4 Chapter 4: Direct and indirect drivers of change in biodiversity and nature's contributions to people

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Executive Summary

The major direct anthropogenic drivers – natural resource extraction, land-use change, pollution, climate change and invasive alien species – all strongly impact on biodiversity and nature's contributions to people in Europe and Central Asia, posing substantial risks for nature and human well-being (*well established*) (4.2.1). Direct drivers act independently and in combination, amplifying and altering their context-specific individual and combined effects on biodiversity and nature's contributions to people (*well established*) (4.2.3, 4.9.1). For example, the impacts of climate change are considerably exacerbated by adverse land-use changes. Direct drivers also impact each other through different feedback systems and alter driver trends (*established but incomplete*) (4.9.1). Indirect drivers – institutional, economic, demographic, cultural & religious and scientific & technological – interactively determine the trends and impacts of direct drivers (*well established*) (4.2.3).

The belief that further GDP growth will facilitate sustainable development is a deeply rooted cultural driver, especially evident in Western and Central Europe, calling for smart, inclusive and sustainable growth. However, this requires an absolute decoupling between GDP growth and degradation of biodiversity and nature's contributions to people which has not generally been observed (*established but incomplete*). Such decoupling is theoretically possible but would require a radical change in policies and tax reforms at the global and national levels (4.3.1, 4.3.2, 4.3.4). Domestic material consumption has increased in almost all European Union countries since the year 2000 (except for the economic contraction following the financial crisis in 2008), supported by growth-oriented policies (4.4.4.2). There is some evidence that human well-being does not increase further once a certain income threshold has been reached. Indeed, the sustainability challenge is to decouple quality of life (well-being) from environmental degradation and pay less attention to GDP (*unresolved*) (4.3.2, 4.3.4). Such decoupling would require new indicators on well-being, equity, environmental quality, biodiversity conservation and nature's ability to contribute to people. Policies for resource efficiency have been implemented, but the tax system continues to impede recirculation and resource efficiency and hence transitions towards a "green economy". For example, the total revenue from environmental taxes in the EU-28 in 2014 was only 2.5% of GDP, or 6.3% of the total revenues derived from all taxes and social contributions. These proportions have decreased since 2002, from 2.6 % and 6.8 %, respectively (*well established*) (4.3.2).

Demography as an indirect driver varies significantly between the subregions, with a dramatic population decrease projected for Central Europe (*established but incomplete*). Urban development will continue to affect natural and semi-natural rural land in large parts of Europe and Central Asia. The population of Europe and Central Asia, 910 million, is stable, but a dramatic population decrease in Central Europe (excluding Turkey) is projected until 2050, from 123 to 104 million, due to currently low fertility rates and high emigration rates (4.3.3). On-going rapid urbanization as people move from rural areas into cities in Central and Eastern Europe and in Central Asia is fuelled by the deterioration of livelihoods in rural areas (4.3.3 and 4.5.6). The consequent urban development results in both urban sprawl and rural land abandonment. In Western Europe, urbanization occurs increasingly as people move from inland areas to coastal cities, which puts further pressure on estuaries and other coastal ecosystems (*well established*). There is a high potential for migration from Turkey and Central Asia to Eastern and Central Europe in the coming decades. Armed conflicts have profound effects on migration; for example, Turkey recently received (by March 2016) over 3 million refugees from Syria, Iraq and Afghanistan. These large migrations may have important effects on other drivers of biodiversity change (*established but incomplete*) (4.3.3).

Conventional intensification of agriculture and forestry has resulted in habitat loss, fragmentation, and degradation and has negative impacts on biodiversity and nature's contributions to people (*well established*) (4.5.1, 4.5.2, 4.5.3). Intensification of agriculture has resulted in conversion of natural and semi-natural habitats on fertile landscapes, with severe negative impacts on biodiversity (*well established*) (4.5.1, 4.5.2, 4.5.3). In marginal lands, the side-effect of agricultural intensification has been the degradation and abandonment of traditionally managed semi-natural habitats and cultural landscapes that support high biodiversity and provide the magnitude of nature's contributions to people (*well established*) (4.5.1, 4.5.2, 4.5.5). Despite agri-environmental schemes and other mitigation measures, conventional intensive agriculture is jeopardizing sustainable land management, biodiversity, and food production (*established but incomplete*) (4.5.1, 4.5.2). Measures including ecological restoration, sustainable approaches to agriculture, e.g. ecological infrastruture that harness nature's contributions to people and inclusion of indigenous and local knowledge, have mitigated some of the adverse effects of intensive agriculture and represent opportunities to simultaneously secure diverse nature's contributions to people and conservation of biodiversity (*established but incomplete*) (4.5.1, 4.5.2).

Production of forest biomass for energy purposes and intensification of forest management have negative impacts on biodiversity and soil quality, as well as an array of material and non-material contributions from nature. The trade-offs between increasing intensity of forestry and delivery of diverse nature's contributions to people are recognized as a major challenge for forestry in Europe and Central Asia. Additionally, there is continuous logging in intact forest landscapes across the region (*established but incomplete*) (4.5.3). Environmental NGOs have played a key role in the adoption of forest certification schemes, which have reduced "wood mining" of remaining intact forests and have led to the inclusion of biodiversity conservation criteria and indicators in intensive forest management systems (*well established*) (4.5.2, 4.5.3).

Abandonment of intensively managed agricultural land has been widespread across Europe and Central Asia (*well established*). However, a comprehensive assessment of the effects of this process on biodiversity and nature's contributions to people is limited by knowledge gaps. In the European Union, cropland area has decreased by almost 1.2 million hectares in recent decades and largely been replaced by forested and urban areas (4.5.2, 4.5.4). Enlargement of the European Union to Central Europe and implementation of the European Union Common Agricultural Policy in new member States have resulted in the reconversion of some of this abandoned farmland to intensive agriculture – a trend that is likely to continue. Eastern Europe and Central Asia are and will remain hotspots of agricultural land abandonment (*well established*) (4.5.2). This has resulted in substantial reduction in livestock, and decline in crop production in these subregions. With the economic recovery and increasing domestic and foreign investments in agriculture after the year 2000, re-cultivation of some abandoned croplands began, particularly in the agriculturally favourable black soil regions in the south of European Russia, Ukraine and northern Kazakhstan (4.5.2, 4.5.3).

Abandonment of extensively managed traditional land-use systems, and loss of associated indigenous and local knowledge and practices, has been widespread in Europe and Central Asia (*well established*) (4.5.5). Cessation of traditional land use has led to loss of semi-natural habitats which support biodiversity of high conservation value (*well established*) (4.5.1). Loss of traditionally managed semi-natural habitats, especially grasslands, has resulted in decline and loss of associated biodiversity and ecosystem functions. Demographic trends, including urbanization, continue to diminish indigenous and local populations, with concomitant negative impacts on traditional land-use knowledge, culture and identities (*established but incomplete*) (4.5.5). In Europe and Central Asia, production-based subsidies driving growth in agricultural, forestry and natural resource extraction sectors tend to exacerbate conflicting land-use issues, often impinging on available territory for traditional users (*established but incomplete*) (4.5.5). In some areas, traditional practices are maintained to a certain extent, and traditional ecological knowledge is adapting to new ecological and socioeconomic conditions. Maintenance of traditional land use and lifestyles in Europe and Central Asia is strongly related to institutional adequacy and economic viability. Traditional land uses and knowledge are becoming increasingly recognized for their value in solving problems related to biodiversity conservation and the sustainable use of natural resources and ecosystems (*established but incomplete*) (4.5.5). The growth of green tourism and demand for products derived from traditional practices and the availability of subsidies for traditional land uses are important factors in ensuring the economic viability of indigenous peoples and local communities (*well established*) (4.5.5).

Protected areas have enormous importance for biodiversity conservation, and the area under protection has been constantly expanding during recent decades across the region (*well established*) (4.5.4). In Europe and Central Asia, the total coverage of areas declared as protected is 10.2%, with 13.5% of the terrestrial area and 5.2% of the marine area being protected. Natura 2000 in the European Union represents a systematic effort to develop new protected areas (4.5.4). Measures to improve environmental status within conservation areas combined with landscape-scale approaches that improve matrix quality for native biodiversity are needed (*established but incomplete*) (4.5.1.7). The prioritization and implementation of adequate legal frameworks for protected area development has largely been driven by the adoption of international agreements, as well as increasing public environmental awareness. The perceived trade-offs with economic development goals, however, have in many cases delayed the development of, or weakened, adequate nature conservation policies. The inadequacy of institutions in navigating local resistance to protected areas and regulating the negative impacts of conflicting land uses outside of protected areas poses important problems for biodiversity conservation. Environmental NGOs have had an important impact in building public awareness of the role of nature protection, leading to shifts in consumer preferences and political priorities. Additionally, Europe and Central Asia is unfortunately the arena for a number of recent and current armed conflicts. Armed conflict has many deleterious effects on protected areas, including multiple direct and indirect environmental impacts, diversion of economic resources from protected area budgets, loss of institutions and human resources, and interruption of long-term monitoring. There is considerable evidence that protected areas alone cannot prevent global biodiversity loss (*well established*) (4.5.4).

Within the present institutional framework, fishing, hunting, and mining pose considerable threats to biodiversity (*well established*). Depletion of local mineral and fish stocks are disguised by global trade, which delays effective responses, and harmful subsidies exacerbating unsustainable extraction levels (*established but incomplete*). Fossil fuels and rare earth minerals are the largest contributors to GDP in Central Asia and the volume of coal mined has doubled in the last decade. The mineral extraction industry in Central Asia has been driven by trade liberalization and increasing world market prices (*well established*) (4.4.4.2). Demand for fish in the European Union continues to exceed the sustainable yield and an increasing proportion of fish is imported. In a closed market economy, the local shortage of material contributions to people due to excessive use would increase prices, drawing attention to the shortage and the reasons for it. However, in a global economy these feedbacks (price signals and awareness) are often masked by substitution. For example, the shortage of cod in Europe has partly been substituted by cod and other white fish from other regions (4.4.1). The more successful globalization and substitution becomes, the longer the delays between declining material contributions to people, e.g. fish stocks, within one region, and policy responses in this region to correct that decline (*established but incomplete*) (4.2.5, 4.4.1). Institutional drivers have changed, e.g. the European Union's Common Fisheries Policy, but economic drivers have not (4.3.1, 4.3.2, 4.4.1.3). Inefficiently low prices of fish are further lowered by harmful subsidies and technological drivers, which result in high harvest levels despite declining stock. Europe, mainly the European Union and Russia, continue to pay about 6 billion USD annually in capacity-enhancing (harmful) fishing subsidies (*well established*) (4.4.1.3).

Despite effective regulations for some forms of pollution, this direct driver still poses major threats to biodiversity, nature's contributions to people and human health *(well established)*. The drivers of pollution are mainly economic, i.e. effects of industrialization and globalization, including conventional intensive agriculture and increases in transportation (*well established*). Pollution is also increased by institutional drivers that foster adverse technological development and the cultural belief that a prosperous life must entail more material consumption (*unresolved*). Pollution is a function of the industrial development model (4.6.6) and in general correlated to GDP (4.3.2) *(established but incomplete)*. However, some pollution problems such as acidification and eutrophication of terrestrial ecosystems have been decreasing in Western and Central Europe since 1990, from 30% and 78%, respectively, of areas exceeding critical pollutant loads of sensitive ecosystems, to 3% and 55%, respectively. This has mainly been accomplished by regulations (*well established*) (4.6.1, 4.6.3). Phosphorous and nitrogen (except ammonia) pollution is decreasing in Europe but, partly due to time lags, many terrestrial systems and a large proportion of lakes and rivers in Western and Central Europe continue to be negatively affected (*well established*) (4.6.1, 4.6.2). Although marine and coastal eutrophication has decreased, the number of marine dead zones due to oxygen depletion resulting from nutrient and organic pollutants has increased markedly (*established but incomplete*) (4.6.1, 4.6.2). Overall, there is evidence that pollution particularly negatively affects freshwater and marine biodiversity and water quality across Europe and Central Asia (*well established*). Global sales by the chemical industry doubled between 2000 and 2009 and continue to increase. Due to synergistic or "cocktail" effects, substances present in concentrations below recognized health threshold values can still be toxic, leading, for example, to human hormone disruption (*well established*) (4.6.4). Two kinds of pollution are increasing rapidly: plastic debris and microplastics affecting a wide array of marine organisms; and artificial light at night affecting terrestrial, aquatic and marine ecosystems (*established but incomplete*) (4.6.5).

There is strong evidence that the climate of Europe and Central Asia is changing towards warmer temperatures and regionally changed precipitation (*well established*) (4.7.2.1, 4.7.2.2), with generally drier summers in the southern and wetter winters in the northern parts of the region and increasing risk and amplitude of extreme climatic events such as droughts and storms (*established but incomplete*) (4.7.2.2, 4.7.2.5). Evidence that climate change impacts biodiversity and nature's contributions to people is emerging rapidly, and climate change is likely to become one of the most important drivers in the future, especially in combination with other drivers (*established but incomplete*) (4.9.2.2). The temperature will increase in the next decades and most units of analysis (biomes and land cover types) will experience an average warming between 1 and 3 °C by 2041-2060 relative to 1986-2005, with larger increases for northernmost biomes such as snow and ice dominated ones and tundras (*well established*) (4.7.1.2). Precipitation patterns are projected to change across Western and Central Europe: drier climates and increased drought risk in their south-west, no change or increased precipitation in their north-west, while trends for Eastern Europe and Central Asia are ambiguous (*established but incomplete*) (4.7.2.2). Effects on biodiversity and nature's contributions to people vary according to the ecosystem itself, in particular depending on whether productivity is precipitation-, radiation- or temperature-limited. Climatic warming and precipitation change are driving shifts in seasonal timing, growth and productivity, species ranges and habitat occupancy with impacts on biodiversity, agriculture, forestry, and fisheries (*well established*) (4.7.1.1). Knowledge of the underlying processes and mechanisms suggests that many species will not be able to respond, migrate or adapt fast enough to keep pace with the projected rates of change in mean climate conditions, threatening ecosystem functioning and livelihoods (*established but incomplete*) (4.7.1.1.2). Across Europe and Central Asia, increased drought results in decreased primary productivity, increased net carbon flux to the atmosphere, nutrient leaching from terrestrial systems and algal blooms, biodiversity loss, and decreased water quality in aquatic systems (*established but incomplete*) (4.7.1.1). The fifth assessment report of the Intergovernmental Panel on Climate Change established that economic growth is the main driver of greenhouse gas emissions and hence climate change in Europe and Central Asia (*well established*) (4.7.3). From 1970–2010, economic growth has been only partially offset by improvements in the energy intensity of the economy and the emissions intensity of energy production, and policies have proved insufficient in influencing infrastructure, technological, or behavioural choices at a scale that curbs the upward greenhouse gas emissions trends (*well established*) (4.7.3). Per capita emissions vary widely, depending on geography, income, lifestyle, and the available energy resources and technologies, leading to differences in climate footprints within Europe and Central Asia (*established but incomplete*) (4.7.3).

Evidence is emerging that indirect climate change effects, such as increased fire and flood risks and loss of permafrost are affecting biodiversity and nature's contributions to people in Europe and Central Asia (*well established*) (4.7.1.3). Increased precipitation, especially in winter, will result in increased flood risk in the northern parts of Western and Central Europe (*established but incomplete*) (4.7.2.1). Floods are a serious hazard to people, and increase erosion, water turbidity and eutrophication, impacting freshwater provisioning (*established but incomplete*) (4.7.1.2). Increased fire risk is projected across large parts of Western and Central Europe (*established but incomplete*), while projected increases in fire danger for Eastern Europe and Central Asia are uncertain. Near-surface permafrost extent at high northern latitudes is projected to decrease by between 37% (RCP2.6) and 81% (RCP8.5) by the end of the 21st century, (*established but incomplete*). In Arctic and alpine regions, permafrost melting may lead to large greenhouse gas emissions, and short-term heat waves negatively impact productivity and may result in reduced food availability for wildlife and livestock (*unresolved*). Climate change further leads to ocean acidification, sea level rise and changes in ocean stratification, generally resulting in biodiversity loss, reduced growth and productivity and hence impaired fisheries and increased release of CO_2 to the atmosphere (*established but incomplete)* (4.7.1.3).

Invasive alien species have increased in number and for all taxonomic groups across all subregions of Europe and Central Asia and this has severe effects on biodiversity and nature's contribution to people (*well established*). For Eastern Europe and Central Asia, the rate of invasion has been less severe than in Western and Central Europe, but is expected to increase at a rate that strongly depends on GDP development (*established but incomplete*) (4.8.1, 4.8.2). Rates of increase in numbers of invasive alien species are strongly correlated with introduction rates. Introduction rates of alien species are strongly related to trade networks and have increased dramatically over the last 200 years in all environments (terrestrial, freshwater and marine), with 37% of first records reported from 1970-2014 (*well established*). Invasive alien species are affected by interactions with other drivers of change such as land-use change and climate change (*established but incomplete*). The invasion process (transportation, introduction, establishment and spread) is influenced by economic factors. Major pathways of introduction in Europe and Central Asia include horticulture and ornamental trade, accidental transportation, creation of commercial paths such as canals, and tourism (*well established*). International, national and sub-national legal instruments targeting invasive and alien species have been developed in Western and Central Europe but are currently lacking in Eastern Europe and Central Asia. In addition, proactive educational outreach programmes as well as trans-boundary legal instruments targeting major introduction pathways have shown promising potential for improved prevention and earlier detection of invasive alien species (*well established*). However, Aichi

Biodiversity Targets 5 and 9 are unlikely to be achieved for Europe and Central Asia because of ongoing habitat conversion and fragmentation (Target 5) and because invasive alien species are not adequately controlled and are still increasing in numbers (Target 9) (*established but incomplete*) (4.5.1, 4.8.2). Invasive alien species generally tend to have negative effects on biodiversity and nature's contributions to people. However, their magnitude and direction vary among types of impact, taxa and environments (*well established*) (4.8.1).

In addition to immediate effects, the individual and combined effects of natural resource extraction, land-use change, climate change, diffuse pollution and invasive alien species can have chronic, prolonged and delayed impacts on biodiversity and the provision of nature's contributions to people, due to considerable time-lags in the response of ecological systems (e.g. extinction debt, colonization time-lags) (*well established*) (4.9.1). For example, species extinctions due to habitat area loss and increasing fragmentation can take decades or centuries due to the slow intrinsic dynamics of populations of many species (*well established*) (4.5.1, 4.9.1). Climate change can have delayed effects on change in species distribution patterns and development of species assemblages under new conditions because of time lags in population response and migrational lags (*established but incomplete*) (4.7.1.1.2, 4.9.1). Nutrient pollution continues to influence terrestrial and aquatic ecosystems for decades after external inputs are reduced (*well established*) (4.6.1). Considerable delays occur between the initial introduction of alien species and their possible spread as invasive alien species (*well established*) (4.8.1). Such time-lags introduce uncertainty and can lead to serious underestimation of the effects of current direct drivers on biodiversity and nature's contributions to people. Decisive and proactive policies would avoid future loss of species and nature´s contributions to people (*established but incomplete*) (4.5.1, 4.6.1, 4.7.1, 4.8.1, 4.9.1).

4.1 Introduction

4.1.1 Aim of the chapter

The aim of this chapter is to assess evidence of the status and trends of the drivers that affect biodiversity and nature's contributions to people. There are three wider categories of nature's contributions to people: regulating, material and non-material contributions, that are similar to, but not identical to classifications of ecosystem services (see Chapter 1). Ecosystems are dynamic interacting networks of animals, plants, fungi, and microorganisms, above and below ground and water-surfaces. These biodiverse networks of interacting organisms respond to a set of environmental factors such as climate, soil, or water conditions. Social-ecological systems also include human activities (direct drivers) that modify almost all of these ecosystem interactions and environmental factors, and the underlying societal (indirect) drivers of these activities. It is thus important to understand the status and trends of the direct and indirect drivers that affect biodiversity, including ecosystems and, thereby, affect nature's contributions to people.

4.1.2 Scope and organization of the chapter

This chapter focuses on the effects of drivers on biodiversity and nature's contributions to people and thereby only indirectly on quality of life, which is dealt with in greater detail in Chapter 2. Section 4.1 describes the scope of the chapter, the role of drivers in the IPBES conceptual framework, and methodological approaches. Section 4.2 explains which system of "drivers of change" is addressed in this assessment. We compare and specify concepts which have been used in earlier assessments to justify the choice of direct and indirect drivers, including their sub-categories. The section also discusses the importance of the temporal and spatial variability of drivers and interregional flows. Section 4.3 assesses major trends in the five individual indirect drivers in Europe and Central Asia. Indirect drivers are then assessed for each direct driver in the subsequent sections.

Chapters 2 and 3 of the IPBES Regional Assessment for Europe and Central Asia identified strong evidence that biodiversity and nature's contributions to people are declining, and that natural resource extraction, land-use change, pollution, climate change, and invasive alien species are the main direct drivers of these changes. Sections 4.4 to 4.8 assess five direct drivers, one in each section. We first assess the overall effects of the direct drivers on biodiversity and nature's contributions to people in Europe and Central Asia (please note that the specific effects of direct drivers on specific taxa and each unit of analysis (i.e. types of ecosystems, see Chapter 1) are the subject of Chapter 3). After establishing the general effects of the direct drivers, we provide an assessment of the trends in each direct driver and sub-categories of the drivers within the different regions and units of analysis over the recent past (20-40 years) and projected into the future (50-85 years). We use the word "projected" rather than "predicted" because in the medium long run, predictions of the future are not possible. Then we assess the indirect drivers that underpin the direct drivers of changes in biodiversity and nature's contributions to people. As described below, the indirect drivers interact considerably and are often context specific, and therefore they should not be assessed in isolation. We use causal loop diagrams (CLDs) to illustrate some of the complex interactions and causal relationships affecting each driver.

Section 4.9 synthesizes the main findings for the overall trends in, and impacts of, drivers on biodiversity and nature's contributions to people across subregions and biomes (the unit of analysis) in the past and projected into the future. For direct drivers, this synthesis is based on an assessment of all sub-categories of drivers and their compound impacts. For indirect drivers, the synthesis in 4.9.3 is based on the empirical sections.

4.1.3 Driver as a concept

The distinction between "indirect" and "direct" drivers was popularized by the Millennium Ecosystem Assessment (MEA, 2005b) and this classification still dominates the debate on ecosystem change (e.g. Pereira *et al.*, 2010). The older DPSIR terminology (drivers, pressures, states, impacts, responses), popular in Western Europe (Stanners & Bourdeau, 1995), divided drivers into "driving forces" and "pressures", with the former corresponding to indirect drivers and the latter corresponding to direct drivers of the Millennium Ecosystem Assessment (Tzanopoulos *et al.*, 2013).

4.1.4 Natural and anthropogenic drivers

Analytically it is sometimes difficult to distinguish whether an element (process, factor, driver) belongs to the natural or the human system. Biogeophysical processes and factors such as volcanic eruptions, tsunamis, El Nino, solar radiation, or storms, are natural and they influence all elements of life on earth. These "natural drivers" and extreme events are not assessed in this chapter. Our analysis is limited to drivers linked to human activities, and are therefore considered anthropogenic or at least anthropogenically influenced drivers. In this context, direct drivers are the result of human interactions with natural processes that directly act upon biodiversity, including ecosystems, by altering natural processes, while indirect drivers are structures and processes governing the human interactions, thereby influencing direct drivers. So, while we would consider climate and weather, habitats, and species' dispersal and range dynamics to be natural processes, anthropogenic climate change, land-use change and invasion by alien species reflect the human influence on climate, land use and biodiversity dynamics, respectively.

However, to unequivocally disentangle natural variability from anthropogenic drivers is often difficult. Human impacts now affect more than half of the Earth's ice-free terrestrial surface (Ellis *et al.*, 2010) and humans now exert a dominant influence on key Earth system processes and on ecosystem change and biodiversity loss (Newbold *et al.*, 2015, 2016; Steffen *et al.*, 2007). This has led to the coining of a new geological epoche, the "Anthropocene" (see Crutzen, 2002). While there is debate over when, exactly, the transition from the Holocene to the Anthropocene occurred, it is often set to when human impacts took over as a dominating influence on the earth system processes, early in the 20th century (see Steffen *et al.*, 2007). Human influences were also present prior to this transition, and the nature and magnitude of the impacts through time and especially in the more distant past 3,000-8,000 years ago, are still debated (Ruddiman, 2013; Scott *et al.*, 2014). This assessment takes a pragmatic approach to this challenge, focusing on assessing the impacts of major modern (i.e., post-industrial) anthropogenic drivers relative to the more-or-less human affected pre-industrial landscapes (see also Nybø *et al.*, 2017). **Box 4.1** exemplifies the analytical challenges in distinguishing between "natural" or "anthropogenic" factors in the past through the example of forest fires.

Box 4.1: Natural or human control over forest fires in northern Europe?

It is sometimes difficult to clearly separate natural (Earth system) and anthropogenic (human activities) drivers and land use often interacts with natural processes. A typical example is the occurrence of forest fires in the northern boreal forests of Europe and Central Asia.

Long term chronologies based on charcoal in sediments, covering 10,000 years after the last glaciation (Holocene), suggest climate has been the main governing factor for fire regimes (Carcaillet *et al.*, 2007). Although humans have been present during most of the Holocene (based on carbon-14 dated archaeological features) on millennial scales, fire history does not show any relationship to human presence in these remote landscapes.

Hence, Carcaillet *et al.* (2007) suggest that natural processes have been decisive for the long-term fire pattern. It has also been argued, based on dendrochronological (tree-ring based) reconstructions that years with many large fires may be controlled by the climate (Drobyshev*et al.*, 2015).

However, several dendrochronological reconstructions (with high temporal resolution) show clear links between fire patterns and human presence in the landscape during most of the last millennium (Granström & Niklasson, 2008; Wallenius*et al.*, 2004). Granström and Niklasson (2008) depict several fairly distinct periods of human influence on forest fires. In the earliest stage (during the millennia after the deglaciation), prehistoric moose hunters may have used fire to open the landscape in order provide better grazing conditions.

When the Sami people in the northern parts of Europe and Central Asia began changing from hunting reindeer to reindeer husbandry in the 17th century (Hahn, 2000), they had an incentive to ensure that ground vegetation conditions (lichens) were suitable for reindeer. Since only an estimated 1 % of the stands naturally burned every year (Zackrisson, 1977) and the repeated burning was important to create open all-aged tree stands optimal for maintaining lichen cover on the ground (Axelsson & Östlund, 2001; Berg *et al.*, 2008; Östlund *et al.*, 1997), fires were probably an important management practice also for the early forms of reindeer husbandry. When commercial forestry was established from the mid-1800s, it resulted in increasingly effective fire suppression. Dense monoculture forests and lack of fires have reduced the extent of lichen covered areas in Sweden by 70 % since 1955 (Sandström *et al.*, 2016).

On the border between current Finland and Russia there is an apparent mismatch between predicted lightning ignition frequency and observed fire history, suggesting that as far back as a millennium ago, a very small human population may have played a role in the fire history of remote boreal forests (Wallenius *et al.*, 2010a). This leads to the conclusion that potentially the boreal forests that developed after the glaciation have to quite some extent been formed by human presence in the landscape.

End of Box 4.1

4.2 Drivers of change in biodiversity and nature's contributions to people

4.2.1 Direct drivers

The Millennium Ecosystem Assessment (MEA, 2005b) distinguished five major classes of direct drivers of biodiversity change, namely habitat change, climate change, invasive alien species, over-exploitation and pollution (mostly nitrogen and phosphorous). Here, we largely follow this classification of direct drivers, although we use "natural resource extraction" instead of "over-exploitation, to avoid using value-laden terms (**Table 4.1**). However, we still use "over-fishing" since this is such an established term in contemporary global fisheries (Worm *et al.*, 2006). Water extraction and fish harvesting are considered here as two sub-categories of natural resource extraction, not as two separate direct drivers as in Pereira *et al.* (2010). Here, we briefly describe the five categories of direct drivers including sub-categories, and explain what is summarized within each of these (see **Table 4.1** for an overview of all classes). None of the sub-categories is uniform in its expected impacts on biodiversity and nature's contributions to people, and we have, therefore, distinguished a number of further elements within sub-categories when analysing the available information for recent and projected future trends.

Table 4.1: Categories of direct drivers of change in biodiversity and nature's contributions to people. The five major categories are composed of two to six subcategories. More details for each sub-category are given in the text.

Natural Resource Extraction	**Climate Change**
▪ Fishing	▪ Temperature change

▪ Hunting
▪ Water use & desalination
▪ Mineral & fossil fuel extraction

Land use change
▪ Changes in agriculture
▪ Changes in forestry
▪ Changes in protected areas
▪ Changes in traditional land use
▪ Changes in urban development

Pollution
▪ Nutrient pollution
▪ Organic pollution
▪ Acidification
▪ Xenochemical & heavy metal pollution

▪ Precipitation change
▪ Sea-Level change
▪ Glaciers & permafrost
▪ Extreme events
▪ Marine circulation and deoxygenation
▪ Atmospheric CO_2 concentration

Invasive alien species
▪ Terrestrial
▪ Freshwater & Brackish
▪ Marine

Natural Resources Extraction: For biotic resources extraction, we distinguish fishing and hunting. Logging is treated as a sub-category of land-use change and therefore not included here. Gathering of plants for human use (e.g. berries, mushrooms) is identified by IUCN as a threat to biodiversity (Maxwell *et al.*, 2016), but not assessed here. For the extraction of abiotic resources, we distinguish water use & desalination, and mineral & fossil fuel extraction.

Land-use change: Changes in five major land-use categories are assessed, namely: changes in agriculture, forestry, protected areas, traditional land use and urban development.

Pollution: Past assessments focused on pollution from nitrogen and phosphorus (MEA, 2005a, 2005b). In this assessment, we distinguish five main categories of pollutants, namely: nutrient pollution, organic pollution, acidification, xenochemical and heavy metal pollution and "other" pollution (including ground-level ozone, light and plastic pollution).

Climate Change: This driver class has been studied prominently in recent IPCC reports, with regards to both its current and projected future trends, and its expected impacts on terrestrial and marine ecosystems (IPCC, 2013b, 2014a, 2014b). Here, we distinguish seven major sub-categories, namely: changes in precipitation, temperature, atmospheric CO_2 concentrations, glacier and permafrost extent, sea-level, extreme events, and marine ocean-atmosphere interchange.

Invasive Alien Species: An alien species (also known as an exotic or introduced species) is a species occurring in an area outside of its historically known natural range as a result of intentional or accidental dispersal by human activities (CBD, 2011). Invasive alien species (IAS) are alien species whose introduction or spread threaten biological diversity or that have other negative effects on ecosystems, economy or society (CBD, 2011; Roy *et al.*, 2014a). In this report, we distinguish three major categories of invasive alien species, namely: terrestrial, freshwater (including brackish waters), and marine.

4.2.2 Indirect drivers

We identify five categories of indirect drivers, adapted from Hauck *et al.* (2015) building on the MEA (2005b) framework. Some scholars call indirect drivers "underlying drivers" (van Vliet *et al.*, 2015), "underlying causes," "fundamental social processes" (Geist & Lambin, 2002), "categories of origin" or "key driving forces" (Brandt *et al.*, 1999). Hence, there are different attempts to conceptualize indirect drivers (**Table 4.2**). If indirect drivers are the underlying causes of, for example, land-use change or pollution, then the tangible results of human activities can be seen as direct drivers, or "proximate causes". For example, for deforestation, proximate causes can include agricultural expansion, wood extraction or the extension of road infrastructure (Geist & Lambin, 2002). Indirect drivers do not directly impact biodiversity, but may have a direct impact on nature's contributions to people, according to the IPBES conceptual framework. For example, some legal restrictions may reduce nature's contributions to people to certain groups of people and some non-material contributions of nature are co-produced by people and nature (Díaz *et al.*, 2015).

The literature on indirect drivers often treats land-use change as the dependent variable and gives less attention to its consequences for biodiversity and ecosystem services. Van Vliet *et al.* (2015) include "location factors" as an underlying (indirect) driver, consisting of accessibility, climate, topography, and soil quality ("EU" in **Table 4.2**). Similarly, Brandt *et al.* (1999) include "natural environment" ("UNESCO" in **Table 4.2**) as a key driving force, consisting of geomorphology, soil, climate and hydrology. Geist and Lambin (2002) also include "pre-disposing environmental factors" such as soil, topography and fragmentation mediating the underlying drivers ("IGBP-IHDP" in **Table 4.2**). Furthermore, they address biophysical and social triggers, which ecologists call "fast variables" or "disturbances".

Table 4.2: Different categorizations of indirect drivers.

UNESCO[1]	IGBP-IHDP[2]	EU[3]	MA 2005a	IPBES
Socioeconomic	Economic	Economic	Economic	Economic
Policy	Policy/Institutional	Institutional	Socio-political	Institutional
Culture	Cultural	Sociocultural	Cultural & religious	Cultural & religious
–	Demographic	Demographic	Demographic	Demographic
Technology	Technological	Technological	Science & Technology	Scientific & Technological
Natural environment	(Environmental factors)	Location factors	–	–

[1] Brandt *et al.* (1999); [2] Geist and Lambin (2002); [3] van Vliet *et al.* (2015)

Based on Geist and Lambin (2002), we identify biophysical triggers including fires, droughts, floods and storms, and social triggers including revolution, social disorder, abrupt displacements, economic shocks, and abrupt policy shifts. These triggers emerge from indirect drivers and may have dramatic effects on direct drivers. The breakdown of the Soviet Union (Baumann *et al.*, 2011; Prishchepov *et al.*, 2013) and the nuclear accident of Chernobyl (Hostert *et al.*, 2011) could not be foreseen, but led to widespread farmland abandonment and decreasing land-use intensity. From a policy perspective, it is important to understand both drivers and triggers, to "accept uncertainty, be prepared for change and surprise, and enhance the adaptive capacity to deal with disturbance" (Folke *et al.*, 2005).

Here, we use categories of indirect drivers similar to previous assessments and studies (**Table 4.2**). However, the sub-categories of indirect drivers have been updated as outlined in **Table 4.3**.

Table 4.3: Categories of indirect drivers that underpin direct drivers of change in biodiversity and nature's contributions to people. More detailed information about, and motivation for selecting the sub-category under each main category is given in the text.

Institutional ▪ Regulations ▪ Institutional capacity ▪ Environmental policy integration ▪ Political/armed conflicts	**Economic** ▪ Material intensity of GDP ▪ Globalization ▪ Taxes and subsidies ▪ Environmental fiscal reform
Demographic ▪ Population growth & density ▪ Urbanization ▪ Migration	**Cultural & religious** ▪ Public awareness, knowledge ▪ Values, beliefs, social norms ▪ Lifestyle, consumption ▪ Social capital ▪ Cultural capital
Scientific & Technological ▪ New technologies ▪ Innovation	

Institutional drivers: Legislation and regulations provide the institutional arrangements (formal institutions, or legal framework) for all natural resource management. We refer to these as "regulations", to distinguish institutional drivers from informal institutions, which are mainly social norms and therefore belong to cultural & religious drivers. Some regulations promote sustainable natural resource management and governance to a greater or lesser extent. However, regulations safeguarding biodiversity and nature's contributions to people may not be enforced or, if they are, may not be effective. This depends on the institutional capacity, or the governability of the state, for example to regulate the private/public sectors and to engage civil society (Breukers & Wolsink, 2007; MEA, 2005b). Important institutional drivers are those sector regulations that impact biodiversity and nature's contributions to people, for example, energy, mining, conventional agriculture and forestry, large-scale fisheries, and tourism. Improving or changing these sectoral policies to better account for biodiversity and nature's contributions to people is sometimes called "mainstreaming" or "environmental policy integration" (Nilsson & Persson, 2003), including consistent multilevel governance (Malayang III *et al.*, 2006; Pahl-Wostl, 2009).

The international discussion has lately emphasized the role of policy integration. For example, Strategic Goal A of the Strategic Plan for Biodiversity 2011-2020 addresses "the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society" (CBD, 2010). Hence, assessing changes in institutional drivers can be framed regarding the extent to which countries and regions have succeeded in environmental policy integration (mainstreaming). Institutional drivers are intertwined with other indirect drivers. For example, since markets are influenced by legislation (Bromley, 1991), global trade as an economic driver is largely the result of trade agreements, which are not always consistent with international environmental regulations. Finally, the literature also refers to the role of international collaboration as well as political or armed conflicts.

Economic drivers: Global GDP is expected to increase from about 50 trillion US$ in 2005 to between 155 trillion (UNEP, 2012) and 300 trillion in 2050 (OECD, 2001). These figures diverge considerably and provide no information about how sustainable the growth of GDP is expected to be. Hence, we see material intensity of GDP, not GDP in itself, as a driver. Global trade increases demand for many nature's contributions to people and changes production and consumption patterns, and therefore ecosystem use, at local, national, and global levels. Economic drivers are strongly linked to institutional drivers, which govern production through regulations, taxes and subsidies, thereby influencing relative prices of, for example, fossil fuel versus renewable energy. Internalising external environmental costs may, however, be difficult due to its effects on global competitiveness. Hence, the environmental fiscal reforms called for by the United Nations Environment Programme to make the economy more efficient, must be seen in a global context.

Demographic drivers**:** This group includes population density and growth, urbanization and migration as well as ageing population (Hossman *et al.*, 2008; Kroll & Kabisch, 2012). Human population growth is one of the most fundamental reasons behind all direct drivers.

Cultural and religious drivers: Public awareness and knowledge about environmental change are fundamental indirect drivers. Filtered by values, beliefs and social norms, public awareness exerts pressure on decision-making about the environment (Nelson *et al.*, 2006). Culture conditions the individual's perceptions of the world, influences what he or she considers important, and suggests courses of action that are appropriate and inappropriate. Although culture is most often thought of as a characteristic of national or ethnic groups, our definition emphasizes the emergence of cultures within professions, organizations and gender, along with the possibility that an individual may be able to draw on or reconcile more than one culture (Nelson *et al.*, 2006). Cultural values are materially manifested in lifestyles and consumption patterns. To enable transitions to sustainability, cultural drivers such as social capital may be mobilized by trust-building (Pretty, 2003).

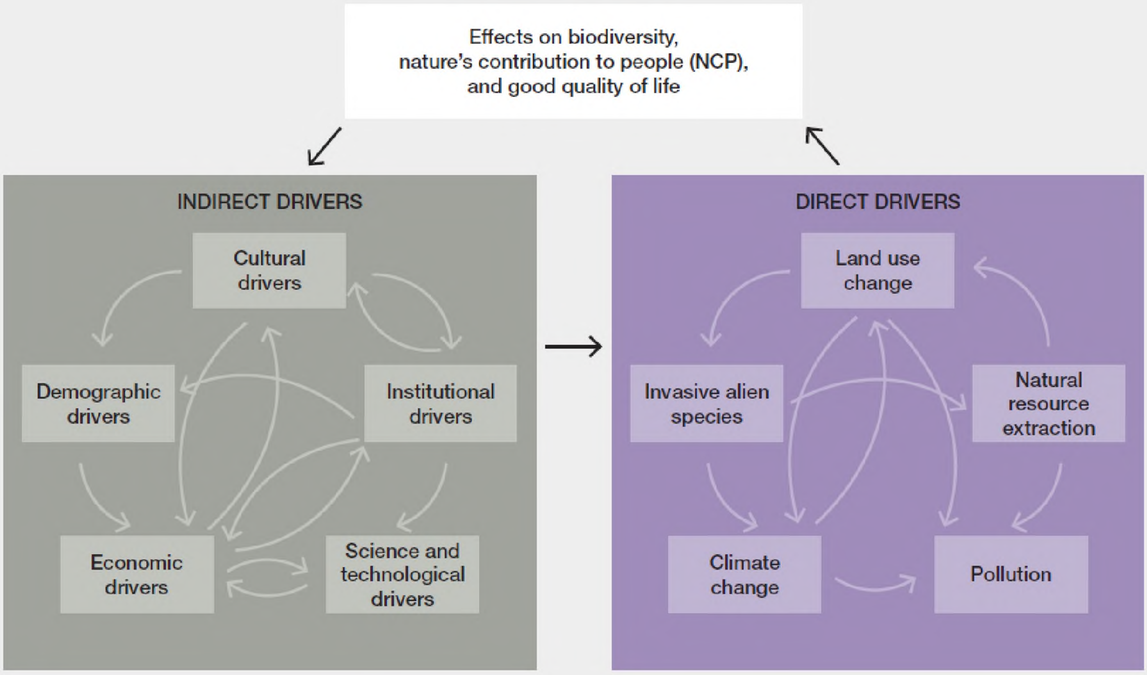
Scientific and technological drivers: Technology is a major driver of economic growth, accounting for more than one third of the GDP growth in the US 1929-1980 (MEA, 2005b) and similar effects might be expected in Europe and Central Asia. Technology also directly influences direct drivers in very tangible ways, for example in forestry, agriculture and fisheries, resulting in intensification of land uses (MEA, 2005b). Technology can be seen as just a "tool", neither good nor bad. Its effects depend on how it is used and developed. For instance, new information and communication technologies might have the potential for both agricultural intensification and disintensification (Grimes, 2000). At the same time, the direction of technological development is a function of price relations, which in turn are influenced by institutions. For example, the "green" revolution has promoted fossil fuel derived inputs to replace natural inputs in agriculture (Perelman, 1972). With different institutions, technological innovations and development can increase resource efficiency and decoupling, being an integrated part of the transformation to a green economy and an important part of the development of the circular economy (European Commission, 2017b; UNEP, 2011). However, technological development resulting in resource efficiency may lower the price of the natural resource, which in turn may increase the consumption of this resource; this is called Jevons paradox or the rebound effect. Taxes on natural resources (e.g. an environmental fiscal reform) are needed to prevent the rebound effect (Polimeni *et al.*, 2012).

