🔴				🔴				🟢				🟢				🟢				🟢				🟡				🟡			
🔴	🔴	🔴	🔴	🟡	🟢	🔴	🔴	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟡	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🟢	🟡	🟡	🟡	🟢
🔴				🔴				🔴				🟡				🟡				🔴				🟡				🟡			
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🔴	🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🟡	🟡	🟡	🔴	🔴	🟡	🟡
🔴				🔴				🟡				🟡				🔴				🟡				🟢				🟢			
🔴	•	🔴	🔴	🔴	•	🔴	🔴	🟡	•	🟡	🟡	🟡	•	🟡	🟡	🟡	•	🔴	🔴	🟡	•	🟡	🟡	🟢	•	🟢	🟢	🟢	•	🟢	🟢
🔴				🔴				🔴				🔴				🔴				🟡				🟡				🟡			
🔴	•	🔴	🔴	🔴	•	🔴	🔴	🔴	•	🔴	🔴	🔴	•	🔴	🔴	🔴	•	🔴	🔴	🔴	•	🟡	🟡	🟡	•	🟡	🟡	🟡	•	🟡	🟡
🟢				🟢				🟢				🟢				🟢				🟢				🟢				🟢			
🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•	🟢	•
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🟢	•	🟢	•	🟡	•	🟡	•	🟢	•	🟢	•	🟡	•	🟡	•	🟢	•	🟢	•	🟡	•	🟡	•	🟢	•	🟢	•	🟡	•	🟡	•
🔴				🟢				🟡				🟢				🟡				🟢				🟡				🟡			
🔴	•	🔴	•	🟢	•	🟢	•	🟡	•	🟡	•	🟢	•	🟢	•	🟡	•	🟡	•	🟢	•	🟢	•	🟡	•	🟡	•	🟡	•	🟡	•
🔴				🟡				🟡				🟢				🟡				🟢				🟡				🟡			
🔴	•	🔴	•	🟡	•	🟡	•	🟡	•	🟡	•	🟢	•	🟢	•	🟡	•	🟡	•	🟢	•	🟢	•	🟡	•	🟡	•	🟡	•	🟡	•
🟡				🟢				🟡				🟡				🟡				🟢				🟡				🟡			
🟡	🟡	🟡	•	🟢	🟢	🟢	•	🟡	🟡	🟡	•	🟡	🟡	🟡	•	🟡	🟡	🟡	•	🟢	🟢	🟢	•	🟡	🟡	🟡	•	🟡	🟡	🟡	•
🟡				🟢				🟡				🟡				🟡				🟢				🟡				🟡			
🟡	🟡	🟡	•	🟢	🟢	🟢	•	🟡	🟡	🟡	•	🟡	🟡	🟡	•	🟡	🟡	🟡	•	🟢	🟢	🟢	•	🟡	🟡	🟡	•	🟡	🟡	🟡	•

Box ③ ①

Table ③ ⑤

		GENERAL TREND								CLIMATE CHANGE							
		Past				Present				Past				Present			
		ECA				ECA				ECA				ECA			
Unit of analysis	Indicator	WE	CE	EE	CA	WE	CE	EE	CA	WE	CE	EE	CA	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	↕				↘				🟡				🟡			

LAND USE CHANGE									
Past					Present				
ECA	WE	CE	EE	CA	ECA	WE	CE	EE	CA
🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡
🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🟡	🟡	🟡	🟡	🟡	🔴	🟡	🔴	🔴	🔴
🟡	🟡	🟡	🟡	🟡	🔴	🟡	🔴	🔴	🔴
🔴	🔴	🔴	🔴	🟡	🔴	🟡	🟡	🟡	🟡
🔴	🔴	🔴	🟡	🔴	🔴	🔴	🔴	🟡	🟡
🔴	•	•	•	🔴	🔴	•	•	•	🔴
🔴	•	•	•	🔴	🔴	•	•	•	🔴
🔴	•	•	🔴	🔴	🔴	•	•	🔴	🔴
🔴	•	•	🔴	🔴	🔴	•	•	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🟡					🔴				
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🟡					🔴				
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POLLUTION									
Past					Present				
ECA	WE	CE	EE	CA	ECA	WE	CE	EE	CA
🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
🟡	🟡	🟡	🟡	🟡	🟢	🟢	🟢	🟢	🟢
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟡	🟡
🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🔴	🔴
🟡	🟡	🟡	🟡	🟡	🔴	🟡	🔴	🔴	🔴
🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🟡	🟡	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🟡	🟡	🔴	🔴
🟢	•	•	•	🟢	🟢	•	•	•	🟢
🔴	•	•	•	🔴	🔴	•	•	•	🔴
🟢	•	•	🟢	🟢	🟢	•	•	🟢	🟢
🟡	•	•	🟡	🟡	🟡	•	•	🟡	🟡
🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🔴					🔴				
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🔴					🟡				
🟢					🟡				
🔴					🟡				
🟢					🟢				

OVEREXPLOITATION									
Past					Present				
ECA	WE	CE	EE	CA	ECA	WE	CE	EE	CA
🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡
🔴	🔴	🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡
🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
🟡	🟡	🟡	🟡	🟡	🔴	🟡	🔴	🔴	🔴
🟡	🟡	🟡	🟡	🟡	🔴	🟡	🔴	🔴	🔴
🔴	🔴	🔴	🟡	🔴	🔴	🔴	🔴	🟡	🔴
🔴	🔴	🔴	🟡	🟡	🔴	🔴	🔴	🟡	🟡
🟢	•	•	•	🟢	🟢	•	•	•	🟢
🔴	•	•	•	🔴	🟡	•	•	•	🟡
🟢	•	•	🟢	🟢	🟢	•	•	🟢	🟢
🔴	•	•	🔴	🔴	🔴	•	•	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🔴					🟡				
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🔴					🔴				
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🔴					🔴				