4.2.3 Relationship between indirect and direct drivers

The previous section suggests that indirect drivers are intertwined and in combination influence direct drivers. The interaction among indirect drivers is highly complex, i.e. they are hard to trace back to a single point of origin, and their impacts are often reciprocal and not unidirectional. Jointly, indirect drivers impact on direct drivers, which in turn also interact in the way they drive ecosystem change (**Figure 4.1**). For example, climate change affects the survival of invasive alien species, and land-use change can have feedback effects on climate. Knowledge about the effects of direct drivers on biodiversity and nature's contributions to people increases public awareness and feedback to the underlying indirect drivers.

Figure 4 1 **Illustration of multiple interactions among indirect and direct drivers within a specific context that have impact on biodiversity and nature's contributions to people.**

The graph illustrates important links; more are possible. Knowledge and awareness of changes in biodiversity and nature's contributions to people influence indirect drivers and make adaptations possible. Source: Own representation.



4.2.4 Spatial and temporal variability

Even though the major direct drivers are known, their specific effects and overall trends over time are not always easy to identify, quantify and assess. This is primarily due to their high spatial and temporal variability. Some drivers are local in nature (e.g., land-use change and point-source pollution of heavy metals or nutrients), while others are regional (e.g., ozone or atmospheric nitrogen pollution from combustion engines) or global (e.g., atmospheric CO_2 or sea-level rise). Some of these drivers affect all species and ecosystems more-or-less equally (e.g., radioactive pollution), while other drivers affect species and ecosystems very selectively (e.g., nitrogen deposition), and therefore often exert complex effects on biodiversity and nature's contributions to people.

While the effect of some drivers is immediate (e.g. mining), others exhibit significant time lags in their effect on biodiversity and nature's contributions to people. While climate and land-use change and invasions by alien species are steadily increasing, their full effect is often visible only much later, since the biodiversity and ecosystem response is slow. This has given rise to the terms invasion debt (Essl *et al.*, 2011) or extinction debt (Dullinger *et al.*, 2012; Tilman*et al.*, 1994), to express the expected time lags until the full effects of drivers are realized. The many facets of climate change rarely affect species and ecosystems without delay, and the climate itself also lags behind the increase in greenhouse gas concentrations (IPCC, 2014a).

While some effects are steadily shifting (e.g., sea-level rise), others are unstable and show high temporal variability. This is especially the case with climate, which includes changes in mean conditions, time course and extremes (such as heat-waves, drought, fire, floods or winds). The biological response can be linked to the changes in means, time courses and in extremes, and the responses can be gradual or they can be in the form of tipping points between alternative stable states (Barnosky *et al.*, 2012; Hoegh-Guldberg *et al.*, 2007), which can be irreversible.

4.2.5 Interregional flows

Inter-regional flows include trade in agricultural commodities, fish and wood, which can be measured as human appropriation of net primary productivity (HANPP) (Krausmann *et al.*, 2013). As a result of international and even interregional trade, and with the exception of northern parts of Western Europe, and Eastern Europe, Europe and Central Asia appropriates a larger amount of nature's contributions to people than it produces. Put differently, their ecological footprints exceed their bio-capacity (Global Footprint Network, 2017). Inter-regional trade of nature's contributions to people has consequences for local ecosystems in the exporting country, but also direct global effects. For example, in 2004, the deforestation embodied in final consumption within the EU-27 was 732,000 ha, which was about 10% of the world's annual deforestation (European Commission, 2013) (see also Section 2.2.4 in this Volume).

Besides these direct biophysical effects, interregional flows of nature's contributions to people also have profound effects on direct and indirect drivers of ecosystem change. First, the pressures on domestic and regional ecosystems can be reduced when nature's contributions to people are imported, i.e. when natural resource extraction, pollution and land-use change are "exported". Second, interregional flows may have repercussions for other sectors, sometimes referred to as telecoupling. For example, biofuel mandates in the European Union contributed to global food shortages in 2008 and subsequent civil unrest in other world regions (Liu *et al.*, 2015).

Inter-regional trade is justified in terms of economic efficiency. Differences in the market price of agricultural or forestry commodities, fish, and minerals can be seen as differences in scarcity, which are levelled out by trade, resulting in increased efficiency. However, if the external costs of production and trade are not taken into account, interregional trade may not enhance efficiency. Without interregional trade, a region consuming more than its biocapacity (or extraction of minerals) would experience increased physical scarcity which, in a market economy, would result in increasing prices. This price signal would, in turn, drive producers and consumers to search for substitutes. It would also raise public awareness of the scarcity, which could become a pressure for institutional change. Inter-regional trade offsets this price signal and thereby inhibits the feedbacks to economic and institutional drivers. This is the purpose of trade, not a side-effect, and it would not be a problem for nature or human quality of life if trade were based on sustainable harvest and extraction levels (Daly & Farley, 2014).

However, if harvest levels or the production methods of these goods are not sustainable, partly because external costs are not included in their price, then inefficient and unsustainable production of material contributions of nature are exacerbated by interregional trade. Policy failures such as inappropriate environmental regulations in producing countries, increase incentives to export these goods. Importing countries subsequently enjoy low prices and offsetting of scarcity. This "organized irresponsibility" (Beck, 2005) has not emerged by accident. On the contrary, export-oriented economic growth has been a common growth strategy for many developing countries, supported by the World Bank and other international organizations. For example, unsustainably produced agricultural commodities such as soy, coffee, and palm oil have flooded the world market, resulting in low and fluctuating prices and thereby increased vulnerability in the producing countries (Adger *et al.*, 2009). In this way, global drivers of market integration become drivers of both local vulnerability and global unsustainability. In a sustainable world, global trade would not be a problem. However, in the contemporary world, unsustainable production methods in the producing countries are reinforced and scaled-up by short-term profits from trade and the lack of environmental regulations in present global and bi-lateral trade institutions (Daly & Farley, 2014).

Natural resource extraction of minerals and fish are also important interregional flows. Western and Central Europe import most of their mineral resources due to the depletion of their own resources, and high extraction costs partly due to environmental regulations (European Commission, 2014). Without cheap imported minerals, there would be pressure to increase recycling and substitution. However, interregional trade softens and delays these economic and institutional feedbacks. Similarly, the depletion of fish stocks in Europe and Central Asia has partly been met by supply of imported fish, preventing increases in the cultural drivers of prices and awareness, respectively. Both reduce public pressure for institutional responses (see Section 4.4.1.3).

4.2.6 Methodological approach

4.2.6.1 Effects of, and trends in, direct drivers

Each of the main five direct drivers (see Section 4.2.1) was assessed focussing on a set of sub-categories of these main driver categories. The literature was screened for effects of direct drivers on biodiversity and nature's contributions to people, and for trends of the recent past and of the projected future within Europe and Central Asia. Most weight was given to literature published after 2005, since earlier literature was largely covered by the Millennium Ecosystem Assessment. For some drivers (e.g. some aspects of natural resource extraction, land-use change, or biological invasions), there is less available information than for others, or it is only available for recent periods. To assess the trends in climate change drivers, more publications are available than for the other direct drivers, and also large databases of spatial data. While we did not perform primary analyses for this assessment, we assessed climate drivers through observational data and data from the CMIP5 (Coupled Model Intercomparison Project Phase 5) climate change simulations used in the IPCC AR5 WGI reports (IPCC, 2012, 2013a, 2013b) and extracted from the KNMI climate change atlas website (IPCC, 2012; van Oldenborgh, 2016). For historical climate data we used five data sets, namely: 1) GISTEMP (GISTEMP Team, 2015; Hansen *et al.*, 2010); 2) HadCRUT version 4.2.0.0 (Morice *et al.*, 2012); and 3) NCDC MOST (Jones & Moberg, 2003; Peterson & Vose, 1997); 4) CRU TS 3.24 (Harris *et al.*, 2014); and 5) GPCC V7 (Schneider *et al.*, 2011). For future climates, we used data used in IPCCs AR5 (IPCC, 2013a, 2013b), using all four representative concentration pathway (RCP) scenarios, indicating levels of radiative forcing by greenhouse gases in the atmosphere), namely: RCP2.6, RCP4.5, RPC6.0, and RCP8.5. Higher numbers indicate a higher greenhouse gas effect and a higher level of change to the atmosphere and climate.

Status and trends of temperature and precipitation were extracted for the whole region and for its four subregions. Values for time series were averaged over land grid. Average historical trend estimates and projected future anomalies were calculated for each unit of analysis within each subregion. Spatial distributions of the units of analysis were derived from multiple datasets (see Chapters 1 and 3) (**Figure 4.2**). Average climate values were computed by overlaying units and subregions with climate data, and calculating mean values for summer (JJA) and winter (DJF). Time series were generated for 1950-2060 as anomalies relative to 1986-2005. As an indication of model uncertainty and natural variability, the time series of each individual model and scenario was included over the analyzed period (see **Figure 4.3** and IPCC (2013b) for more info). Future anomalies were estimated as 20-year means for the time period 2041-2060. Maps of projected trends were generated for Europe and Central Asia similarly to the IPCC AR5 WG1 Annex 1 (IPCC, 2013a), using the KNMI climate change atlas. Two representative concentration pathway scenarios were used to generate maps: scenario RCP 4.5 and 8.5 (IPCC, 2013b). Spatial averages over complex regions provide general trends, but may not explain the details for particular locations.

Figure 4 2 **Spatial distribution of units of analysis for Europe and Central Asia A and additionally important land cover elements B for Chapter 4 (see Chapter 1). Source: Own representation.**

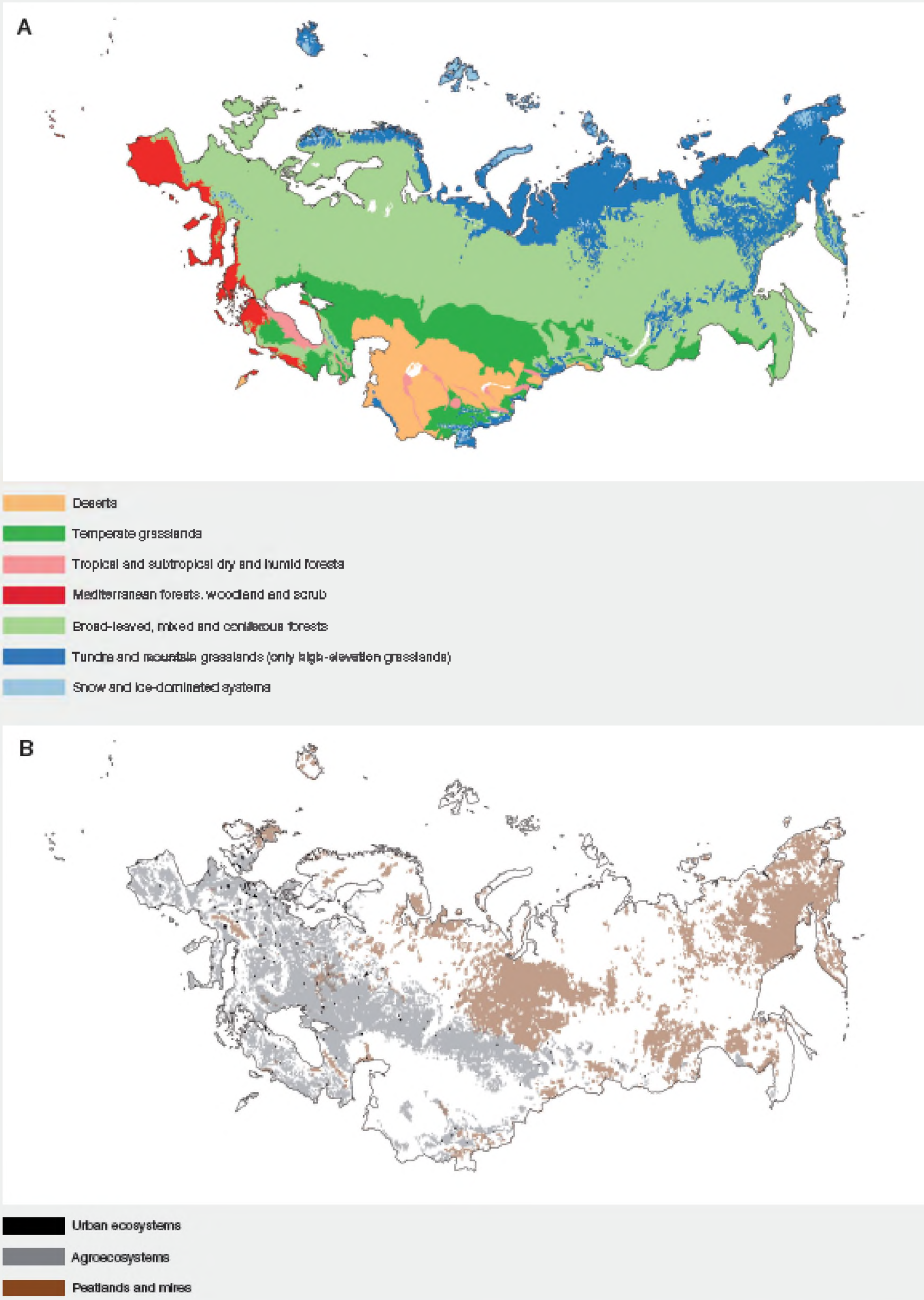
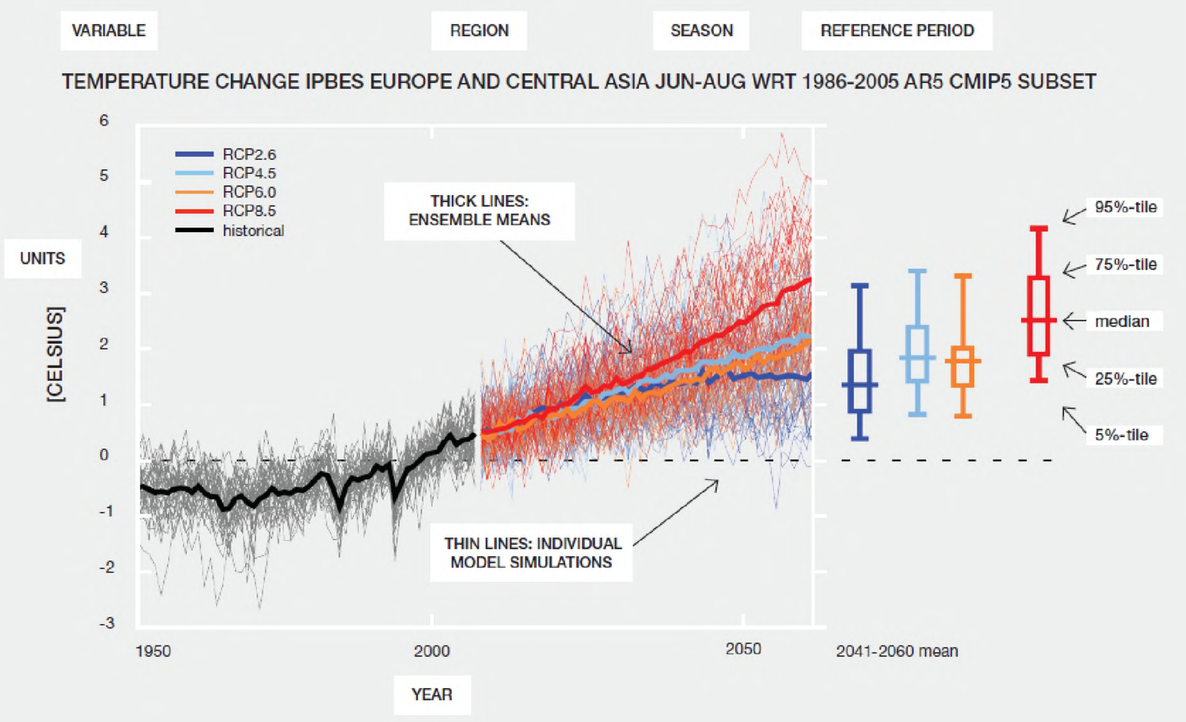


Figure 4 3 Explanation of a typical time series figure used to present projected temperature and precipitation changes for Europe and Central Asia. Source: Modified from IPCC (2013b).



4.2.6.2 Indirect drivers

Various methods were employed to assess indirect drivers. We used a combination of key words in English and several native languages in the Europe and Central Asia region (such as French, Italian, Swedish, Albanian, Russian, Ukrainian, Hungarian) to retrieve peer-reviewed articles in Scopus, e-library and Google Scholar. We also made use of the grey literature published in native languages of countries from different subregions of Europe and Central Asia. Indigenous and local knowledge and practices were assessed through analysis of traditional land uses of indigenous peoples and local communities and their drivers of change.

We applied qualitative systems modelling methods (e.g., Wolstenholme & Coyle, 1983) using causal loop diagrams to structurally map the dynamic inter-relationships within and between indirect and direct drivers of change in biodiversity and nature's contributions to people. Causal loop diagrams provide a concise format for describing complex interconnected system structures and behavioural directionality. They use arrows to indicate direct causal relationships between independent and dependent variables. These relationships can be either in the same direction, represented by a positive (+) sign, or in the opposing direction, represented by a negative (–) sign. Thus, if independent variable A connects to dependent variable B by an arrow with a plus (+) sign, the underlying logic of the causal loop diagram is that an increase (decrease) in A's behaviour will lead to an increase (decrease) in B's behaviour. If the arrow connecting A to B is accompanied by a negative (–) sign then the diagram indicates that an increase (decrease) in A will lead to a decrease (increase) in B. In some cases, variable concepts have been amalgamated or broadly aggregated, or otherwise relationships between independent and dependent variables have been strongly simplified, in such a manner as to impair the clear directionality of a relationship. In these cases, arrows are not represented by a sign. For example, several arrows in overview causal loop diagrams do not carry directional signs, as these arrows are aggregates of multiple, variously signed, relationships.

The causal loop diagrams (CLDs) provided in this chapter are intended to convey the major dynamic relationships identified via the literature review process. Each variable and link is thus based on explicit evidence from one, or several, references. Although expert opinion was gathered during a series of workshops to guide an iterative modelling process, no dynamics have been included in the finalized CLDs without substantiation in published materials. No representation of a fully interconnected model of all identified dynamics is provided. Such a model would be too complex, and would defeat the purpose of using CLDs as communicative devices. Rather, we present a set of nested models throughout the chapter, each providing a level of detail regarding identified trends and major driver dynamics. As such, the CLDs unpack the indirect and direct driver boxes of the IPBES conceptual framework into an overview model of indirect and direct driver categories (see **Figure 4.1**). This overview model is then further unpacked at a variety of levels of detail to examine the major dynamics influencing the indirect and direct driver interactions. Indirect and direct driver categories are colour-coded in each of the CLDs according to the legend. Boxes around variables are used either to signify stocks or to contain a variety of identified sub-variables within an overarching variable. Variables in bold text are used to help guide readers in linking the CLDs with the central themes discussed in respective texts. Grey diamond-shaped boxes around variables are used similarly to aid readers in locating the major trends in land-use change within the diagrams.

4.3 General trends in indirect drivers in Europe and Central Asia

As described in Section 4.1.2, a more specific assessment of indirect drivers in relation to each direct driver is conducted in Sections 4.4-4.8. General trends are assessed in this section.

4.3.1 Institutional drivers

Regulations, including legislation and detailed institutional arrangements, shape all direct drivers and also to some extent all the other indirect drivers. Regulations are the result of purposeful collective political action and reflect the power balance between conflicting interests. Therefore, political and economic conflicts (cultural and economic drivers), influence institutional drivers. Knowledge about the effects of direct drivers on biodiversity and nature's contributions to people increases public awareness and the prices of material contributions from nature and thereby acts as a feedback to institutional drivers (**Figure 4.1**).

In general, the institutional capacity to make and enforce regulations is strong in Western and Central Europe (see Chapter 6). For example, the European Union's Common Fisheries Policy (CFP) illustrates institutional capacity in that regulations have been passed to restore and maintain fish stocks above biomass levels capable of producing maximum sustainable yield, although half of the fish stocks exploited by the fishing fleet of European Union countries are still overexploited (Guillen *et al.*, 2016), see Section 4.4.1.2.

On the other hand, for the European Union's Common Agricultural Policy (CAP) the new environmental prescriptions – including maintaining existing permanent grasslands, crop diversity, and establishing ecological focus areas – maybe "so diluted that they are unlikely to benefit biodiversity"(Pe'er *et al.*, 2014) (Section 4.5.2.2). The most significant recent change in environmental institutional drivers in Europe and Central Asia is arguably the transformation of the energy sector in the European Union.

Here, political leadership, new policies and economic incentives have catalysed technological advancements resulting in lower prices for solar and wind power. These lower prices have subsequently become economic drivers for decreased pollution and greenhouse gas emissions (Bürer & Wüstenhagen, 2009).

For example, the German Renewable Energy Act (EEG) from 2000 has become a major driver for transforming the energy sector, increasing generation of renewable energy from 29 TWh in 1999 to 161 TWh in 2014 (Lauber & Jacobsson, 2016). However, substantial trade-offs may result from a lack of mainstreaming. In a scenario for energy crops, (Gutzler *et al.*, 2015) project substantial reduction in biodiversity and landscape scenery, and increased soil erosion and need for water protection. These three examples from the fishing, agriculture and energy sectors suggest that strong institutional capacity is not sufficient to safeguard biodiversity and nature's contributions to people.

4.3.2 Economic drivers

We have identified the material intensity of GDP, rather than GDP itself, as a main economic driver. The relationship between GDP and resource use has long been debated. The contributions of many scholars – for instance Carson (1962), Boulding (1966), Georgescu-Roegen (1993) – highlighted that serious problems arise from both the quality of the waste (ecotoxicity) and the scale of human activities. The amount and the rate at which matter passes through society (the material throughput) and becomes waste is a major indirect driver of biodiversity loss. This is also called the industrial and socioeconomic metabolism (González de Molina & Toledo, 2014). Despite some evidence that prosperity or human well-being does not increase further once an average income threshold has been reached (Kubiszewski *et al.*, 2013 suggest a threshold as low as 7,000 USD/year and person), Governments in countries with much higher per capita GDP strive hard to increase it further. A growing body of literature suggests that the challenge is to decouple quality of life (well-being or prosperity) from environmental degradation and pay less attention to GDP (Jackson, 2009; Raworth, 2017; Røpke, 2016; van den Bergh, 2010, 2011; Victor, 2008).

Fundamentally, economic growth is largely explained by investments in real capital and there is a near-linear relationship between GDP growth and physical capital accumulation in most countries (Malmaeus, 2016). There are, in turn, clear correlations between investments in physical capital, and resource use including metals (Chen & Graedel, 2015; Kondo *et al.*, 2012), gravel and sand (UNEP, 2014), and biomass.

Growth-oriented policies aim to enhance production and consumption and, except for the economic contraction following the financial crisis in 2008, domestic material consumption (DMC) has not recently decreased in general in most countries in Western and Central Europe (4.4.4.2) (Eurostat, 2017b). Target 8.4 of Sustainable Development Goal 8 ("decent work and economic growth") requires governments to "endeavour to decouple economic growth from environmental degradation". However, GDP growth will have a negative effect on ecosystems unless countries succeed in absolute decoupling, sufficiently large to achieve the environmentally-oriented Sustainable Development Goals. Relative decoupling, where resource use increases, but at a slower pace compared to GDP, is no longer an option except for low-income countries (Raworth, 2017).

Decoupling of GDP growth from resource use and environmental impacts is a requirement for sustainable growth (Bithas & Kalimeris, 2013; OECD, 2011; van den Bergh, 2010). This can be achieved theoretically, but has proved difficult to accomplish empirically. For example, global modelling suggests that absolute decoupling, in terms of 50% reductions of CO_2 emissions and resource use, would require very strong abatement and resource efficiency policies. Because of economic adaptations and technological development this would have negligible effects on economic growth and employment until 2050 (Schandl *et al.*, 2016). The lack of absolute decoupling has been observed empirically (UNEP, 2011). It is often explained by the so-called rebound effect, stating that less demand for natural resources arising from increased productivity results in lower prices and therefore higher demand for natural resources (Sorrell, 2007).

The most well documented case of economic growth as a driver of environmental impact is between GDP growth and CO_2 emissions (e.g. Raftery *et al.*, 2017). Lægreid (2017) found a very robust connection between economic growth and larger greenhouse gas emissions, hence no absolute decoupling. However, in a study of 131 countries, Szigeti *et al.* (2017) found absolute decoupling between GDP and ecological footprint for 40 countries and relative decoupling for 77 countries. Although the evidence is inconclusive, there are signs of relative decoupling occurring in Europe and Central Asia, and sometimes also absolute decoupling, but this is rarely sufficient to achieve climate goals.

The European Union has recently adopted several policies to promote resource efficiency (EEA, 2014e) and sustainable growth (European Commission, 2017b). Economic drivers have been altered, for example by new legislation and the emission trading system for carbon. However, the challenges of decoupling and the rebound effect require more profound changes in economic drivers, especially taxes (Font Vivanco *et al.*, 2016; Polimeni *et al.*, 2012). The tax system is of fundamental importance as an institutional and economic driver since it modifies all market prices and therefore changes incentives for producers and consumers. Despite proposals for environmental fiscal reform (EFR) by UNEP's "Green Economy" (2011) and the United Nations' Strategic Plan for Biodiversity 2011-2020, little progress is evident in Europe and Central Asia. For example, the total revenue from environmental taxes in the EU-28 in 2014 was 2.5 % of GDP, or 6.3 % of the total revenues derived from all taxes and social contributions. These proportions have *decreased* since 2002, from 2.6 % and 6.8 % respectively (Eurostat, 2017b).

Decoupling is an important issue only if growth in GDP is assumed. This is the case for the European Union and its growth strategy Europe 2020, in which economic growth and job creation are top priority goals expressed as smart, sustainable and inclusive GDP growth (European Commission, 2017b). However, if a more "agnostic" approach to GDP growth is taken, resource efficiency and meeting the Sustainable Development Goals can be targeted directly without too much attention to whether GDP increases or decreases a few per cent (Raworth, 2017; van den Bergh, 2011). The literature on "degrowth" aims at decoupling human well-being and quality of life from GDP growth: the prefix smart, sustainable and inclusive are kept, but "growth" is replaced by "development" (Martinez-Alier, 2016).

While sustainability transformations would result in growth in sustainable technologies, it would also shrink non-sustainable technologies (van den Bergh, 2010, 2011). Targets for GDP growth (or de-growth) obfuscate the idea of transformation. For example, sustainable consumption in high-income countries is more about reducing the unsustainable aspects of consumption than increasing the more sustainable aspects of consumption. Focusing on such transformations or transitions represents different policy goals compared to pleas for green or sustainable growth (Geels *et al.*, 2015; Lorek & Spangenberg, 2014; Spangenberg, 2014).

Global trade exposes ecosystems as being part of global supply and demand. This impacts interregional flows (see 4.2.5) and prevents price signals from responding to local scarcity of natural resources (see 4.4.1.3). A third aspect of global trade is the institutional competition it entails. National Governments are reluctant to internalize external costs from natural resource extraction and pollution because that may impede the international competitiveness of taxed corporations (Ayres *et al.*, 2013). Globalized financial markets, including commodity derivative markets and algorithmic trade, have increasing impacts on the world's ecosystems (Galaz *et al.*, 2015).

4.3.3 Demographic drivers

Europe and Central Asia is home to approximately 910 million people or 14.5% of the total world population (United Nations, 2015), almost half of whom live in Western Europe (**Table 4.4**). Although the population in the region is projected to be stable until 2050, there are important differences within subregions. For example, the population growth rate of Turkey is 1.69% and, without Turkey, the rate of population decline in Central Europe is much greater (-0.25%) than illustrated in **Table 4.4** (-0.14%). The population decline projected for 2015-2050 in Central and Eastern Europe due to low birth rates, coupled with emigration and moderate mortality due to low life expectancy, is unprecedented in recent history (Lutz, 2010). Because human populations are increasing in Central Asia and Turkey and decreasing in Central and non-Caucasus Eastern Europe, it is likely that a high potential for migration from Turkey and Central Asia to Central and Eastern Europe will develop until 2050 (Lutz, 2010). Armed conflicts have profound effects on migration, for example, Turkey recently received (by March 2016) over 3 million refugees from Syria, Iraq and Afghanistan (UNHCR, 2017).

Table 4.4: Population trends in Europe and Central Asia. Source: United Nations (2015). ECA: Europe and Central Asia, WE: Western Europe, CE: Central Europe, EE: Eastern Europe, CA: Central Asia.

	ECA	WE	CE	(Turkey)	EE	CA
Population 2015 (million)	910	423	202	(79)	218	67
Fertility rate (children/woman)	-	1.71	1.54	-	1.67	2.83
Net migration (per 1,000 inhab.)	-	2.98	-1.35	-	-1.55	-1.47
Population growth/year (%)	-	0.39	-0.14	(1.69)	0.02	1.65
Population 2050 (million)	913	441	192	(88)	192	88

The age distribution of the population is also changing. With improvements in health care, life expectancy is increasing in each subregion and their populations are aging, meaning a higher proportion of older age groups (Lutz *et al.*, 2008). This has several consequences for biodiversity and nature's contributions to people. First, total consumption may further increase as the consumption of energy, food, medicine and others by elderly people increases even if population size decreases. Ecological footprints may therefore increase even in subregions currently showing human population declines (Hossman *et al.*, 2008). Second, aging in rural areas will lead to a decrease in the number, capacity and effectiveness of the rural workforce, which will ultimately create the socio-economic conditions for intensified use of natural resources (mainly by agriculture, forestry, or fishery) by large corporations rather than by private farmers (Gentile, 2005). Third, age profile also strongly influences where people choose to live, which affects urban growth patterns and subsequent impacts on biodiversity and nature's contributions to people (Fontaine *et al.*, 2014).

Fast population growth in Central Asia, with further expected increase in urbanization, will present risks to the already overpopulated lowland and riparian areas of the subregion and will influence biodiversity and ecosystem services (Osepashvili, 2006). Human population growth will take a heavy toll on water use. This is likely to result in a decline in water-related services, which may trigger water conflicts (e.g. in Fergana valley in Uzbekistan, Tadzjikistan and Kirgizistan) or water-use regulations. In other areas, the collapse of irrigation-based agriculture due to water shortages may cause desertification, such as the complete drying up of the Aral Sea (Gentile, 2005).

4.3.4 Cultural and religious drivers

In democratic societies, public awareness and knowledge of environmental change are the underlying drivers for both institutional change and consumer demand (Nolan & Schultz, 2015). Hence, the feedback to indirect drivers often starts with the cultural driver that we call "public awareness". The cultural belief that further GDP growth will facilitate sustainable development is deeply rooted in Europe and Central Asia, calling for smart, inclusive and sustainable growth (European Commission, 2010). In recent years, many studies have shown that, if biodiversity and nature's contributions to people are to be used sustainably, growing anthropogenic pressures paralleled by environmental degradation would require a radical change in our political value system, with a reorientation of fundamental policy goals from GDP growth towards well-being, environmental quality, employment and equity (Hardt & O'Neill, 2017; Jackson, 2009; Kallis *et al.*, 2012; Martínez-Alier *et al.*, 2010; Røpke, 2016; Victor, 2008).

All regional cultures are increasingly becoming part of a global cultural process. With increasing access to media, information and exchange among regions, the cultural changes taking place in Europe and Central Asia form part of the general globalization trend. Although distinct local cultures, with their beliefs and specific relation to nature may well persist, they will do so in parallel to global cultural trends (Harari, 2014). Cultural and religious beliefs are often exploited politically, which has been evident in the region in recent years. However, it is not clear how these changing beliefs and opinions affect biodiversity and nature's contributions to people.

Central to the effect on biodiversity is how cultural identity and religious beliefs influence lifestyles in terms of consumption patterns. Values promoting a vegetarian diet are, for example, likely to reduce the land-use area needed to produce food, and thus the impacts on nature Alexander *et al.* (2016).

Heterogeneous agricultural landscapes provide biodiversity and are therefore supported by agri-environmental schemes in the European Union. An increasing focus on recreation and eco-tourism in Western Europe has become a further justification for, and therefore driver of, political and economic support to heterogeneous landscapes (Hahn *et al.*, 2017; Beilin *et al.*, 2014; Navarro & Pereira, 2012). Beyond eco-tourism, the increasing popularity of spiritual refreshment and other spiritual experiences have considerable potential for the recognition of nature's contribution to people.

4.3.5 Scientific and technological drivers

If population is constant and affluence, measured in GDP per capita, is increasing, the equation I = P*A*T (Impact = population * affluence * technology) suggests very high expectations of technology to ensure sustainable growth. However, technological innovation is not a driver, which in itself ensures lower negative environmental impact. Scientific and technological innovation is a double-edged sword (Westley *et al.*, 2011), which could have positive or negative effects on biodiversity. As mentioned in Section 4.2.2, innovation is not a neutral process driven mainly by the curiosity of researchers and innovators. The general pattern of world market prices of natural resources is a sharp decline during the past fifty to one hundred years. At the same time the price for labour has increased dramatically, augmented by the tax system (Eurostat, 2017b). Technological innovation has therefore not targeted resource efficiency, but instead labour productivity (Lorek & Spangenberg, 2014).

Energy and resource efficiency have become political targets. The literature suggests a very high potential for, for example, energy supply and storage, green information technology transportation, foodstuffs, agricultural engineering, design strategies, lightweight construction, as well as the concept “using instead of owning” (Rohn *et al.*, 2014). Realising this potential requires support from institutional and economic drivers (Ayres *et al.*, 2013), and ultimately cultural-religious drivers. For example, if cultural beliefs support “modern” high-input agriculture and if new European Union member States in the Baltic Sea drainage area adopt the same use of fertilizers as Denmark, Sweden and Finland, the eutrophication of the Baltic Sea will accelerate (Larsson & Granstedt, 2010). If the Baltic countries want to achieve the Baltic Sea Action Plan, then climate smart and “Baltic Sea smart” technologies and farm systems are needed.

4.4 Drivers of natural resource extraction and its effects on biodiversity and nature’s contributions to people

This category of driver is often called “overexploitation,” focusing on overfishing (MEA, 2005b). As mentioned before we have chosen a more neutral term, extraction. This section assesses two biotic forms of natural resource extraction: fishing and hunting; and two abiotic forms: mineral and fossil fuel extraction, and water use and desalination. Agriculture, forestry (logging) and traditional land use (gathering wild plants, berries and mushrooms) are assessed under land-use change.

Natural resource extraction is, according to a synthesis based on the IUCN Red List data, “by far the biggest driver of biodiversity decline” (Maxwell *et al.*, 2016). However, that conclusion only holds if unsustainable logging is included. Still, hunting, fishing and mining together pose a considerable threat to biodiversity (Maxwell *et al.*, 2016).

4.4.1 Fishing

4.4.1.1 Effects of fishing on biodiversity and nature’s contributions to people

Fishing affects 1,118 of the 8,688 assessed red list species (Maxwell *et al.*, 2016). Both marine and inland fish stocks in Europe and Central Asia have declined over recent decades. Trawling is a fishing technology with adverse effects on biodiversity, through habitat destruction. Over recent decades, trawlers have become dominant among fishing boats, especially vessels greater than 100 gross registered tons (Anticamara *et al.*, 2011). Despite regulations, half of the fish stocks exploited by the fishing fleet of the European Union are still overexploited (Guillen *et al.*, 2016). Overfishing affects genetic diversity and the age structure of the targeted fish population. Furthermore, removal of top predators through overfishing may disrupt ecological relationships, food webs structure and energy flow pathways (García-Charton *et al.*, 2008; Pérez-Ruzafa *et al.*, 2006; Pérez-Ruzafa *et al.*, 2008).

Inland waters have received less attention than global fisheries. One of the symptoms of intense over-fishing in inland waters is the collapse of particular stocks. Such collapses constitute a biodiversity crisis rather than a fisheries crisis. However, intensive fishing frequently acts synergistically with other pressures, and its consequences for inland fisheries and ecosystems are poorly understood and documented (Allan *et al.*, 2005).

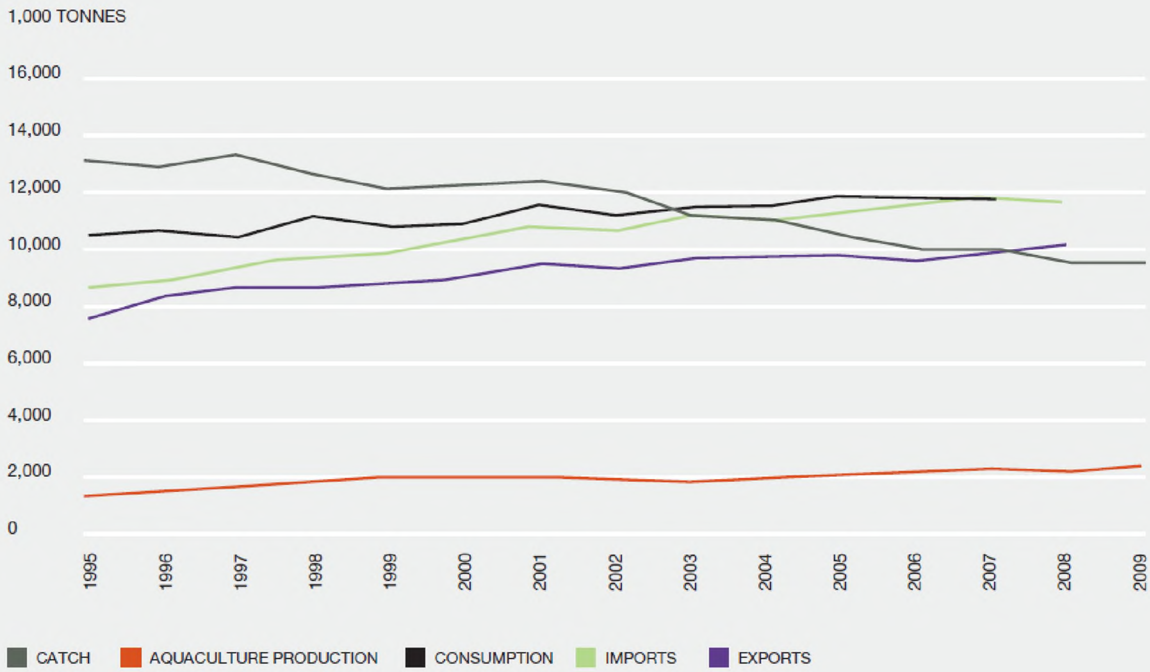
Nevertheless, more stocks are recovering, and the combined effects of climate warming and reduced fishing mortalities have resulted in record large stocks of e.g. mackerel in the Norwegian Sea, plaice in the North Sea and cod in the Barents Sea. The recovery of these major stocks now impacts other parts of the ecosystems through both predation and competition. For instance, a recent collapse in the capelin stock in the Barents Sea was likely partially due to cod predation and competition with cod likely impacts the condition of marine mammals (Bogstad *et al.*, 2015).

4.4.1.2 Trends in fishing

The marine area under the jurisdiction of European Union member States is substantial — larger than the total land area of the European Union— and supports industries such as shipping, fishing, offshore wind energy, tourism, and oil, gas and mineral extraction (EEA, 2012d). Fishing effort has increased over recent decades in Western Europe and Central Europe. However, some analyses suggest a stagnating or even decreasing trend in fishing effort in European marine waters (Gascuel *et al.*, 2016). Fishing effort is a combination of fleet capacity (number of vessels or engine power) and the amount of time spent at sea. Reduced fishing effort may however be counteracted by an increase in the efficiency in detecting and catching fish.

Despite recent attempts by the fishing sector to ensure sustainable practices and recovery of fish stocks, the industry is still characterized by overfishing and declining volumes of fish catch (EEA, 2012d). During recent decades, aquaculture production has been increasing. However, some aquaculture species, like salmon and tuna, are carnivores that feed on other fish that are, themselves, overfished (Naylor *et al.*, 2000; Pauly *et al.*, 2002). Demand for fish in the European Union continues to exceed the sustainable yield and a significant proportion of the fish consumed in the European Union is imported from, for example, Norway, China, Morocco, and the USA (**Figure 4.4**).

Figure 4 4 Total fish catches, aquaculture production, consumption, imports and exports for European Environmental Agency (EEA)-33 countries (except Liechtenstein) and the western Balkans*, 1995 to 2009. Consumption refers only to human consumption. Source: EEA (2012d).

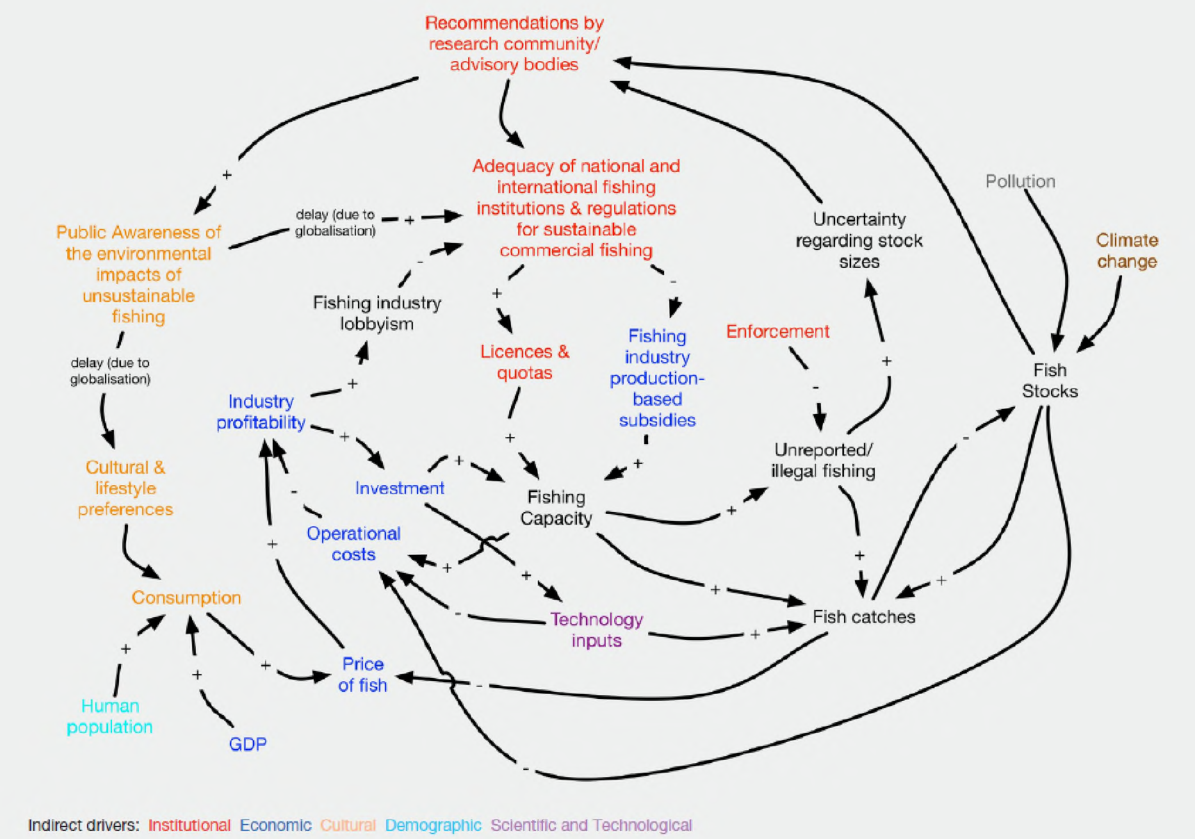


* The EEA currently has 33 member countries and six cooperating Balkan countries. The 33 member countries include the 28 European Union member States together with Iceland, Liechtenstein, Norway, Switzerland and Turkey. The six West Balkan countries are cooperating countries: Albania, Bosnia and Herzegovina, the former Yugoslav Republic of Macedonia, Montenegro, Serbia as well as Kosovo.