INVASIVE ALIEN SPECIES									
Past					Present				
ECA	WE	CE	EE	CA	ECA	WE	CE	EE	CA
🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴	🔴
🟡	🟡	🟡	🟡	🟡	🔴	🔴	🔴	🔴	🔴
🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🔴	🔴	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡
🟢	•	•	•	🟢	🟢	•	•	•	🟢
🟡	•	•	•	🟡	🟡	•	•	•	🟡
🟢	•	•	🟢	🟢	🟢	•	•	🟢	🟢
🔴	•	•	🔴	🔴	🔴	•	•	🔴	🔴
🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴
🔴					🔴				
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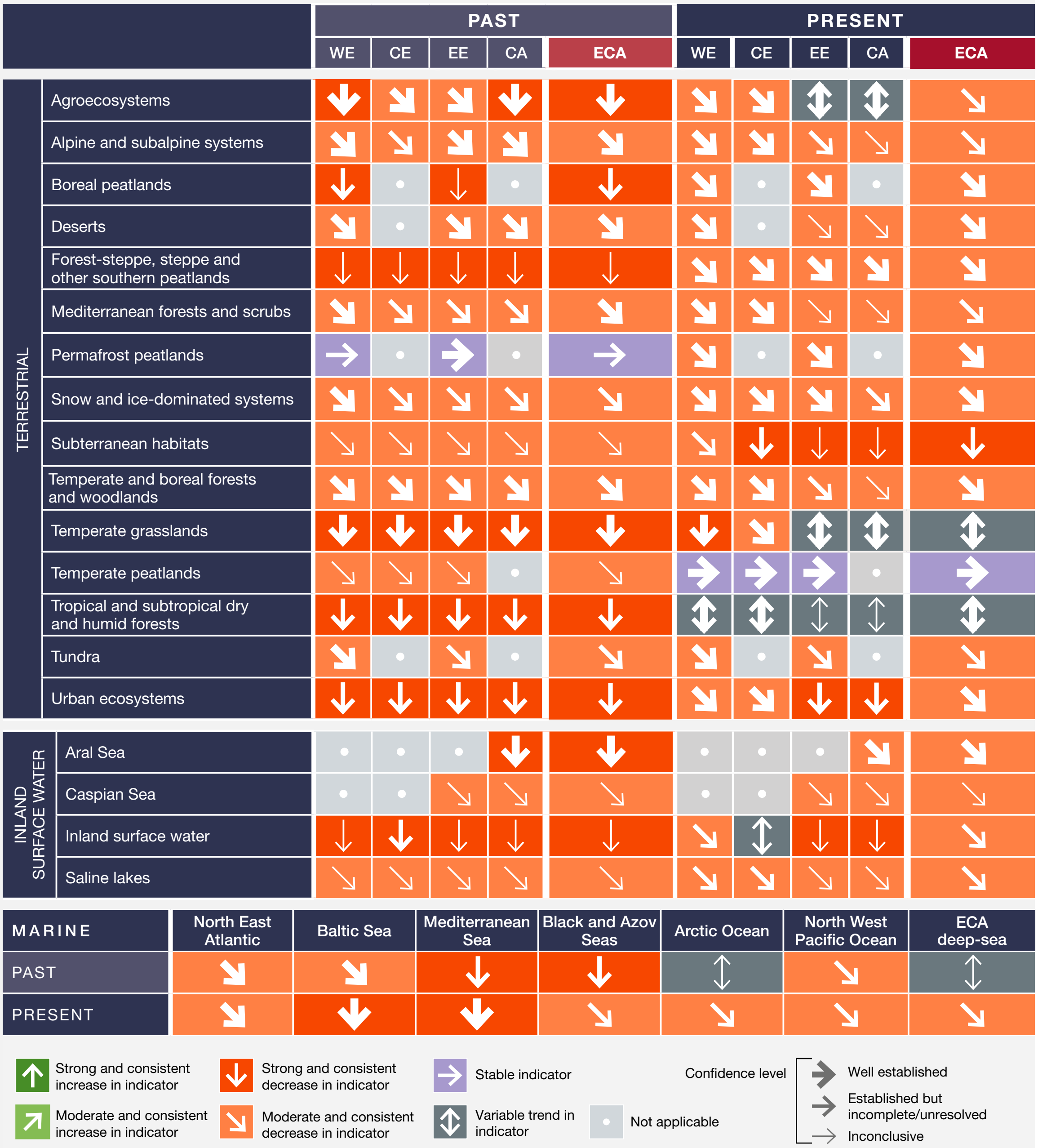
Box 3 1

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	↘	↓	↓	↘	↘	↘	↘

↑ 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

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).**

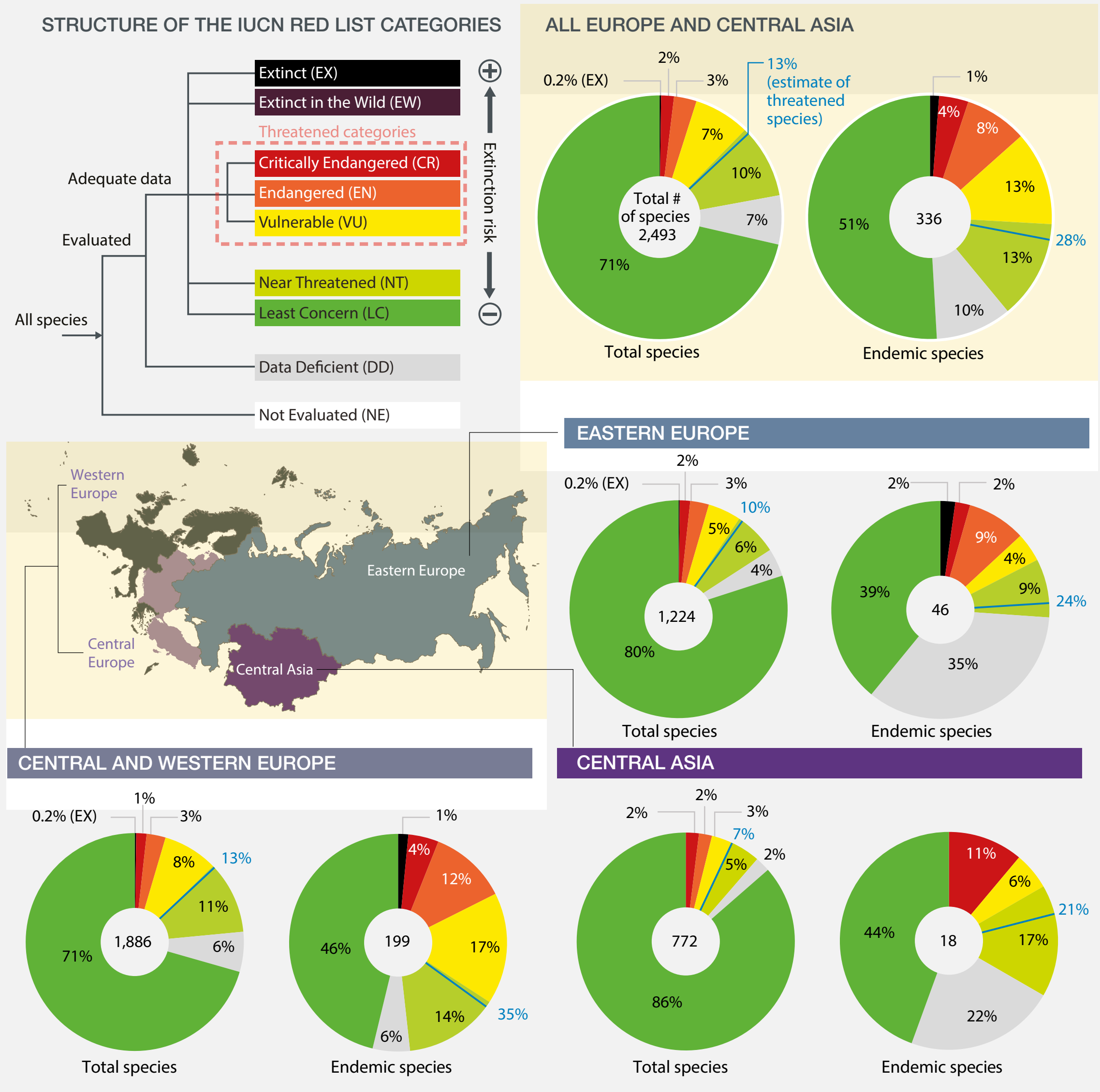


Figure 3 45 **Red List Indices of species extinction risk weighted by the fraction of each species' distribution occurring within Europe and Central Asia and its subregions.**

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).

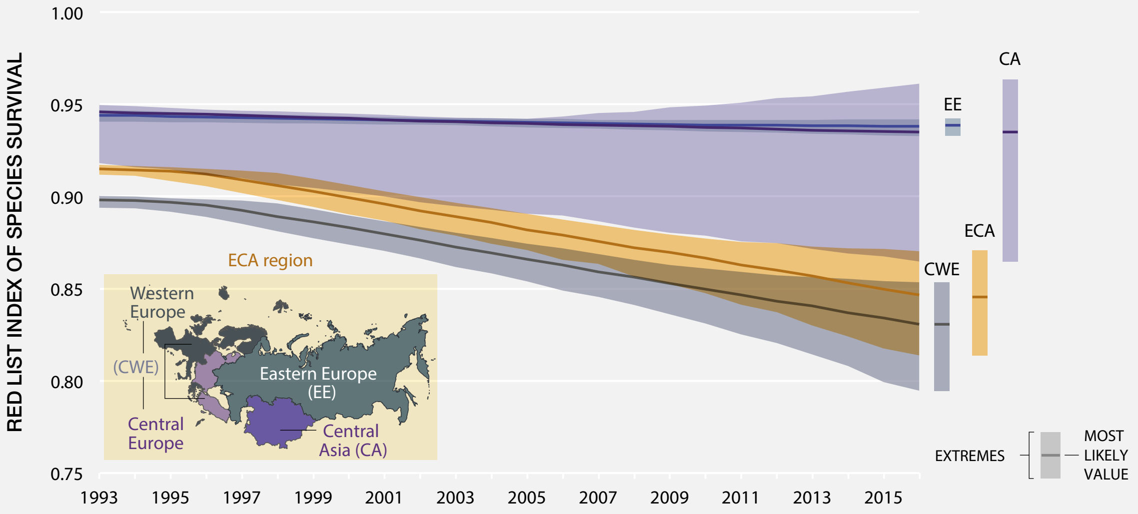


Table 3 6 **Long-term and recent trends of bird species breeding in EU-27 countries (EEA, 2015a). Short-term covers the time period 2001-2012, and long-term the period 1980-2012. The total number of species is 456.**

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.

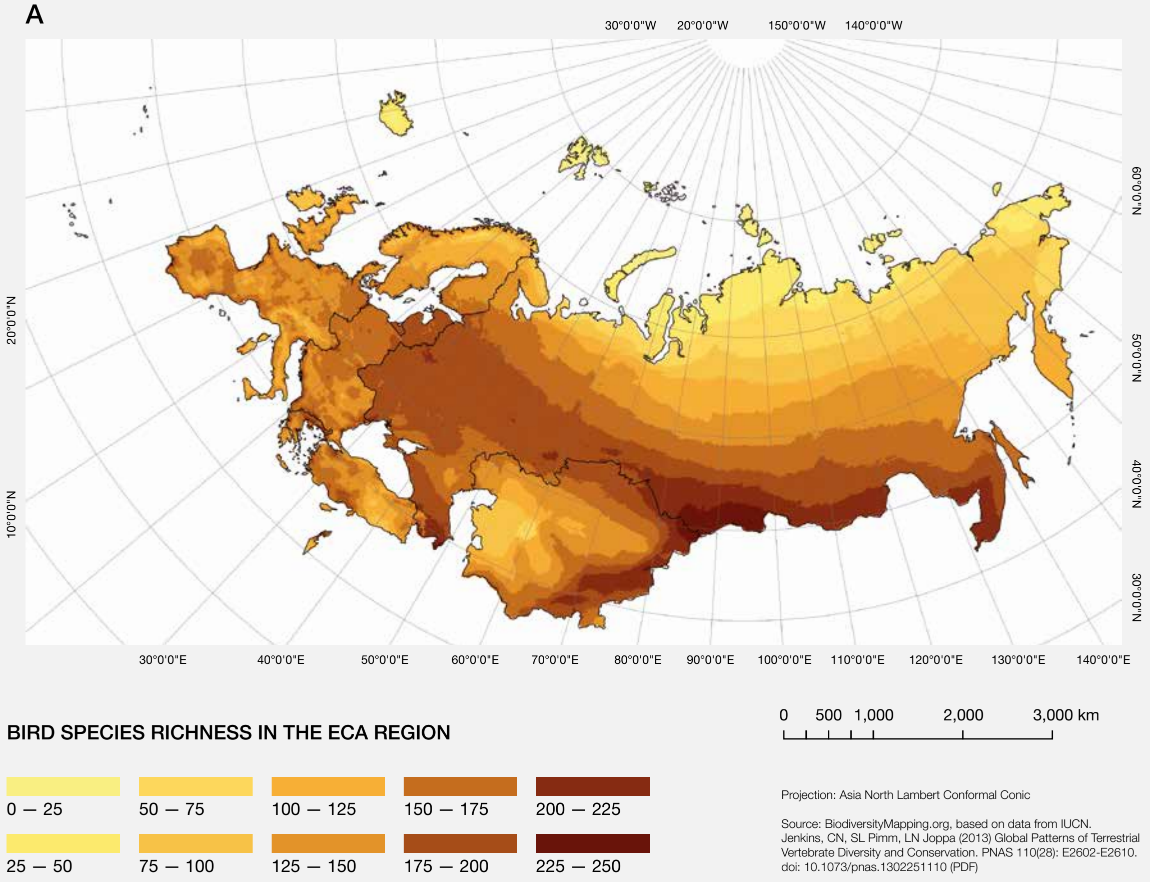
Invasive alien species and invasive native species (e.g. rats, domestic cats), have been shown to threaten the reproductive success of many birds, particularly colonial seabirds (BirdLife International, 2015), and have been linked with declines of some species (e.g. Skorka *et al.*, 2010).

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).

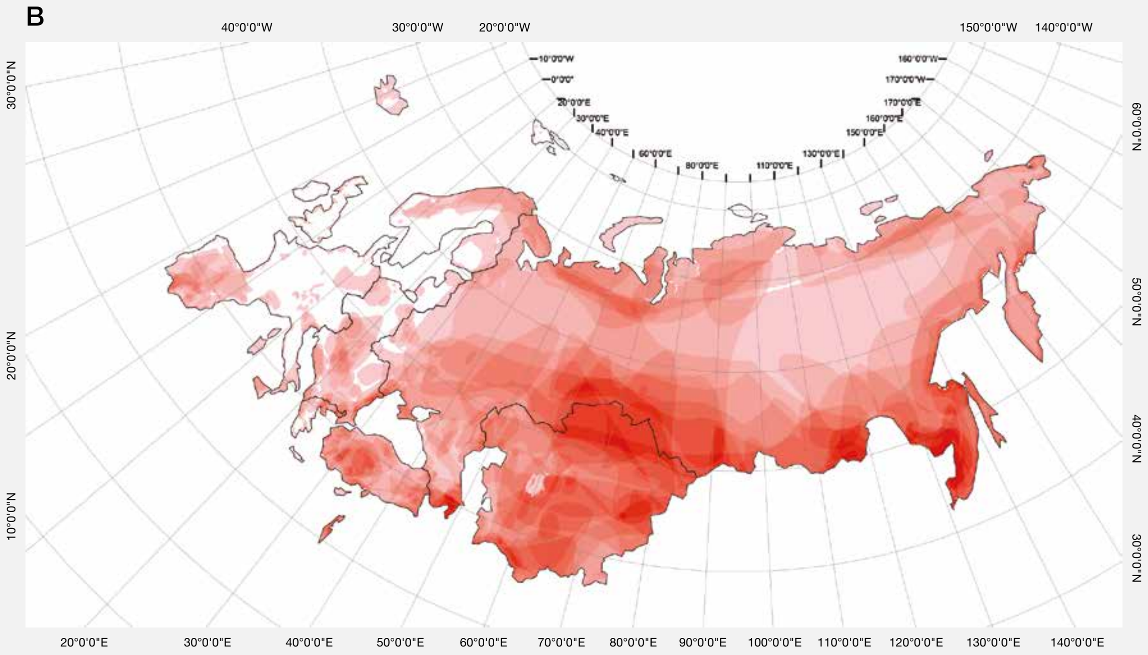
In many cases, it is the combination of drivers that put bird species at risk. Seabirds, for instance, have declined strongly due to a multiplicity of threats. Conservation

Figure ③ ㊻ **Ⓐ 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.**
Ⓑ 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.