4.4.1.3 Drivers of fishing

The drivers of fishing are summarized in **Figure 4.5**. Overcapacity accompanied by non-compliance (illegal, unregulated and unreported fishing) are the most common immediate causes for overfishing (Boonstra & Österblom, 2014). Knudsen *et al.* (2010) identify eight main drivers of overfishing, most of them economic in nature. Fishing costs (including operational costs and fuel prices) and incomes (including fish prices and demand) are important drivers. Fishing costs increase when stocks become over-exploited, but the subsequent increase in fish price motivates investments and continued fishing. Furthermore, tax exemptions and government subsidies, especially for fuel, are very important drivers to offset the increased costs and to maintain a high fishing capacity (**Figure 4.5**). Despite changes in fishing policies, Western Europe pays about six billion US dollars annually (of which four billion by the European Union and almost two billion by Russia) in capacity-enhancing ("harmful") fishing subsidies, which is the second most after Asia (Sumaila *et al.*, 2016).

Figure 4.5 Causal loop diagram of fishing, illustrating causes of the present situation of overfishing. See text for further clarification. Source: Own representation.



The adoption of new technologies leading to overcapacity in vessels and engines is also a major driver for increased fishing (Knudsen *et al.*, 2010; Österblom *et al.*, 2011). Human population growth, associated demand for fisheries products, and multiple effects of pollution, coastal degradation and climate change are other important factors in the analysis of trends in fisheries (Garcia & Rosenberg, 2010). Small changes in temperature affect distribution and abundance of fishes, but can be positive or negative for local fisheries depending on the species and regions (Pörtner & Peck, 2010; Roessig *et al.*, 2004).

For the majority of the stocks, political decision-makers have not followed recent scientific advice, for example from the International Council for the Exploration of the Sea (ICES) and the Mediterranean Advisory Council (MEDAC), and have set Total Allowable Catch (TAC) to levels higher than the scientific recommendation (Voss *et al.*, 2017). Institutional drivers are beginning to change, thanks to information and recommendations from universities, consultative councils and international organizations, with a resulting increase in public awareness. This in turn drives both the market, by avoiding red listed fish, and the political system to regulate fishing and enforce illegal fishing (**Figure 4.5**). For example, the European Union's 2014 update of The Common Fisheries Policy (CFP) illustrates institutional capacity in that regulations have been passed for restoring and maintaining fish stocks above levels capable of producing maximum sustainable yield. However, half of the fish stocks exploited by the European Union fishing fleet are still overexploited (Guillen *et al.*, 2016).

Resource users who are limited to local resources have an incentive to sustain these resources because they do not have a substitute, while users who can access global resources have no such incentives. Therefore, good stewardship depends on institutions where users are held accountable for sustaining the local resources (Berkes *et al.*, 2006). Consumers and citizens are not reached by the feedbacks of natural resource depletion such as price signals and physical scarcity. Three reasons why price fails to provide an accurate signal of declining fish stocks to globally distributed consumers have been proposed by Crona *et al.* (2016). First, the costs of depleting the resource through habitat damage by trawling and by-catch of endangered megafauna have little effect on yield or revenue, as these costs are not reflected in the market price. Second, better fishing technologies can maintain or even increase harvest levels despite declining fish stocks. Third, when declining stocks are substituted by global tradefrom other regions, market signals to consumers also fail. All of these factors result in “masking” and “dilution” of the feedbacks to consumers and citizens by preventing increases in prices and hence in awareness (economic and cultural drivers), thereby reducing public pressure for institutional responses (Morato *et al.*, 2006).

4.4.2 Hunting

4.4.2.1 Effects of hunting on biodiversity and nature’s contributions to people

Hunting is practiced across a wide spectrum of cultural, institutional, economic and environmental contexts within Europe and Central Asia. Whilst hunting clearly impacts the populations of the hunted species, the effects on biodiversity and nature’s contributions to people vary. Hunting takes several forms and is done for various reasons, including management, subsistence, and recreation (Fischer *et al.*, 2013). Under “management hunting” the population densities of certain large-bodied game species are controlled by hunters with potentially positive impacts on biodiversity and forestry (Brainerd, 2007). On the other hand, these game species are sometimes kept at high densities for recreational hunting purposes, resulting in overgrazing, over-browsing and trampling of forest ecosystems by large herbivores, leading to reduced diversity of the understory vegetation and stunted or no regrowth of forest trees and understory plants. Browsing and grazing by wild ungulate game species (such as several deer species or wild boar) are a significant cause of plant species loss regardless of the type of forest management (Pollock *et al.*, 2005; Schulze *et al.*, 2014). Beyond the direct mortality impact on hunted animals, therefore, altered vegetation dynamics can also change animal communities, and the current high densities of ungulate populations in Germany, and Romania and other Central and Eastern European countries, are a major threat to the biodiversity of deciduous forests (Schulze *et al.*, 2014). Hence, the hunting sector and its management is also a main driver of forest change.

Management hunting also provides material (meat) and non-material contributions to people, for example by maintaining traditions and promoting social relations (Fischer *et al.*, 2013). This is also the focus in indigenous or subsistence hunting where cultural identity is emphasized. However, there are signs that indigenous or subsistence hunting is declining, for example in traditional communities in Faroe Islands due to the changing cultural values of younger generations (Nieminen *et al.*, 2004).

Sport and trophy hunting are not motivated by ecological objectives (if so, we would call it management hunting). Sport hunting, including the trapping of individuals, has been mostly aimed at large game species in Europe and Central Asia. These include predatory mammals such as bears, wolves and lynx; herbivorous mammals such as red deer, moose, elk, ibex and chamois; omnivorous mammals such as wild boar; and birds (mainly ducks, geese, waders, doves and several passerines). This has resulted in the extinction of, for example, Caucasian moose and wisent, Carpathian wisent, and ibex on the Iberian Peninsula. The hunting, trapping and poaching of migratory birds is a chronic conservation problem, particularly in the Mediterranean countries, where birds, even small passerines, have been traditionally hunted and trapped for human consumption or for sport (Vickery *et al.*, 2014).

4.4.2.2 Trends in hunting

In the European Union, the Birds Directive currently allows the hunting of 82 species (25 ducks and geese, 15 gallinaceous species, 22 waders, shorebirds and gulls, five doves, 12 passerine species and three rallied species) of which 24 can be hunted in all member States (Annex II of Birds Directive). Many species that are declining at an alarming rate, may still be hunted in several European countries (for example skylark, lapwing, curlew, black-tailed godwit, garganey, taiga bean goose, pintail, snipe, quail and turtle Dove). In addition to hunting, selective trapping is allowed in several European Union member States (Art. 9 of Birds Directive), where net traps and cage traps lead to the killing of tens of thousands of skylarks, Ortolan buntings, golden plovers, turtle doves, quail and lapwings in France, and millions of thrushes in Malta, Spain (Catalonia) and Italy annually (Fenech, 1992; Hirschfeld & Heyd, 2005).

Since the 1950s there has been a general decrease in the size of the annual wild bird hunting bag (total catch) in Western Europe. Hunting pressure is still high, although uncertain in the south and east of the region (Weinbaum *et al.*, 2013). In 2005, the total hunting bag in the EU-27 was around 102 million individuals of 82 bird species (Hirschfeld & Heyd, 2005). Hunting bag data also suggest a recent short-term increase in hunting pressure for mammals. For instance, the hunting of red deer (*Cervus elaphus*) has increased exponentially in eight of 11 Western and Central European countries studied (Milner *et al.*, 2006).

4.4.2.3 Drivers of hunting

The culture of hunting is based on a value system that is deeply rooted in traditions in Europe and Central Asia. However, traditions emerging from subsistence hunting are today based on identity and life-style, expressed as sport and trophy hunting or management hunting, with wild meat as a bonus (Fischer *et al.*, 2013). Demographic drivers like urbanization do not seem to change these cultural drivers; there is still a high density of hunters per km^2, 50 in Cyprus, followed by 47 in Malta, 5.0 in Ireland, 3.8 in Denmark, 3.3 in the UK, 2.5 in Italy and Portugal, 2.4 in France, and 2.0 in Greece (Hirschfeld & Heyd, 2005). Illegal hunting and trapping is still common in the south and east of the Europe and Central Asia region (Arizaga & Laso, 2015; Michel, 2008).

Hunting is well regulated in most countries in Europe and Central Asia, however, law enforcement is lagging behind in many Central Asian countries and the southern parts of Western and Central Europe (Michel, 2008). Hunter associations are powerful interest groups in many countries and the governance trend is to foster stewardship and sustainable management hunting for vulnerable species rather than imposing hunting bans (Dusseldorp *et al.*, 2004). Tensions between hunters and anti-hunting groups have escalated, e.g. in Malta, with rural surveillance systems and local raids by anti-hunting groups, physical fights between anti-hunting activists and hunters or poachers, use of drones for observations, and police or army interventions (Veríssimo & Campbell, 2015).

4.4.3 Water use and desalination

4.4.3.1 Effects on water use and desalination on biodiversity and nature's contributions to people

Water is extracted from streams, rivers, lakes and wetlands for drinking and bathing, irrigation for agriculture, cooling for energy production (power plants), as coolant or reagent in various industries and as a leaching agent in mining. Freshwater ecosystems host disproportionately high numbers of species relative to their surface area, yet their biodiversity is declining faster than either terrestrial or marine biodiversity (Dudgeon *et al.*, 2006; Strayer & Dudgeon, 2010; WWF, 2008). In addition, ecosystem services provided by freshwater systems (streams, rivers, lakes, wetlands) were estimated to contribute to 20% of the value of all ecosystem services (Costanza *et al.*, 1997).

Groundwater overexploitation, often due to irrigation, results in lowering the groundwater table, which increases the risk of desertification. In addition, the chemical composition of groundwater is often suboptimal for irrigation due to its high salt/mineral or metal content and irrigation with groundwater often leads to salinization or alkalinization of the soils. This is a problem in Estonia, Latvia, Poland, Hungary, Romania, Moldova and Spain (EEA, 2007) and in many areas of Eastern Europe and Central Asia. The intrusion of salt water from the sea in the place of groundwater is an acute problem in coastal areas of Denmark, and in coastal Mediterranean areas of Spain, Italy and Turkey, mostly due to the water needs of mass tourism facilities and irrigation (EEA, 2007).

Desalination of seawater is increasing to satisfy demand for water due to the present water shortage, mainly in semiarid and arid coastal regions (Llamas *et al.*, 2015). It has a long history in the Middle East and Mediterranean (Einav & Lokiec, 2003; Roberts *et al.*, 2010), but studies on the impacts of desalinization on biodiversity are recent and still scarce. The greatest environmental and ecological impacts have occurred around older multi-stage flash plants discharging salt into water bodies with little flushing. Effects include substantial increases in salinity and temperature and the accumulation of metals, hydrocarbons and toxic anti-fouling compounds in receiving waters (Al-Taani *et al.*, 2014; Höpner & Lattemann, 2003; Roberts *et al.*, 2010) and sediments (Alharbi *et al.*, 2012).

Effects on ecosystems range from no significant impacts on benthic communities, to reduced leaf growth and higher incidence of leaf necrosis, drop in photosynthetic performance and mortality in seagrasses, and widespread alterations to community structure in seagrass, coral reef and soft-sediment ecosystems when discharges are released to poorly flushed environments (Del-Pilar-Ruso *et al.*, 2015; Del-Pilar-Ruso *et al.*, 2008; Pagès *et al.*, 2010; Roberts *et al.*, 2010).

4.4.3.2 Trends in water use and desalination

Water management has become one of the main concerns for humanity, also in areas where water has until now been considered an unlimited resource. The availability of freshwater resources in a country is determined by geology, climate, land use and external (transboundary) water flows. In Western and Central Europe, the largest freshwater resources are in Norway, Turkey, Germany, France and Sweden (Eurostat, 2015). Many countries receive the majority of freshwater resources externally, with Serbia, Hungary, the Netherlands, Slovakia and Bulgaria receiving over 80% of their freshwater from upstream areas in other countries. The amount of potable freshwater per inhabitant is highest in Iceland, Norway, Serbia, Croatia and Finland, whereas low levels (<3,000 m^3 per inhabitant per year are found in Denmark, Romania, Belgium, the Czech Republic, Cyprus and Malta, and in countries with large human populations (France, UK, Spain, Germany, Italy and Poland) (Eurostat, 2015). In 2012, total water extraction from surface waters was highest in Turkey, Spain, Germany and France (over 24 billion m^3 from surface waters, over 5 billion m^3 from groundwater) (Eurostat, 2015). Between 2003 and 2013, the amount of freshwater extracted increased most in Malta (43%, mostly groundwater), Slovenia (36%, mostly surface water) and decreased the most in Lithuania (80%, mostly surface water) and Slovakia (39%, mostly surface water) (Eurostat, 2015).

Around 63% of desalinated water worldwide is used for satisfying urban demand for drinking water, 26% for industrial uses, and 6% in power stations for electricity generation (Ziolkowska & Ziolkowski, 2016). The cost of desalinated water is decreasing, thanks to technological and efficiency improvements of the membrane filters, which reduces energy demand (Semiat, 2000; Ziolkowska & Ziolkowski, 2016).

4.4.3.3 Drivers of water use and desalination

Water regulations are cornerstones of national environmental regulations. The rapid decrease in water use in Lithuania and Slovakia mentioned above is mainly a result of institutional drivers (regulations and a better price system). Other important Institutional drivers are regulations and investments in, or subsidies for, wastewater and desalination ("recycling" in **Figure 4.7**). Depletion (unsustainable use) of ground and surface water is driven by high domestic material consumption (DMC) fuelled by urbanization and GDP growth.

Similarly, seawater desalination is driven by growth in human population, income and domestic material consumption in general and growth and urban development, agriculture and tourism in particular (EEA, 2007; Gladstone *et al.*, 2013).

4.4.4 Mineral and fossil fuel extraction

4.4.4.1 Effects on biodiversity and nature's contributions to people

The minerals industry is divided into four sectors: fossil fuels (e.g. coal and oil), metallic minerals (e.g. iron, copper and zinc), construction minerals (e.g. natural stone, sediments and other aggregates, gravel) and industrial minerals (e.g. borates, talc, silica and limestone).

In Western and Central Europe, extraction of abiotic resources is highly dominated by construction and industrial minerals, and to a more limited extent fossil energy (Bahn-Walkowiak *et al.*, 2012). Dredging and pumping operations have a direct effect on the local biological communities and cause changes in the composition of fauna (Pérez-Ruzafa *et al.*, 2007), and reduction in species diversity, abundance, and biomass (Bolam *et al.*, 2015; Sutton *et al.*, 2009). This changes food webs, particularly lower trophic levels including detritivores, with impacts on carbon cycling (Tecchio *et al.*, 2016).

In Central Asia the extraction and processing of minerals, including poor governance practices and economic pressures (Honkonen, 2013), leads to various environmental impacts including depletion of non-renewable resources and consequent disturbance of the landscape, biodiversity and nature's associated contributions to people (Azapagic, 2004; Starikova, 2014), particularly in vulnerable arid and mountainous territories (Lukashov & Akpambetova, 2012).

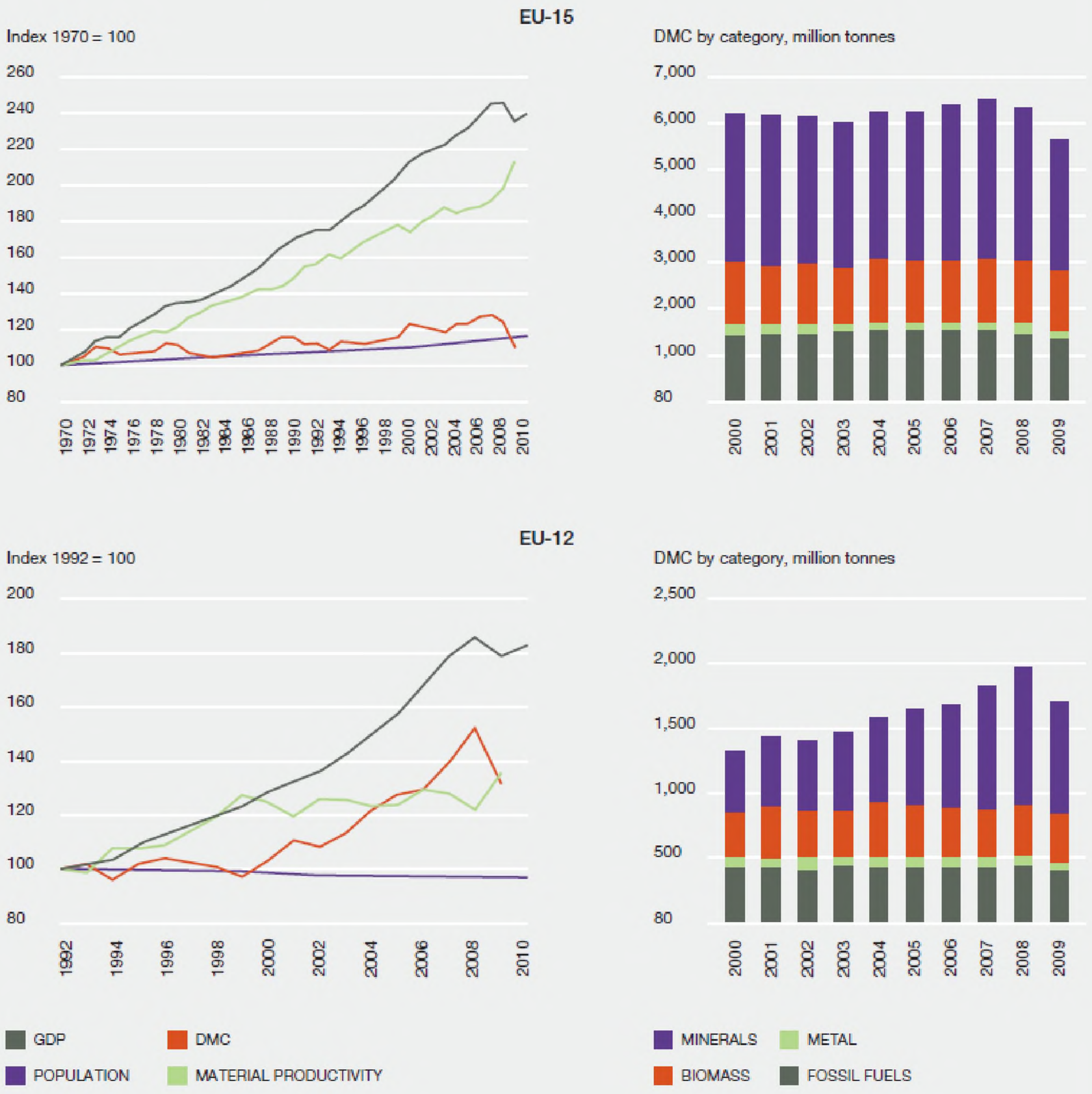
The environmental effects of the mining and minerals industry include gas emissions, discharge of liquid effluents (including acidification of waterways) and generation of large volumes of solid waste, as well as direct destruction or disturbance of natural habitats. Additionally, contamination of water can continue when mining or mineral extraction activity ceases due to acid mine drainage and other toxic leachates. Large water bodies and land are being polluted by natural resource extraction in Central Asia (Jakupov, 2013; Kalmenova, 2014) and methane leaks from gas infrastructure and coal mines pollute soil and the Caspian Sea (Dahl & Kuralbayeva, 2001; Karenov, 2006; Mukanova, 2015). Oil production in the Caspian Sea has had a direct impact on ecosystem functioning through pollution (Netalieva *et al.*, 2005). An increase in the rate of glacier melting has been observed as a consequence of dumping of mine spoil on receding and thinning glacier snouts in Kyrgyzstan (Evans *et al.*, 2015; Jamieson *et al.*, 2015; Kronenberg, 2014). Uranium mining sites pose a threat to biodiversity exposed to high radiation doses (Oughton *et al.*, 2013; Bekbolotova & Toychubekova, 2014; Jolboldiev, 2016; Karsenov, 2011).

4.4.4.2 Trends in mineral and fossil fuel extraction

Fossil fuels and rare earth minerals are the largest contributors to GDP of Central Asia and in the last decade the volume of coal mining has doubled in this subregion (Kabirova, 2009; Plakitkina, 2014). The mineral extraction industry in Central Asia has been driven by trade liberalization and increasing world market prices. The largest share of foreign direct investment in Central Asia is in the natural resource extraction industry. Central Asian Governments seek foreign direct investments as a way to boost local incomes while the high environmental risks and lack of transparent governance have had little effect on economic development (Dikkaya & Keles, 2006; Doroshenko *et al.*, 2014).

Since the 1950s most metallic mineral resources have been imported into Western and Central Europe (Calvo *et al.*, 2016; Schaffartzik *et al.*, 2016; Schoer *et al.*, 2012). However, while domestic extraction of metallic mineral has been reduced, extraction of sediments is increasing mainly in coastal areas. Domestic material consumption (DMC) is defined as the annual quantity of raw materials extracted from the domestic territory, plus all physical imports minus all physical exports. It has increased since 1970 (**Figure 4.6**) but decreased from 7.7 to 6.7 billion tonnes in the period 2000-2016 in EU-28. Greece, Spain and Italy almost halved their domestic material consumption since the financial crisis in 2008 and without these countries the domestic material consumption in the European Union has been stable since 2000 (Eurostat, 2017a). Recently there has been an increase in prospecting for resources in previously unexploited and fragile environments such as the Arctic and on the ocean floor, which consequently increases the pressure on ecosystem resilience in sensitive environments (Martin *et al.*, 2012).

Figure 4 6 Domestic material consumption (DMC) in EU-12 and EU-15 has increased historically except as a result of the financial crisis in 2008. Source: Martin *et al.* (2012).



4.4.4.3 Drivers of mineral and fossil fuel extraction

The European Union is aiming to become a resource-efficient, green and competitive low-carbon economy through absolute decoupling of economic growth and environmental degradation (EEA, 2014e). Recent changes in indirect drivers include climate and energy policies, natural resource taxation, subsidizes to recycling schemes (Sӧderholm, 2011) and regulating producers' responsibility for the waste (Ekvall *et al.*, 2016). This has resulted in reduced use of fossil fuels for energy production, improved energy efficiency and increased resource efficiency. However, domestic material consumption is only beginning to decline from a very high level and the increases in environmental taxes have only kept pace with other taxes and the GDP, hence the environmental tax reforms called for by the Green Economy (UNEP, 2011) and Convention on Biological Diversity (Aichi Biodiversity Targets) have not progressed since 2002 (Section 4.3.2). On the contrary, some aspects of public support to mineral extraction can be seen as harmful subsidies. In 2010, the metal mining sector in Sweden received subsidies of € 40 million compared to only € 0.6 million for the metal recycling sector (Johansson *et al.*, 2014). Furthermore, mining companies only pay 0.2% of the revenues from mining as resource tax (Koh *et al.*, 2017).

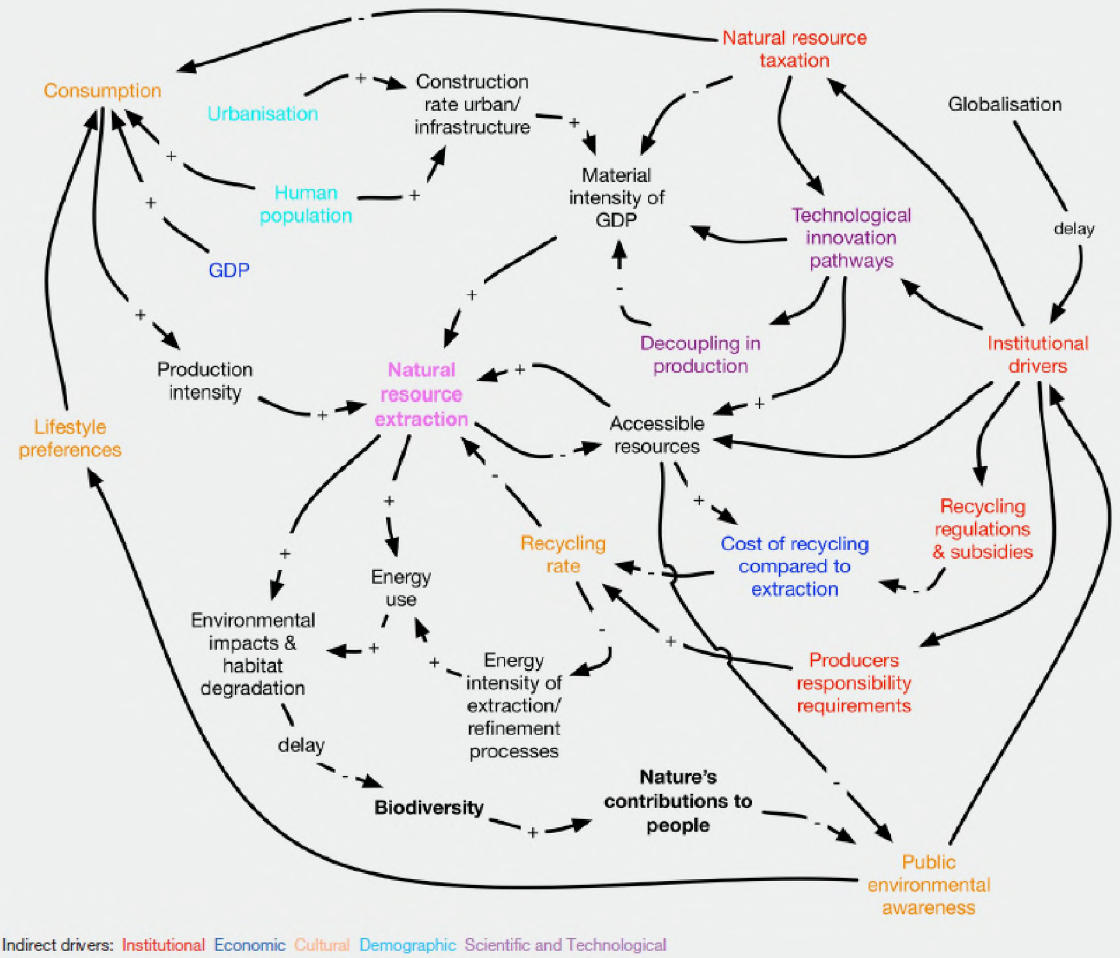
Conversely, in Central Asia, economic growth is currently closely associated with mineral and fossil fuel extraction (Ondash, 2011), which is anticipated to continue in the future (Doroshenko *et al.*, 2014). Central Asia has initiated policies for increased resource efficiency, mainly targeted at energy efficiency (Government of Kyrgyzstan, 2014; Pomfret, 2011). However, global initiatives (e.g. the Extractive Industries Transparency Initiative) have so far had a limited effect on sustainable use of natural resources (Furstenberg, 2015).

4.4.5 Drivers of natural resource extraction

Drivers of natural resource extraction are indirect drivers of biodiversity change, as synthesized in **Figure 4.7**. Natural resource extraction basically follows increases in GDP and human population growth (Peet & Hartwick, 2015). Urban sprawl increases this pressure (Schewenius *et al.*, 2014). GDP growth is still the goal of the European Union, but recently the goal has been reformulated toward smart, sustainable and inclusive growth (European Commission, 2017a).

The ecological footprint is an area-based measure of material consumption driving natural resource extraction. Western and Central Europe's ecological footprint is twice the size of its area and consumption patterns remain very high by global standards (EEA, 2014a). Due to institutional drivers, increasing productivity and the financial crisis of 2008, Western and Central Europe's domestic material consumption has decreased recently (4.4.4).

Figure 4 7 Causal loop diagram illustrating drivers of natural resource extraction.
Source: Own representation.



Population changes influence GDP, which drives the rate of production intensity and thereby extraction of natural resources (EEA, 2012b). However, formal institutions drive the taxation of natural resources, which influences the rate of domestic material intensity and the material intensity of GDP (domestic material consumption divided by GDP), affecting extraction rates. Institutional drivers also regulate producers' responsibility and influence the rate of recycling through regulations and economic incentives. Environmental regulations may restrict availability of natural resources but technological innovation typically increases availability by facilitating extraction (Litovitz *et al.*, 2013). Finally, institutions like the German energy transformation also influence technological innovation pathways, impacting the material intensity of GDP (**Figure 4.7**).

Natural resource extraction may result in depletion of natural resources as well as unintended environmental impacts and habitat degradation. These effects may increase public awareness which in turn influences lifestyle preferences and becomes a driver of institutional change (Nolan & Schultz, 2015). Global trade, on the other hand, may disguise these effects and thereby delay institutional responses (4.4.1.3).

In summary, cultural drivers (growth oriented development), demographic and economic drivers (urban sprawl, tourism, consumption etc.) continue exerting a pressure on natural resource extraction in Europe and Central Asia. Institutional drivers have been used to reduce this pressure. However, economic drivers in terms of environmental taxes have so far not been employed to support these advances in institutional drivers and therefore the technological innovative potential is not realized.

4.5 Drivers and effects of land-use change

4.5.1 Effects of land-use change on biodiversity and nature's contributions to people

In Europe and Central Asia land-use change is one of the most important drivers of changes in biodiversity and the provision of nature's contributions to people (Aguilar *et al.*, 2006; CBD, 2014; EEA, 2015c; Fischer & Lindenmayer, 2007; Frankham *et al.*, 2014; Garibaldi *et al.*, 2011; Gil-Tena *et al.*, 2015; Gonthier *et al.*, 2014; Humphrey *et al.*, 2015; IPBES, 2016a, 2016b; Leimu *et al.*, 2010; Rusch *et al.*, 2016; Tscharntke *et al.*, 2007). Mitigating the adverse effects of land-use change is crucial to halting the loss of biodiversity and nature's contributions to people (Alkemade *et al.*, 2009; CBD, 2014; Dirzo & Raven, 2003; Hoekstra *et al.*, 2005; MEA, 2005a).

4.5.1.1 Effects of conventional agricultural intensification

Intensification of conventional agriculture has a multi-factorial impact on biodiversity and nature's contributions to people. The actual impacts often vary with an organisms' taxonomic or functional group and hence evolutionary history, within and between geographic regions (Báldi *et al.*, 2013; Billeter *et al.*, 2008; Flohre *et al.*, 2011; Gabriel *et al.*, 2013; Gonthier *et al.*, 2014; Guerrero *et al.*, 2011; IPBES, 2016a; Le Féon *et al.*, 2010; Redhead *et al.*, 2015; Sjödin *et al.*, 2008; Tsiafouli *et al.*, 2015; Woodcock *et al.*, 2005). In Europe and Central Asia, the impact of conventional intensification of agriculture has been manifest through loss of (semi-) natural habitats, landscape homogenization and intensive use of agri-chemicals (Gonthier *et al.*, 2014, see Chapter 3). The focus on maximising agricultural production since World War II has transformed and modified natural habitats and traditional semi-natural ecosystems physically, biologically and chemically with profound implications for biodiversity and nature's contributions to people (Gabriel *et al.*, 2013; Gil-Tena *et al.*, 2015; IPBES, 2016a; Sanderson *et al.*, 2013; Stoate *et al.*, 2009; Tscharntke *et al.*, 2005; UNEP, 2016). For example, a negative relationship between crop yield and most elements of biodiversity (plants, bumblebees, solitary bees, butterflies, epigeal arthropods) were found in a study of eight paired landscapes of organic and conventional management farms (Gabriel *et al.*, 2013). In Europe and Central Asia agricultural intensification (defined as the number of pesticide applications, tillage operations, fertilizer levels or crop types) relates to reductions in species richness and diversity of plants, wild bees and birds, but not ground beetles, at scales from field to region (Billeter *et al.*, 2008; Flohre *et al.*, 2011; Le Féon *et al.*, 2010). Among grazed and mown grasslands biodiversity of plants, animals and microorganisms declines with increasing mean land-use intensity, while this decline is at least ameliorated by variation in land-use intensity between years (Allan *et al.*, 2014).

Landscape homogenization is an outcome of conversion of semi-natural habitats based on traditional land-use practices into intensively managed arable or grazing land, which has reduced biodiversity and a number of nature's contributions to people across Europe and Central Asia (Billeter *et al.*, 2008; Flohre *et al.*, 2011; Le Féon *et al.*, 2010; Munteanu *et al.*, 2014; Newbold *et al.*, 2016; Pe'er *et al.*, 2014; Stoate *et al.*, 2009; Tscharntke *et al.*, 2005; Van Zanten *et al.*, 2014; Vanbergen *et al.*, 2006; Vanbergen, 2014; Yoshihara *et al.*, 2008; Zhu *et al.*, 2012). Although large-scale, intensively-managed agricultural monocultures can provide food and habitat resources for organisms adapted to exploit it, this resource is insufficient to cater for most elements of biodiversity (Diekötter *et al.*, 2014; IPBES, 2016a; Kovács-

Hostyánszki *et al.*, 2013, 2017; Riedinger *et al.*, 2015; Rundlöf *et al.*, 2014; Schweiger *et al.*, 2007; Tscharntke *et al.*, 2005; Vanbergen *et al.*, 2010; Westphal *et al.*, 2009).

Intensive use of agri-chemicals (such as herbicides, insecticides, or inorganic fertilizers) is linked to transformation of ecological communities and directly contributes to declines of species, some of which providing important contributions to people (Brittain *et al.*, 2010; Chiron *et al.*, 2014; Deguines *et al.*, 2014; Dormann *et al.*, 2007; Gabriel *et al.*, 2013; Gonthier *et al.*, 2014; Hawes *et al.*, 2003; IPBES, 2016a; Rundlöf *et al.*, 2015; Storkey *et al.*, 2012; Woodcock *et al.*, 2016). For example, intensive use of herbicides and inorganic fertilizers act as environmental filters eliminating wild plant species, especially those adapted to conditions of intermediate fertility, with implications for the higher trophic levels, such as insect pollinators and seed feeding birds, which depend on such wild plant species for food resources (Chiron *et al.*, 2014; Hawes *et al.*, 2003; IPBES, 2016a; Storkey *et al.*, 2012). Further, agricultural insecticides target pest populations, they also pose a direct hazard to non-target insects, such as pollinators, that are crucial for the maintenance of biodiversity in natural ecosystems and deliver important services to pollinator-dependent crops (Deguines *et al.*, 2014; IPBES, 2016a).

Genetically modified crops can possess traits for herbicide tolerance or resistance to pests and their large-scale cultivation may drive changes to species and populations in agricultural landscapes either directly on gene pools or indirectly on dependent biodiversity. The direct hazard for biodiversity and nature's contributions to people is relatively low, although where lethal impacts of insect-resistant crops on biodiversity occur they tend to be on species closely related to the targeted pest (IPBES, 2016a, 2016b; Marvier *et al.*, 2007; Mommaerts *et al.*, 2010; Nicolia *et al.*, 2014; Potts *et al.*, 2016). Reductions of pesticides that may accompany the use of insect-resistant crops could lower overall pesticide pressure on non-target organisms, but the emergence of secondary outbreaks of non-target pests or primary pest resistance can lead to a resumption of pesticide use (Barfoot & Brookes, 2014; IPBES, 2016a, 2016b; Lu *et al.*, 2010). Most risk to biodiversity and nature's contributions to people from genetically modified crops comes both from their management and direct impacts per se. Intensive herbicide use on herbicide-tolerant crops will eliminate wild plants, with concomitant effects on other biodiversity components through the network of interactions, although this and the effects on nature's contributions to people remains little-studied (Bohan *et al.*, 2005; IPBES, 2016a, 2016b; Morandin & Winston, 2005).

If continued, conventional intensive agriculture will jeopardize both sustainable land management and food production. Erosion of natural capital (such as pollinators, natural enemies of pests, soil biodiversity and others) poses a substantial risk to the sustained and resilient production of food (IPBES, 2016a; Kovács-Hostyánszki *et al.*, 2017; Tsiafouli *et al.*, 2015). Studies focusing on the comparison of conventional intensive management with less-intensive agricultural systems indicate that there is considerable potential for alternative approaches to management that secure farm production and conservation of nature's contributions to people (Bommarco *et al.*, 2013; Kovács-Hostyánszki *et al.*, 2017; Pywell *et al.*, 2015).

4.5.1.2 Effects of agri-environment schemes

A wealth of studies shows that diversity or activity densities at local (field to farm) scales can be enhanced through agri-environment schemes (Albrecht *et al.*, 2007; Batáry *et al.*, 2011; Carvell *et al.*, 2011; Doxa *et al.*, 2010; Fuentes-Montemayor *et al.*, 2011; Gonthier *et al.*, 2014; Haaland *et al.*, 2011; Hiron *et al.*, 2013; Holzschuh *et al.*, 2007; Kovács-Hostyánszki *et al.*, 2011; Krauss *et al.*, 2011; Pywell *et al.*, 2012; Scheper *et al.*, 2013). Certain agri-environmental schemes in the European Union clearly benefit target organisms (e.g. wildflower strips and bees - Carvell *et al.*, 2017; organic farming and plants - Batáry *et al.*, 2013; Henckel *et al.*, 2015; Tuck *et al.*, 2014). Evidence for increasing biodiversity is sometimes equivocal, complex and unpredictable (e.g. organic farming effects on insects and mammals - Bengtsson *et al.*, 2005; Gabriel *et al.*, 2010, 2013; Krauss *et al.*, 2011; Ponce *et al.*, 2011; Tuck *et al.*, 2014).

Landscape complexity and the ecological contrast with other habitats at field scales influence the efficacy of agri-environment schemes, with typically the greatest uplift in local biodiversity in highly homogenized landscapes that lack remnant semi-natural habitats providing resources for wildlife (Batáry *et al.*, 2011; Gabriel *et al.*, 2010; Heard *et al.*, 2007; Hiron *et al.*, 2013; Holzschuh *et al.*, 2007; Kleijn *et al.*, 2011; Rundlöf & Smith, 2006; Scheper *et al.*, 2013, 2015; Tscharntke *et al.*, 2005; Tuck *et al.*, 2014). Agri-environment schemes have a less expected impact in high-diversity cultural landscapes where they slow down, prevent or even reverse the abandonment process and thus help maintain high nature-value grasslands (Babai *et al.*, 2015).

Recent evidence points to population-level increases when diverse habitat resources are provided and sustained at the landscape scale (Carvell *et al.*, 2017; Carvell *et al.*, 2015; Doxa *et al.*, 2010; Tschumi *et al.*, 2016; Wood *et al.*, 2015). Emerging evidence suggests that targeted habitat creation or protection of ecological infrastructure in the landscape can contribute towards achieving a more sustainable agriculture (Bommarco *et al.*, 2013; IPBES, 2016a, 2016b; Kovács-Hostyánszki *et al.*, 2017; Potts *et al.*, 2016; Pywell *et al.*, 2015; Tittonell, 2014). There are concerns about the efficacy of agri-environment schemes for conserving rare, specialized species and there is a level of geographic bias in the available evidence (Batáry *et al.*, 2015; Scheper *et al.*, 2013; Sutcliffe *et al.*, 2015). Effectiveness of agri-environment schemes' interventions could be improved by tailoring to targets for biodiversity or nature's contributions to people considering local ecological and landscape context, and different socio-economic settings (Babai *et al.*, 2015; Batáry *et al.*, 2015; Bright *et al.*, 2015; Dicks *et al.*, 2016; Ekroos *et al.*, 2014; IPBES, 2016a; Mccracken *et al.*, 2015; Molnár & Berkes, 2017; Pe'er *et al.*, 2014; Pywell *et al.*, 2012; Scheper *et al.*, 2013; Sutcliffe *et al.*, 2015).

4.5.1.3 Effects of increasing intensity of management on forest land

Long-term human pressures during the last centuries (Kolář *et al.*, 2016) have resulted in the deforestation and fragmentation of forests in Europe and Central Asia (e.g., Wallenius *et al.*, 2010b) and will continue to cause species extinctions (Hanski, 2000; Niemelä *et al.*, 2005). More than 35% of European forests are in mosaic landscapes that are significantly fragmented by agricultural and artificial lands (EEA, 2016b).

One key indicator of high quality habitats is the amount of dead wood. Overall, natural forests normally harbour around 100 m^3/ha of dead wood (Jonsson & Siitonen, 2012). Currently, in Westen, Central and Eastern Europe, the volume of dead wood in forests is estimated to be 20.5 m^3/ha (including the Russian Federation) and 10 m^3/ha (without the Russian Federation; Forest Europe (2011). Among individual countries the volumes vary considerably (**Figure 4.8**).