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)



NUMBER OF THREATENED BIRD SPECIES IN THE ECA REGION

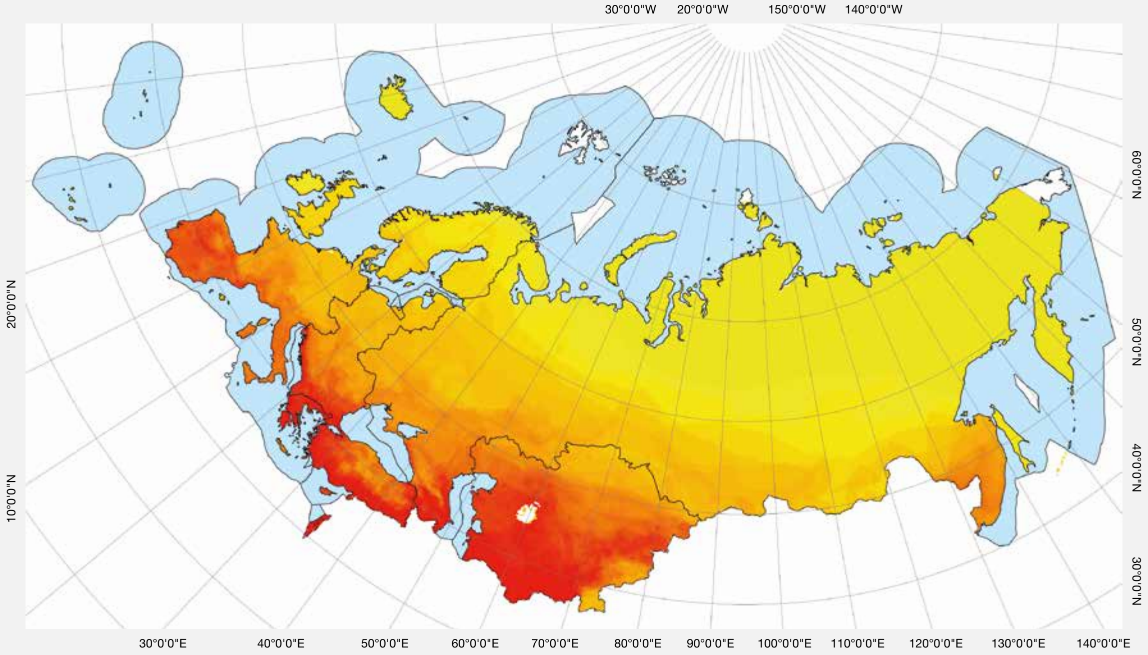
Number of bird species per 10 km²

HIGH: 12 LOW: 1

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

Projection: Asia North Lambert Conformal Conic

Source: Data from IUCN 2017, published in Roll, Uri, et al. Nature ecology & evolution (2017).

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

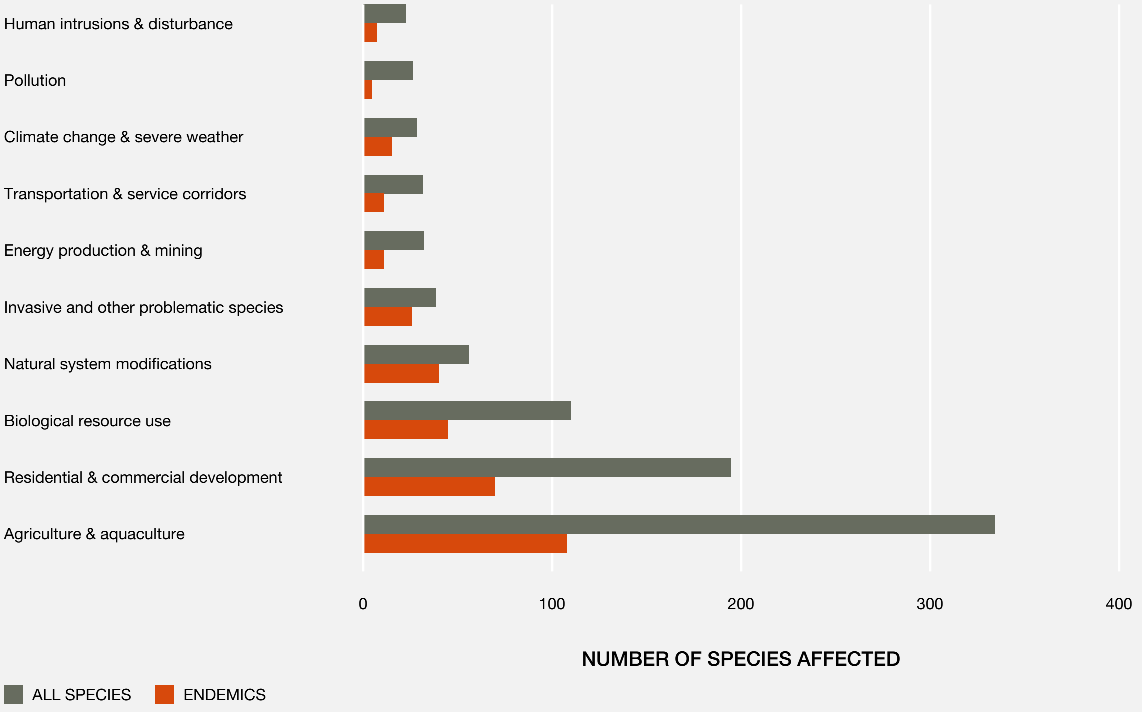
The main threats to reptiles in Europe and Central Asia, according to the IUCN Red List, are agriculture, residential/commercial development, and biological resource use **(Figure 3.48)**. These threats primarily cause habitat fragmentation and loss.

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).

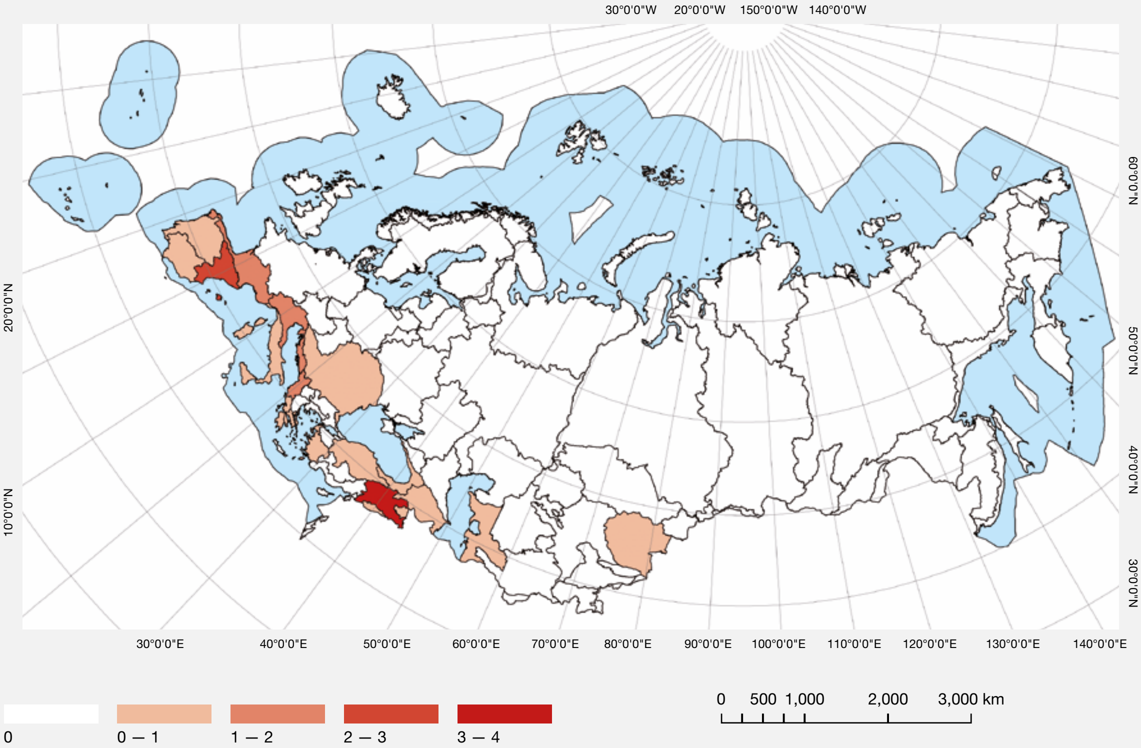
Amphibians represent the third most endangered group of vertebrates in the European Union, with 23% of species (19 species out of the 83 assessed) considered as threatened (Temple & Cox, 2009) **(Figure 3.49)**. According to the Habitats Directive, more than two-thirds of the amphibian species assessed 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).**

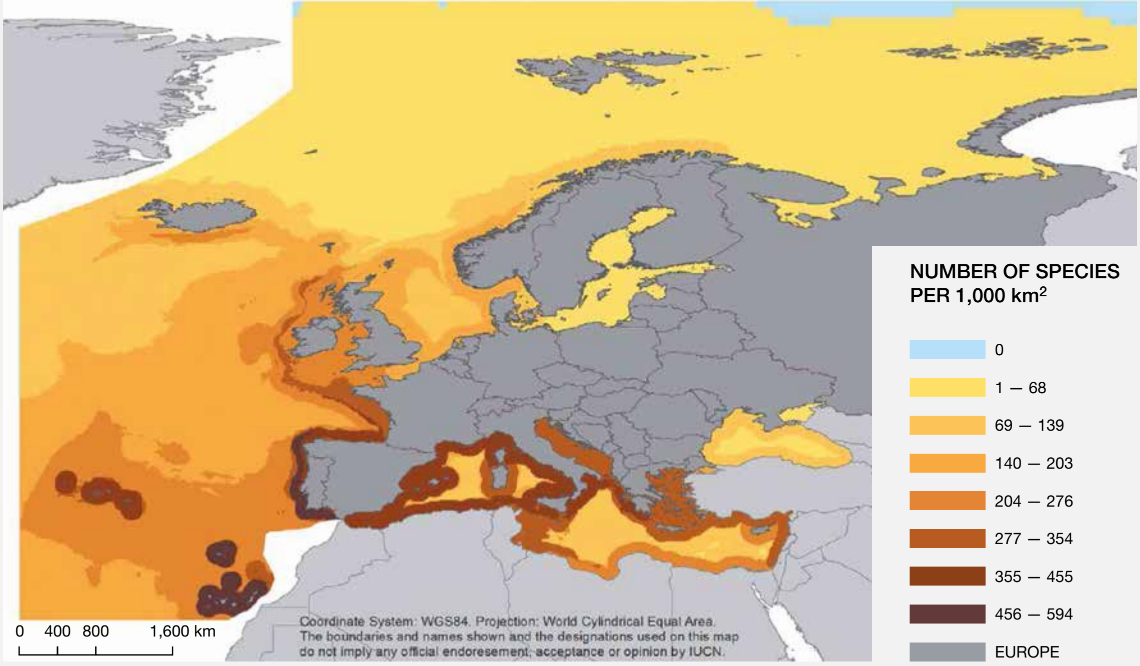
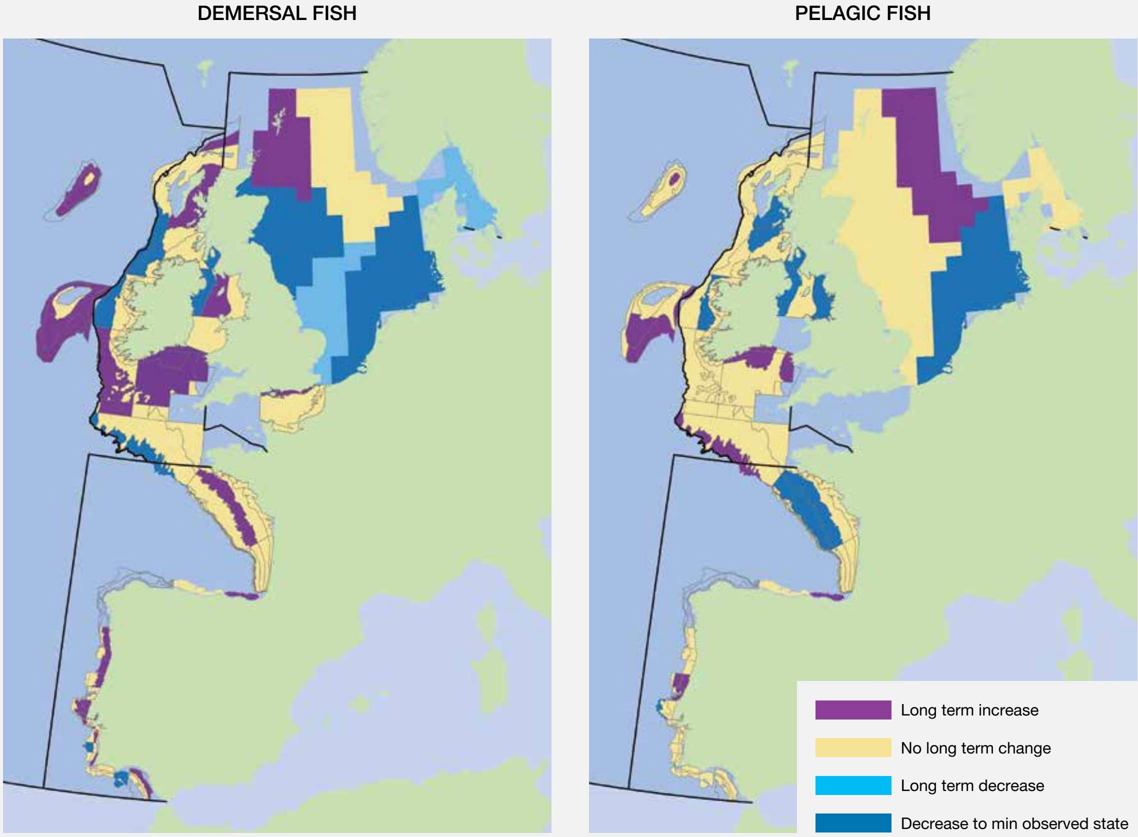


Figure ③ (51) **Spatial pattern of trends in the "typical length" of fish caught in surveys. Source: OSPAR (2017).**



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

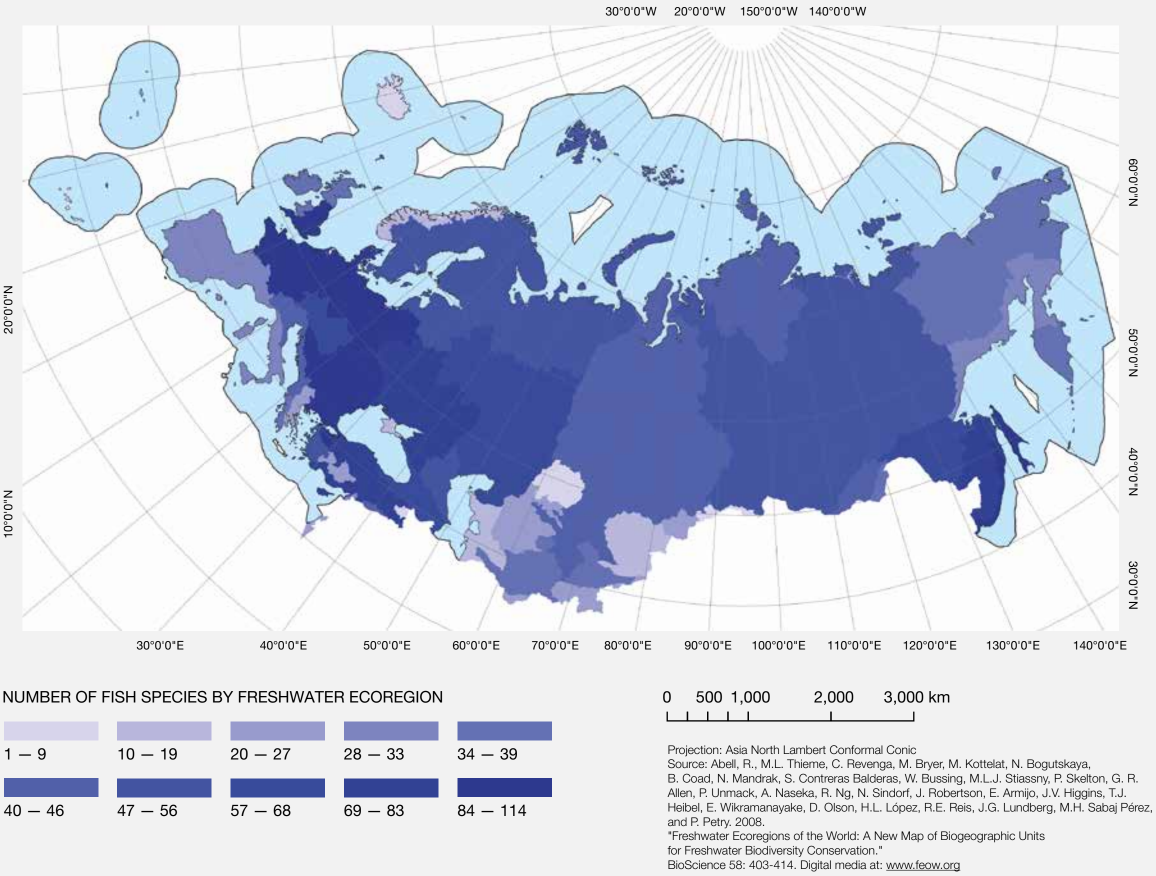
The European Union contains 546 native species of freshwater fish of which, according to IUCN assessments, at least 37% are threatened and 4% are considered near threatened (Freyhof & Brooks, 2011). This is currently the second most threatened taxonomic group assessed, after freshwater molluscs. The 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**).