Forest management intensification reduces natural forest area and degrades habitat quality due to loss of structural components (e.g. dead wood), simplified spatial stand structure (e.g. uneven age-structure) and simplification of natural processes (e.g. gap formation, decomposition) (**Table 4.5**). This greatly impacts associated biodiversity, particularly specialist species (Brockerhoff *et al.*, 2008; Brumelis *et al.*, 2011; Esseen *et al.*, 1997; Kuuluvainen, 2002; Paillet *et al.*, 2010). For example, species richness of multiple taxa (bryophytes, lichens, fungi, saproxylic beetles, carabids) were considerably lower in managed forests than in unmanaged forests with effects most pronounced for forests that underwent clearcutting and historic changes in tree species composition (Paillet *et al.*, 2010).

Figure 4 8 Average volume of standing and lying deadwood in 2010 (m³/ha). Source: Forest Europe (2011).

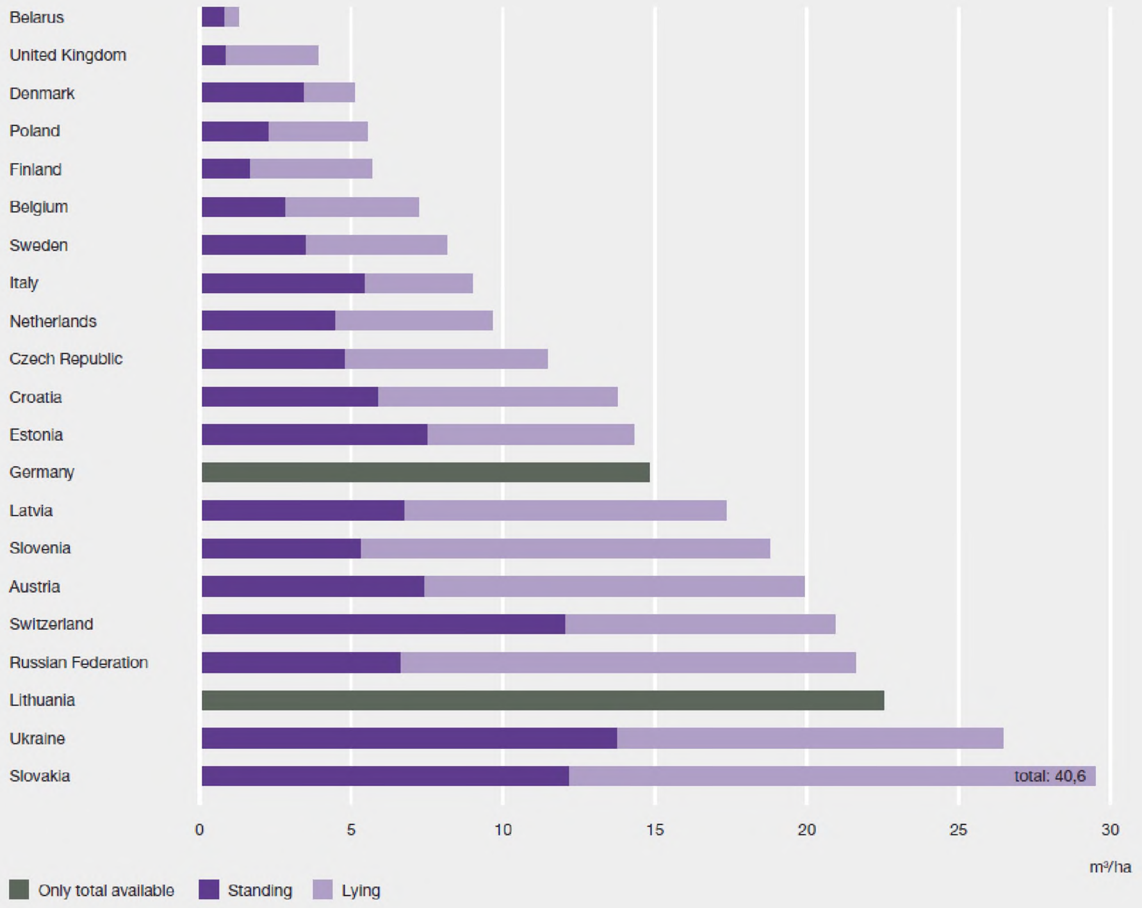


Table 4.5: Components that are important for biodiversity in natural forests that are negatively influenced by forestry. Source: Modified after Esseen *et al.* (1997).

Structural components
Very old trees
Trees with abundant growth of epiphytes
Broken, stag-headed and leaning trees
Trees with holes, cavities and other microhabitats
Dead standing trees (snags)
Fire-scarred trees, snags and stumps
Large downed logs in various stages of decomposition
Spatial patterns
A developed understory of tree saplings and shrubs
Mixed stands, with both conifers and broad-leaves
Uneven-aged stand structure
Multi-layered tree canopies
Patchy distribution of trees, gaps
Processes

Post-fire succession
Succession with tree-species replacement
Self-thinning
Gap formation
Snag and log formation
Decomposition of coarse woody debris

Intensive forest management also includes conversion of non-forested lands to managed forest plantations, which often have detrimental effects on in situ biodiversity due to loss of habitat and associated species turnover (Brockerhoff *et al.*, 2008). However, afforestation of agricultural land can assist biodiversity conservation by providing ecotones and increasing forest connectivity (Brockerhoff *et al.*, 2008). Where forest cover is low, plantation on marginal land can provide habitats for rare forest adapted species (Humphrey *et al.*, 2003). Additional conservation efforts to improve forest structure can correspondingly improve the situation for biodiversity (Humphrey, 2005). Traditionally managed and used forest ecosystems such as traditional agro-silvicultural systems with wood-pastures and coppicing also support and maintain suitable conditions for many forest species (Bollmann & Braunisch, 2013; Kirby & Watkins, 2015; Plieninger *et al.*, 2015). As different types of forest management promote different facets of biodiversity or nature's contributions to people, heterogeneity of forest management practice at the landscape scale, also including unmanaged forest, is likely to maximize landscape-level forest biodiversity and forest contributions to people (Elbakidze *et al.*, 2017; van der Plas *et al.*, 2016).

4.5.1.4 Effects of decrease in land area with traditional land use and loss of traditional ecological knowledge

Much of the biodiversity in Europe and Central Asia relies on traditionally managed semi-natural habitats (EEA, 2015b; Kirby & Watkins, 2015; Liira *et al.*, 2008; Plieninger *et al.*, 2015; Stoate *et al.*, 2009; Tscharntke *et al.*, 2005; UNEP, 2016). The loss and abandonment of traditionally managed systems due to multiple factors has been an important driver of decline in biodiversity and nature's contributions to people (Bergmeier *et al.*, 2010; Bubová *et al.*, 2015; Fuller, 1987; Helm *et al.*, 2006; Henle *et al.*, 2008; MacDonald *et al.*, 2000; Middleton, 2013; Plieninger *et al.*, 2015; Rotherham, 2015; van Swaay *et al.*, 2006). Abandonment of traditional land management allows reassertion of successional and other ecological processes (e.g. increase in interspecific competition) leading to loss of specific habitats that support biodiversity of high conservation value (Bergmeier *et al.*, 2010; Middleton, 2013; Rotherham, 2015). The evidence for a negative impact of abandonment is particularly strong for semi-natural grassland systems (Bergmeier *et al.*, 2010; Dengler *et al.*, 2014; Rotherham, 2015), mountainous areas (MacDonald *et al.*, 2000) and for particular taxa (e.g. butterflies, farmland birds and plants - van Swaay *et al.*, 2006; Bubová *et al.*, 2015; Liira *et al.*, 2008). Since World War II, the cover of open cultural woodlands in Western and Central Europe has rapidly declined and been replaced with agricultural fields and closed forests. This led to a decline in light-dependent specialist species of open woodland and increases in species typical for mesic and closed forest (Bütler *et al.*, 2013; Hédl *et al.*, 2010; Miklín & Čížek, 2014; Nieto & Alexander, 2010; Plieninger *et al.*, 2015; Saniga *et al.*, 2014; van Swaay *et al.*, 2006). There are cases where taxa benefit from abandonment of semi-natural habitats (Gulvik (2007) for Oribatid mites, Sitzia *et al.* (2010) but these seem to be the exception. Abandonment and loss of semi-natural habitats have also significant negative impacts on cultural and social capital and results in loss of traditional and local knowledge (Csergo *et al.*, 2013; Molnár & Berkes, 2017; Rotherham, 2015). The precise outcome often depends upon the direction of the succession (e.g. to steppe vs to forest - Dengler *et al.*, 2014; or above vs below the treeline -

MacDonald *et al.*, 2000; the spatial context - Sitzia *et al.*, 2010; the time since abandonment - Lasanta *et al.*, 2015; and the pattern of farming - MacDonald *et al.*, 2000).

4.5.1.5 Effects of urban development

The expansion of urban areas and its pressure on natural and semi-natural land will continue to be one of the major land-use factors in large parts of Western and Central Europe (EEA, 2016d), and is also likely to result in considerable land take in Central Asia in coming decades (Angel *et al.*, 2011). With increasing urbanization, direct destruction of habitats, reduction of habitat areas, increasing fragmentation and degradation in both terrestrial and aquatic habitats can lead to significant negative impacts on biodiversity (Güneralp & Seto, 2013; McKinney, 2006, 2008). Urbanization affects different habitats and species groups disproportionately and often its effects are related to intensity of urbanization and regional biodiversity patterns (McKinney, 2008). In regions with less effective governance of land use, there is a greater possibility of development affecting areas with high biodiversity (Güneralp & Seto, 2013). In Western and Central Europe, urban development and its associated land take poses a major threat to soil and could have significant effects on agricultural production and food (Gardi *et al.*, 2015).

4.5.1.6 Effectiveness of landscape and habitat restoration

Restoration success depends on the ability to encompass the important ecological mechanisms that underpin ecosystem functioning (Török & Helm, 2017). For example, forest restoration success has been shown to mainly depend on time since restoration initiation, disturbance type (secondary or selectively logged forests) and landscape context (e.g. forest patch size and isolation; (Crouzeilles *et al.*, 2016). It is important to restore the genetic diversity contained within an ecosystem to assure evolutionary potential and to avoid adverse effects caused by management, e.g. founder effects, where only few individuals contribute to initial genetic diversity (Brudvig, 2011; Mijangos *et al.*, 2015; Wortley *et al.*, 2013) (see **Box 4.2**).

There are numerous links between restoration, economic development, and human well-being (Aronson *et al.*, 2010; Benayas *et al.*, 2009) (see **Box 4.2)**. Successful engagement of local community and other social attributes are considered highly important in determining the feasibility and cost of restoration, as well as the success of restoration and sustainability of the restoration outcome (Shackelford *et al.*, 2013; Wortley *et al.*, 2013).

Rewilding is a particular approach that aims to restore ecosystems toward a state of wilderness (Carey, 2016). The effects of rewilding on biodiversity and nature's contributions to people likely depend on initial conditions before rewilding and on the success of development of self-sustainable ecosystems. Sufficient evidence for suitable solutions is not yet available (Cerqueira *et al.*, 2015; Dixon *et al.*, 2016; Götmark, 2013; Smit *et al.*, 2015; Ziolkowska & Ziolkowski, 2016). There is a potential for conflict as many proposed rewilding areas (see Ceaușu *et al.*, 2015) lie in regions where indigenous peoples and local communities live (e.g. Carpathians, Balkan). Considering human rights and the rights of these communities during the establishment of rewilding areas is of vital importance.

Box 4.2: Restoration of grasslands has brought people back to the countryside.

Sheep on recently restored alvar grassland in Saaremaa, Estonia. Following the abandonment of traditional land use, large part of Estonian semi-natural grasslands overgrew with shrubs. Restoration of those species rich grasslands and subsequent grazing, supported by agricultural subsidies, has led to positive impacts on both the local livelihood and on biodiversity. Photo: Ants Animägi, Estonian State Forest Management Centre.

Alvars, dry calcareous thin-soiled semi-natural grasslands once covered ca 50,000 hectares in western part of Estonia, especially on its scenic islands Saaremaa, Muhu and Hiiumaa (Helm *et al.*, 2007). By 2013, only 2500 hectares were managed by grazing – a traditional management method necessary for the persistence of these high nature value habitats (Government of Estonia, 2013). In an effort to save high biodiversity and traditional land-use practices related to Alvar grasslands, 600 land-owners and 41 local farmers and farming companies in 25 regions all over western part of Estonia are participating in the LIFE+ programme project "LIFE to Alvars" (LIFE13NAT/EE/000082) from 2014 to 2020. Project aims to double the area of managed Alvars in Estonia by restoring 2500 hectares of grasslands and encouraging local people and farmers to take up grazing in those areas. Already by 2017, restoration activities, vastly changing landscapes and awareness-raising activities have had considerable impact both on the public knowledge about the value of grasslands, as well as on economic and lifestyle choices among local people. Implementation of the infrastructure necessary for grazing (fences, animal drinking places and shelters, gates), coupled with the support system for managing semi-natural areas have created incentives for local farmers to increase their livestock and move animals from cultural grasslands to restored Alvars. By 2017, following the restoration of the first 1500 hectares of traditional grassland landscapes, 270 head of cattle and 400 sheep were added to the herds of local farmers. In addition, four families moved back to the countryside and changed their profession to livestock farmers. Open Alvar grasslands have great aesthetic, cultural heritage and recreational value and several nearby tourism facilities noted the positive effect of grassland restoration on their activities, by boosting visitor numbers and by increasing the opportunities on offer for scenic nature tours (Prangel, 2017).

End of Box 4.2

4.5.1.7 Effectiveness of protected areas

Designated conservation areas are highly important in safeguarding biodiversity and nature's benefits to people, but there is considerable evidence that protected areas alone cannot prevent biodiversity loss (e.g. Mora & Sale, 2011), particularly for migratory species (e.g. Saiga antelope; Bull *et al.*, 2013) or habitats or species particularly sensitive to environmental change (Bull *et al.*, 2013; Dudley *et al.*, 2014; Strayer & Dudgeon, 2010). A global systematic review shows that individual protected areas were effective at protecting habitats, particularly forests, but less effective at conserving populations of species (Geldmann *et al.*, 2013). There is great variation across Europe and Central Asia in the efficacy of formally protected areas for biodiversity conservation. Recent evaluations of the European Union Natura 2000 network of protected areas found it to be effective in providing coverage to most species listed in Annex II of the Habitats Directive (http://ec.europa.eu/environment/nature/legislation/fitness_check/docs/nature_fitness_check.pdf).

Natura 2000 sites do not only therefore serve the purpose of protecting Annex 1 (Birds Directive) and Annex 2 (Habitats Directive) species, but also protect certain more common (non-Annex) species, in particular breeding birds and butterflies, but less so amphibians and reptiles (van der Sluis *et al.*, 2016). However, the Natura 2000 network is not completely effective because there are exceptions for particular taxa (e.g., Zehetmair *et al.*, 2015a, 2015b) or in the way different member States implement it. For example, certain species (e.g. those dependent on traditional land management) or ecological zones (e.g. lowland versus upland) were either over- or under-represented, and gaps exist in the protection of biodiversity in certain habitats (e.g. marine and temporary freshwater habitats) (Gruber *et al.*, 2012; Maiorano *et al.*, 2007; McKenna *et al.*, 2014; van der Sluis *et al.*, 2012).

Natura 2000 represents one of the most systematic efforts for developing new protected areas, but the effectiveness of implementation of Natura 2000 for biodiversity conservation has not been sufficiently evaluated (Gaston *et al.*, 2008). A major challenge for forest protected areas is that current conditions are not pristine due to past management and suppression of natural disturbance processes (Hedwall & Mikusiński, 2015; Lõhmus *et al.*, 2004; Meyer *et al.*, 2011). Protected areas also tend to be too small to accommodate the full range of natural processes and hence unable to maintain sufficient ecological memory to re-organize after disturbances (Bengtsson *et al.*, 2003). In some forest reserves, “natural” state is contingent on traditional management (e.g. livestock grazing, coppicing, pollarding or small-scale felling). The introduction of such methods may be needed to secure forest biodiversity, but is so far rarely implemented mainly for economic reasons (Bernes *et al.*, 2015; Götmark, 2013; Sebek *et al.*, 2013). Measures to improve environmental status within conservation areas, combined with landscape-scale approaches that improve matrix quality for native biodiversity, are therefore urgently needed if their efficiency is to be improved.

The degree of monitoring and enforcement of protected areas can be critical for their efficacy in protecting biodiversity (e.g. Wendland *et al.*, 2015). For instance, almost 40% of the protected areas in the Barents Euro-Arctic region remain vulnerable to disturbance from human activities, including logging, mining, drilling and construction (Aksenov *et al.*, 2014). The efficacy of protection often varies among countries. For example, protected forest areas in the eastern Carpathians have proved effective at halting illegal logging in Poland and Slovakia but have been less so in Ukraine (Kuemmerle *et al.*, 2007) and protection in one country can lead to displacement of adverse impacts to adjoining territories (Mayer *et al.*, 2006). In certain circumstances, proximity to humans can sometimes affect the efficacy of protected areas in conserving biodiversity. For example, in Kyrgyzstan proximity of villages to protected areas was linked to a lowering of effectiveness for conserving non-ungulate large mammals (McCarthy *et al.*, 2010). In eastern Russia, Siberian tiger survival was inversely linked to roads bordering or crossing the strictly protected Sikhote-Alin State Biosphere Zapovednik (Kerley *et al.*, 2002). In summary, the proportion of protected areas is an important indicator of conservation efforts, although it needs to be combined with other indicators to fully assess the efficacy of measures aiming to conserve biodiversity (e.g. management plans, restoration actions, population indices of target species etc.).

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. There is a progress on those targets in the European Union, but in insufficient rate: 16% of the habitats assessments are favourable, 4% are unfavourable, but improving, 33% are unfavourable and stable, 30 % are unfavourable and deteriorating, 10% are unfavourable with unknown trend and 7% are unknown (European Commission, 2015). The network of Natura 2000 sites has progressed and is largely completed for terrestrial habitats, since 2010 it has grown for 1,4% and covering in 2015 18,1% of the European Union land. Overall, the mid-term assessment indicates the progress as the one with an insufficient rate (European Commission, 2015). Therefore, the European Union Biodiversity Targets 1 and 2 may not to be fully met by 2020 if the rate of the progress remains at the current level.

Regarding the Aichi Biodiversity Targets, reaching Target 11 ("protected areas increased and improved") for terrestrial ecosystems implies an increase in terrestrial protected areas, with an increased focus on representativeness and management effectiveness (Leverington *et al.*, 2008). A focus on representativeness is crucial as current protected area networks have gaps, and some fail to offer adequate protection to many species and ecosystems. These gaps include many sites of high biodiversity value such as Alliance for Zero Extinction sites and Important Bird Areas (Butchart *et al.*, 2010; Ricketts *et al.*, 2005). The global data sets statistically prove the progress in the increased coverage of protected area and sites of significance that ensure ecosystems connectedness in Europe and Central Asia. The data includes the World Database on Protected Areas (WDPA) (UNEP-WCMC & IUCN, 2014). Data on sites contributing significantly to the global persistence of biodiversity, or "key biodiversity areas" (KBAs) are provided by BirdLife International (2017) for Important Bird & Biodiversity Areas and by the Alliance for Zero Extinction sites holding the entire population of at least one highly threatened species (Brooks *et al.*, 2006; Ricketts *et al.*, 2005).

4.5.2 Trends and indirect drivers of changes in agricultural land use

4.5.2.1 Trends in agricultural land use

Agricultural land-use changes are constrained and driven by biophysical conditions and sets of inter-related indirect drivers (e.g. policies, political changes in Eastern Europe and Central Asia, demands for food and ecological products etc.). Across the region there are two principal trends: (1) intensification of conventional agriculture; and (2) decreasing land-use intensity and abandonment of conventional agricultural land.

Trend 1: Intensification of conventional agriculture.

This model of agricultural production is characterized by large-scale monocultures specializing on few crops that are supported by high levels of agrichemical inputs or irrigation, management of high livestock densities, and mechanization to increase production (Foley *et al.*, 2005; Goldewijk, 2001; Henle *et al.*, 2008; Robinson & Sutherland, 2002; Tilman *et al.*, 2001; Tscharntke *et al.*, 2005; van Vliet *et al.*, 2015; EEA, 2015a). Agricultural intensification has been a dominant driver of land-use changes in Europe and Central Asia since the 1950s (Jepsen *et al.*, 2015; EEA, 2015a). The area of agricultural holdings and their role in the agricultural sector are constantly growing (BEFL, 2016; Petrick *et al.*, 2013; Visser *et al.*, 2012; Visser *et al.*, 2014). They are especially large in the most favourable regions for agriculture. For example, in Russia 43 companies cultivate in total 10.4 million hectares (BEFL, 2016). Although the temporal and spatial patterns and intensity of agricultural land use vary considerably within the region, intensive agriculture is expected to remain among the prevailing land-use practices in the region into the future (Jepsen *et al.*, 2015).

In Western Europe, conventional intensive agriculture has been the prevailing model of agricultural production since the 1950s (EEA, 2015a). This has led to considerable landscape homogenization (Curado *et al.*, 2011; Stoate *et al.*, 2009). Landscape homogenization also occurred in many parts of Central Europe during the socialist period (1945 – 1990) (Fraser & Stringer, 2009; Munteanu *et al.*, 2014). In Eastern Europe and Central Asia, Russia, Ukraine and Kazakhstan, conventional agricultural intensification happened mainly after the break-up of the USSR in 1991; and these three countries became major exporters of agricultural products products (Liefert *et al.*, 2009). Currently, Russia is among the major world grain exporting nations due to an increase of land-use intensity and partial re-cultivation of abandoned lands after 2000 (Medetsky, 2016).

In Central Asia irrigated agricultural areas have increased at the expense of natural pastures since the 1960-1970s, for instance in the vicinity of the Syr Darja and Amur Darja rivers. This is mainly due to cotton production (Kaplan *et al.*, 2014), which has doubled since the 1960s and now accounts for nearly half of all irrigated arable land. Irrigation is currently used for 33% (13 million ha) of cultivated areas in Central Asia. However, poor maintenance of drainage systems has resulted in millions of hectares of irrigated areas suffering from salinization and waterlogging. In Uzbekistan 51% (2.1 million ha) and Turkmenistan 68% (1.3 million ha) of irrigated areas are salinized and further widespread degradation of agricultural land is expected in these countries (Frenken, 2013; Horion *et al.*, 2016).

Trend 2: Decrease of land-use intensity and abandonment of conventional agricultural land

Agricultural land abandonment is widespread in Europe and Central Asia (Benayas *et al.*, 2007; Díaz *et al.*, 2011; Prishchepov *et al.*, 2013; Wang *et al.*, 2016). The available evidence indicates that it is reasonable to expect that farmland abandonment will continue over the next few decades, particularly in the case of extensively grazed lands (Keenleyside & Tucker, 2010). However, some projections of land-use change are limited by a lack of appropriate data on historical legacies, local conditions and drivers (Biró *et al.*, 2013; Feranec *et al.*, 2010; Hatna & Bakker, 2011; Temme & Verburg, 2011). Overall, the largest abandonment extent is in the East European forest steppe and Pontic steppe zones, in Sarmatic mixed forests and in the boreal zone (Schierhorn *et al.*, 2013).

Agricultural land abandonment leads to complete termination of agricultural activity and reforestation through silviculture or natural succession to shrubs and forest (Alcantara *et al.*, 2012; Baldock *et al.*, 1996; Baumann *et al.*, 2011; van der Zanden *et al.*, 2017). For example, in Western and Central Europe an increase of forest and semi-natural habitats after abandonment of agricultural land occurred widely in Italy, Hungary, Poland and Germany and to a lesser extent in France and Greece, while in Spain the transition was in the opposite direction (Petersen, 2006). Agricultural land abandonment has tended to be concentrated in areas that are marginal for agriculture, for example, on unproductive soils and areas limited by other biophysical conditions (temperature, high precipitations etc.) (Ioffe, 2005; Meyfroidt *et al.*, 2016; Prishchepov *et al.*, 2013, 2016). Abandoned farmland was converted to urban residential areas or infrastructure in some places or, more often, became forested or afforested (Grădinaru *et al.*, 2015; Plutzar *et al.*, 2015; Schierhorn *et al.*, 2013).

In the European Union cropland area has decreased by almost 1.2 million hectares in recent decades (Dixon *et al.*, 2009; Grădinaru *et al.*, 2015; Munton, 2009). In Central European countries, including Poland, Czech Republic, Slovakia, Hungary, Romania and Bulgaria, during the 1990s and 2000s the prevailing land-use trend was abandonment of arable land and grassland, reductions of livestock densities and agrochemical use and reforestation (Biró *et al.*, 2013; Sutcliffe *et al.*, 2013). For example, in Poland 17.6%, in Estonia 10.1% and in Latvia 21.1% of agricultural land was abandoned by 2002 (Keenleyside & Tucker, 2010). However, expansion of the European Union and implementation of its Common Agricultural Policy in new member States has resulted in the reclaiming of abandoned farmland for intensive agriculture – a trend that is likely to continue (Keenleyside & Tucker, 2010; Kuemmerle *et al.*, 2009; Sutcliffe *et al.*, 2013).

Eastern Europe and Central Asia have been hotspots of cropland abandonment since the 1990s (Keenleyside & Tucker, 2010; Kuemmerle *et al.*, 2009; Sutcliffe *et al.*, 2013). The collapse of the socialistic collective farming system resulted in the abandonment of more than 58 million hectares of former croplands in Russia and Kazakhstan (Kurganova *et al.*, 2015) (**Figure 4.9** and **Figure 4.10**). This was mirrored by substantial reductions in livestock (e.g. >30% reductions from 1990 levels in cattle densities in 2005 and 2015) (Chibilyov, 2016; Lescheva & Ivolga, 2015; Rosstat, 2017). In Kazakhstan and in stock farming steppe regions of Russia the collapse of livestock populations and state farms were combined with the private acquisition of former state assets, including livestock (Kerven *et al.*, 2016; Robinson *et al.*, 2016; Suleimenov & Oram, 2000). Livestock declines of up to 80% in sheep and cattle took place in Kazakhstan (Kamp *et al.*, 2011), creating a vast area of un-grazed grasslands (Kerven *et al.*, 2016). Grazing patterns changed significantly, and intensive grazing became restricted to areas around villages, which have been rapidly degrading due to overgrazing (Kamp *et al.*, 2011, 2012; Kandalova & Lysanova, 2010; Kitov & Tsapkov, 2015; Kühling *et al.*, 2016; Morozova, 2012; Suleimenov & Oram, 2000).

Figure 4 9 Decrease in sown area from 1990 to 2013 across Russia, Ukraine and Kazakhstan. Source: CISSTAT (2017).

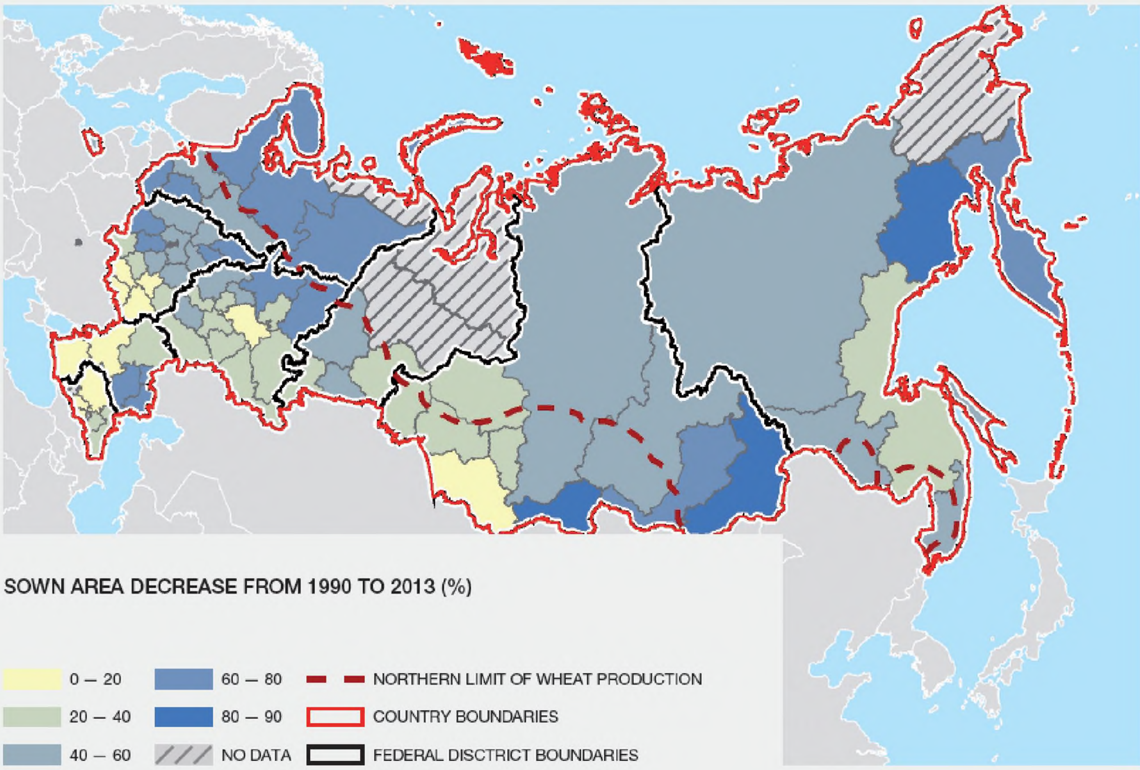
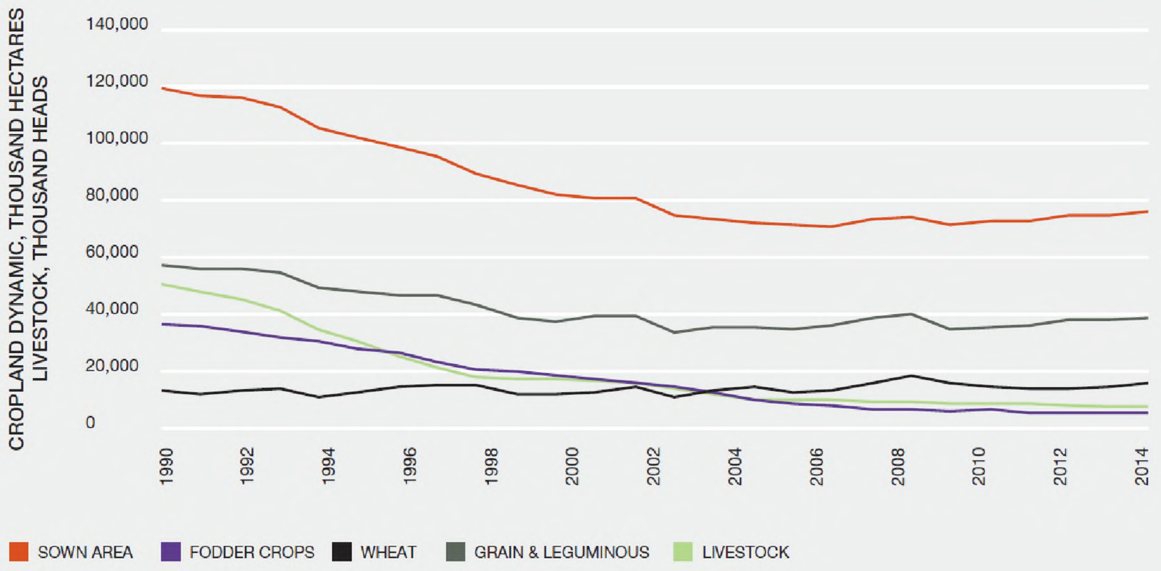


Figure 4 10 Change in cropland areas and livestock numbers in Eastern Europe and Central Asia. Source: Data from CISSTAT (2017); ESTstat (2017); LATVstat (2017); LITHstat (2017).

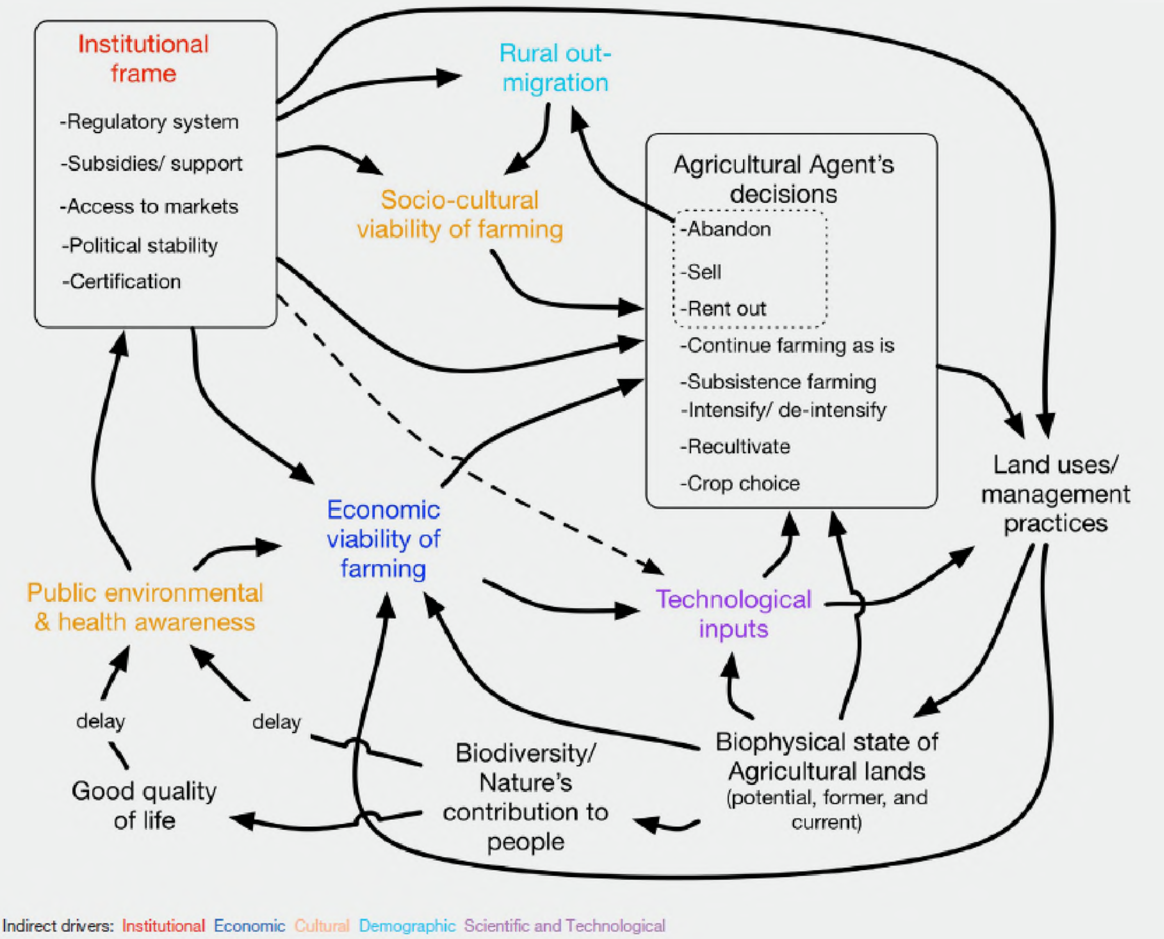


4.5.2.2 Indirect drivers of trends in agricultural land use

Changes in agriculture are driven by multiple interconnected institutional, economic, cultural and technological drivers (**Figure 4.11**).

Figure 4 11 An illustration of the key drivers and systemic interconnections dominating agricultural land-use change in Europe and Central Asia. Source: Own representation.

The specific ensemble of land uses and agricultural practices employed on a given farm are largely the aggregate of a set of decisions made by the agricultural agent, which in turn are shaped by the economic and socio-cultural viability of farming in the region, the availability of technological inputs, as well as the institutional frame in which agriculture is situated. These inter-relationships are further unpacked in a series of sub-models below (see further figures).



4.5.2.2.1 Institutional drivers of trends in agricultural land use

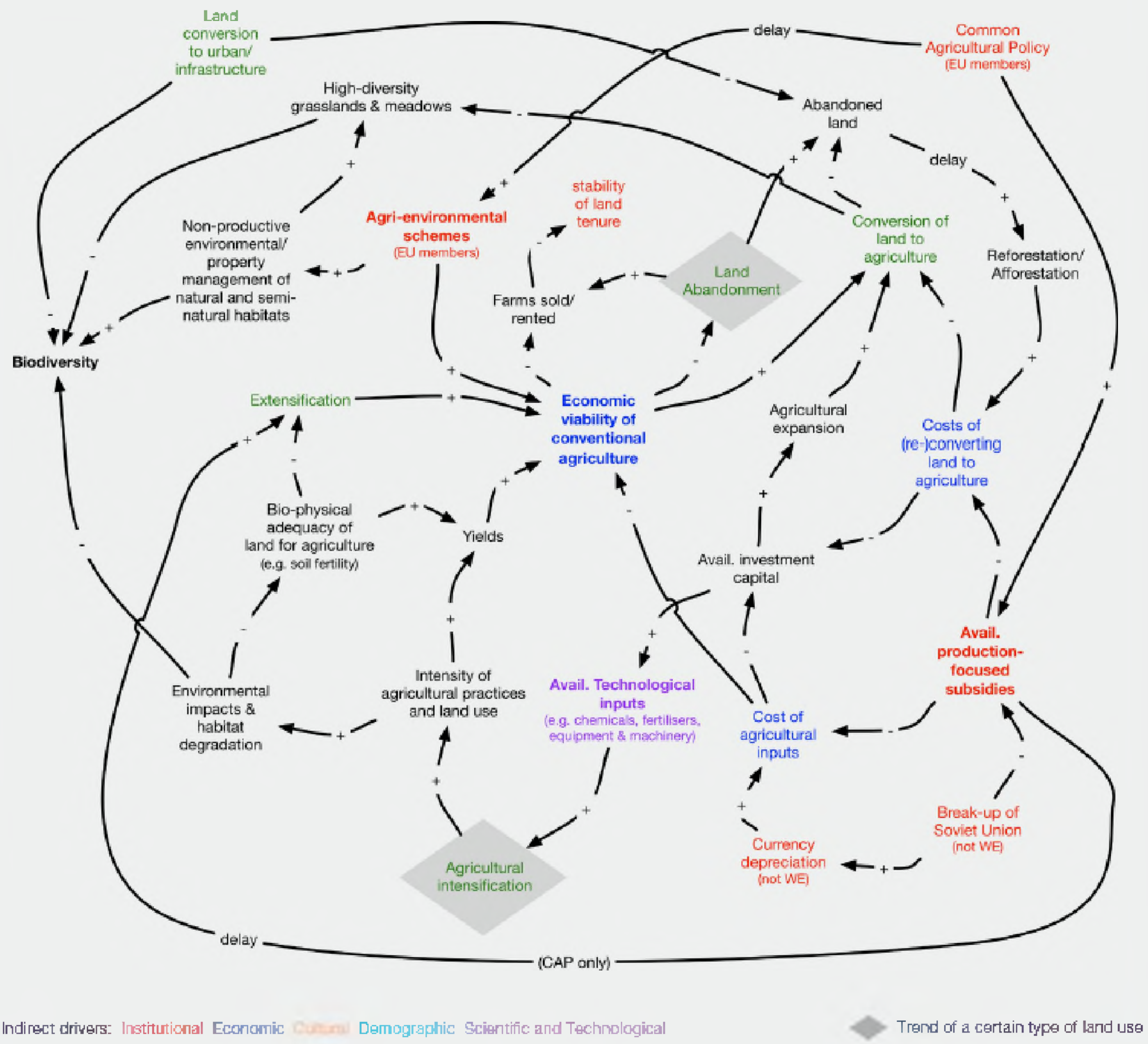
Until the 1980s, the European Union Common Agricultural Policy became the major incentive for the conventional intensification of agriculture (Van Zanten *et al.*, 2014) through a subsidy scheme based primarily on price pegging. Since 1992, it has increasingly been adapted to better serve the aims of sustainability by means of a fundamental reform process designed to move away from a policy of price and production support to a policy of direct income aid and rural development measures, including agri-environmental schemes. The 2003 Common Agricultural Policy reform (Council Regulation (EC) No. 1782/2003) brought forward environmental concerns in agriculture. It reinforced a number of measures that encourage land use and practices compatible with the protection of environmental resources. Agri-environmental schemes became compulsory for every member State.

From the 1990s, in Central Europe radical changes in political, social and economic systems brought about the restitution of private property and the land market with consequential economic drivers.

State support diminished, former export markets within the socialist sphere of influence disappeared, prices were liberalized, and farmers suddenly faced strong external competition even though they often lacked the necessary inputs (e.g., fertilizer) and technology (e.g., access to machinery) to sustain high yields (Lerman *et al.*, 2004; Rozelle & Swinnen, 2004; Skokanová *et al.*, 2016). These politico-economic drivers instigated further agricultural intensification in fertile regions, and abandonment of less fertile or less accessible land (Fonji & Taff, 2014; Jepsen *et al.*, 2015; Skokanová *et al.*, 2016; van Vliet *et al.*, 2015). During the transition from planned to market economy the agricultural cooperatives were dismantled and much of their land was privatized to new owners or re-privatized to the former owners, which led to establishment of numerous smallholder farms. Many smallholders had no interest or knowledge, or adequate financial resources and equipment to profitably cultivate the agricultural land and thus quit farming or resorted to subsistence farming on small parcels of land scattered across the landscape (Biró *et al.*, 2013). The rapid privatization (Skokanová *et al.*, 2016), ownership insecurity (e.g. in Romania, see Kuemmerle *et al.*, 2009), and a lack of interest or knowledge in agriculture of the new landowners resulted in large-scale land abandonment and decreased management intensity in large areas of Central Europe (Liira *et al.*, 2008; Palang & Printsmann, 2010; van der Sluis *et al.*, 2015; van Vliet *et al.*, 2015). After 2004, when many Central European countries joined the European Union, the land tenure system stabilized through the introduction of the European Union Common Agricultural Policy, which has helped to restore farming activities in many areas, especially mountain regions, and has stabilized agricultural development in some countries of Central Europe (Bezák & Mitchley, 2014; Ruskule *et al.*, 2013; van der Sluis *et al.*, 2015). Agricultural subsidies introduced with the accession to the European Union increased the economic viability of agricultural land, leading to agricultural expansion and intensification. Agricultural subsidies, however, also caused problems as they enhanced regional inequality by excluding small-scale farmers in remote areas (Bezák & Mitchley, 2014) or by causing damage to areas of conservation interest (e.g. ploughing high-diversity grasslands and meadows) (**Figure 4.12**).