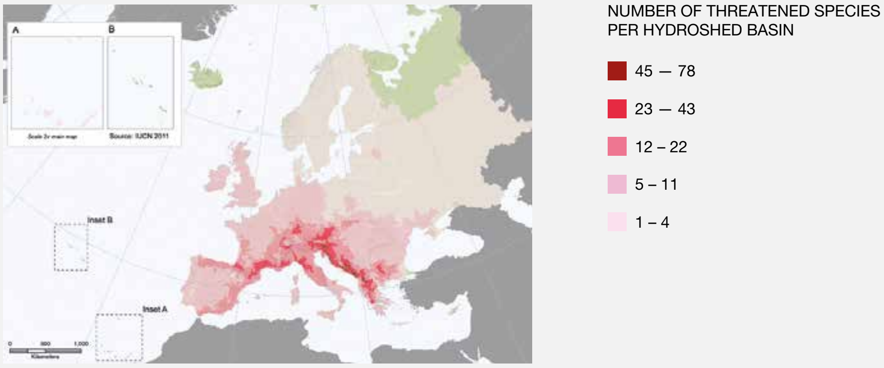
Villéger and co-authors (2014) have also shown that among current European fish assemblages, functional homogenization (reduction in diversity of functional traits over space and time) exceeds taxonomic homogenization (reduction in species diversity) six-fold. In addition, non-native species originating 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).**



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).**



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 km^3), 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 km^2) 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).

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[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 incluing 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

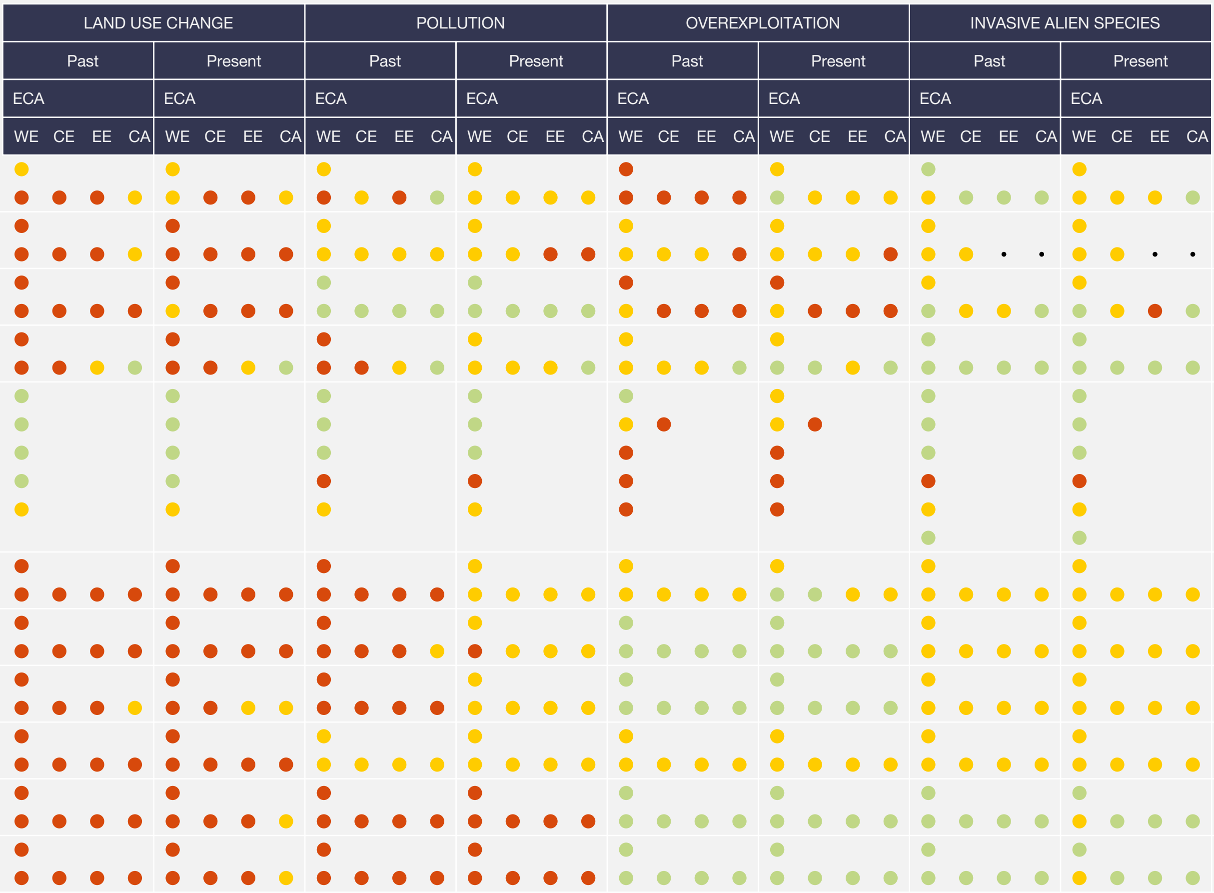
		GENERAL TREND										CLIMATE CHANGE									
		Past					Present					Past					Present				
		ECA					ECA					ECA					ECA				
TAXON			WE	CE	EE	CA		WE	CE	EE	CA		WE	CE	EE	CA		WE	CE	EE	CA
Mammals			↘	↘	↘	↘		↘	↓	↓	↘	🟢	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🟡	🟡
Birds		↘	↘	↘	↓	↘	↘	↘	↘	↓	↘	🟢	🟢	🟢	🟢	🟢	🟡	🟡	🟡	🟡	🟡
Reptiles		↑	↗	↗	↗	↗	↑	↔	↗	↗	↔	🟡	🟢	🟢	🟡	🔴	🟡	🟡	🟡	🟡	🔴
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.



LAND USE CHANGE								POLLUTION								OVEREXPLOITATION								INVASIVE ALIEN SPECIES							
Past				Present				Past				Present				Past				Present				Past				Present			
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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
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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, 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

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[11] http://iucn.ekoo.se/en/iucn/welcome

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.