Figure 4 12 The availability, or unavailability, of agricultural subsidies has been a major driver of agricultural land-use change in Europe and Central Asia. Source: Own representation.

Production-based subsidies have generally led to the intensification of farming. The sudden unavailability of subsidies in large parts of the region following the dissolution of the Soviet Union was a major driver of agricultural land abandonment, as well as intensification on the most fertile soils. The introduction of non-production-based subsidies, for example agri-environmental schemes in the European Union, has improved the ability of small-scale farmers to maintain lower-intensity agricultural practices with long-term benefits for biodiversity.



In Eastern Europe and Central Asia, the political changes since the 1990s were accompanied by radical large-scale land reforms, involving the elimination of the state monopoly and division of land ownership (state, collective, and private) (Lerman *et al.*, 2004; Liefert & Liefert, 2012; Liira *et al.*, 2008; Rozelle & Swinnen, 2004; Smelansky, 2003; Swinnen *et al.*, 2017) (**Figure 4.12**). Since then, the areas of large private agricultural companies owned by agro-holdings and their role in the agricultural sector has constantly expanded (BEFL, 2016; Nefedova, 2016; Petrick *et al.*, 2013; Visser *et al.*, 2012, 2014), especially in the most favourable regions for agriculture, (e.g., south-western Russia, south-eastern Ukraine, and Kazakhstan). However, subsistence farming has played the main role in the food security of citizens in villages, towns and cities. For example, in Russia, the economic crisis in 2008 led to a 2-fold increase in the number of rural residents engaged in subsistence farming, and in the cities the number increased by 2.8 times. Subsistence farms produced 98% of potatoes, vegetables and fruit crops, 82% of milk, 68% of meat and 54% of eggs (Martyn & Yevsiukov, 2009; Swinnen *et al.*, 2017). Appropriate legislation is important for biodiversity conservation on agricultural lands. However, currently for the majority of Eastern European countries there is a lack of links between environmental legislation and legislation related to land, territorial development and agriculture (Smelansky, 2003). For example, Russian legislation does not identify grasslands as a separate category of agricultural land. Several legal regulations address pastures (grazing lands) and hay-making lands (Bakirova, 2011; Smelansky, 2003; Smelansky & Tishkov, 2012); the federal law provides some legal framework for constraining grassland transformation into other land uses but it has been insufficient to protect grasslands outside protected areas.

Land ownership is another institutional driver that has changed across Europe and Central Asia from having many small landowners in the agricultural sector to increasingly larger areas of land being managed by fewer farmers – either after being purchased by farmers or based on an increase in rented land. For the latter, concern is raised that managers' connection with, and sense of responsibility to the land is decreasing, especially in the case of short term rental agreements. This may result in poor management, including less environmental considerations (Forbord *et al.*, 2014; Lobley & Potter, 2004; Stokstad, 2010). This problem is especially vital in some countries of Central Europe (e.g. in the Czech Republic) where the original small owners sold or leased their land to new owners from elsewhere after restitution in the 1990s (Skokanová *et al.*, 2016).

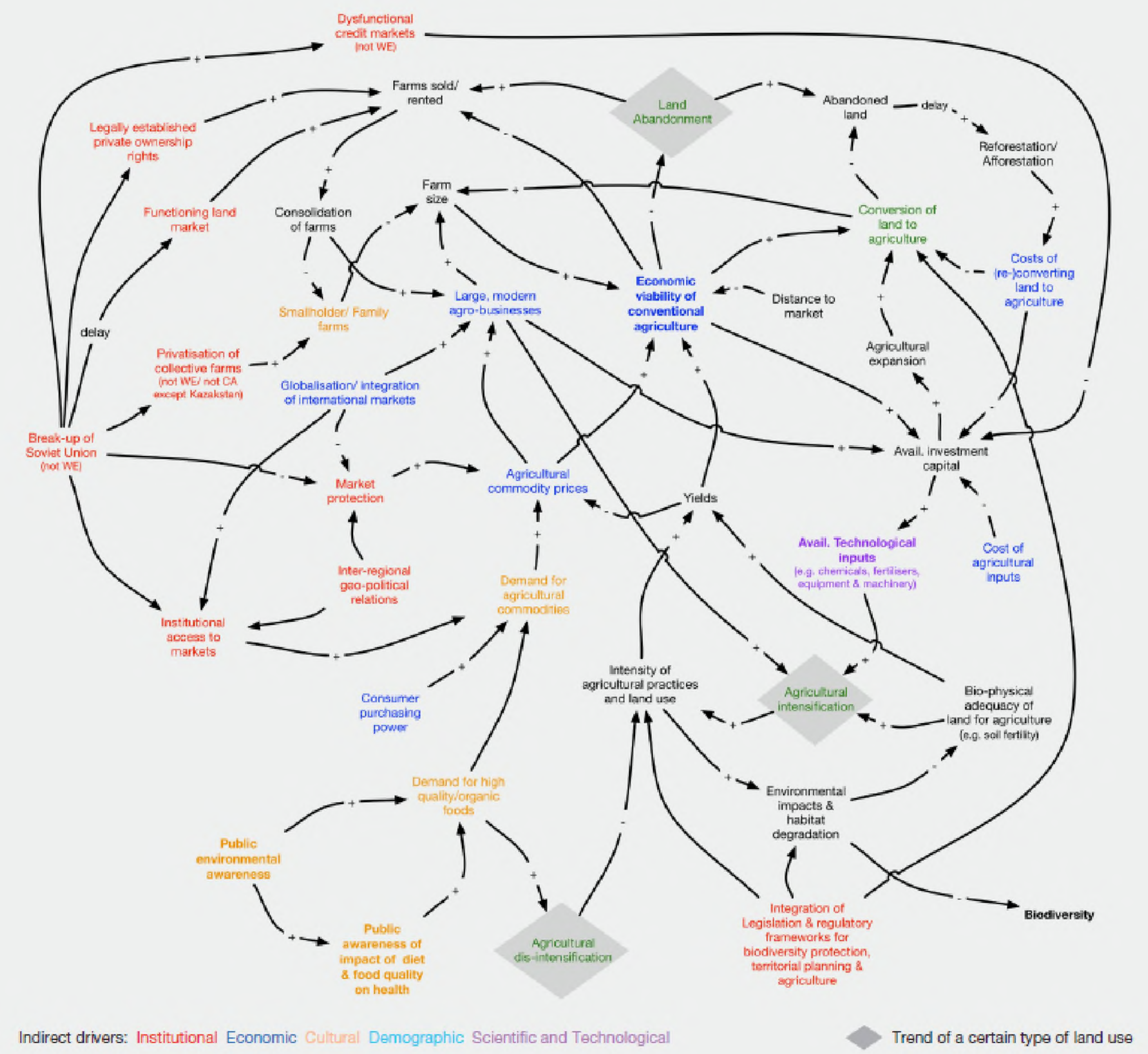
4.5.2.2.2 Economic drivers of trends in agricultural land use

Economic factors often underpin decisions about cultivation or termination of agricultural production (**Figure 4.12**). Agricultural expansion and intensification on fertile, productive land often coincides with land abandonment on marginal land (Beilin *et al.*, 2014; Skokanová *et al.*, 2016). These two trends are largely driven by global trade of the agricultural market since the 1950s (van Vliet *et al.*, 2015). As a result of global trade, the size of farms and their specialization have increased (Lobley & Butler, 2010).

In Eastern Europe and Central Asia, economic factors include prices for agricultural products (outputs), and the parity of the prices between inputs and outputs. This was most likely one of the primary reasons for widespread termination of farming and livestock production in these subregions (Rozelle & Swinnen, 2004; Schierhorn *et al.*, 2016) (**Figure 4.13**). In some countries of Eastern Europe and Central Asia, an inability to fill budget gaps led governments to reduce subsidies for agricultural production and consumption by 95% (Prishchepov *et al.*, 2013). The agricultural sector, and particularly livestock sector, immediately faced a mismatch between increased prices for inputs and output production (Sedik, 1993). Additionally, the removal of subsidies led to domestically produced beef and milk becoming non-competitive compared to subsidized imported goods (Schierhorn *et al.*, 2016). Lack of cash flow to cover production costs led to a reduction in livestock numbers and concomitant reduction in fodder crop production (Prishchepov *et al.*, 2017; Rozelle & Swinnen, 2004; Schierhorn *et al.*, 2013; Sedik, 1993) which resulted in widespread agricultural land abandonment (Ioffe *et al.*, 2004; Liefert & Liefert, 2012; Prishchepov *et al.*, 2012; Schierhorn *et al.*, 2013). The wheat production sector was also affected, but to a lesser extent than other grain and fodder crops. Maize, sunflower and beets continued to be cultivated at almost the same levels as in 1990. Availability of the domestic market and accessibility to international markets, distances to the markets and transportation costs may have determined the decision to abandon or re-cultivate agricultural land (Prishchepov *et al.*, 2013). For instance, discovery of new markets for Russia's wheat most likely stimulated partial re-cultivation in the Russian south, in proximity to water ways and sea ports (FAO, 2009). Similarly, growing demand in the Chinese market triggered re-cultivation of abandoned lands for soya production in the Amur region of Russia (Rosstat, 2016). The economic advantages or disadvantages of the agricultural sector compared to other sectors, may drive the decision to quit farming or to pursue alternative income sources. For instance, the value added by agriculture to total GDP declined by 32% from 1990 to 2000 (Prishchepov *et al.*, 2017). Additionally, low taxation, which was based on normative average yields during the Soviet era, did not stimulate either the cultivation of lands, or a concentration on yield increases. Land transaction costs and legal burdens themselves preclude fast transactions and limit incentives for the re-cultivation of abandoned land (Meshkov, 2014; Uzun, 2011).

Figure 4 13 **Land, credit and commodity markets have had important impacts on the economic viability of conventional agriculture across Europe and Central Asia, and as such have been key factors in regional intensification and abandonment trends. Source: Own representation.**

The dissolution of the Soviet Union was a source of considerable institutional turmoil affecting the function of these markets in many countries in the region. The recent trend towards agricultural de-intensification in some parts of Europe and Central Asia is largely driven by growing public environmental and health awareness.



4.5.2.2.3 Cultural drivers of trends in agricultural land use

Socio-cultural attributes of individual farmers have had a bearing on the extent of land-use intensification or change, thereby slowing down the effects of specialization and global trade (Lobley & Butler, 2010). "Property management" or property-related issues play a vital role in the farmer's practice; and values related to family and individual strategies may often explain why landowners undertake land-use changes that are not profitable (Kristensen, 2016) (**Figure 4.14**).

In Western Europe, farm and farmer characteristics have been particularly important drivers of specialization (Breen *et al.*, 2005; Gorton *et al.*, 2008; van Vliet *et al.*, 2015). Choices of crops and farming systems are largely controlled by economic and legal factors (markets and state subsides/regulation); however, local traditions may still moderate the rate of change (Beilin *et al.*, 2014; Curado *et al.*, 2011; Elmhagen *et al.*, 2015; Forbord *et al.*, 2014) (**Figure 4.14**). Ethical and cultural trends have gradually brought changes in diet and food consumption as well as leisure activities. There is a growing number of consumers who are particularly interested in how and where food was produced and, sometimes, in participating in the production process (Guarino *et al.*, 2015). Additionally, there is also a growing interest in leisure farming and hobbyhorses. Quality products, related to high natural value farmlands and often linked to community-led local development policies, and in 2010 reached a substantial weight, worth about 18% (excluding wine) of the gross saleable production in the European Union agricultural sector (http://ec.europa.eu/agriculture/quality/schemes/index_en.htm). At the same time, in Western and Central Europe, organic products comprise about 1% of total food sales, but trends are increasing (FiBL, 2015).

In Central Europe since the 1990s many livestock farms have collapsed due to termination of subsidies for external inputs (e.g. fertilizers) for fodder production during the socialist period. This resulted in the large-scale movement of people with agricultural experience into cities or abroad (Bell & Muhidin, 2009), leading to widespread abandonment of farmland (Munteanu *et al.*, 2014; van Vliet *et al.*, 2015).

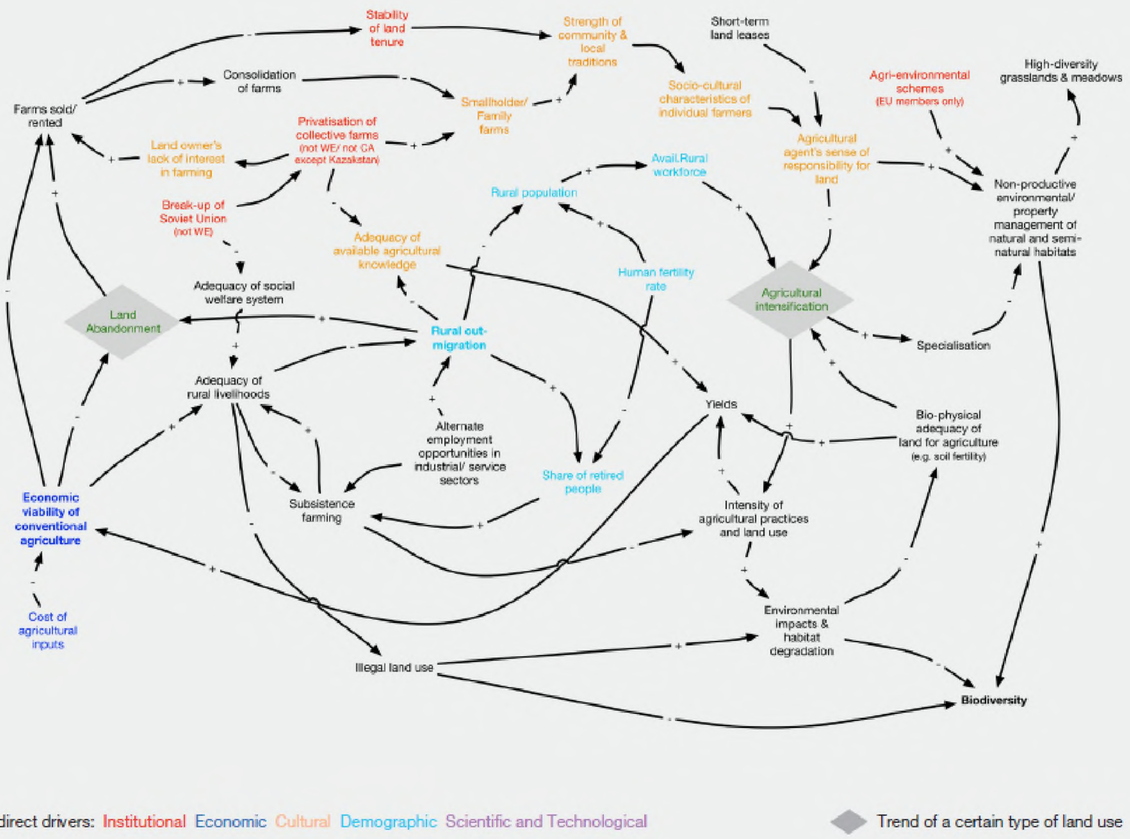
In Eastern Europe and Central Asia during the Soviet era, agricultural enterprises fulfilled many obligations related to providing jobs and services to local population (e.g., schools, shops, centres of culture, libraries etc.) (**Figure 4.14**). Since the 1990s these obligations have been transferred to local governments, which have not had resources to fulfil them (Ioffe *et al.*, 2012). This led to a sharp increase in the burden on the biological resources of rural areas (e.g. through poaching and illegal logging); destructive extraction of soil and mineral resources (e.g. through sale of fertile topsoil and illegal mass extraction of building materials and coal); as well as growing poverty in rural areas (Allina-Pisano, 2007; Ovcharova & Pishnyak, 2003; Petrick *et al.*, 2013; Visser & Schoenmaker, 2011). Since the 1990s a large proportion of agricultural land has been freely transferred to multiple private owners who had a share in the property of former collective farms. This has led to the appearance of a significant number of non-agricultural enterprises (Lerman & Shagaida, 2007; Petrick *et al.*, 2013; Shagaida, 2005) operated by managers often with a lack of adequate professional knowledge in agriculture (Maslak, 2015; Sabluk *et al.*, 2015). For example, in Ukraine land reform has led to the privatization of 12,000 collective or state farms; and the majority of the agricultural land (27 million ha, 66% of all agricultural land of the country) was distributed among 6.9 million citizens (http://land.gov.ua; Khodakivs'ka, 2015). This has created a precondition for widespread land abandonment.

In general, quantitative studies confirm that agricultural land abandonment is strongly linked to a decrease in rural population density, ageing population, and lower birth rates (Ioffe *et al.*, 2004; Meyfroidt *et al.*, 2016; Prishchepov *et al.*, 2017). Demography legacies also played a crucial role in explaining the patterns of land abandonment, such as reduced population due to World War II in western Russia (e.g., Smolensk province), and outmigration in the 1960s and 1970s from the non-Chernozem region (Ioffe, 2005; Prishchepov *et al.*, 2013). It has been proposed that agricultural production looses its economic feasibility when rural population density falls below five people/km^2

(Ioffe *et al.*, 2004). The regions with higher birth rates and higher population density were found to be more favourable for re-cultivation (Meyfroidt *et al.*, 2016; Shagaida, 2005).

Figure 4 14 **Agricultural intensification and abandonment trends in Europe and Central Asia are also influenced by socio-cultural and demographic factors. Source: Own representation.**

Whilst the consolidation of farms has led to some improvements in economic viability, it is also linked to the erosion of local traditions and of a sense of long-term custodial responsibility for the land, which are important for the continuation of non-productive management practices for biodiversity. Rural outmigration is also linked to the loss of key forms of agricultural knowledge.



4.5.2.2.4 Technological drivers of trends in agricultural land use

Technological drivers such as biotechnology and mechanization (e.g. tractors) are important drivers of change in the agricultural sector (Jepsen *et al.*, 2015). Better production technology, for instance application of high power tractors and other machinery, may encourage farmers to cultivate more land, thus stimulating re-cultivation of abandoned plots. However, improvement of technological production can also be strongly influenced by whether economic factors favour investment in technological advances (Jepsen *et al.*, 2015).

4.5.3 Trends and indirect drivers of changes in forestry

4.5.3.1 Trends in forestry

Forest management systems vary across Europe and Central Asia. In the boreal zone forest management with clear-cuts followed by intensive silviculture dominates in Fennoscandia (Granhus *et* *al.*, 2015), and wood mining without silviculture in boreal Russia (Naumov *et al.*, 2016) (**Figure 4.15** and **Figure 4.16**). Forestry in the temperate zone utilizes a wider spectrum of management systems. This includes different harvest and regeneration systems ranging from clear-cut management with tree plantations to continuous cover forestry with single-tree harvest and natural regeneration (Kuuluvainen *et al.*, 2012; Pommerening & Murphy, 2004). Almost all forest management systems result in simplified forests with loss of structural complexity and biodiversity at multiple spatial scales. In the Mediterranean, agroforestry systems are widespread, which incorporate combinations of trees, grasslands and rotation cereal cropping. In Western, Central and Eastern Europe, traditional agroforestry systems have been key elements in the European cultural landscapes throughout history (Eichhorn *et al.*, 2006; Erixon, 1960) (**Figure 4.17**). As an example, the Spanish dehesas and Portuguese montados form extensive agro-silvo-pastural savannahs, which cover about 5 million ha in south-western Spain and Portugal (Joffre *et al.*, 1988; Plieninger *et al.*, 2003).

Figure 4 15 **Even-aged A Scots pine (*Pinus sylvestris*) and B Norway spruce (*Picea abies*) forests with simplified vertical and horizontal structures are the outcomes of forest management systems aiming at maximum sustained yield with even-aged silvicultural system (Bergslagen region, Sweden). Photo: Per Angelstam.**

Generally, forest management is based on silviculture using the gradient between even-aged and uneven-aged systems. There are three general types of age-class structures that are managed for: (1) even-aged systems that include clear-cutting or seed tree systems; (2) the intermediate double-cohort systems with shelterwood cutting, and (3) uneven-aged systems with single tree and group selection. The different systems can be understood better if considered as located in a continuum of proportion trees removed at each treatment and the size of the treatment unit.

A B

Figure 4 16 **Remaining intact forest landscapes in Western, Central and Eastern Europe are subject to on-going wood mining. Photo: Intact forests in the Komi Republic, Russian Federation. Marine Elbakidze.**

Initially single high value trees, and later entire stands in naturally dynamic forests are harvested without plans for future forest development. This leads to frontiers of wood felling that develop as market demands spread into increasingly remote regions. In Fennoscandia, this process began about 150 years ago and in remote parts of north-western Russia it began in the 1960s (Naumov *et al.*, 2016), and is still on-going (Potapov *et al.*, 2017).

Figure 4 17 **Dehesa and montado agroforestry systems integrate use of forest, grasslands, and fields (Pardo & Gil, 2005). Photo: Montado system in Portugal. Marine Elbakidze.**

These cultural landscapes host outstanding biodiversity (Bugalho *et al.*, 2011; Diaz *et al.*, 2013) and provide multiple contributions to people that enhance quality of life. The importance of traditional agroforestry landscapes has been recognized at the European Union level and the relevance of traditional practices to deliver multiple contributions of nature to people has been acknowledged (Bergmeier *et al.*, 2010; Eichhorn *et al.*, 2006; Marañón, 1988; Rackham, 2003). However, at present these landscapes are deteriorating due to farmland abandonment, intensification of agriculture or creation of forest plantations (Garrido *et al.*, 2017).

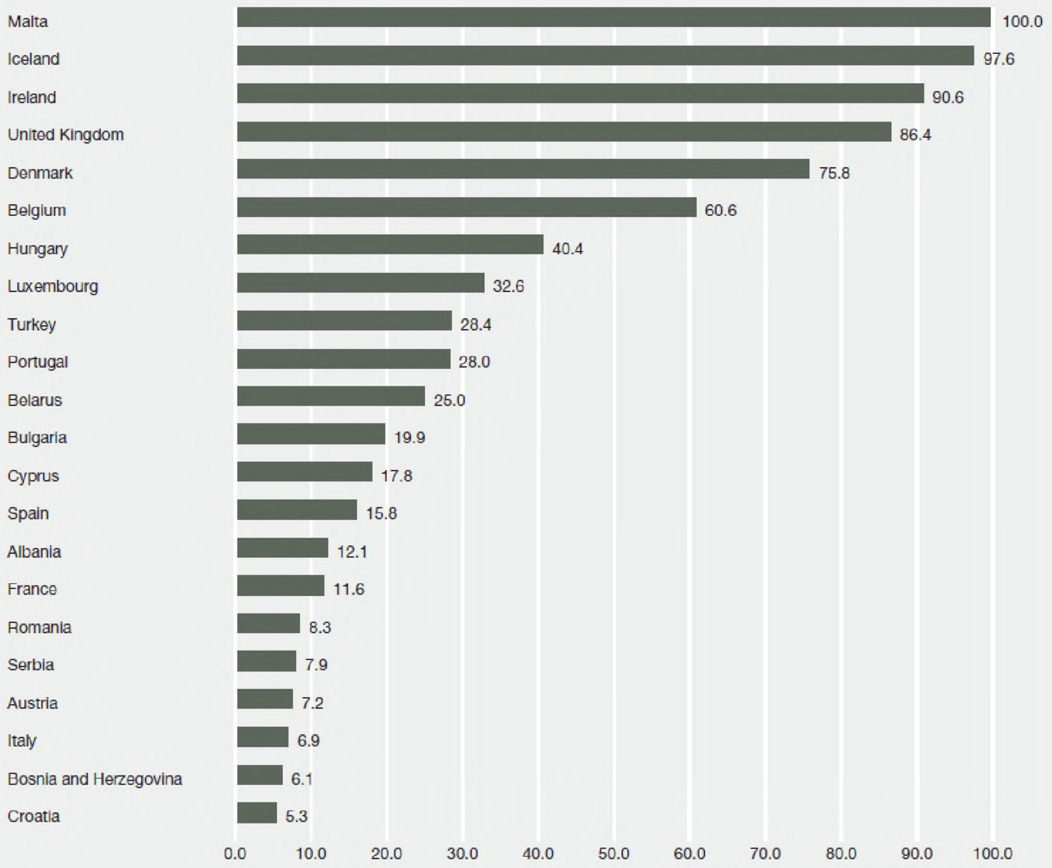
The main trends in forestry across Europe and Central Asia are as follows: (1) increasing intensity of management on forested land; (2) continued logging of intact forests; (3) rehabilitation of forest land after overgrazing, overexploitation, and desertification; and (4) efforts to implement sustainable forest management. These trends are assessed in more detail below.

Trend 1: Increasing intensity of management on forested land

Increasing intensity of management of forested land includes: (i) increasing extraction of bioenergy resources; (ii) increasing area of plantations; and (iii) intensification of forest management. Production of forest biomass for energy purposes includes increasing use of more intensive management methods and extraction of a larger fraction of biomass during harvest operations, including tree-tops, branches and roots (Bouget *et al.*, 2012). It is theoretically possible to increase the availability of forest biomass significantly beyond the current level of resource utilization (Verkerk *et al.*, 2011). Intensification of biomass removals from forests has raised concerns about its environmental impacts on forest productivity, biodiversity, soil quality, and climate change mitigation potential, as well as social values (Aherne *et al.*, 2012; Bouget *et al.*, 2012; Forsius *et al.*, 2016; Triviño *et al.*, 2015). The trade-offs between increasing biomass output and delivery of diverse contributions of nature to people are recognized as a major challenge for forestry in Europe and Central Asia (Verkerk *et al.*, 2011). These concerns have resulted in the development of sustainability criteria for bioenergy production (European Commission, 2009). However, several studies have pointed to the need to include landscape-scale segregated approaches to define appropriate indicators for long-term sustainability, including energy wood production (Fu *et al.*, 2013; Nelson *et al.*, 2009; Vihervaara *et al.*, 2015). This applies in particular to resolving potential impact on biodiversity, soil carbon, nutrient store and leaching (Forsius *et al.*, 2016), but also to forests as an asset for recreation and nature-based tourism.

Plantation forestry in Europe constituted 9% of the forested area in 2015 with an increase during the last 20 years of 3.8 million hectares (Forest Europe, 2015). The fraction of plantation forests varies among countries in Western Europe and Central Europe (**Figure 4.18**).

Figure 4 18 Western, Central and Eastern European countries with a share of over 5 per cent of plantations of the total forest area. Source: Forest Europe (2015).



The growing stock of forests in continental Europe has increased at an annual average of 1.4% or, in absolute terms, by 403 million cubic meters per year over the last 25 years (Forest Europe, 2015). Growing stock has increased despite a significant increase in annual felling. During the period 1990 – 2010, annual felling increased by more than 20% (from 216 to 263 million cubic meters) in Europe. Thus, only over half of the growth is harvested. Additionally, there are combined effects of increased CO_2 concentration and nitrogen deposition. Sweden and Finland are viewed as role models for the development of maximum sustained yield wood production in Europe (e.g. Elbakidze *et al.*, 2013a; Lindahl & Westholm, 2010). However, there are arguments that sustained yield forestry as a single-use management (Behan, 1990) focused on wood, changes forest composition and structure, and alters the natural dynamics in forest landscapes (Bawa & Seidler, 1998; Holling & Meffe, 1996; Luckert & Williamson, 2005). As a consequence, forest ecosystems lose native species, habitats, and ecological processes, which affect ecological integrity and resilience (Farrell *et al.*, 2000). The Russian Federation currently aims to increase the sustained yield of wood by intensifying wood production in accessible areas previously harvested by wood mining (Naumov *et al.*, 2017). This requires changes in forest management that include silvicultural methods, for example, scarification, planting or seeding, pre-commercial thinning and even fertilization (Elbakidze *et al.*, 2013a).

Trend 2: Continuous logging of intact forest landscapes

Industrial forestry has expanded throughout Europe over the centuries, basically from south-west to north-east (Lotz, 2015; Lundmark *et al.*, 2013). According to Potapov *et al.* (2017), industrial timber extraction, resulting in forest landscape alteration and fragmentation, was the primary global cause of intact forest landscape area reduction. Three countries comprise 52% of the total reduction of intact forest landscapes area: Russia (179,000 km^2 of IFL area lost), Brazil (157,000 km^2), and Canada (142,000 km^2). In Europe and Central Asia clear-cutting was the main intact forest landscape loss cause in the temperate and southern boreal zones (54%). Proportional to the year 2000 IFL area, the highest percentages of intact forest landscape area reduction were found in Romania (Central Europe), which lost all of its intact forest landscapes. Russia has approximately 20% of the world's forests, and human influence on forests has been growing over recent decades, mainly as a consequence of logging activities including both clear-felling and selective logging (Achard *et al.*, 2006; Naumov *et al.*, 2017). Easily accessible Russian forest resources are being exhausted (e.g., Naumov *et al.*, 2016). Despite a huge forested area there is a serious shortage of accessible wood resources demanded by the forest industry. Large sawmills, pulp and paper enterprises, especially those focused on output with low added value, are heavily reliant on low transportation costs for the delivery of raw materials from the forest. Thus, increasingly, forest logging companies harvest in protective forests and other valuable forests (Naumov *et al.*, 2017) that support biodiversity conservation and rural development.

Trend 3: Rehabilitation of forest land after overgrazing, overexploitation, and desertification

This trend is prominent in Central Asia, where forest cover is about 5% of the subregion. Distribution of forested areas is uneven with the largest forested areas in Turkmenistan and the smallest in Tajikistan. Due to overall arid environments, the wood production in this subregion is low, and its economic/monetary contribution is insignificant (Kleine *et al.*, 2009). However, forests deliver diverse contributions to people, including water regulation, soil protection, climate mitigation, fire wood, and recreational value at multiple scales. Nevertheless, significant degradation of forests has taken place since World War II, while not necessarily decreasing forested area. Main causes include converting forested area into agricultural land, overgrazing and overexploitation, including illegal logging, and fires (Baizakov, 2014; Toktoraliev & Attokurov, 2009). Major concerns are related to the disappearing Aral Sea, leaving a large area of degraded land. Attempts to afforest this area are being made to increase the area of land defined as forests in Kazakhstan. The forest management in this subregion is mainly focusing on rehabilitation of degraded forested land. This includes reforestation and afforestation as well as planting trees and shrubs to combat desertification (Meshkov, 2014).

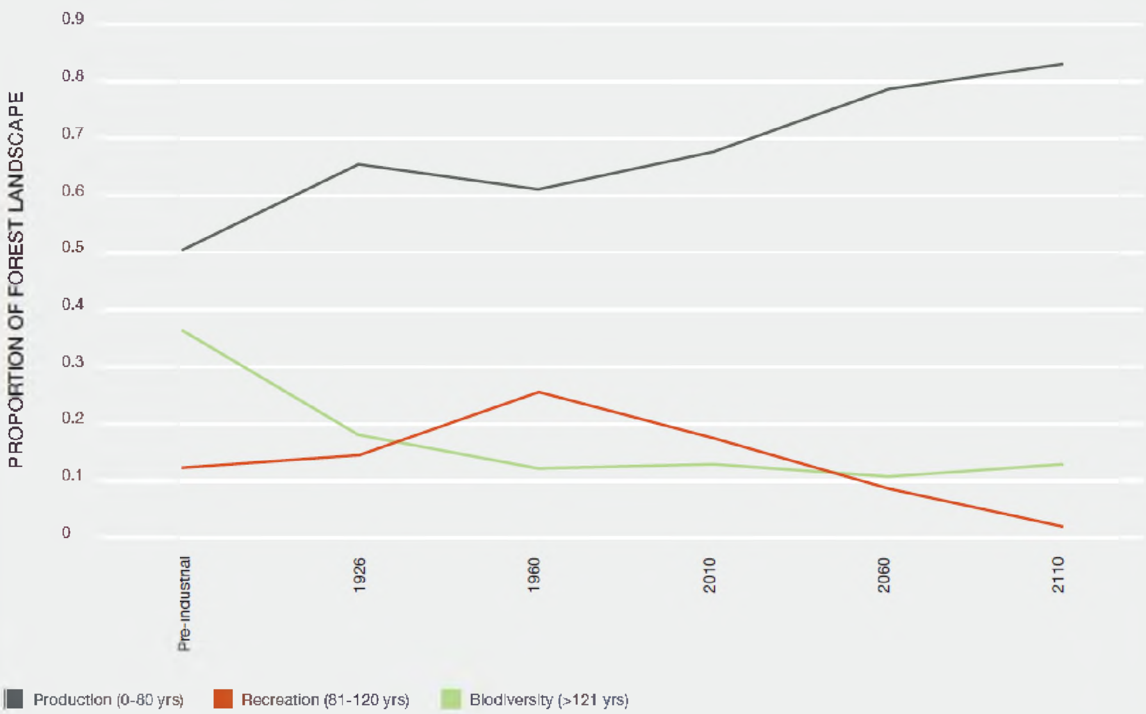
Trend 4: Multifunctional forestry

For the past four centuries sustained yield forestry has been focused mainly on wood for construction, fibre, or fuel. However, the normative interpretation of sustainability in forestry became broader when sustainable forest management policies appeared at the end of the twentieth century (MCPFE, 1998, 2001; Wang & Wilson, 2007). Sustainable forest management aims at maintaining, now and in the future, sustainable ecological, economic, social, and cultural functions of managed forests through multi-stakeholder participatory approaches (Hahn & Knoke, 2010; MCPFE, 1998, 2001; Wiersum, 1995). This requires that forest managers consider the use of a broad range of nature’s contributions to people through adaptive management and governance to be able to handle potentially conflicting demands at multiple spatial scales (Bawa & Seidler, 1998; Behan, 1990; Bouthillier, 2001; Farrell *et al.*, 2000; Hahn & Knoke, 2010; Sandström *et al.*, 2011; Wiersum, 1995). Lindahl *et al.* (2017) noted that this pathway is influenced by ideas of ecological modernization and the optimistic view that existing resources can be increased, thus prioritizing the economic dimension of sustainability. At present, society’s interest in sustainable forest management is growing. This is mainly linked to bioenergy production and energy security as well as climate change adaptation and mitigation (Spittlehouse & Stewart, 2003). There are arguments that timber supply-oriented sustained yield concept is no longer appropriate (Wiersum, 1995), and forest managers need to “develop from being crop managers to ecosystem managers” (Farrell *et al.*, 2000).

Countries in Europe and Central Asia have diverse natural, historical, societal, and economical legacies and thus have different starting points in their trajectories of development toward sustainable forest management (Angelstam *et al.*, 2011; Lehtinen *et al.*, 2004). For example, recent analyses of the future development of boreal forests in Western Europe (Claesson *et al.*, 2015) indicate that this process will divide forest landscapes into intensively managed stands with harvest return intervals of 60-80 years and only scattered remnants of old growth forests set aside for biodiversity conservation purposes (**Figure 4.19**). To counteract this segregated trend there is an increasing focus on integrative approaches in forest management (Kraus & Krumm, 2013). These initiatives include green tree retention, identification of small valuable forest habitats, and promotion of mixed forest stands (e.g. Brang *et al.*, 2014; Johansson *et al.*, 2013). Similarly, integrative approaches may benefit the protection of wooded grasslands - habitats that have declined dramatically during the 20th century (Axelsson *et al.*, 2007) – with their ecological and social values (Hartel & Plieninger, 2014).

Figure 4 19 Past, current and projected proportions of three forest stand age classes in Sweden representing maximum sustained yield wood production (0–80 yrs), recreation (81–120 yrs) and biodiversity conservation (>121 yrs).

Source: Pre-industrial data from Angelstam and Kuuluvainen (2004), and other periods from Claesson et al. (2015). Note, however, that for biodiversity conservation these statistics overestimate the functionality of areas >121 yrs as functional habitat networks. Three reasons are loss of the ecologically most important forest, say >180 yrs with dead wood in different decay stages, small patch sizes and limited functional connectivity.

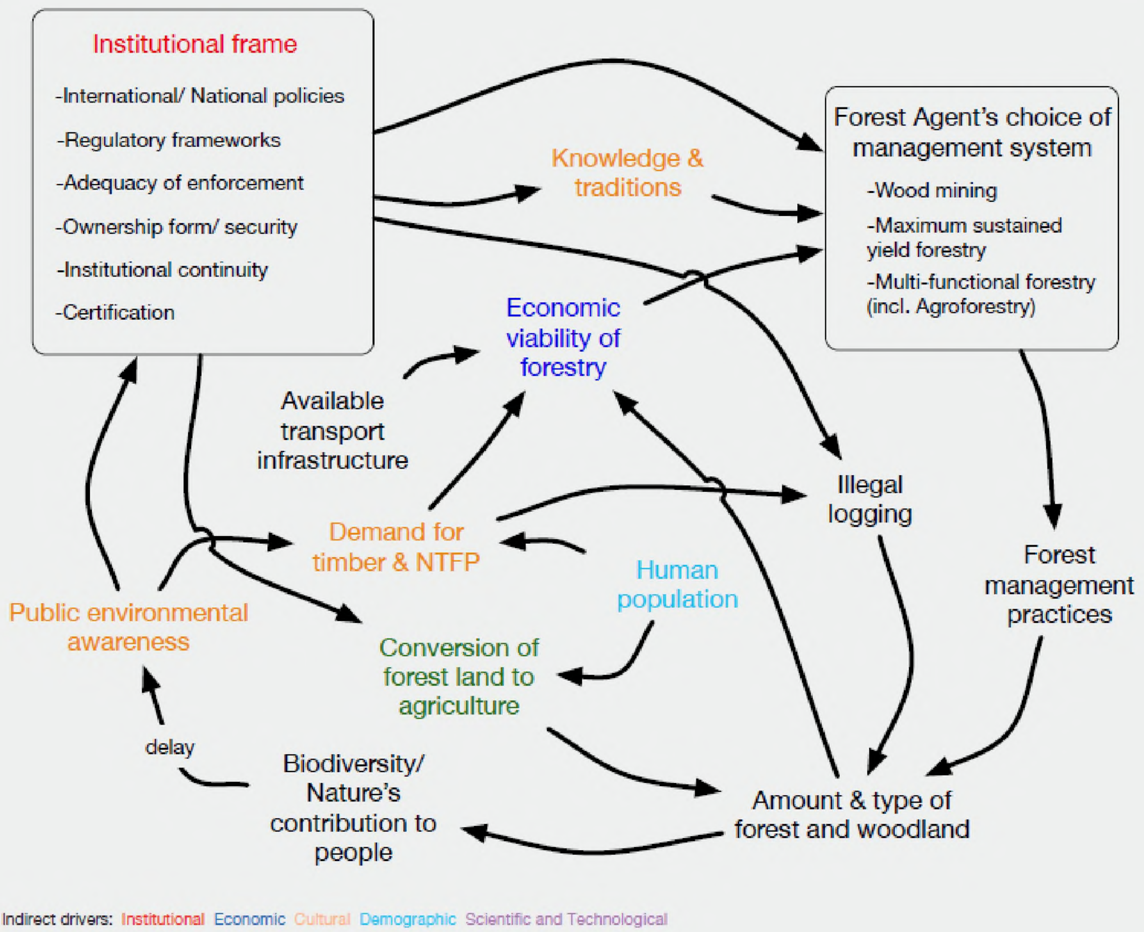


4.5.3.2 Drivers of trends in forestry

The overview model (**Figure 4.20**) shows the dynamic inter-relationships within and between indirect and direct drivers of change in forestry identified via the literature review process. This overview model is then further unpacked at a variety of levels of detail to examine the major dynamics of indirect drivers of changes in forestry.

Figure 4 20 An overview of the key drivers and systemic interconnections leading to changes in forestry in Europe and Central Asia. Source: Own representation.

Given the long lag times for forest regeneration, past choices of forest management systems continue to exert a major influence on the amount and types of forest and woodland available in Europe and Central Asia. The choice of forest management system is influenced by a broad set of institutional drivers, the availability of relevant knowledge, regional forestry traditions, and considerations of economic viability. Multi-functional forestry, for example, is dependent on extant demand for non-timber forest products, such as wild foods, or for social values, such as recreation and tourism. Illegal logging and the conversion of forested land to agricultural land are also important drivers in parts of Europe and Central Asia. These inter-relationships are further unpacked in the following sections.



4.5.3.2.1 Legal frameworks

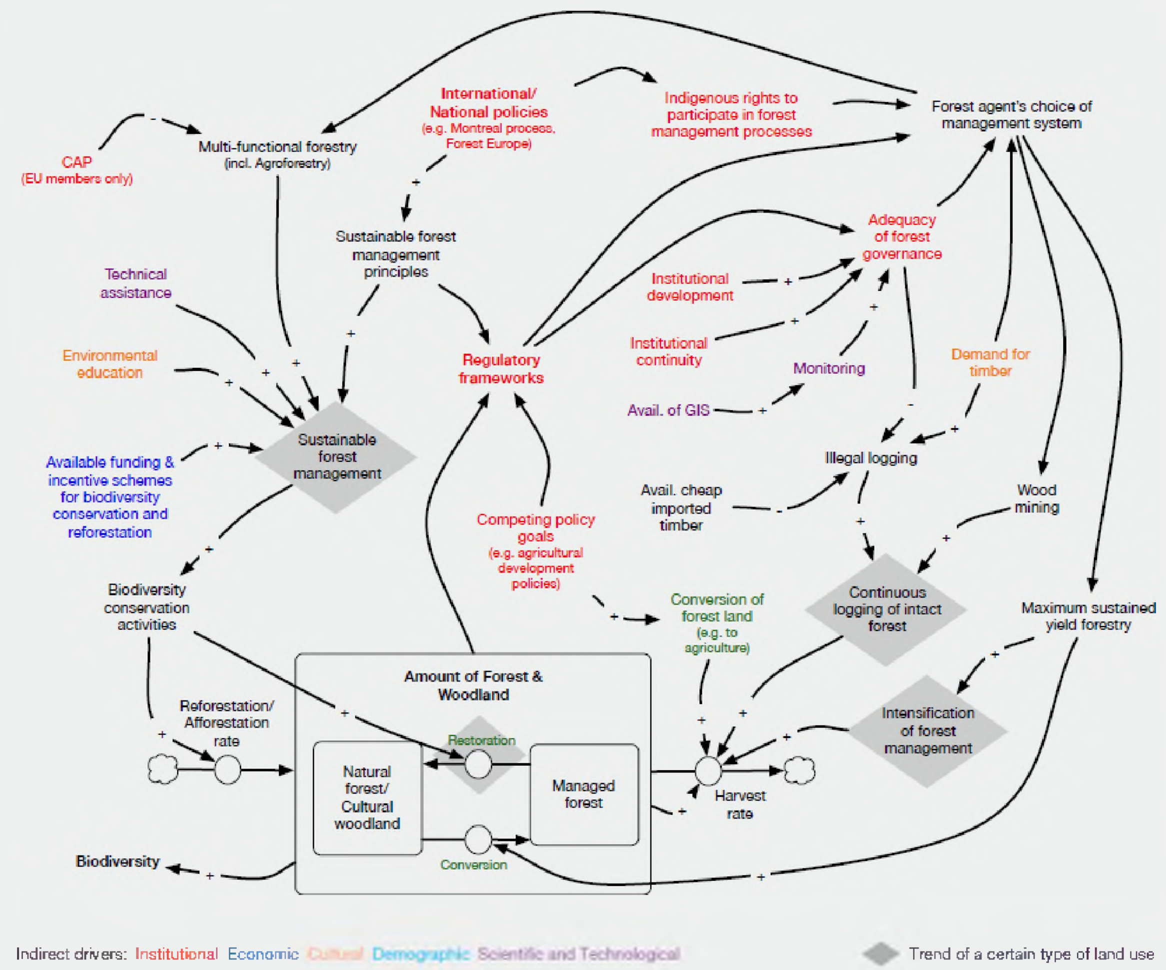
Regulatory frameworks for forest management have a long history in Europe and Central Asia. In Western Europe, they date back to at least the 17th century. Starting already in the beginning of the 19th century, forest management for wood production was regularly taught at forestry schools aiming for efficient silviculture. After two centuries of maximum sustained forestry yield, in recent decades the international policy pendulum (e.g. CBD, 2010; European Commission, 2013; MCPFE, 1998, 2001) has swung towards multiple use and benefits. Initially, this was focused on biodiversity conservation and later also on rural development (Kennedy *et al.*, 2001).

The Montréal Process developed sustainable forest management principles for the temperate and boreal forests; and the Ministerial Conference on Protection of Forest in Europe (Pan-European process or Forests of Europe) for countries in Western, Central and Eastern Europe (Forest Europe, 2011). The Pan-European criteria and indicators provide guidelines for sustainable forest management at the national and sub-national levels, and to operationalize and complement the existing (MCPFE, 1998, 2013). There is a common strategy for 46 countries in Western, Central and Eastern Europe on how to sustainably manage their forests (Forest Europe, 2015). The sustainable forest management concept is an overarching guiding principle at the policy level. However, there is considerable variation in how this concept is implemented among countries (Lehtinen *et al.*, 2004), different forest owner categories, and over time in a given country. In 2011, Forests Europe presented "European forests 2020 Goals and Targets" (Forest Europe, 2011) that requires the sustainable management of all European forests, including multiple forest functions and enhanced use of forest goods and services (**Figure 4.21**).

Regarding agroforestry systems, the agricultural subsidy regime within the European Union is considered unfavourable towards silvo-arable practices (e.g. Fragoso *et al.*, 2011; Plieninger *et al.*, 2004); and there is a need to reinforce and promote alternative agricultural and non-agricultural economic activities in rural areas. New functions include leisure and recreation (García Pérez, 2002; Pinto-Correia, 2000; Surová & Pinto-Correia, 2009). Indeed, Gaspar *et al.* (2009) showed that mixed livestock dehesa farms made optimal use of resources, and had little dependence on external subsidies. Given uncertainties about the European Union subsidies, this type of farm might be a goal for dehesa farmers. Thus, the maintenance of the traditional agroforestry systems in Spain and Portugal is a good example of how a diversity of forest and woodland management regimes sustains multiple goods, services and landscape values (Linares, 2007). However, Pinto-Correia (2000) and Plieninger *et al.* (2004) pointed out that this requires a holistic landscape approach including conservation-incentive schemes, environmental education, and technical assistance.

Figure 4 21 Legal frameworks are key drivers of change in forestry in Europe and Central Asia. Source: Own representation.

A number of recent international processes have led to the evolution and adoption of a set of principles supporting sustainable forest management, and strengthening regulatory frameworks, which govern norms and practices associated with other forms of forest management. However, these new institutions require further integration to reduce conflicts with other policies and institutional legacies.



In Central Asia at the beginning of the 1990s, national agricultural policies such as converting forests into arable land and pasture continued to reduce forested areas. For instance, in Uzbekistan the area of tugay forests – a form of riparian forest or woodland associated with fluvial and floodplain areas in arid climates – decreased to less than one-tenth of the original area. Walnut forests in Kyrgyzstan decreased by 50% while mountain slope desertification increased by 31% (Toktoraliev & Attokurov, 2009). Since the 1990s, forest management organizations at different levels have gone through many reforms (Baizakov, 2014) and political and economic uncertainties, and severely weakened forest governance had caused growth of illegal logging and forest fires. The stabilization of economies in the region has shifted the attention to the forest crisis in the region. For example, Kazakhstan has prohibited cutting of saxaul forests, and Kyrgyzstan has announced a moratorium on cutting of walnut forest. The import of wood from Russia was renewed, and the pressure on forests has declined. Institutional development strengthened forest protection in the region. Also, introduction of GIS technologies enabled forestry to collect and monitor the forest data more effectively (Government of Kyrgyzstan, 2007; Karibayeva *et al.*, 2008).

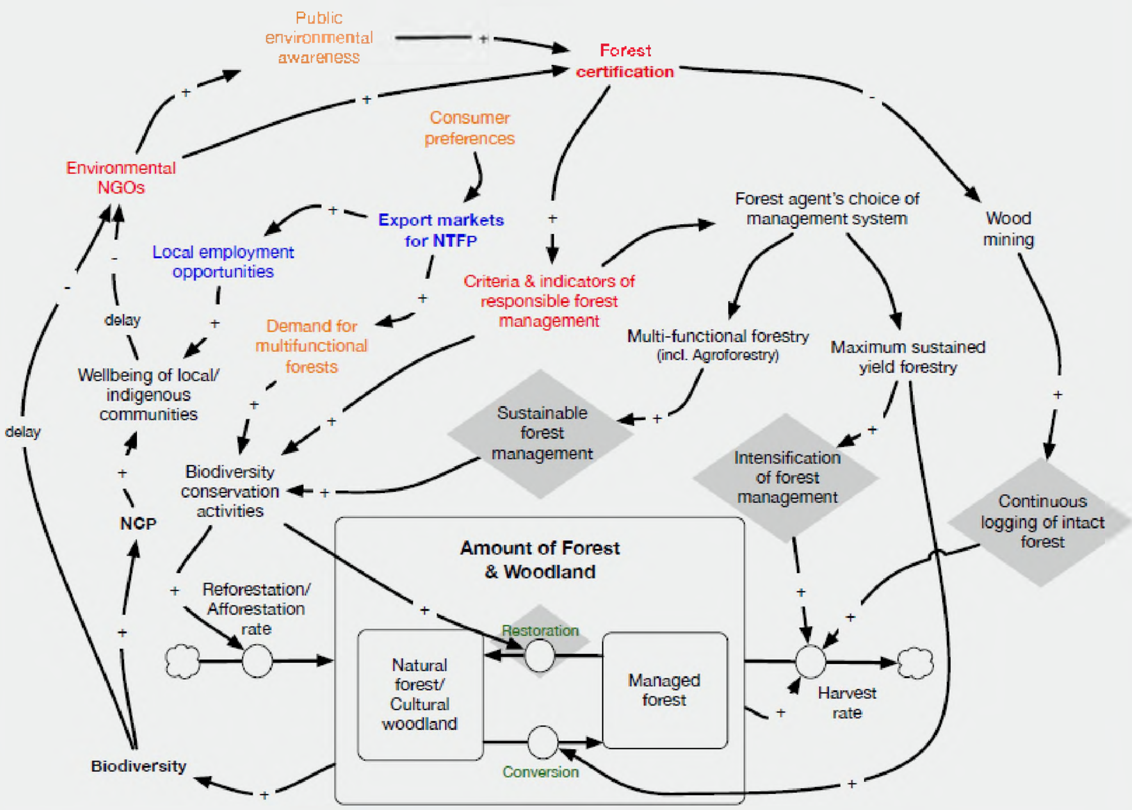
4.5.3.2.2 Forest certification

Forest certification is a market-driven instrument that is becoming increasingly important in forest management in Europe and Central Asia. The Forest Stewardship Council certification operates in 33 countries in Europe and Central Asia, covering almost 96 Million hectares (FSC, 2016); and the PEFC (the Programme for the Endorsement of Forest Certification) operates in 23 countries in the region, covering almost 84 Million hectares (PEFC, 2016). While many Governments in Europe and Central Asia have favoured command and control mechanisms to address policy targets, a growing number of private and civil society actors have pioneered non-state voluntary instruments as a means to achieve responsible forest management that aims at maintaining, protecting and sustaining ecological, economic and social-cultural values of forests. This complex landscape of state and non-state governance has shifted the power dynamics of environmental governance, raising questions about whose interests and priorities are being served, in which contexts, and with what consequences for social equity and biodiversity conservation (Cashore *et al.*, 2003, 2005).

Forest certification growth has provoked considerable public debate (Angelstam *et al.*, 2013; Elbakidze *et al.*, 2011; Lindahl & Westholm, 2010; Sandström *et al.*, 2011), highlighting how the design and implementation of certification inevitably involves struggles for power amongst diverse interests with differing standards and impacts across countries (see **Figure 4.22**).

Figure 4 22 Forest certification and the development of export markets for non-timber forest products are increasingly important drivers of forest management in Europe and Central Asia. Source: Own representation.

Environmental NGOs have played a key role in the adoption of forest certification schemes, which have reduced "wood mining" of remaining intact forests and have led to the inclusion of biodiversity conservation criteria and indicators in forest management systems. The establishment of functioning markets for non-timber forest products is important to grow demand for multi-functional forests, which promote forest biodiversity.



Indirect drivers: Institutional Economic Cultural Demographic Scientific and Technological

Trend of a certain type of land use

4.5.3.2.3 Markets of non-timber products

Emergence of new export markets for non-timber products, primarily medical plants and walnuts, has increased reliance of local populations on forests. The evidence shows that the health of forest ecosystems that produce these products greatly improves the quality of life of local households (Fisher *et al.*, 2004).

4.5.3.2.4 Forest ownership

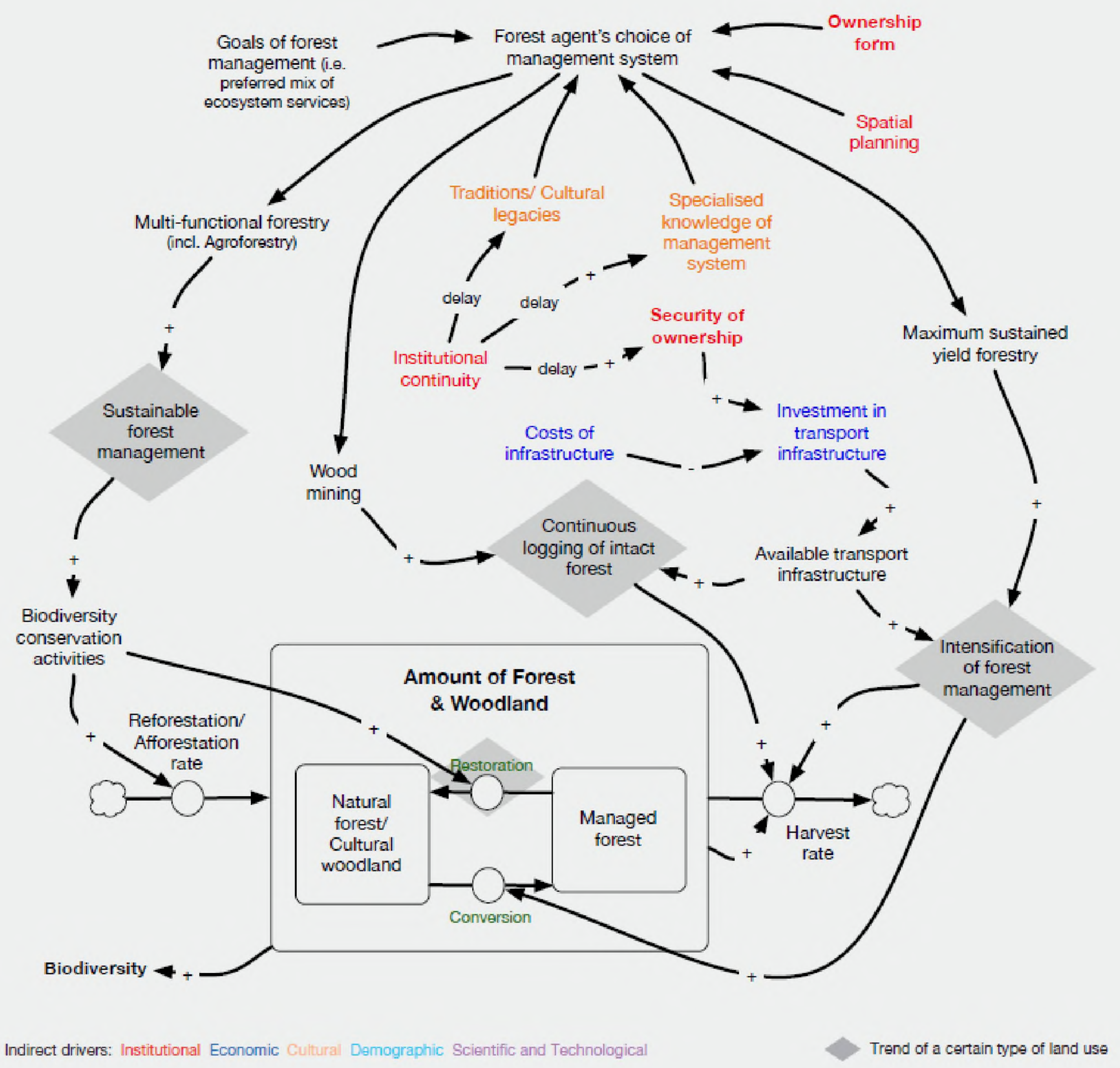
The forest harvest rate, land conversion and management system is decided by the choices of forest managers – individual managers and land owners – but a wide range of drivers influence their decisions in an interrelated and complex way. These drivers are depicted in **Figure 4.23** summarizing the causal links influencing forest ecosystems. The choice of management system is influenced primarily by cultural legacies (managers' world views of forestry), laws and policies (institutional drivers), the demand for specific forest products (e.g. increasing use of biofuels) as well as by costs, e.g. related to development of infrastructure. For example, the opportunity for introducing intensified forest management in Eastern Europe based on pre-commercial and commercial thinning is hampered by short forest leasing periods (Naumov *et al.*, 2017). A permanent transport infrastructure, which is available not only for harvesting, but for silviculture is also necessary. To pay for these costs, commercial thinning usually delivers inadequate financial net values (Brukas & Weber, 2009). Additionally, transport cost to remote, not yet harvested, areas need to be considered when investing in roads for harvest only, or also for silvicultural treatments. However, the costs of investing in transport infrastructure are high, and there are uncertainties regarding ownership and long-term maintenance (Naumov *et al.*, 2016).

4.5.3.2.5 Urban development

Urban development has had profound effects on forest. For instance from 1930 to 2000 in Central Asia, overharvesting decreased the area of spruce forests in Kyrgyzstan by 50% (Musuraliev *et al.*, 2000; Toktoraliev & Attokurov, 2009). Growing industrialization and population and rise of collective farming increased human-caused fires in forests. In 1954-1960 only 31% of fires were caused by man, while in 1981-1990 this number increased to 66% (Baizakov, 2014). Future change in forested area in Central Asia is likely to be strongly linked to the direct and indirect effects of ongoing climate change in combination with effects from changing demography, economy, technology, lifestyle, and policies (Moss *et al.*, 2010) (**Figure 4.23**).

Figure 4 23 **The choice of forest management system is influenced by the preferred mix of nature's contribution to people, the form and security of land ownership, and the knowledge and forestry traditions embodied in individual managers. Source: Own representation.**

Investment in transport infrastructure, vital to intensive silviculture, is also highly dependent on secure, long-term land ownership.

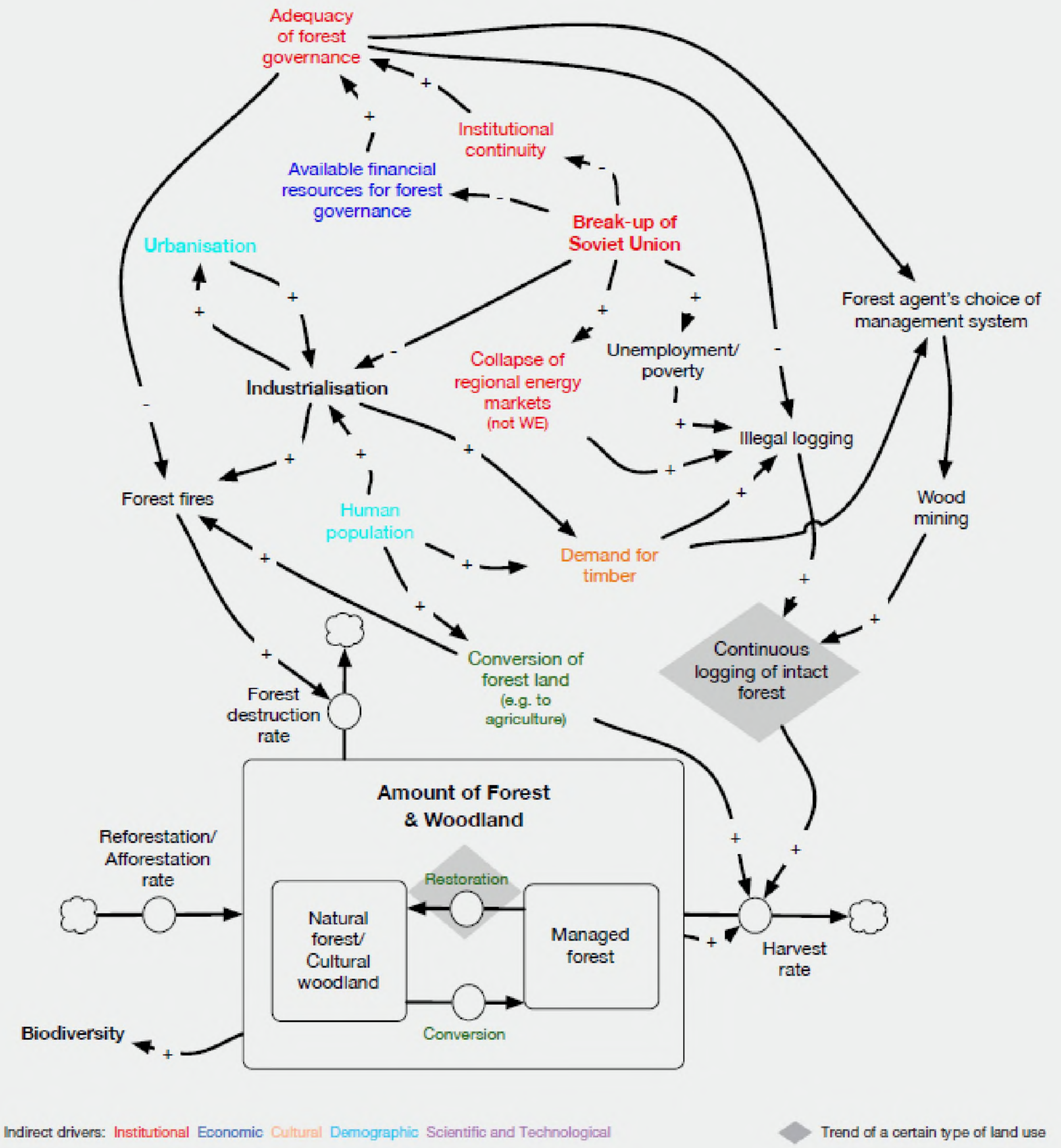


4.5.3.2.6 Radical changes in political, economic and social contexts as triggers of changes in forestry

Since 1991, after the dissolution of the Soviet Union, radical changes in political, social and economic contexts put pressure on forest areas in Eastern Europe and Central Asia, causing a decline in financial resources for forest management, and a decline in control measures. Forest management institutions lacked financial and political support (Baizakov, 2014). At the same time, local households experienced shortages in the supply of oil, firewood and coal, which led to increased illegal logging in rural areas. The regional market for coal and oil collapsed, which increased the use of forest wood for heating purposes. Rise of unemployment and poverty contributed further to forest destruction. For the past 20 years, forest area with tree species such as saxaul, pistache, almond and walnut have been reduced considerably (Demidova, 2013) (**Figure 4.24**).

Figure 4.24 **Population growth, industrialization and urban development are drivers of demand for timber. Source: Own representation.**

As a result, a "wood mining" frontier has slowly moved from south-west to north-east Europe during recent centuries, leaving a long-standing legacy of impaired forest biodiversity. The dissolution of the Soviet Union had a variety of institutional impacts on forests, and led to increases in illegal logging in Central and Eastern Europe as well as Central Asia.

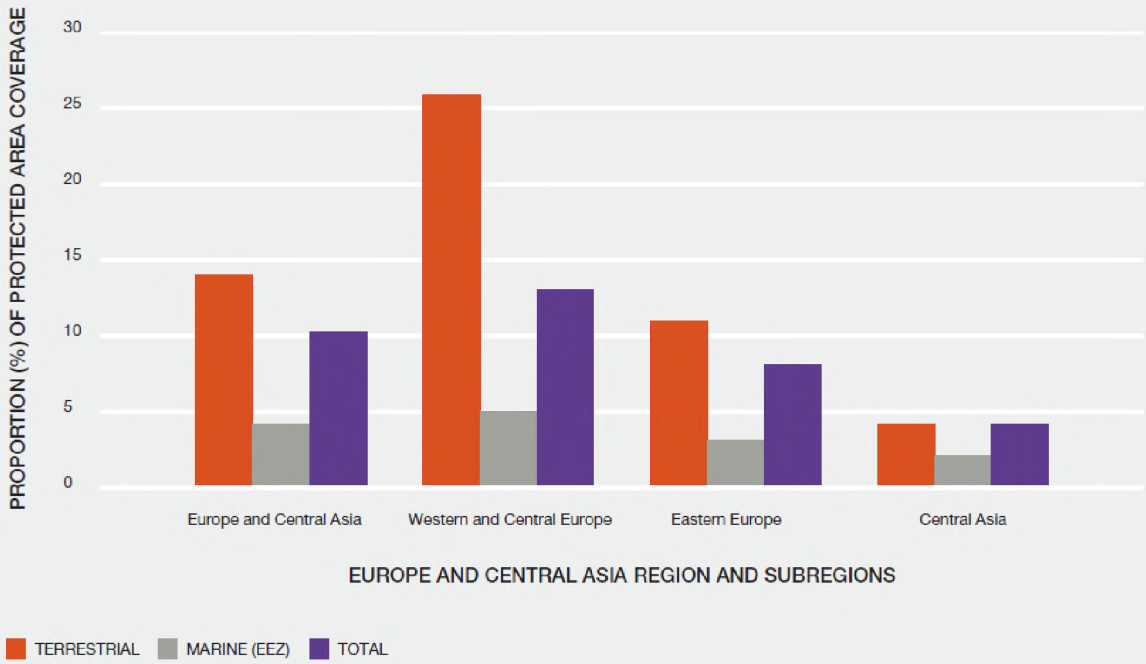


4.5.4 Trends and indirect drivers of changes in protected area development

4.5.4.1 Trends in protected area development

In Europe and Central Asia, the total coverage of protected areas is 10.2%, with 13.5% of its terrestrial area and 5.2% of its marine area (within the Exclusive Economic Zone) being protected (**Figure 4.25**) (UNEP-WCMC & IUCN, 2014). Key biodiversity areas cover 5.5% of Europe and Central Asia for Important Bird & Biodiversity Areas and 0.01% for the Alliance for Zero Extinction sites. As of 2017, the proportion of Key biodiversity areas fully covered by protected areas in Europe and Central Asia is 33.3% of Alliance for Zero Extinction sites and 28.1% of Important Bird & Biodiversity Areas.

Figure 4 25 Proportion of protected area coverage in Europe and Central Asia and subregions. Source: Own representation based on data from UNEP-WCMC & IUCN (2017).



In Western and Central Europe, the total coverage of protected areas is 14.9%, with 26.7% of the terrestrial area and 6.8% of the marine area being protected (**Figure 4.25**). These subregions have the highest proportion of terrestrial and marine areas, and also the highest proportion of protected area coverage in Europe and Central Asia. Key biodiversity areas cover 6.4% of Western and Central Europe for Important Bird and Biodiversity Areas, and only 0.01% for Alliance for Zero Extinction sites. As of 2017, the proportion of Key biodiversity areas fully covered by protected areas in Western and Central Europe is 14.3% of Alliance for Zero Extinction sites and 35.5% of Important Bird and Biodiversity Areas (**Figure 4.26**). In Eastern Europe, the total coverage of protected areas is 7.5%, with 9.5% of the terrestrial area and 2.9% of the marine area (within the Exclusive Economic Zone) being protected (**Figure 4.25**). Key biodiversity areas cover 4.8% of Eastern Europe for Important Biodiversity Areas, and 0.01% for Alliance for Zero Extinction sites. As of 2017, the percentage of Key biodiversity areas fully covered by protected areas in Eastern Europe is 100% of Alliance for Zero Extinction sites and 5.42% of Important Bird and Biodiversity Areas (**Figure 4.27**). In Central Asia, the total coverage of protected areas is 4.1%, with 4.2% of the terrestrial area and 2.4% of the marine area (within the Exclusive Economic Zone) being protected (**Figure 4.26**). Key biodiversity areas cover 5.4% of Central Asia for Important Bird and Biodiversity Areas, and there are no Alliance for Zero Extinction sites in the subregion. As of 2017, the proportion of key biodiversity areas fully covered by protected areas in Central Asia is 4.65% (**Figure 4.27**).

Figure 4 26 Growth in the proportion of key biodiversity areas completely covered by protected areas in Europe and Central Asia. Source: Brooks *et al.* (2016).

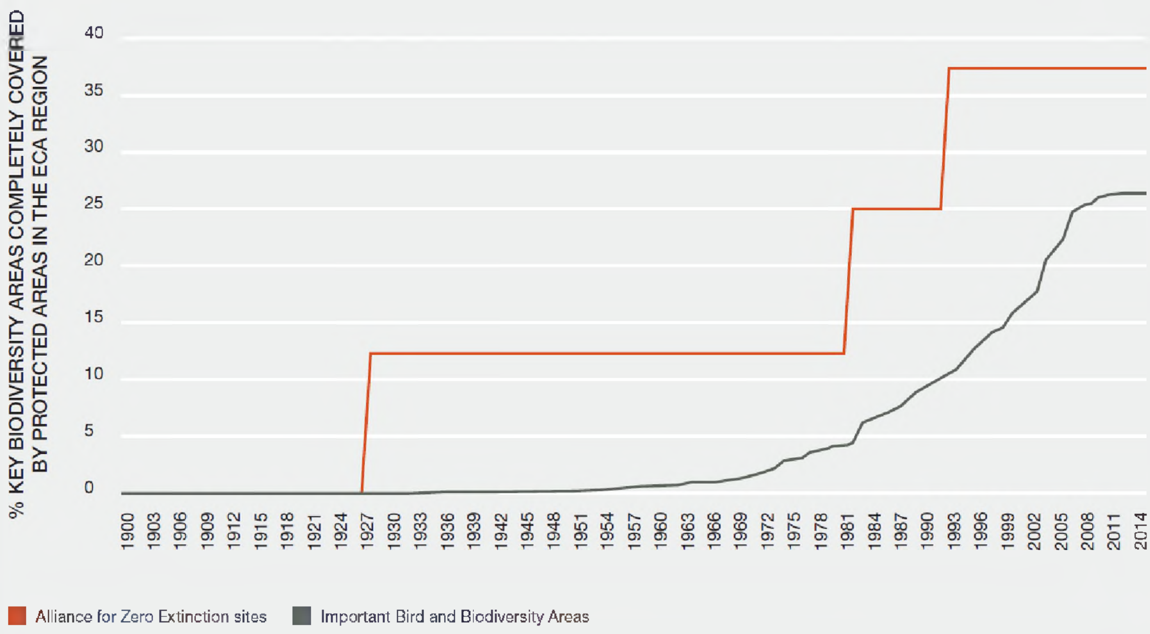
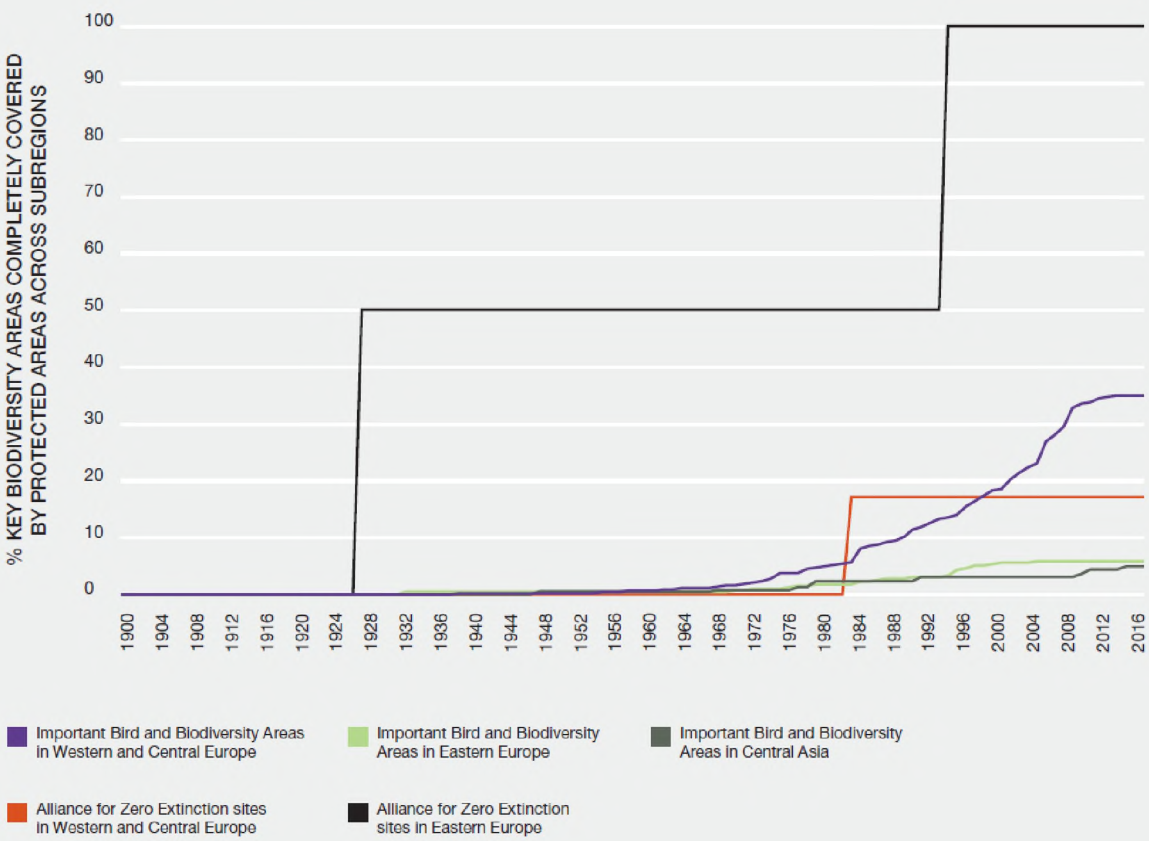


Figure 4 27 Growth in the proportion of key biodiversity areas completely covered by protected areas in the subregions of Europe and Central Asia. Source: Brooks *et al.* (2016).



The main trend in protected area development in Europe and Central Asia is increasing area under protection. Increase within the European Union has been significant, amounting to about 25% of land cover (UNEP-WCMC & IUCN, 2016). Superficially, this suggests that the European Union has already met Aichi Biodiversity Target 11 of 17% protected terrestrial area. However, the bio-geographical and ecological representativeness as well as connectivity (e.g., Angelstam *et al.*, 2011) of protected area needs further research. Consequently, tools for monitoring and analytic prioritization are clearly needed (Branquart *et al.*, 2008; Rosati *et al.*, 2008; Schultze *et al.*, 2014).

Analysis of the development of protected areas in the boreal zone in Western and Eastern Europe over the last 100 years (Elbakidze *et al.*, 2013b) shows that the areal extent of protected areas has increased from approximately 1500 ha in 1909 to 23 million ha in 2010 (**Figure 4.28**). The area proportion, size and management profiles of protected areas were very different over time among boreal countries. Throughout this 100-year study period, the least productive northern boreal forest was preferentially protected (**Figure 4.28** and **Figure 4.29**). The uneven representation of protected areas among boreal zone in Western and Eastern Europe was maintained over almost the entire previous century and presents a big challenge for boreal forest conservation (e.g. Hanski, 2011; Uotila *et al.*, 2002; Virkala & Rajasarkka, 2007). Another challenge for ecological sustainability is that the vast majority of boreal protected areas are small. According to many studies concerning the requirements of species with different life histories (Belovsky, 1987; Biedermann, 2003; Edenius & Sjoberg, 1997; Jansson & Angelstam, 1999; Jansson & Andrén, 2003; Linnell *et al.*, 2005; McNab, 1963; Meffe & Carroll, 1994; Menges, 1991; Roberge & Angelstam, 2004), it is evident that many protected areas are too small for focal and umbrella species such as specialized birds and area-demanding mammals.

Figure 4 28 **Cumulative growth of terrestrial protected areas in northern, middle, and southern boreal forests in Western and Central Europe and the European part of Russia by decade. Source: Elbakidze *et al.* (2013b).**

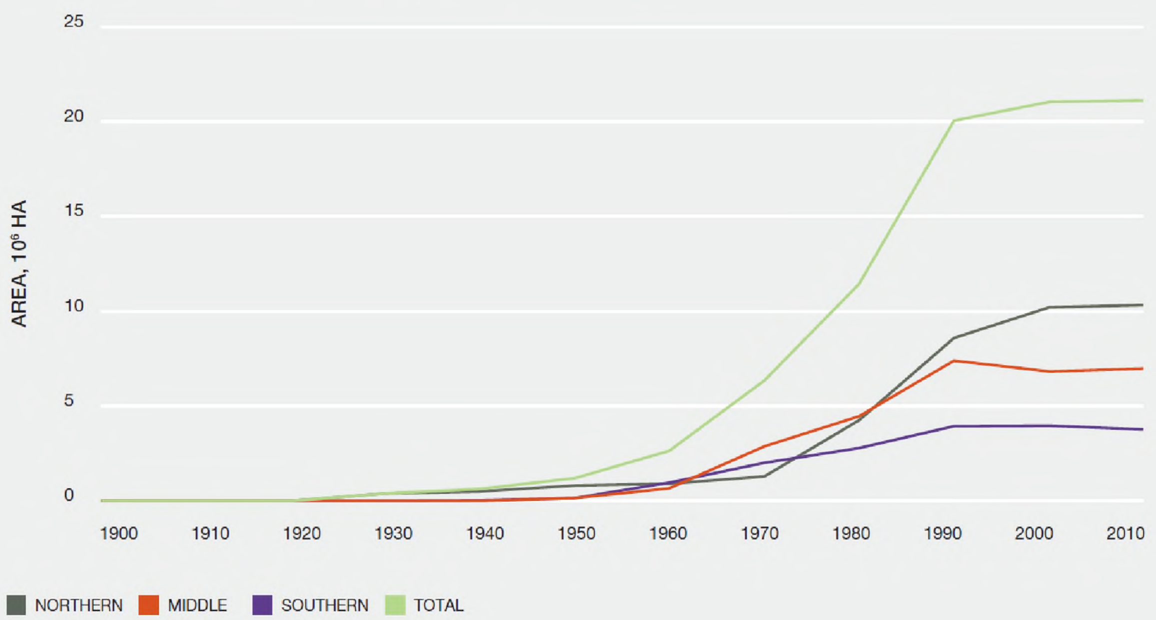
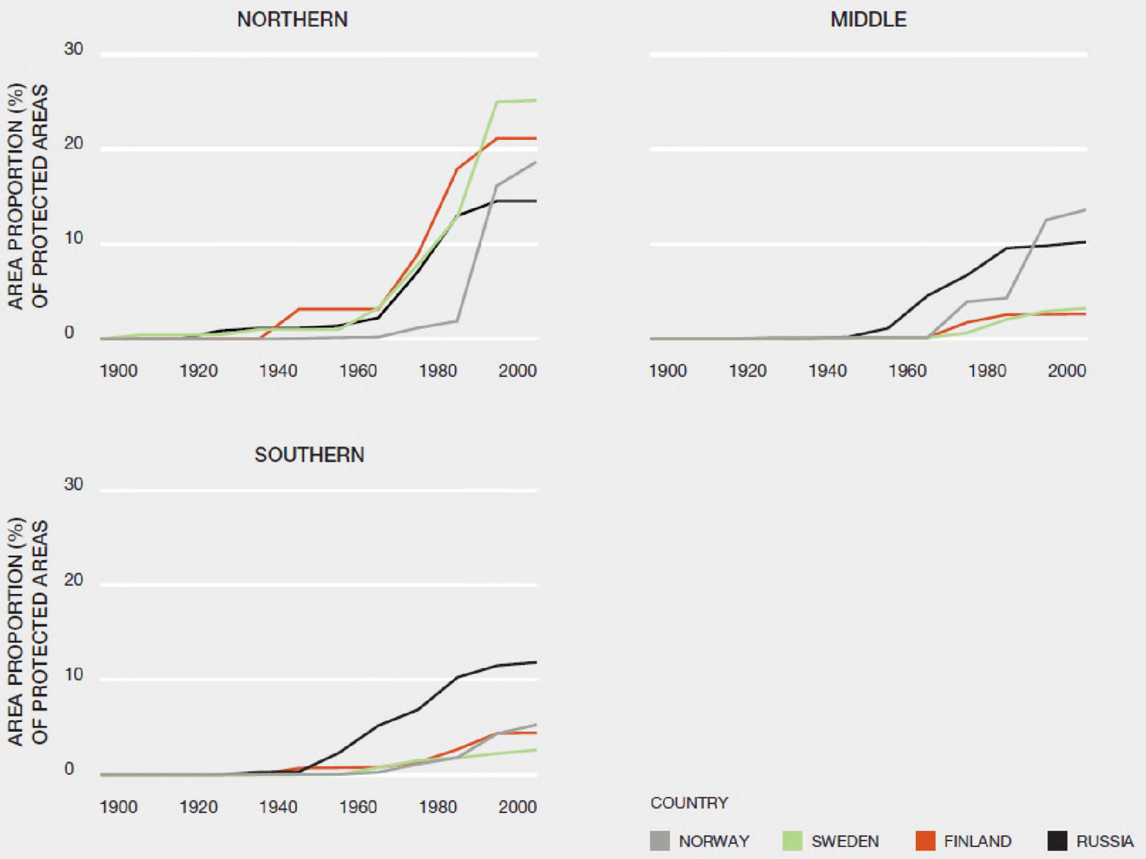


Figure 4 29 Cumulative growth of the area proportion of terrestrial protected areas in northern, middle and southern boreal forests in Western and Central Europe and the European part of Russia. Source: Elbakidze *et al.* (2013b).



The number of marine protected areas around the world has increased in recent decades, including in the European Union, aiming at the enhancement of local fisheries (Jones *et al.*, 1993; Lubchenco *et al.*, 2003) following the failure of traditional management measures (Batista & Cabral, 2016; Devillers *et al.*, 2015; Fenberg *et al.*, 2012; Jones *et al.*, 1993; Lubchenco *et al.*, 2003; Waters, 1991). Marine protected areas are generally strongly advocated as an ideal tool for resource management – specifically of coastal fisheries, as well as for preserving biodiversity (Agardy & Tundi Agardy, 1994; Costello & Ballantine, 2015; Dugan & Davis, 1993; Gaines *et al.*, 2010; Lubchenco & Grorud-Colvert, 2015; NOAA, 1990; Roberts & Pollunin, 1991). In 2016, Mediterranean Marine Protected Area Network and Regional Activity Centre for Specially Protected Areas reports 1231 marine protected areas in the Mediterranean covering 18 million hectares, or 7.1% (MAPAMED, 2017) (http://www.medpan.org/en/mapamed) (**Figure 4.30**). The expectation is that marine protected areas will continue to increase in number and area across the Mediterranean and North East Atlantic (**Figure 4.31**).

However, marine protected areas design differs between Atlantic and Mediterranean areas (Pérez-Ruzafa *et al.*, 2017). Northern marine protected areas (the so-called fish boxes or fisheries closures; Pastoors *et al.*, 2000) generally cover hundreds of thousands of hectares, and are intended to protect one or more target or by-catch species (e.g., plaice, sole, cod, herring, sprat, haddock). Mediterranean marine protected areas (Fenberg *et al.*, 2012; Planes *et al.*, 2006), meanwhile, usually over hundreds of hectares or less (Gabrié *et al.*, 2012; Portman *et al.*, 2012), are in general located in areas that are biologically unique. Both types include differences in management strategies that can affect their efficiency as fisheries and biodiversity conservation tools (Pérez-Ruzafa *et al.*, 2017).

Figure 4 30 Distribution of marine protected areas (MPAs) in Western and Central Europe. Source: Pérez-Ruzafa *et al.* (2017).