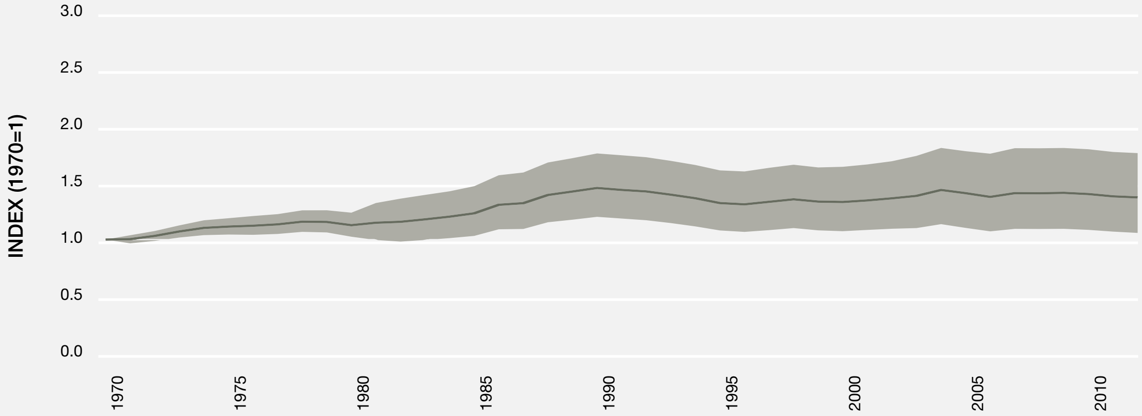
[13] Number obtained by intersecting IUCN data on direct threats to species with population trends from the IUCN Red List Database version 2017.1 (IUCN, 2017c) on the subset of species listed in the CITES Annexes and whose range overlap with the Europe and Central Asia region. The list of threats considered 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 Ⓐ and species can disperse one mean dispersal distance per generation Ⓑ. Source: Visconti *et al.* (2016).**

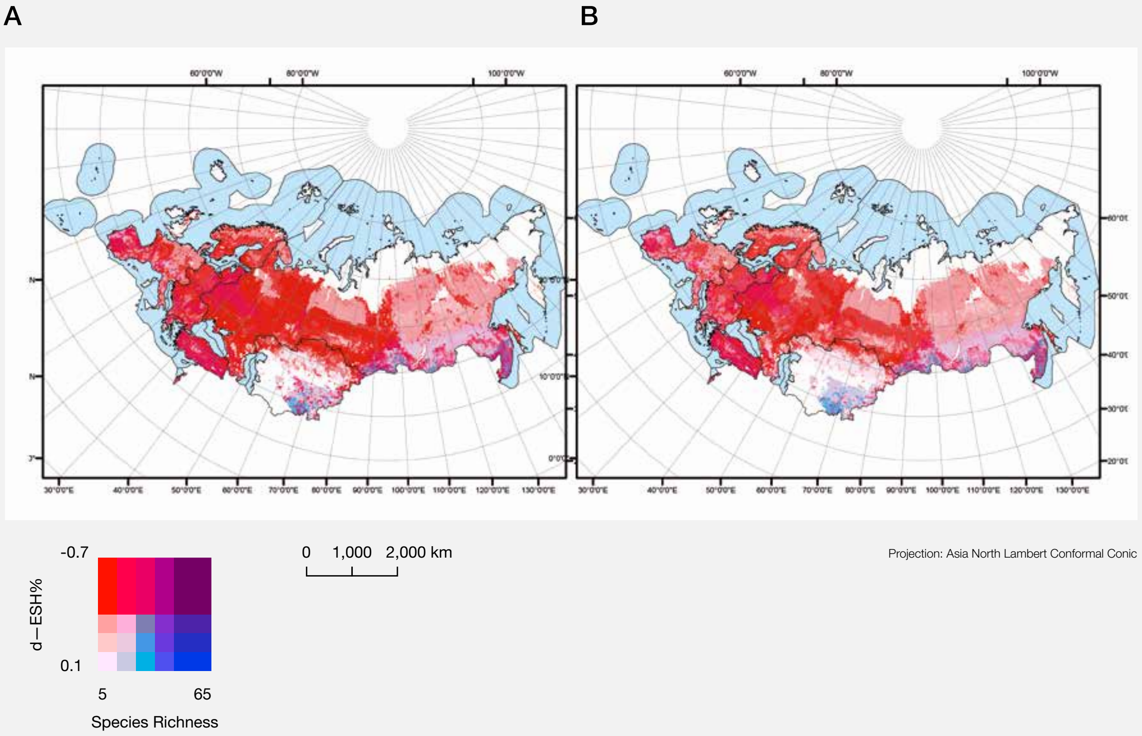
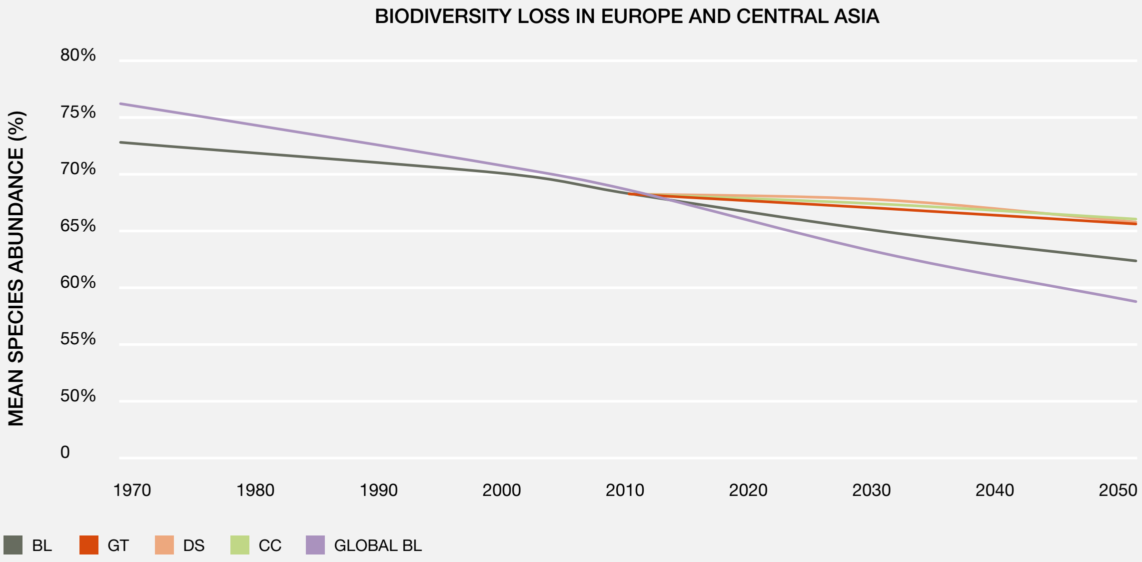


Figure 3 (56) **Trends in mean species abundance across animal and plant groups for four future scenarios for Europe and Central Asia and the world.**

BL: baseline; GT: global technology, DS: decentralized solutions, CC: consumption change. Storylines for each scenarios are in Chapter 5. Source: PBL (2014).



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°C 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).