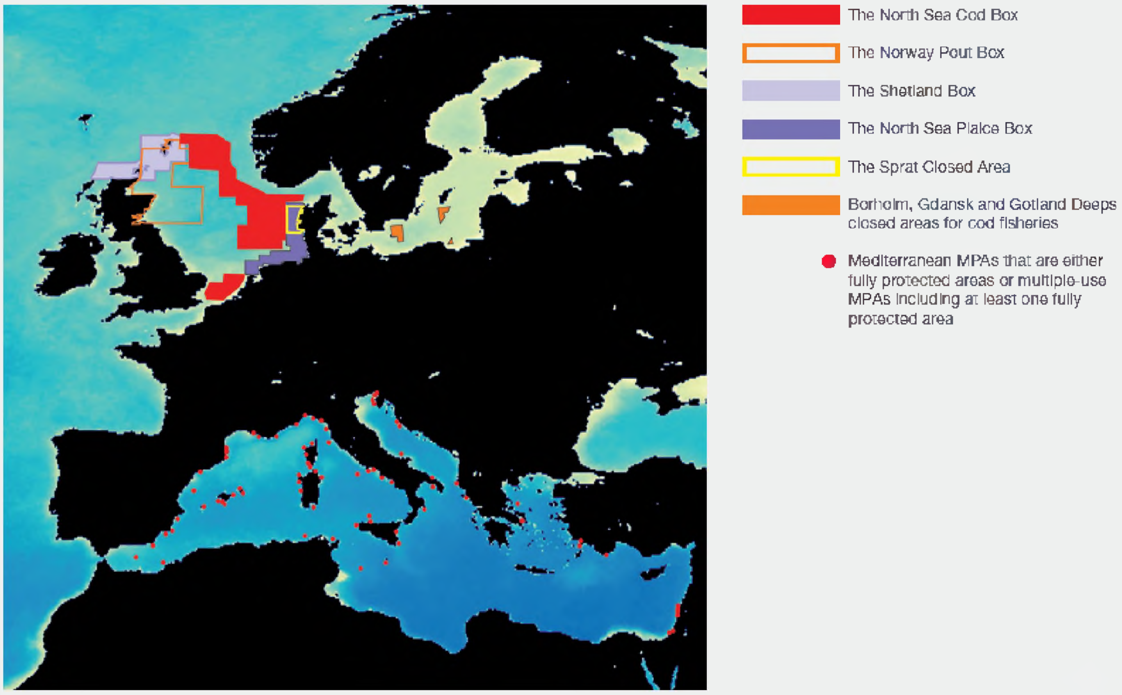
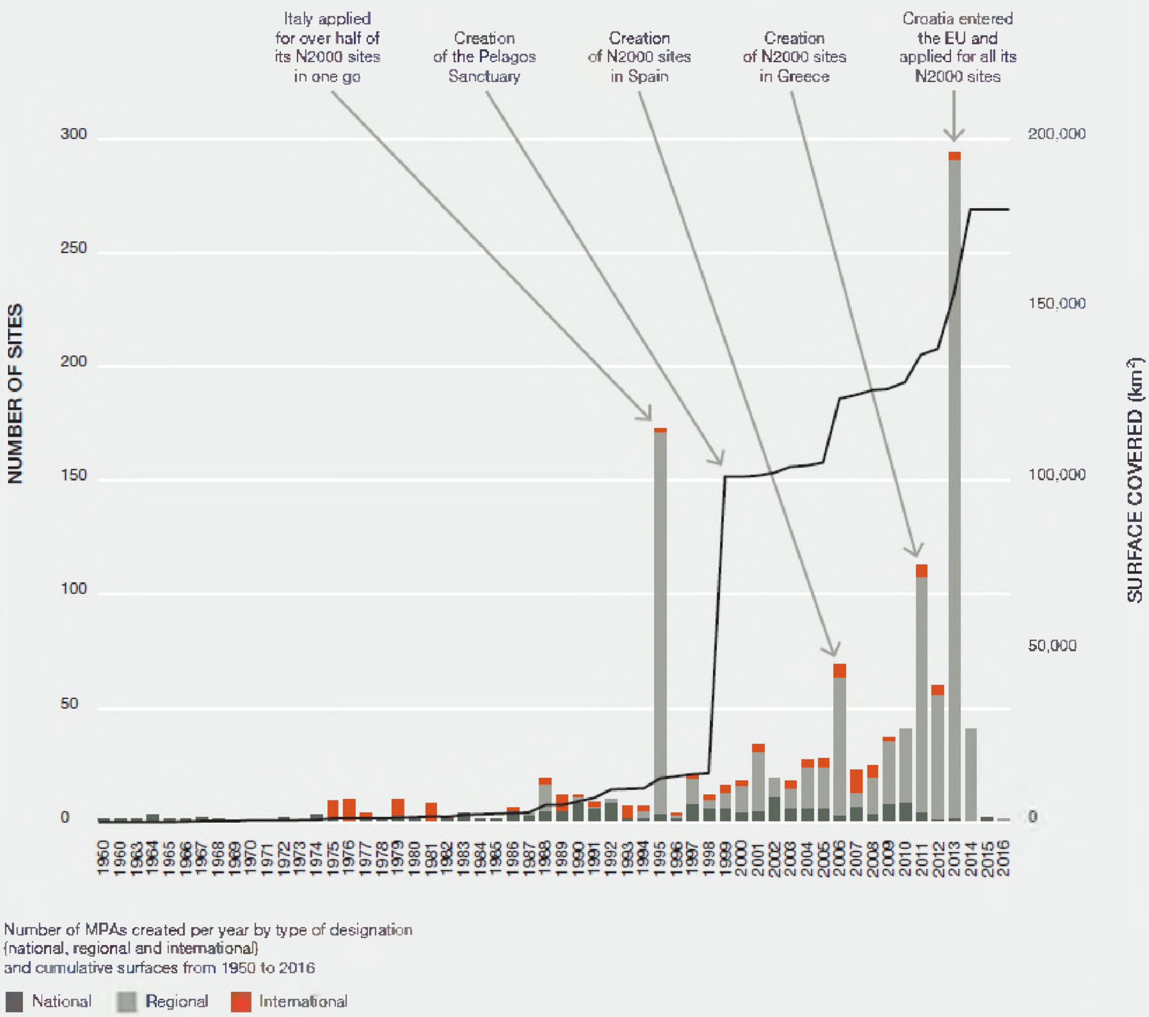


Figure 4 31 Trends in the number and cumulative surfaces of marine protected areas (MPAs) created per year and type of designation (national, regional and international) from 1950 to 2016.

Source: MEDPAN (2017). N2000: European Union Habitats and Birds Directives Natura 2000 sites.

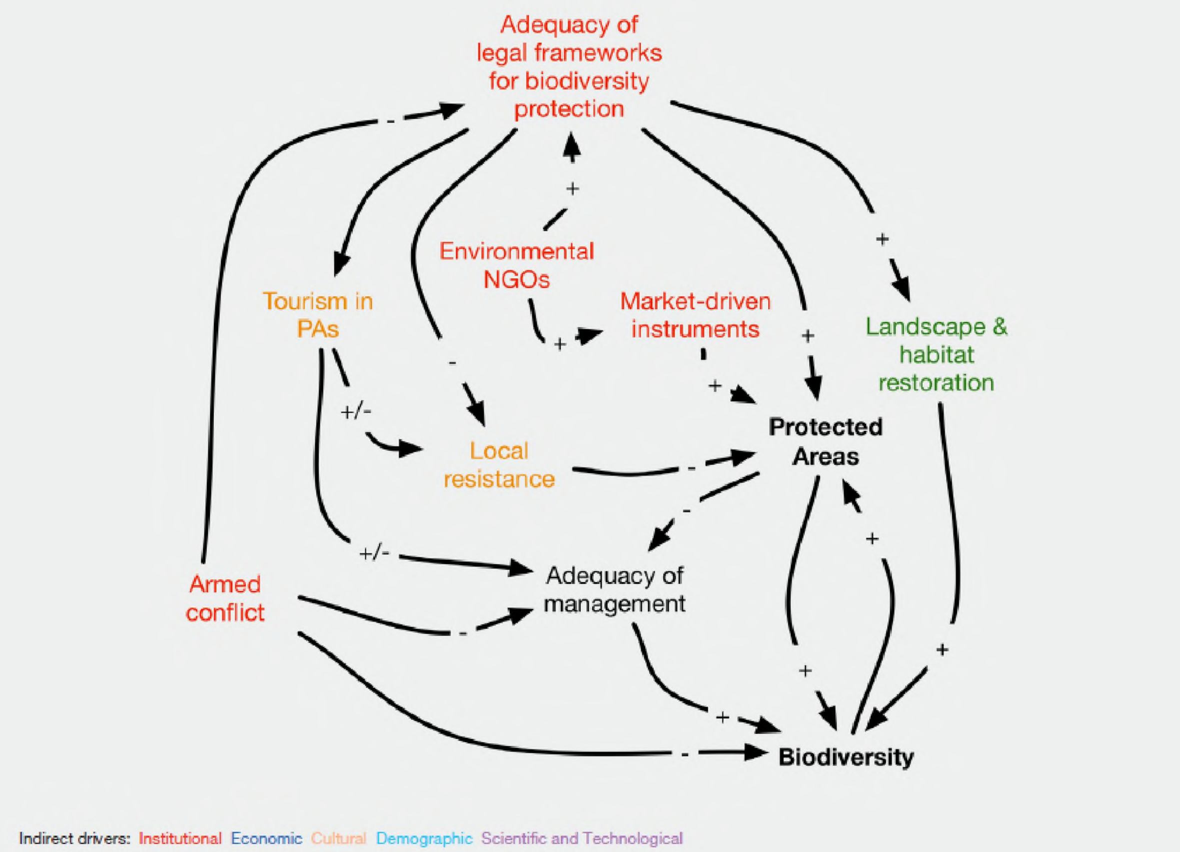


4.5.4.2 Indirect drivers of trends in protected area development

There are several key drivers of protected areas in Europe and Central Asia (**Figure 4.32**) that are unpacked below in Figures 4.33 – 4.39.

Figure 4 32 An overview of causal interconnections between the major drivers of protected areas (PAs) development in Europe and Central Asia. Source: Own representation.

Legal frameworks, particularly international agreements, have led to increased levels of biodiversity protection throughout the region. However, the general inadequacy of institutions for navigating local resistance to protected areas and regulating the negative impacts of tourism poses important problems. These drivers and interconnections are further unpacked in sub-model structures below.



4.5.4.2.1 Legal frameworks

An increasing number of global, regional and national legal frameworks have been a key driver of protected area development in Europe and Central Asia. Agreements such as the Convention of Biological Diversity (CBD, 2010) and associated Aichi Biodiversity Targets, have led to the adoption of a number of strategic plans and quantitative targets for protected areas. Underpinning these agreements is a growing public environmental awareness, which has influenced policy on nature protection. Another key factor has been the growing body of scientific knowledge on biodiversity and nature's contributions to people. Improved understanding regarding the negative effects of habitat fragmentation on ecological functionality, for example, has led to the consideration of functional networks of protected areas, at multiple scales, as a means of addressing biodiversity loss (e.g., European Commission, 2013; Hodge *et al.*, 2015) (see also Aichi Biodiversity Target 11 – "protected areas increased and improved").

In response to international agreements, most countries in Europe and Central Asia have developed national biodiversity strategies, in most cases including quantitative targets for protected areas (cf. https://www.cbd.int/nbsap/). In Western Europe, international plans and targets are mirrored in the EU Biodiversity Strategy (EU Parliament, 2012) and directly linked to the European Union Species and

Habitats Directive. These are subsequently enacted through national legislation. There is strong evidence that supranational conservation policy can bring measurable conservation benefits, although future assessments will require the setting of quantitative objectives and an increase in the availability of data from monitoring schemes (Donald *et al.*, 2007).

As a result of various bilateral agreements, a number of Eastern European countries (e.g. Ukraine, Belarus) are also in the process of harmonising national biodiversity protection legislation in line with European Union directives (e.g. regarding Natura 2000, and the Pan European Ecological Network). However, European Union policies are primarily based on Western European experiences. Numerous studies have shown cases where nature conservation legislation has underperformed when transplanted into new regional or local contexts (e.g. Aksenov *et al.*, 2014; Kuemmerle *et al.*, 2007; Wendland *et al.*, 2015) and a risk remains that European Union-developed approaches will prove either inefficient or inappropriate for supporting biodiversity associated with cultural landscapes in Eastern Europe and Central Asia. Additionally, in some cases the adoption of national strategies has led to unforeseen transboundary consequences. For example, forest protection in China and Finland have both resulted in increased harvest of old-growth forests in neighbouring regions of Central Asia and north-western Russia (Mayer *et al.*, 2006) respectively. Also, some countries have weakened national and sub-national protection regulations largely in favour of regional economic development (see **Box 4.3** and **Figure 4.33**).

Regarding the Marine Protected Areas, for example, the European Union Marine Strategy Framework Directive requires that member States should reach Good Environmental Status of their waters by 2020. The strategy sees establishment of a coherent network of Marine Protected Areas as one of the approaches to fulfil this aim. It specifically refers to Maritime Spatial Planning based on ecosystem based approach as a key tool to reinforced the objectives of the European Union Marine Strategy (Douvere & Ehler, 2009; Ehler, 2008).

Experience shows that these are not empty words. A study published in Marine Policy earlier this year assessed plans in Western and Central Europe, Australia and the USA. They found that planning led to a host of benefits for the environment: it increased marine protection, ensured that industrial uses avoided sensitive habitat, cut carbon emissions, and reduced the risk of oil spills.

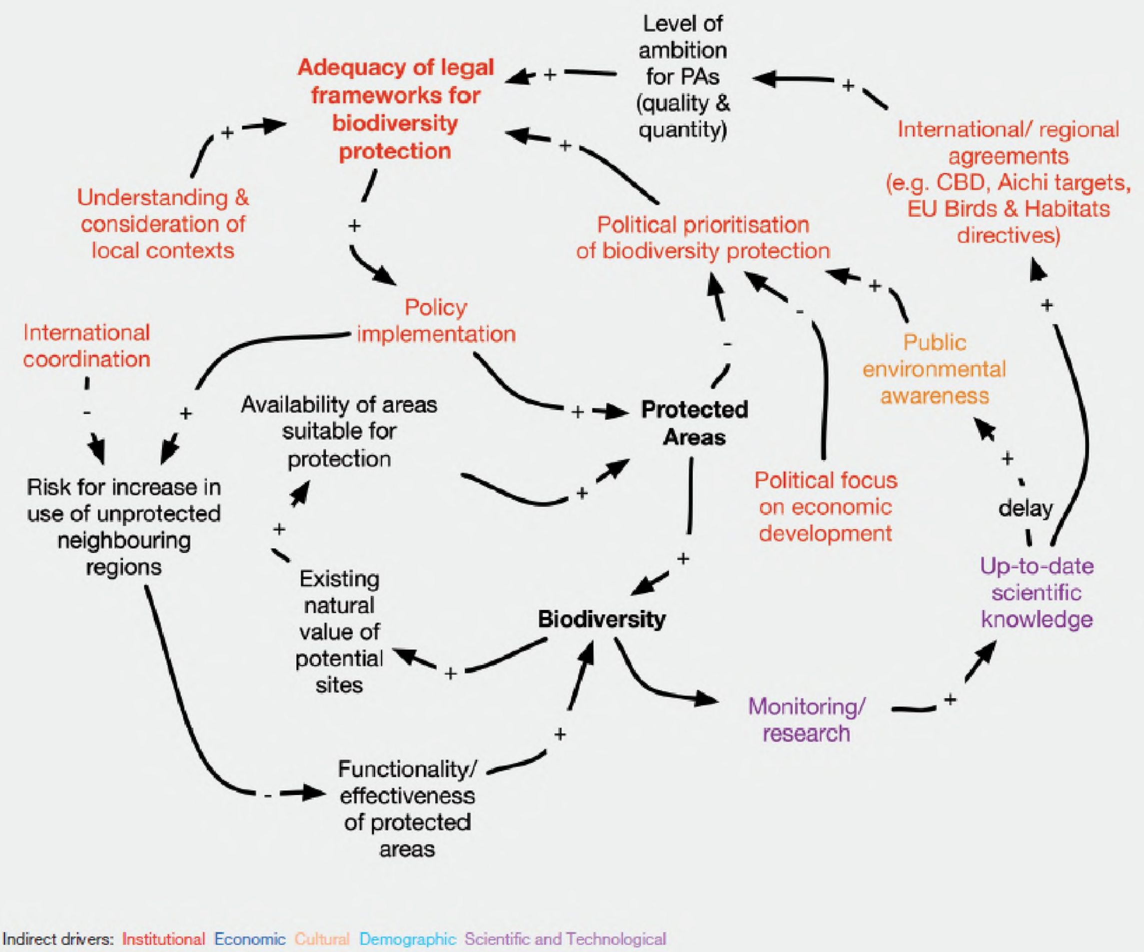
Box 4.3: Example of dynamics in legal frameworks from Eastern Europe.

In the Russian Federation, despite adopting several fundamental legal documents, and subsequent rapid growth in protected areas during the 1990s, numerous laws or amendments have recently been passed to weaken the protection status of existing protected areas, primarily in favour of increased economic activity (Brynych, 2016; NIA-Priroda, 2016. For example, in preparation for the Sochi Olympics an amendment was made in the law "On Specially Protected Natural Areas" allowing the construction of sports infrastructure in national parks. This amendment set legal preconditions for use of lands within national parks by new ski resorts. The governmental programme "The main directions of the state policy on the development of the system of state nature reserves and national parks in the Russian Federation for the period until 2015", adopted by the Ministry of Natural Resources of Russia in 2003, was not able to stop the subsequent degradation of protected areas. Recent changes in water and forest legislation led to a weaker legal regime in the areas of water protection zones and protective forests (Naumov *et al.*, 2017). In 2013, a law was passed that eliminated the principle of perpetuity of existence of protected areas and initiated transformation of strict nature reserves into national parks. In 2016, another law was adopted allowing the allocation of biosphere polygons within the boundaries of biosphere reserves, which legalized economic development (Brynych, 2016; NIA-Priroda, 2016). Other amendments were made to the federal law "On Territories of Traditional Nature Use of the Indigenous Minorities of the North, Siberia and the Far East of the Russian Federation" (2001), according to which such territories are not considered any more as Specially Protected Natural Areas; currently it creates new challenges in the procedure of their creation. Since 2001, not a single territory of traditional land management of indigenous people of Federal importance has been created (NIA-Priroda, 2016). At national and regional levels, there are no legal frameworks that take into account the specific nature of conservation of steppe landscapes (Chibilev, 2015).

End of Box 4.3

Figure 4 33 The prioritization and implementation of adequate legal frameworks for protected areas (PAs) development has largely been driven by the adoption of international agreements, as well as an increasing public environmental awareness. Source: Own representation.

A strong political focus on economic development goals, however, has in many cases delayed the development, and in some cases, has led to the weakening, of adequate policies.



4.5.4.2.2 Forest certification

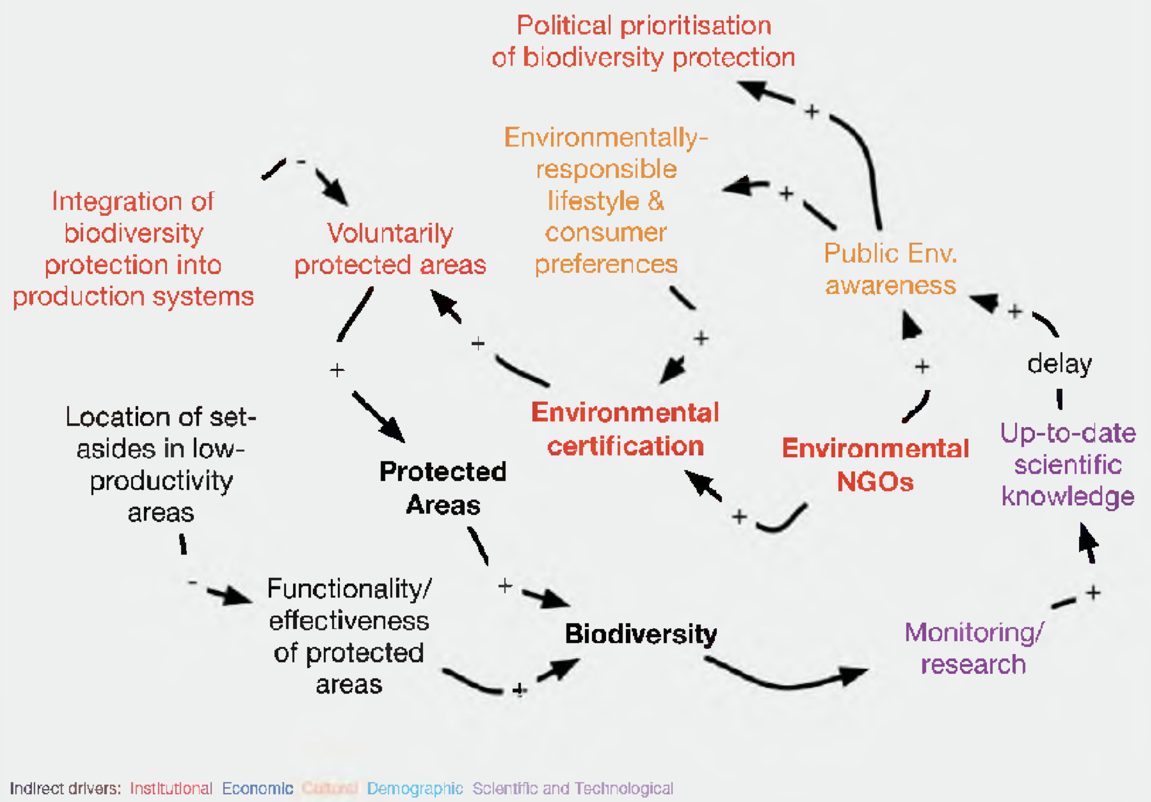
Industries have largely adopted certification requirements in response to increased consumer demand for environmentally responsible products, as a result of heightened public environmental awareness globally and across Europe and Central Asia. Voluntarily set-asides, driven in large part by the requirements of various production certification schemes, are also important for protected areas in Europe and Central Asia. For example, market-driven forestry certification schemes require that a certain fraction of the certified forest holding is set-aside for biodiversity conservation (often around 5% of the land holding, e.g. www.fsc.org). Certification systems highlight protection of forest areas as a means to maintain forest biodiversity (FSC, 2016; PEFC, 2010) and hence their national standards regularly include targets for voluntary set-asides. Both increased forestry certification as well as the adoption of national and global targets for protected areas have resulted in an increased area of formally protected forests and voluntary set-asides for biodiversity conservation purposes.

4.5.4.2.3 Activity of environmental non-governmental organizations

Environmental NGOs are among the key actors in shaping general public environmental awareness across Europe and Central Asia (Cashore *et al.*, 2003; Meidinger, 2003; Tysiachniouk, 2012; Tysiachniouk & McDermott, 2016). Public awareness has proved influential in creating a greater political prioritization of nature protection, as well as steering consumer preferences towards environmentally certified products. NGOs have also actively and directly lobbied industries and decision-makers to develop stricter (self-)regulatory frameworks for nature protection (e.g. marine protected areas) and to otherwise engage with various certification systems. In Eastern Europe, environmental NGOs – largely supported by foreign donors – contributed considerably to protected area development and management during the long post-Soviet transition towards a market economy. The Centre for Wildlife Conservation (1994), developed strategic and management plans for protected areas (e.g. Nature protected areas, 1998), designed regional protected areas and ecological networks, coordinated ecological restoration projects, as well as carrying out many other activities. However, the last decade has seen an increase in legal and administrative pressure on the activities of environmental NGOs in some Eastern Europe countries. For example, in Russia the official list of foreign NGOs permitted to operate in the country has been reduced by 7 times, and since 2008 consists of only 12 organizations; NGOs receiving any form of foreign funding are frequently classified as "foreign agents" (Shevchenko, 2016). Russian environmental NGOs have seen funding liquidated, or have otherwise been forced to gradually cease their activities (Yablokov & Zimenko, 2009). Under such circumstances, the activity of many NGOs cooperating with protected areas in Russia has decreased considerably (Bishop *et al.*, 2000; Brynych, 2016; Buivolov & Grigorian, 2006; Steppe fires and management of fire situation in steppe PAs, 2015; Stepanytskyy & Kreyndlin, 2004; Shtilmark, 2003) (**Figure 4.34**).

Figure 4 34 Environmental NGOs have had an important impact in building public awareness of the role of nature protection, leading to shifts in consumer preferences and political priorities. Source: Own representation.

The drive to environmental certification has been important provider of voluntary protection.



4.5.4.2.4 Adequacy of management resources for protected areas

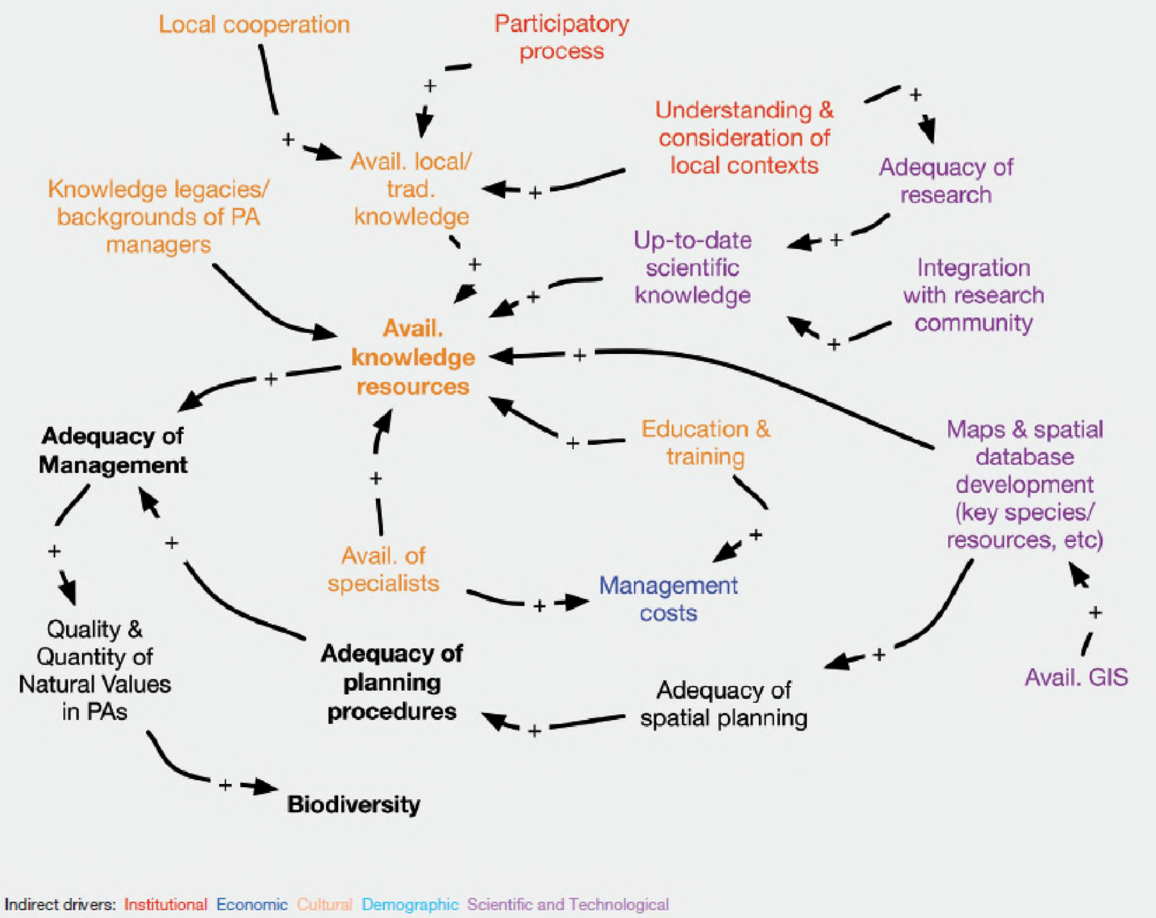
The availability of state-based funding for protected areas varies across Europe and Central Asia. In some countries state funding is insufficient for adequate management (Stepanytskyy, 1999, 2000). Funding from external bodies, for example, European Union environmental funds and international NGO funding, has in some instances bolstered protected area management budgets. In some countries there are, however, a number of institutional impediments to accessing such funding. Recent changes to laws in Russia (see **Box 4.3**), for example, have had a negative influence on protected areas funding (Shevchenko, 2016). Many protected areas also seek to augment management budgets by generating income opportunities based on protected area resources, for example through forestry or tourism. Managing these kinds of use often requires additional resources, and has an adverse impact on the natural values provided by the protected areas. In addition, acquisition costs for protected areas are generally much higher than annual management costs and have a strong impact on the financial resources available for protection (James *et al.*, 1999). As such, high land prices can present barriers for biodiversity protection in areas where land must be purchased prior to the establishment of protected areas, with intensive land uses generally associated with higher prices (Naidoo *et al.*, 2006).

Protected area management also requires sufficient training of managers or the procurement of a variety of specialists, both of which represent additional costs. Inadequate training of young specialists has been identified as a barrier to good management in some Eastern European countries (e.g. Mashin *et al.*, 2001), where the previous, Soviet-trained generation of managers is beginning to retire. Up-to-date scientific knowledge is partly dependent on taking local contexts into account in high-quality research. In addition to formal knowledge and training, the inclusion of local knowledge is seen as an important component in ensuring adequate management (Vdovin, 2016; Shulgin, 2007) (**Figure 4.35**).

Whilst staff are often driven by a strong desire to preserve unique natural values, low salaries (Ivanov & Chizhova, 2003) together with often poor working conditions and a general lack of focus on long-term capacity building, this has led to the demotivation of staff (Mashkin, 2007; Sidenko, 2010). Many protected areas are also reliant on the contribution of civil sector volunteers (e.g. members of NGOs or local communities). However, the degree to which these human resources are permitted to contribute to protected area management is partly dependent on the inclusion of suitable participatory mechanisms in the overall governance and management approach.

Figure 4 35 **Knowledge and planning resources are essential to adequate protected area (PA) management. Source: Own representation.**

The education and training of staff and integration of management with both research and local communities are key strategies.



Specialized equipment (e.g. GIS, computerized species-monitoring systems) is often required to establish the baseline data for, or monitor the impacts of, protected area strategies and plans. Other more generic forms of technology, such as suitable vehicles, and infrastructure, such as protected area management offices, are also required inputs. In broad terms, many Eastern European and Central Asian protected areas suffer from poor quality or out-dated equipment, infrastructure and vehicles, or lack these entirely.

4.5.4.2.5 Local resistance

A major factor affecting the establishment or successful management of protected areas in Europe and Central Asia relates to the manner in which they navigate local use conflicts arising as a result of protection status and management (Babai *et al.*, 2016). Protected area governance and management regimes are often characterized as top-down with low levels or quality of public participation; inflexible responsible authorities and insufficient consideration of the local context; engendering negative public perceptions; and resistance amongst members of local communities (Blicharska *et al.*, 2016; Carrus *et al.*, 2005; Elbakidze *et al.*, 2013c; Grodzinska-Jurczak & Cent, 2011; Mathevet *et al.*, 2016). These factors pose significant challenges to the functionality of protected area networks (Blicharska *et al.*, 2016; Elenius *et al.*, 2017; Stenseke, 2009).

Local resistance to protected areas can be related to in-group/out-group identity processes, e.g. local communities vs central governmental authorities (Bonaiuto *et al.*, 2002; Stoll-Kleemann, 2001), or from the perceived loss of user rights as a result of protected areas' restrictions (James *et al.*, 1999). For the latter reason, land owners tend to oppose establishment of protected areas to a greater extent than other stakeholders (Brescancin *et al.*, 2017; Kamal & Grodzinska-Jurczak, 2014), particularly in countries where social values are strongly linked to long histories of private ownership. At the same time, local identity in some cases is also linked to reduced local resistance due to strong socio-cultural links to nature (Carrus *et al.*, 2005; Uzzell *et al.*, 2002).

The mutual dependence of extensive land use and conservation management has become apparent in the last 20-30 years. Small-scale extensive land use often survives in protected areas only, in the form of conservation management, and is largely side-lined in regulatory frameworks. Regulations introduced to protect such areas often apparently do not consider local world views, or the effects of local practices. This results in the restriction of local people's activities (Babai *et al.*, 2016; Molnár *et al.*, 2016) and conflict between locals and the protected area's authority (Kelemen *et al.*, 2013). The adoption of a more integrated, participatory approach to the governance and management of protected areas is suggested as a potential remedy to local use conflicts, particularly in protected areas established in cultural, small-scale, or indigenous landscapes. There is a need for "hybrid people" who have knowledge of traditional practices and world views, as well as of mainstream nature conservation ideas (Molnár *et al.*, 2016). Additionally, the introduction of agro-environmental schemes in protected areas can mitigate the loss of traditional management practices and so prevent biodiversity loss accompanying land abandonment (Babai *et al.*, 2015). One approach might be for landscape- and culturally-specific agricultural regulatory frameworks and subsidy systems that include local and traditional knowledge to produce tailored local solutions that respect the strong link between natural and cultural capital (Molnár & Berkes, 2017) (**Figure 4.36**).

Marine protected areas appear to have been more successful than terrestrial ones in combining conservation plans and management practices with visible economic benefits in terms of long-term fishery management and diving-based tourism. Marine protected area design takes greater account of geographical and cultural contexts in which users are situated (Fenberg *et al.*, 2012; Gabrié *et al.*, 2012; Pastoors *et al.*, 2000; Planes *et al.*, 2006; Portman *et al.*, 2012). However, while aiding local acceptance of marine protected areas, a strong consideration of the needs of multiple users within the local context has potentially led to the protection of areas of lower inherent conservation value (Coll *et al.*, 2012).

Figure 4 36 Consideration of local contexts, knowledge and identities in the design and establishment of protected areas (PAs) helps to avoid local resistance. Source: Own representation.

4.5.4.2.6 Armed conflicts

Armed conflicts have multiple negative impacts on biodiversity and nature's contributions to people. Europe and Central Asia is unfortunately the arena for a number of recent and current armed conflicts (Vasyliuk *et al.*, 2017). Whilst few studies have been conducted in the region on the specific effects of armed conflict on protected areas, the environmental effects are presumed to be identical to those in non-protected areas and include the various forms of direct environmental damage associated with the use of heavy weapons and military equipment, as well as a number of effects resulting from sudden changes in land-use regimes (see **Box 4.4**). It is apparent that legal protection status is not well-respected during times of armed conflict. Studies of conflicts outside of Europe and Central Asia suggest that protected areas, which are often remote or difficult to access, serve as refuges for fighting forces, and as such are key targets for opposing forces (D'Huart, 1996). In addition, armed conflicts exacerbate poaching pressure and other illegal use, immediately eliminate tourism activities, and drain financial and human resources from ecosystem management (Baumann *et al.*, 2015; D'Huart, 1996; de Merode *et al.*, 2007; Dudley *et al.*, 2002) (**Figure 4.37**).

Box 4.4: Consequences of armed conflicts for biodiversity and nature's contributions to people - example from Ukraine.

Since 2014, armed conflict in the eastern region of Ukraine (Luhansk and Donetsk), in addition to a large number of human casualties and the destruction of infrastructure, has led to extensive habitat loss in existing protected areas, largely due to:

(1) Heavy military machinery driving or otherwise operating in protected areas.

(2) Explosions of munitions, resulting in the destruction of vegetation and accumulation of debris and chemicals in soils - primarily sulphur and heavy metals, e.g. experts counted about 15,500 craters from explosions in the regional landscape park "Donetsk ridge".

(3) Construction of military infrastructure, e.g. training grounds and trenches, within protected areas.

(4) Illegal logging for military purposes and fires. Pine forests of the steppe zone of Ukraine are extremely fire-prone. About 3000 fires occurred in the military zone within protected areas during 2014. Roughly half of all protected areas in the war zone are fire-damaged.

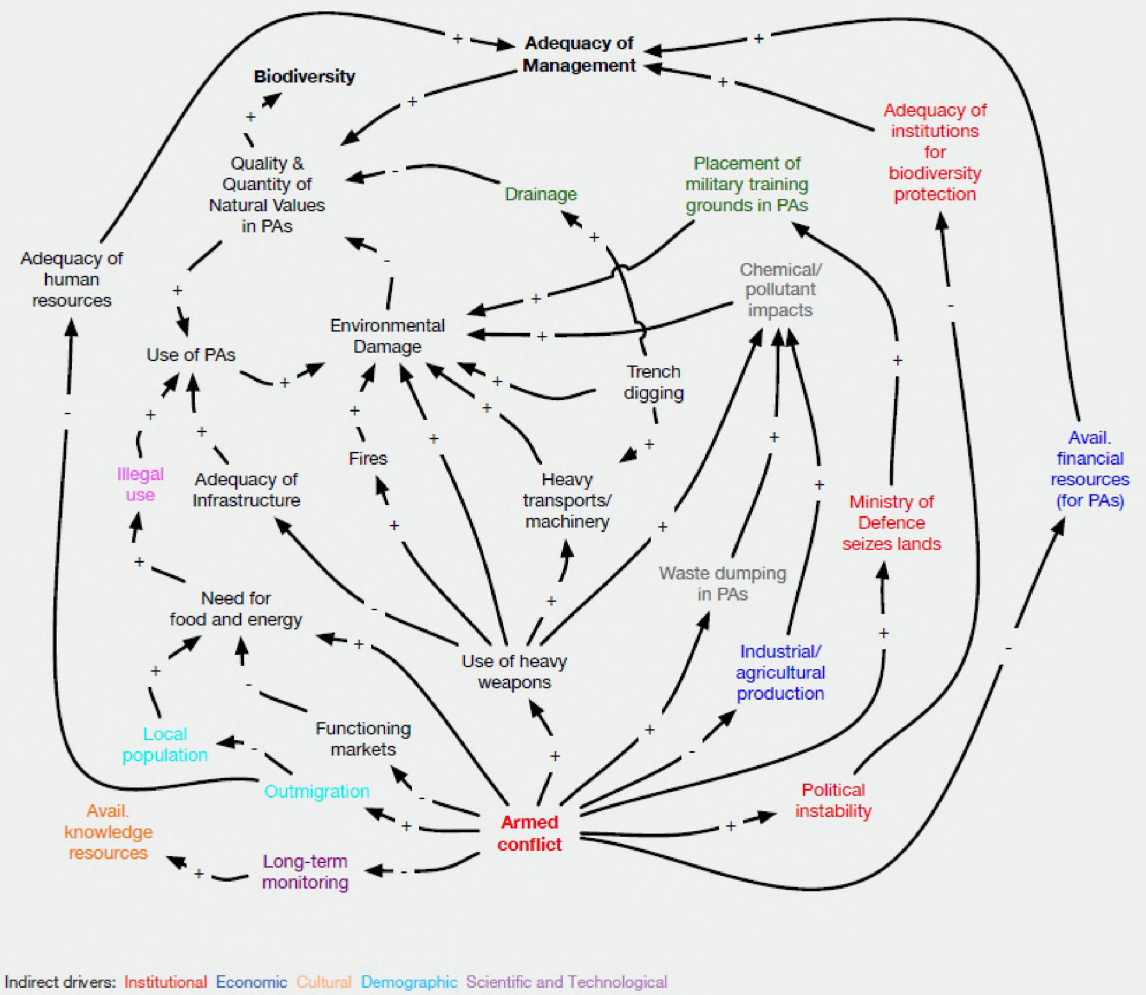
(5) Illegal logging by local people for domestic needs, associated with the destruction of regional heating systems and gas supply; as well as for the construction of defensive infrastructure. This has resulted in intensified wind erosion and dust storms.

(6) Use of protected areas for waste storage/ dumping.

In addition, much of the institutional framework underpinning protected area governance and management in the annexed areas has been lost, and many employees have resigned. The war has also indirectly led to major reductions in national budgets for protected areas, both within and outside of the conflict zone (Melen'-Zabramna *et al.*, 2015).

End of Box 4.4

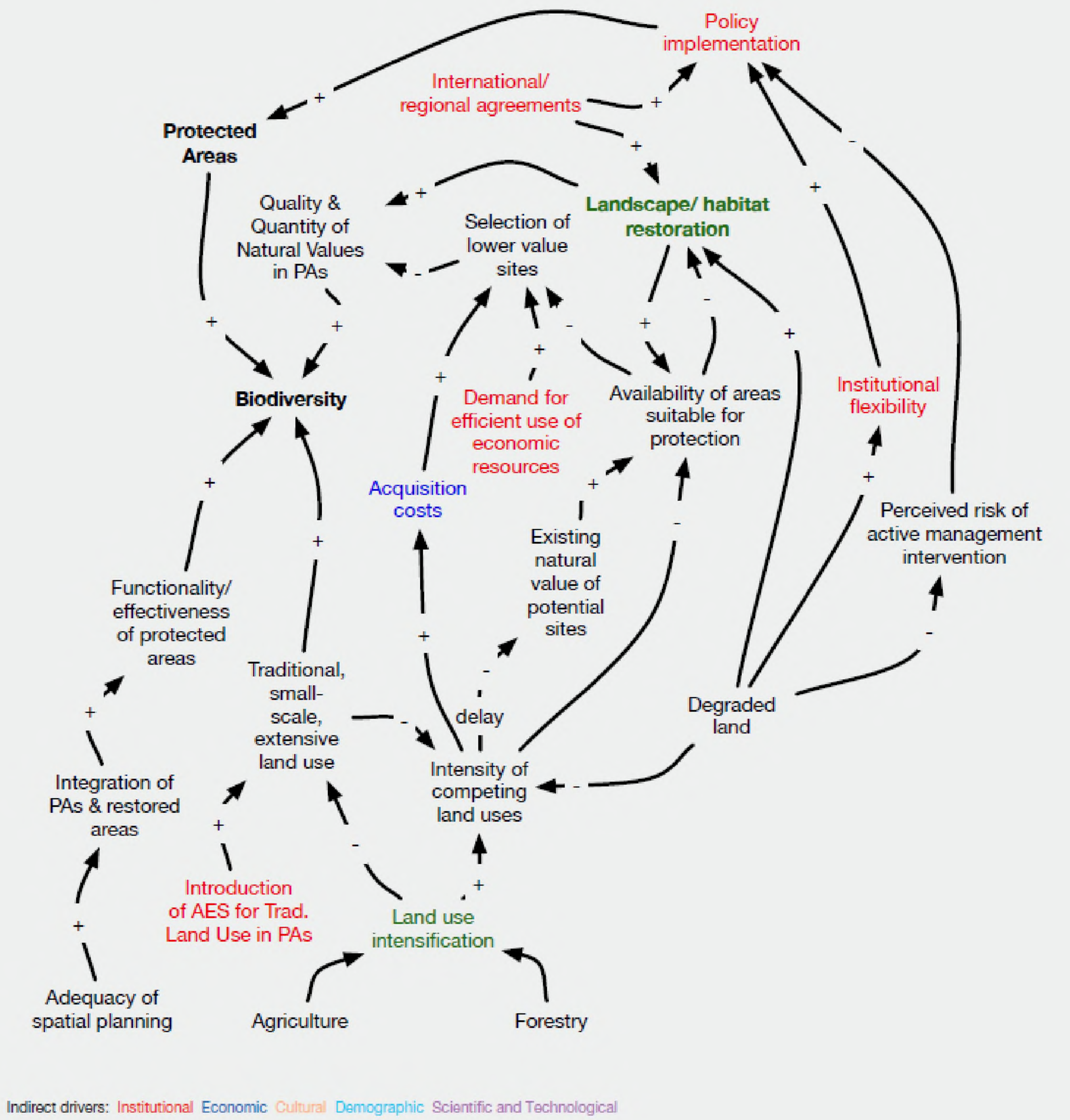
Figure 4.37 Armed conflict has many deleterious effects on protected areas (PAs), including multiple direct and indirect environmental impacts, diversion of economic resources from protected area budgets, loss of institutions and human resources, and interruption of long-term monitoring. Source: Own representation.