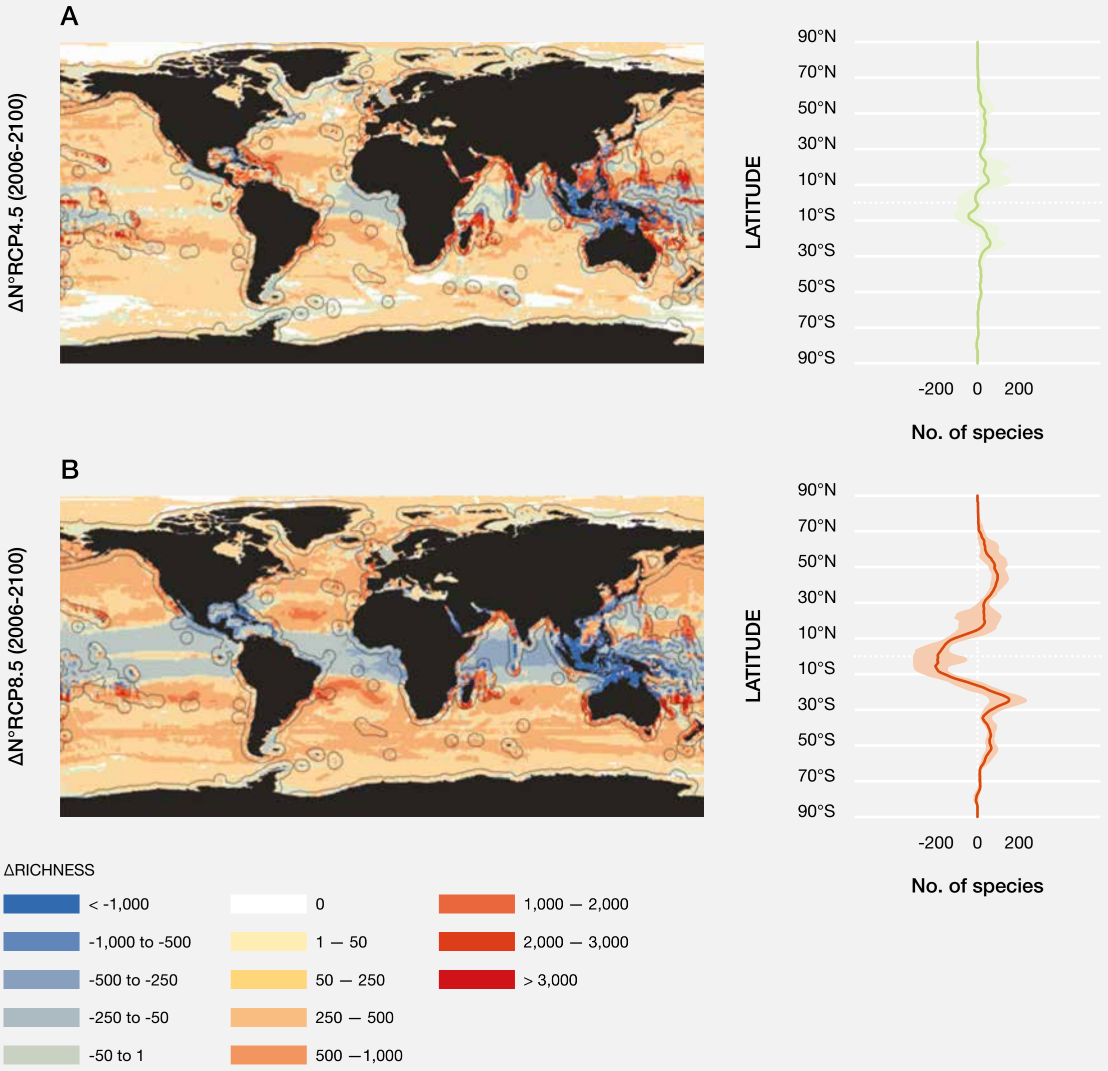
Species range changes, phenological reactions, and variations in production, is expected to cause the Eurasian Arctic Seas ecosystem structure and functions to change (Larsen *et al.*, 2014).

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 ㊼ **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 minimalize 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)**.

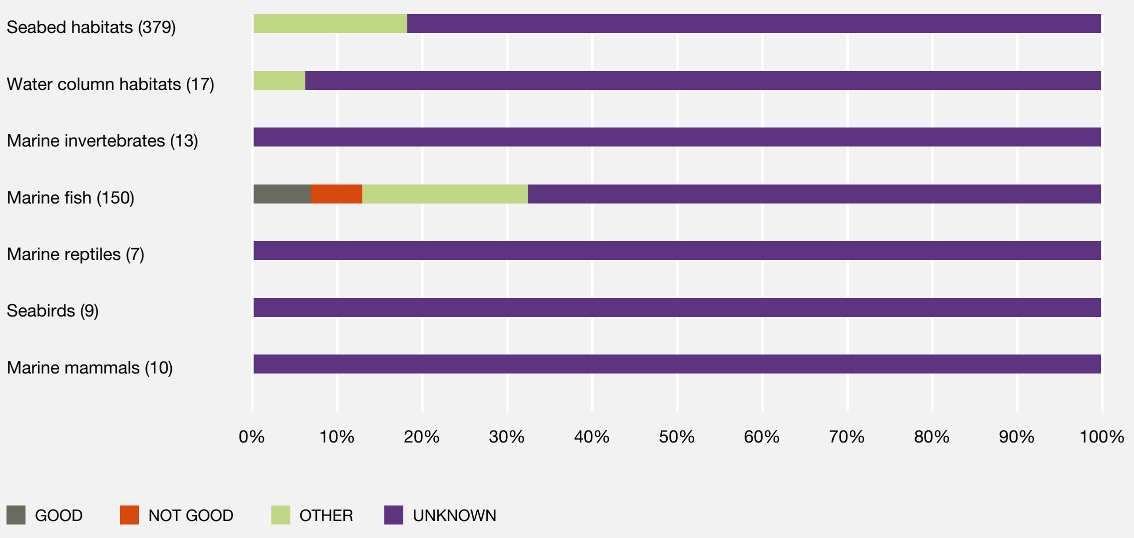
Most long-term marine datasets (since the 1950s) concern pelagic ecosystems (e.g. Beaugrand *et al.* 2002), intertidal rocky shores (e.g. Mieszkowska *et al.*, 2006), or specific taxa or taxonomic groups (in particular fishes, marine mammals or seabirds). Almost no data are available to document changes 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 58 **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.

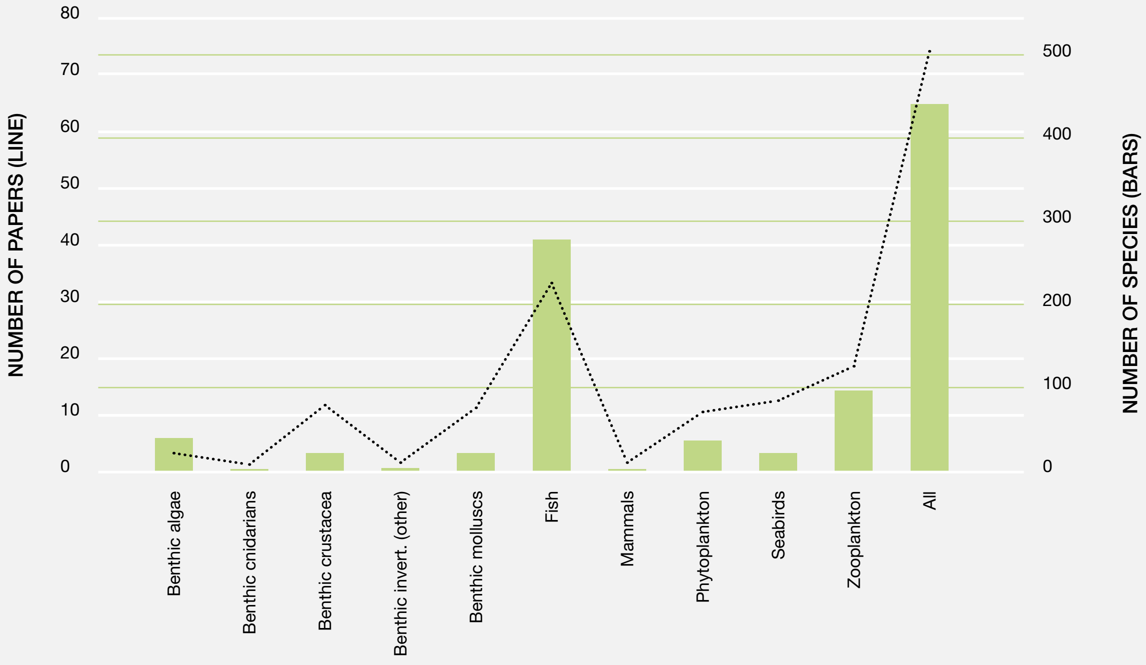
Similarly, almost a quarter of all European freshwater molluscs are data deficient and many might prove to be threatened once enough data become available to evaluate their extinction risk. However, the number of data-deficient species may well increase, since 76% of freshwater fishes and 83% of freshwater 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.

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