4.5.4.2.7 Landscape and habitat restoration

Landscape and habitat restoration offers opportunities for nature conservation and protected area development. For certain habitat types, restoration activities are prescribed to secure sufficient areas for protection and for meeting Aichi Biodiversity Target 11. High rates of land conversion, including loss of cultural landscape habitats dependent on traditional land use (Hartel & Plieninger, 2014) and the expansion of modern forestry into remnants of natural forests in the northern part of Western Europe (Naumov *et al.*, 2017), implies the continued loss of high-quality areas suitable for protection. Lack of suitable areas combined with demands for efficient use of limited resources for protection leads to the consideration of sites/areas of lower natural values in terms of biodiversity and nature's contributions to people. This includes, for example, expanding existing reserves with adjacent areas of lower conservation value, but providing long-term benefits by succession or active conservation (Mazziotta *et al.*, 2016; Polasky *et al.*, 2008) (**Figure 4.38**).

Figure 4 38 The restoration of degraded land can be an important opportunity for nature protection in some regions, particularly where land prices are high or land uses are intensive. PAs: Protected areas. Source: Own representation.



The restoration of degraded land is a part of the Strategic Plan for Biodiversity 2011-2020 (specifically Aichi Biodiversity Target 15 – "ecosystems restored and resilience enhanced") and is included in the European Union's biodiversity strategy; both calling for restoration of at least 15% of degraded ecosystems. Degraded lands may offer multiple opportunities for restoration projects, including lower land prices, fewer current users and greater support for active management interventions, lower perceived risks, and greater institutional flexibility (Dawson *et al.*, 2017).

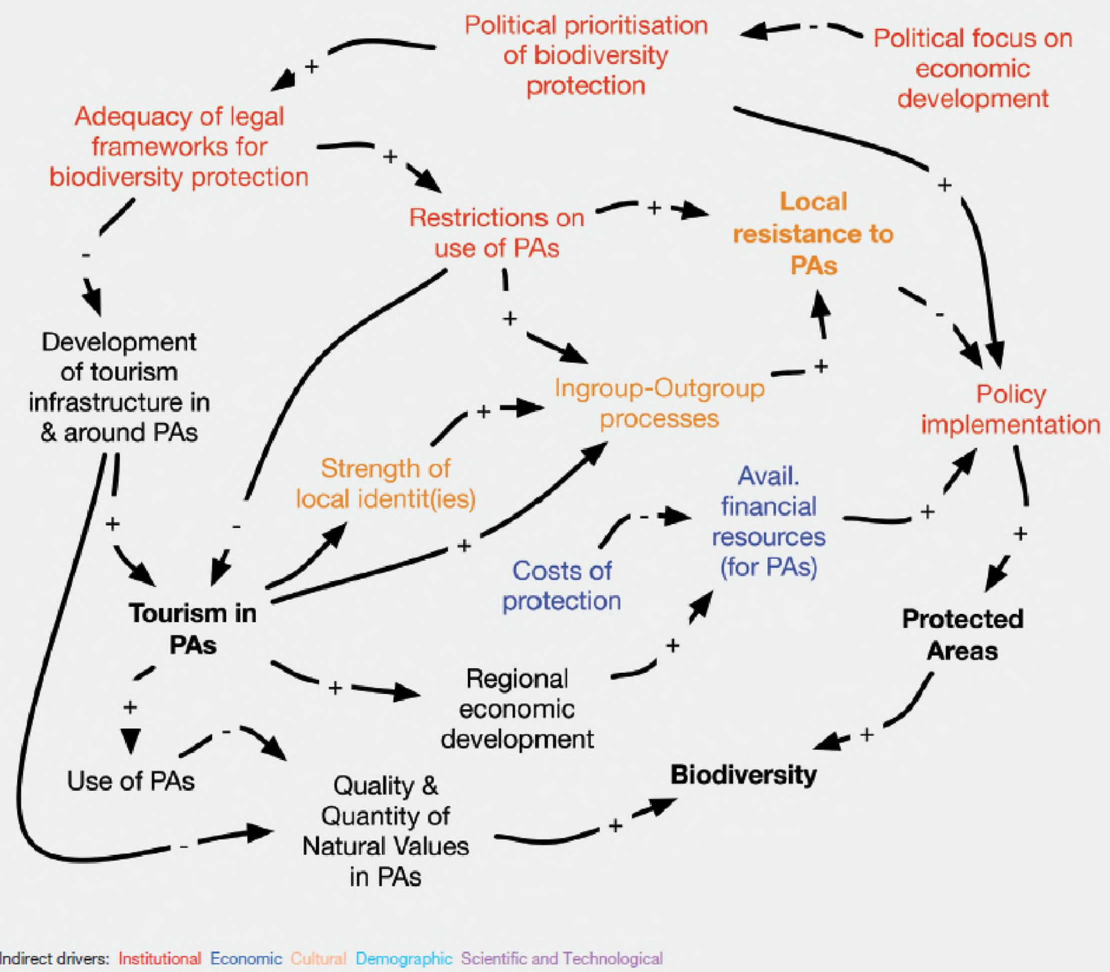
4.5.4.2.8 Tourism

Tourism opportunities can provide a political incentive for protected area establishment, due to the possibility of offsetting protection costs with sought-after rural socio-economic development

(Sevastiyanov *et al.*, 2014; Svoronou & Holden, 2005; Zachrisson *et al.*, 2006). An example is the creation of new diving tourism opportunities associated with marine protected areas. However, the introduction of new user restrictions for local residents, while at the same time opening up the area for new users (tourists), may reinforce insider-outsider dynamics or otherwise engender local resistance and conflicts (Colchester, 1997; Cortes-Vazquez, 2014). For example, a number of studies note that urban populations tend to adopt a more dualistic perspective regarding human-nature relations, supporting calls for more protected areas with less human intervention in their management (Coleman & Aykroyd, 2009; Cortes-Vazquez, 2014; Linnell *et al.*, 2015). Additionally, the transition from a staple economy to jobs based on amenity values, outdoor recreation and tourism can also be challenging for many local rural communities (Westlund & Kobayashi, 2013). Recent legislative amendments in Russia (see above) have opened protected areas up for tourism, ostensibly as a means to improve their economic situation (Boreyko *et al.*, 2015; Chibilev, 2014; Shtilmark, 2014). The engagement of strict nature reserves, for example, in commercial activities (primarily tourism) has led to numerous attempts to violate the nature protection regimes in both federal and regional protected areas, and UNESCO World Heritage sites, including illegal construction of tourism-related infrastructure (Stepanytskyy & Kreyndlin, 2004) (**Figure 4.39**).

Figure 4 39 **Tourism offers opportunities for regional economic development offsetting some of the costs of protected areas (PAs). Source: Own representation.**

However, tourism and related infrastructure have negative impacts on the natural values under protection, as well as encouraging the erosion of protection policies. Encouraging tourist use while imposing restrictions on local use can result in conflicts.



4.5.5 Trends and indirect drivers of changes in traditional land use

4.5.5.1 Trends in traditional land use

Traditional land use encompasses multiple non-intensive, locally adapted land-use practices based on local and indigenous knowledge that have played a significant role in the development of diverse, productive and sustainable food and material production systems (Molnár & Berkes, 2017; Parrotta & Sunderland, 2015; Parrotta *et al.*, 2016; Plieninger *et al.*, 2006). In Europe and Central Asia, traditional land-use practices, including forest management, agricultural activities, and agroforestry, have influenced nature over millennia, leading to the development of diverse ecosystems and cultural landscapes favouring a range of semi-natural and natural habitats and associated plant and animal species (Aitpaeva *et al.*, 2007; Fedorova, 1986; Kile, 1997; Laletin, 1999; Saastamoinen, 1999; Saastamoinen *et al.*, 2000; Taksami & Kosarev, 1986; Turnhout *et al.*, 2012; Dmitriev, 1991) (**Figure 4.40**).

Figure 4 40 **The traditional village system found in Eastern Europe's forest and woodland landscapes is characterized by a centre-periphery zoning from houses, gardens, fields, mowed and grazed grasslands to forests (i.e. the ancient system with domus, hortus, ager, saltus and silva), as illustrated in this view of the village Volosyanka in the Skole district of Ukraine's west Carpathian Mountains.**

Beginning in the left part of the picture, the church in the very village centre is surrounded by houses that are located in the bottom of the shallow valley. The private gardens have many fruit trees and shrubs. Further to the right there is a fine-grained mixture of grasslands, some individual fields of which some have been mowed and have hay-stacks and some not yet, and fields, like the potato field in the foreground. Further, to the right there is forest, which is grazed by cattle moving in and out along specially designed fenced trails from the farm houses in the valley bottom. In addition, above the tree line on the top of the mountains, there are open grazed pasture commons. Photo: Per Angelstam.

Since the 1950s, agricultural practices across the region and traditional land use have undergone substantial changes (EEA, 2015a; Van Zanten *et al.*, 2014). There are two main trends in traditional land-use systems in Europe and Central Asia: (1) substantial decrease in land area with traditional land use and loss of traditional ecological knowledge; and (2) maintenance of traditional practices and adaptation of traditional ecological knowledge to new ecological and socio-economic conditions.

Trend 1: Substantial decrease in land area with traditional land use and loss of traditional ecological knowledge

The land area, where traditional practices are still applied has substantially decreased in many regions of Europe and Central Asia (Rotherham, 2007) as a result of socio-economical changes and land-use intensification However, many practices have survived on marginal lands, in protected areas, or as a result of socio-cultural preferences (Juler, 2014; Lieskovský *et al.*, 2014; Molnár *et al.*, 2016). For example, transhumant herding, once dominant practice in most mountainous areas in Western and Central Europe, has undergone a sharp decline but has still survived some regions due to cultural traditions (e.g. in Romania - Juler, 2014) or as a part of organic farming activities (Evans, 1940; Juler, 2014; Thompson *et al.*, 2006). Other, more sedentary forms of herded grazing have for example survived in the vast steppe areas of Hungary (Kis *et al.*, 2016; Molnár, 2014). Traditional agro-silvicultural systems, including wood-pastures and coppicing, have almost completely disappeared in Western and Central Europe, as well as management of forest commons according to ancient regulations (Kirby & Watkins, 2015; Rigueiro-Rodríguez *et al.*, 2009). Traditionally managed wood-pastures have partly been preserved in Romania (Hartel *et al.*, 2015), but are also in decline. For example, traditional multi-species fruit orchards with ancient varieties and a species-rich semi-natural grazed herb layer are also in decline, but have begun to revive over the past two decades in Romania (Antofie *et al.*, 2016). Semi-natural grassland ecosystems in Western, Central and Eastern Europe have been largely converted to agricultural fields, afforested or abandoned, depending on the region, though agri-environmental schemes of the European Union may help some to survive. For example, mountain meadows in the Carpathians (examples of the most species rich grasslands on Earth) are mostly abandoned (Babai *et al.*, 2015; Dengler *et al.*, 2014; Ivaşcu *et al.*, 2016) (**Box 4.5**). In Estonia, traditionally managed semi-natural grassland habitats (wooded meadows, coastal grasslands, floodplain meadows, dry and mesic grasslands) covered about 1.5 million hectares (35% of the country) in 1950s (Kukk & Kull, 1997). Since then, some areas have been turned into cultivated land but most overgrew with forest following the abandonment. By 2010, only 60'000 hectares of semi-natural habitats (4% of their coverage in 1950s) remained, of which only 30'000 ha was under appropriate management. However, the area under management has been increasing in past decades with the help of targeted subsidies (Management Plan for Estonian Semi-natural habitats 2014-2020).

Trend 2: Maintenance of traditional practices and adaptation of traditional ecological knowledge to new ecological and socioeconomic conditions

The essence of traditional practices and traditional ecological knowledge has been preserved or adapted with new ecological and socioeconomic conditions in many marginal areas (e.g. mountains, dry areas, taiga-tundra) across Europe and Central Asia. For example, in Eastern Europe, land-use systems based on beliefs, customs, norms, bans, and rules of natural resource use are maintained by numerous indigenous and local communities (Kile, 1997; Taksami & Kosarev, 1986; Turaev *et al.*, 2005). In a survey of more than 500 respondents from Central Siberia Vladyshevskiy *et al.* (2000) have shown that in the last years of the twentieth century the use of wild mushrooms and Siberian pine nuts increased from two- to threefold; the use of wild onion three- to fivefold; and berries one and a half to two times. In forest depending communities, non-timber forest products are often the main source of food and income for village populations, representing as much as 30–40% of family income (Laletin *et al.*, 2002).

Box 4.5: Nature is becoming wild – local perceptions of loss of traditional land use and its drivers in European cultural landscapes.

Box 4 5 If traditionally managed hay meadows are abandoned, pioneer forests develop B. Forced abandonment in national parks is often a source of conflict between locals and conservationists. In the case of the eastern Carpathians, however, both of them prefer the managed cultural landscape which is «in order» A according to locals and rich in species according to conservationists. Photo: Ábel Molnár.

A B

Traditional small-scale farmers developed fine-scale multifunctional cultural landscapes all over Europe (Agnoletti, 2006). With global changes, cultural landscapes are often abandoned or transformed into urban or more intensively managed agricultural areas. If abandoned, natural processes may accelerate, native shrubs and trees and invasive alien species may spread. Local farmers often perceive these changes as a landscape-in-order where "each corner had a role" is changing into a landscape-in-disorder. Independently whether succession is going through more and more natural or degraded stages, locals perceive the process as "getting wild" meaning the intensity of ecosystem service use decreases (Babai & Molnár, 2014; Molnár, 2014). Wild place is a specific folk habitat: under this expression local people understand an area with no or little human utilization. Wild places are e.g. narrow steep valleys where no livestock can graze and timber is difficult to get out, or marshes dominated by tall tussock sedges, which are difficult to cross, impossible to cut for hay and where livestock can drown (Babai & Molnár, 2014; Kis *et al.*, 2016; Molnár, 2014). Abandoned pastures with accumulating litter and encroaching shrubs also are areas that turn into wild. National parks manage their lands in many different ways to help protected species and natural regeneration. If cultural landscapes in national parks are managed in a way where agricultural use is abandoned, local people often argue: the park manages the landscape improperly by letting it turn wild (Bérard *et al.*, 2005). These differences in understandings of "proper" landscape management may cause conflicts between authorities and locals (Babai *et al.*, 2016; Kelemen *et al.*, 2013).

End of Box 4.5

Pastoralists in mountainous regions of Central Asia practice so-called vertical and horizontal migrations (transhumant) of livestock (Alimaev *et al.*, 2008; Kanchaev *et al.*, 2003). Livestock mobility, which is a main feature of traditional pastoralist patterns, is key for the sustainability of pasture management (Galvin *et al.*, 2008; Robinson *et al.*, 2016). Traditional knowledge in Central Asia has been widely used to control desertification and soil erosion in mountain areas. In Tajikistan, where the use of stepped terraces has a 1,000-year history, planted forests are widely used for stabilization of hill slopes (Civil Initiatives Support Fund, 2006). Methods for slope terracing and cultivation of fruit and nut gardens, especially in the traditional system of land and water management known as boghara, has been known to inhabitants of mountains since ancient times. Throughout the region, traditional techniques including shelterbelts have been used to control windblown sands in the vicinity of settlements.

In the forest regions of the Caucasus, where pastures and haymaking resources are limited, local people use a traditional "pasture turnover" system for regulated forest grazing. The creation of cultural pastures in open areas within forests increases animal productivity while preventing damage to sprouts and seedlings of valuable species (i.e., oak, ash, maple, beech) due to grazing in young, naturally regenerating forest stands. Once regenerating trees attain heights sufficient to prevent their damage by livestock, these forests are used on a temporary basis for grazing, while previously used pastures are managed to encourage restoration of forest cover and growth of valued tree species through natural regeneration (Eganov, 1967).

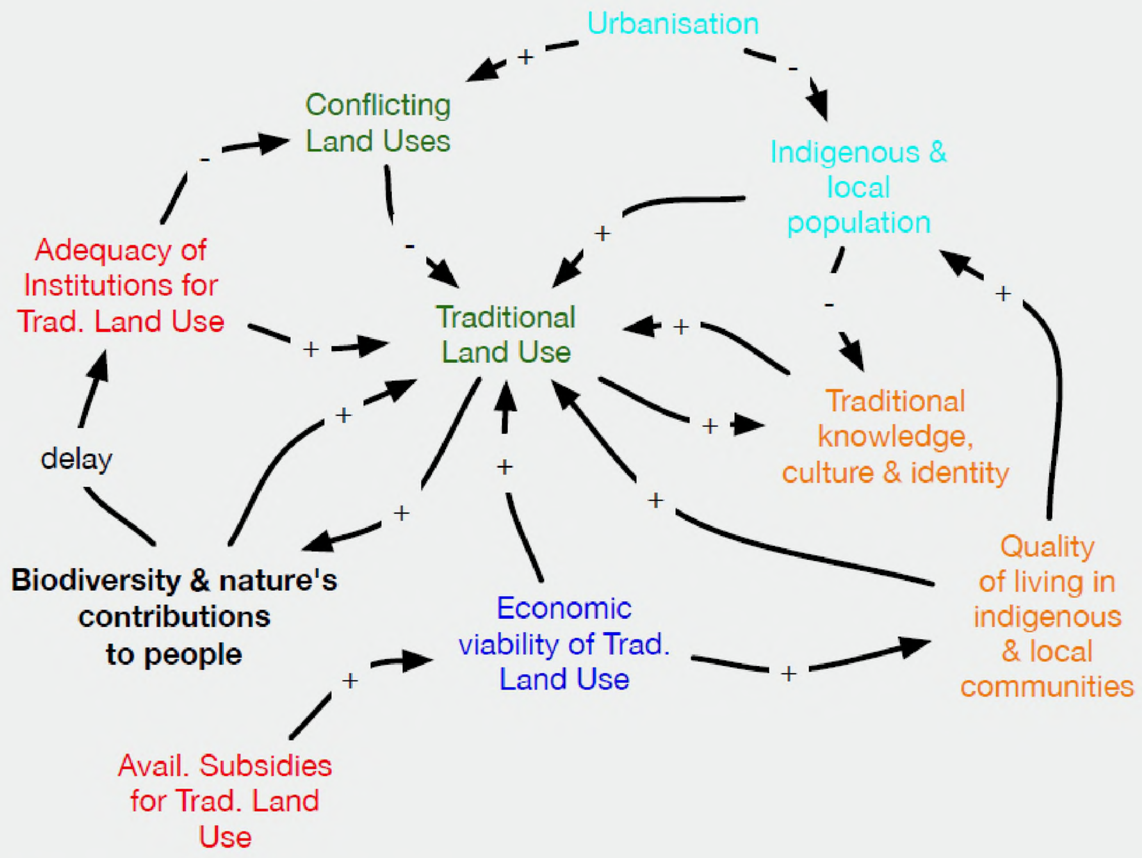
Sacred sites are common throughout Europe and Central Asia where indigenous and local communities still thrive (Bocharnikov *et al.*, 2012). Such sites may range in size from small groves or even individual trees to extensive forested landscapes. Some areas are considered sacred because they provide major habitats for species with ritual or medicinal values. The protection of such sites is important for the health and spiritual well-being of local communities (Samakov & Berkes, 2016). Protection of forest resources based on religious beliefs is characteristic for Central Asia, where the sacralization of nature is expressed in cultural traditions and practices connected with particular species and sites (Aitpaeva *et al.*, 2007).

4.5.5.2 Drivers of trends in traditional land use

Multiple drivers have underpinned traditional land-use change across Europe and Central Asia. These drivers are mainly context specific and differ across the region (**Figure 4.41**).

Figure 4 41 Institutional adequacy and economic viability are key drivers of traditional land use in Europe and Central Asia. Source: Own representation.

Demographic trends, including urbanization, continue to diminish indigenous and local populations, with concomitant negative impacts on traditional knowledge, culture and identities. These inter-relationships are further unpacked in a series of sub-models below (**Figure 4.42** and **Figure 4.43**).



4.5.5.2.1 Institutional drivers of trends in traditional land use

In Central Europe, the European Union's agricultural subsidies have a positive effect on grassland management; many areas abandoned in the 1990s (after collapse/dis-integration of the Soviet Union) are now grazed, mown and cleared of shrubs. In marginalized villages of Central Europe agri-environmental payments are a vital source of income for farmer families. However, culturally and ecologically less adapted regulations for traditional management have diverse side effects – hay meadows are turned into sheep pastures (Csergo *et al.*, 2013), or old trees are cut on wood-pastures (Hartel & Plieninger, 2014). Revival of folk traditions among the youth in cities (e.g. folk singing, folk dancing) may provide a background for the maintenance of traditional practices in rural areas. Back-to-the-country movements are, however, hindered by ecologically and culturally inappropriate regulations (Babai *et al.*, 2015). Recognition of and respect for viable and useful traditional management practices is vital, otherwise farmers may be reluctant to maintain or reintroduce them in their everyday management (Sereke *et al.*, 2016) (**Figure 4.42**).

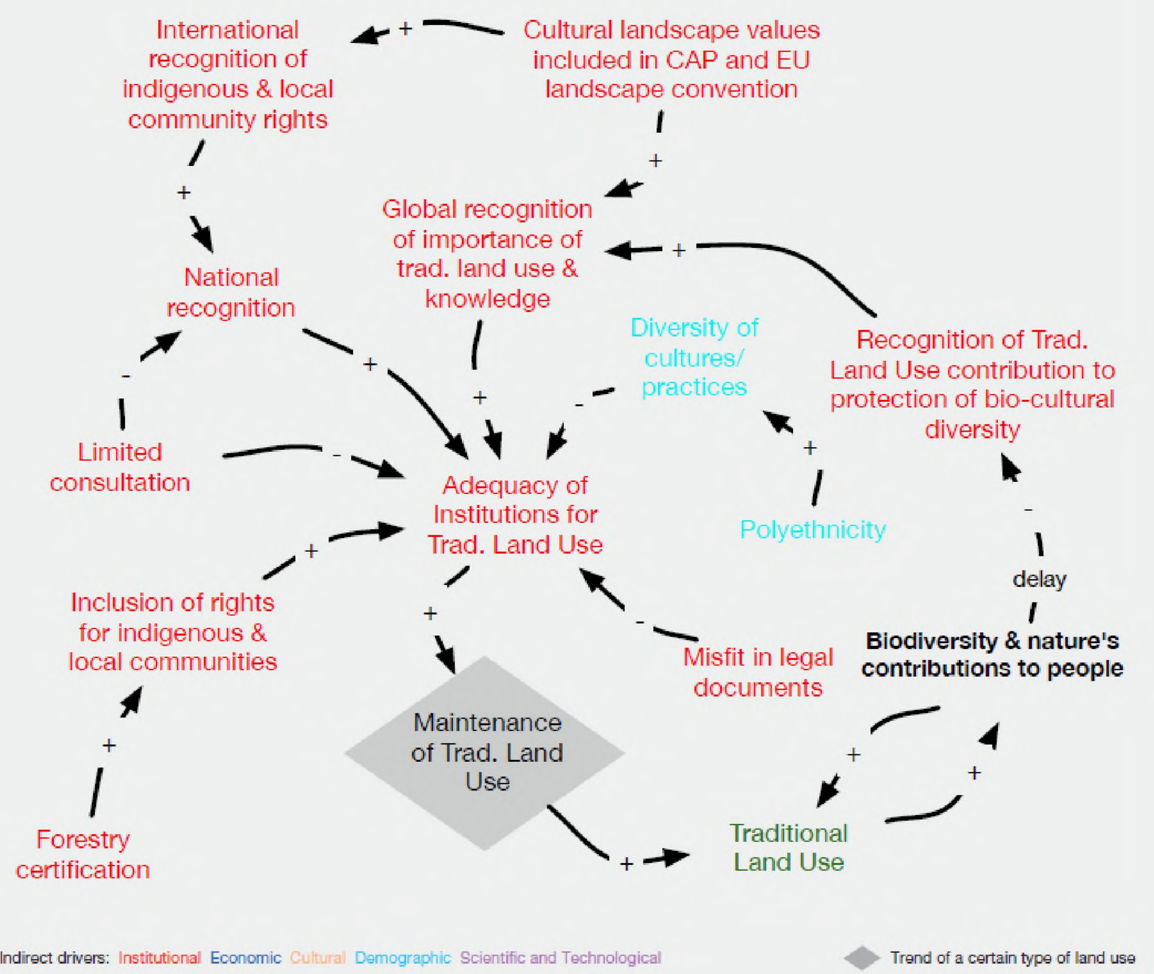
In Eastern Europe and Central Asia traditional land use has been especially affected by radical changes in the political system. In recent years, the indigenous peoples of Russia have been trying to restore their traditional livelihoods through legal efforts. The Russian Constitution contains the concept of "indigenous minorities", whose rights are guaranteed by the Russian Federation in accordance with the generally acknowledged principles and norms of international rights and international agreements. The Russian legislation ensures a new status for indigenous peoples by providing enabling conditions for traditional nature resource use within the so-called Territories of Traditional Nature Resource Use for indigenous peoples. These territories are designated to ensure environmental protection and to support indigenous livelihoods, religion, and culture. The legal norms for these territories are related to the various natural resource uses, such as reindeer breeding, hunting, fishing, and non-timber forest product collection, within different territories (Sulyandziga & Bocharnikov, 2006). However, there are no norms ensuring the preservation and use of traditional knowledge, especially in the management of traditional natural resources. During the preparation of the Strategy and Executive Plan for the Conservation of Biodiversity within the Russian Federation the new goal was formulated to ensure the maintenance of traditional lifestyles and the sustainable use of biodiversity by indigenous peoples, including consideration of traditional knowledge in the planning and implementation of activities related to use of biological resources (Ministry of Natural Resources and Environment of the Russian Federation, 2014). In Central Asia, a shift from state command-and-control economy to market-based economy led to the concentration of a large number of livestock in few hands, which left the majority of households in possession of small numbers of animals (Robinson *et al.*, 2016; Vanselow *et al.*, 2012). To make the use of migratory routes economically viable, the households with a small number of animals revived the traditional models of pooling animals from many households and shepherding them on a rotational basis or hiring a shepherd among themselves (Robinson *et al.*, 2016). As livestock numbers have started to recover following the hardship of early independence years, pasture management issues are becoming more urgent. Having recognized the value of traditional migratory grazing patterns and importance of livestock mobility in sustainable use of pastures, for example, countries in Central Asia have designated pastures as common property, and management of common pastures is exercised by a locally elected pasture users committee.

4.5.5.2.2 Economic drivers of trends in traditional land use

In Western Europe, the traditional practice of collecting non-timber forest products for wild food and medicine has been declining due to emigration to urban areas to pursue economic opportunities, to mass production of food and to modern synthetically produced medicines (Łuczaj *et al.*, 2012; Quave *et al.*, 2012; Schulp *et al.*, 2014). In some places, however, there are markets for wild plants and mushrooms (Richards & Saastamoinen, 2010; Sitta & Floriani, 2008). For these products, a market demand and viable industry exists, although affected by variation in harvests from year-to-year and sensitive to labour costs. Some of these markets are also dominated by imports (e.g. the Italian market of *Boletus*; Sitta & Floriani, 2008). Estimates of the value of non-timber forest products indicate that these may be in the same order of magnitude as traditional timber products. For instance, if forest management would take bilberry production into account in Finnish forests the economic gain during the rotation period could, theoretically, more than double (Miina *et al.*, 2016). It has also been shown that urban citizens demand a market for non-timber forest products and there can be a considerable demand for such products among urban consumers. This especially concerns food where quality and environmental friendliness is seen as important attributes (Kilchling *et al.*, 2009) (**Figure 4.43**).

Figure 4 42 **Growing international recognition of the rights of indigenous and local communities, and of the contribution of traditional land-use practices in protecting bio-cultural diversity, have improved a number of institutions concerning traditional land use in Europe and Central Asia. Source: Own representation.**

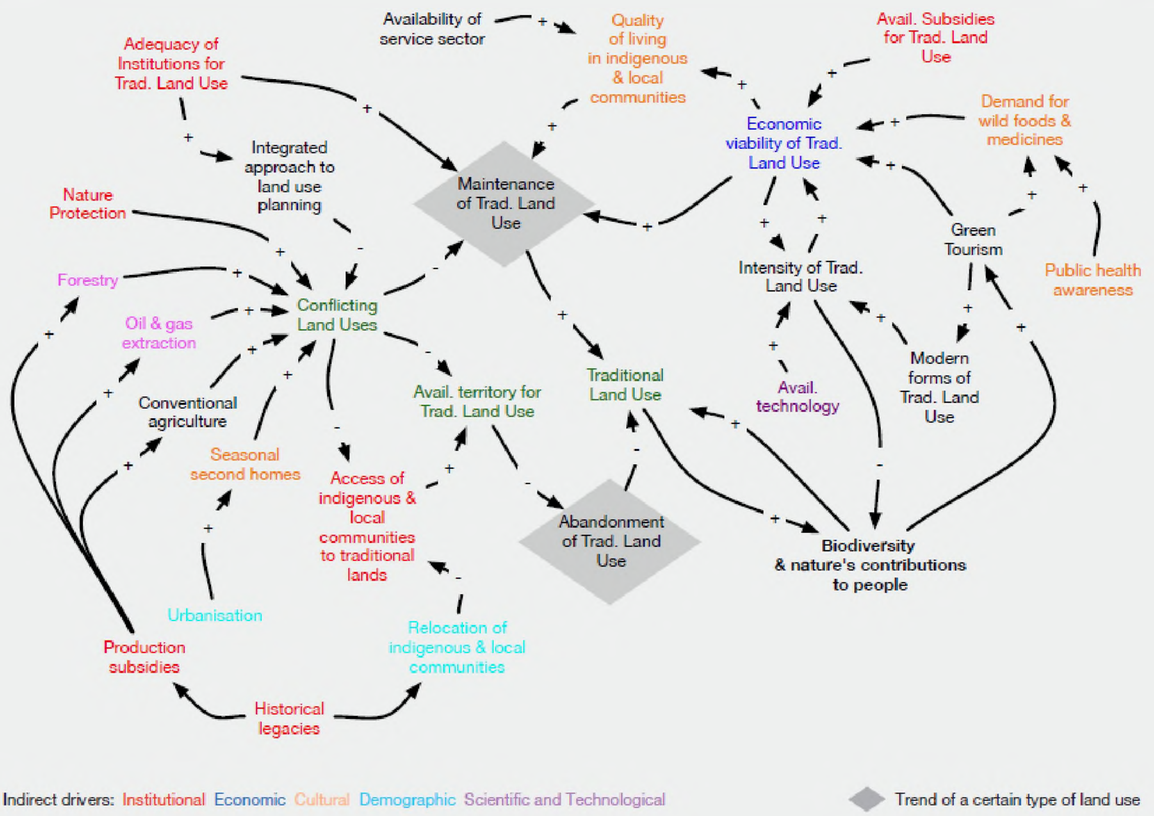
Certification schemes have led to greater inclusion of indigenous and local rights in forest management systems. Poly-ethnic regions, with a broader diversity of cultures and practices to account for, face additional challenges in developing adequate institutions for traditional land use.



In the Russian Federation among the economic drivers that negatively affect the traditional land use of indigenous people is reduction of areas of traditional indigenous settlements due to industrial development. The Committee on the Affairs of the Indigenous Peoples of the Russian Federation in the materials for the Parliamentary hearings on Legal provision of technological expertise (2007) stated "From the 1930s the structure of nature resource use and the concept of development of the North gave priority to industrial development instead of the traditional land use, which resulted in severe pollution and degradation of the natural environment that led to the disruption and retirement of the most valuable agriculture land. First of all, significant damage was done to reindeer pastures. One factor that destabilizes the ecological situation in the area of traditional land use is stressful influence of industrial facilities on deer pastures and hunting grounds, covering up to 40% of the area of "traditional land use". Due to industrial development and pollution by industrial emissions of the traditional land-use area, the rural population lost not only pastures and hunting grounds, but also traditional fishing areas and areas for gathering wild plants" (Ayzan *et al.*, 2011).

Figure 4 43 **Production-based subsidies driving growth in agricultural, forestry and natural resource extraction sectors tend to exacerbate conflicting land-use issues, often impinging on available territory for traditional users. Source: Own representation.**

Maintenance of traditional land use is dependent on the quality of living that traditional lifestyles afford to indigenous and local communities, in turn dependent on the economic viability of traditional land-use practices and the availability of services. The growth of green tourism and demand for products derived from traditional practices, the modernization of practices, and the availability of subsidies for traditional land-use practices are important factors in ensuring the economic viability of traditional practices.



4.5.5.2.3 Social drivers of trends in traditional land use

Often, people leave rural areas for higher education and higher salaries in cities. Even people living in villages pursue an urban lifestyle. In Western Europe, the last few decades' health industry development and alarm about unhealthy additives in mass-produced food have resulted in a renewed interest in wild food and medicine (Mabey, 2001; Reynolds Whyte *et al.*, 2002). Wild food is considered pure, naturally healthy and rich in vitamins and antioxidants (Łuczaj *et al.*, 2013). Moreover, wild plants and mushrooms play an important role as spices and accompaniments in traditional cuisines in the region (Łuczaj *et al.*, 2013; Sõukand *et al.*, 2013; Stryamets *et al.*, 2015; Svanberg, 2012). There is also a growing interest in folk medicine in different parts of the region (DuBois & Lang, 2013; Ghirardini *et al.*, 2007; González-Tejero *et al.*, 2008; Łuczaj *et al.*, 2013; Vitalini *et al.*, 2009), even where collecting plants for medicinal purposes is no longer a widespread practice (Łuczaj *et al.*, 2012; Molina *et al.*, 2009; Quave *et al.*, 2012; Rigat *et al.*, 2007).

4.5.6 Trends in urban development

Urban populations are foreseen to increase considerably across Europe and Central Asia (United Nations, 2014), which may cause further urban sprawl, depending on urban planning policies. In Europe, urban sprawl has increased considerably over the past decades. Between 2006 and 2012, semi-natural and natural areas were converted into artificial surfaces at a rate of 107,000 ha/year in 39 European countries (EEA, 2016d). Urban land expansion has mostly taken previous arable areas and, to a lesser extent, semi-natural habitats and forests (EEA, 2016d). In Central Asia, the rate of urban sprawl was reduced following independence of its constituent States. This was because of economic reasons, but also because migration from rural areas to urban areas increased the density of urban populations rather than the expansion of urban areas (Osepashvili, 2006). Unusually, Kazakhstan experienced a decline in urban populations coupled with increases in the rural population between 1990 and 2014 (United Nations, 2014). A further example of urban sprawl is growing migration to coastal areas, especially in the Mediterranean in Western and Central Europe (**Box 4.6**).

Box 4.6: Urban sprawl on the Mediterranean coast.

Human pressures on the Mediterranean coast are further exacerbated by urbanization, resulting in the decline of rural areas (Giacanelli *et al.*, 2015; Kelly *et al.*, 2015). The general result is a spatial dichotomy between strong, heavily populated coastal areas and thinly populated inland areas, with lower urban density and a less dynamic economy (Parcerisas *et al.*, 2012). The Mediterranean coasts also host a large seasonal tourist population and, even if the fortunes of Mediterranean destinations have fluctuated in recent years, the whole region remains among the most popular destinations of the global tourist market (UNWTO, 2015). Tourism is the main source of foreignincome in the Mediterranean region, representing as much as 25% of GDP in some countries (WTTC, 2015). Projected tourist arrivals in the Mediterranean basin for 2030 are estimated as 350 million (WWF, 2004). The environmental impacts of tourism are far-ranging and include land-use changes, pollution and waste production. Both resident and seasonal human populations are dependent on the availability of resources, infrastructures and services. These economic and demographic shifts also brought radical changes in agricultural, industrial and commercial sectors, all with their own share of environmental implications, ranging from soil degradation (Guerra *et al.*, 2015), land abandonment (Reino *et al.*, 2010), habitat loss (Monteiro *et al.*, 2011), waste production and disposal (Tatsi & Zouboulis, 2002), land-use changes (Celio *et al.*, 2014; Serra *et al.*, 2008) and pollution of water resources, both freshwater and marine (Zalidis *et al.*, 2002). With the help of new technologies enabling the harvest of higher yields, many initially traditional livelihood activities, like subsistence fishing, turned into new, capital-driven economic sectors. Mediterranean fisheries are also the subject of political controversies due to territorial disputes and degradation of marine habitats (Hofrichter, 2003).

The impacts of people moving to the coast are both direct and indirect, with direct impacts including emissions of effluents and pollutants, and indirect impacts including locational factors, where urbanization and industrial areas often serve as hubs for further urban sprawl (Salvati, 2013). Maritime transport also presents a key environmental pressure, with several major commercial routes crossing the Mediterranean Sea. On average, there are about 60 maritime accidents in the Mediterranean annually, of which about 15 involve fuel or chemical spills (EEA, 1999).

Water is also becoming a scarce and valuable commodity in the Mediterranean region, either because of decreasing quantities or inadequate quality. Today it is evident that damming cannot be considered a long-term and large-scale solution to water shortage, while desalination with reverse osmosis technology requires vast amounts of energy (Teixeira *et al.*, 2014). The water conflict in the Middle East and North Africa already provided ample examples of the volatile nature of negotiations over water resources, particularly across national boundaries (Poff *et al.*, 2003).

Climate change will also play a major role in the future evolution of the Mediterranean Basin. Potential impacts related to climate change include drought, floods, sea level rise, changes in the marine currents, and increased storm frequency. All of these changes will affect most coastal regions, with likely repercussions on national economies, particularly where those are directly dependent on natural resources and tourism. The critical factor for implementing future strategies in the Mediterranean region is cooperation, as environmental threats are not constrained by national boundaries.

End of Box 4.6

4.6 Drivers and effects of pollution

By extracting resources and returning them to the environment as waste, humans alter the biogeochemical cycles that have evolved for millennia. Pollution arises when humans introduce new substances that are toxic to species, or when the rate at which humans generate and deposit waste is faster than nature's own rate of re-absorbing and effectively neutralizing these resources.

Pollution is often categorized according to its effect in a certain medium i.e., air, water or soil/land. In this chapter, we categorize pollution according to pollutant or problem/effect (**Table 4.6**) and focus on five categories: nutrient pollution, organic pollution, acidification, xenochemical and heavy metal pollution and "other pollution" (i.e. ground-level (tropospheric) ozone, light and plastic pollution). Gene pollution, noise pollution, thermal pollution and radioactive pollution were also identified as relevant, but generally to a lesser extent, and are therefore not included in this assessment. Greenhouse gas emissions causing climate change and the introduction of invasive alien species can also be considered as pollution (Spangenberg, 2007; Weale, 1992) and have therefore been included in **Table 4.6**, which provides an overview of pollutants, problems/effects, and their drivers.

Table 4.6: Categorization of pollutants, problems/effects and main drivers. Source: Own compilation.

Pollutants	Problem/Effect	Main drivers
Bio-accessible nitrogen and phosphorus in terrestrial, freshwater and marine ecosystems	Eutrophication (hypertrophication)	Agriculture, industrial air pollution, wastewater
Organic pollutants	Oxygen-depleted systems, eutrophication, soil erosion, brownification	Wastewater, land-use change
Sulphur dioxide and nitrogen oxides from high temperature energy release, ammonia	Acidification	Electricity production, agriculture, incineration and industrial processes, transportation
PBT, POPs, pesticides, PCBs, dioxins, furans, PAHs, heavy metals	Xenochemical and heavy metal pollution	As above plus mining, chemical production
Nitrogen oxides, volatile organic compounds including methane, carbon monoxide	Ground-level ozone	Electricity production, industry, transportation
Light pollution	Disruption of species reproduction and survival	Material intensity of GDP, low variable cost of LED

Plastic debris	Life of marine organisms	Polymer production
Carbon dioxide, nitrous oxide, methane, etc. (Section 4.7)	Climate change	Energy use and agriculture
Invasive alien species (Section 4.8)	Biological invasion, biodiversity loss	Globalization

Pollution is influenced by natural resource extraction. In turn, it also influences some forms of resource extraction. For example, local fishing communities on the Faroe Islands, Denmark, who are pressed by international opinions to stop killing pilot whales, are more worried that the whales are too polluted to consume and that the whales will become extinct due to pollution (Nieminen *et al.*, 2004).

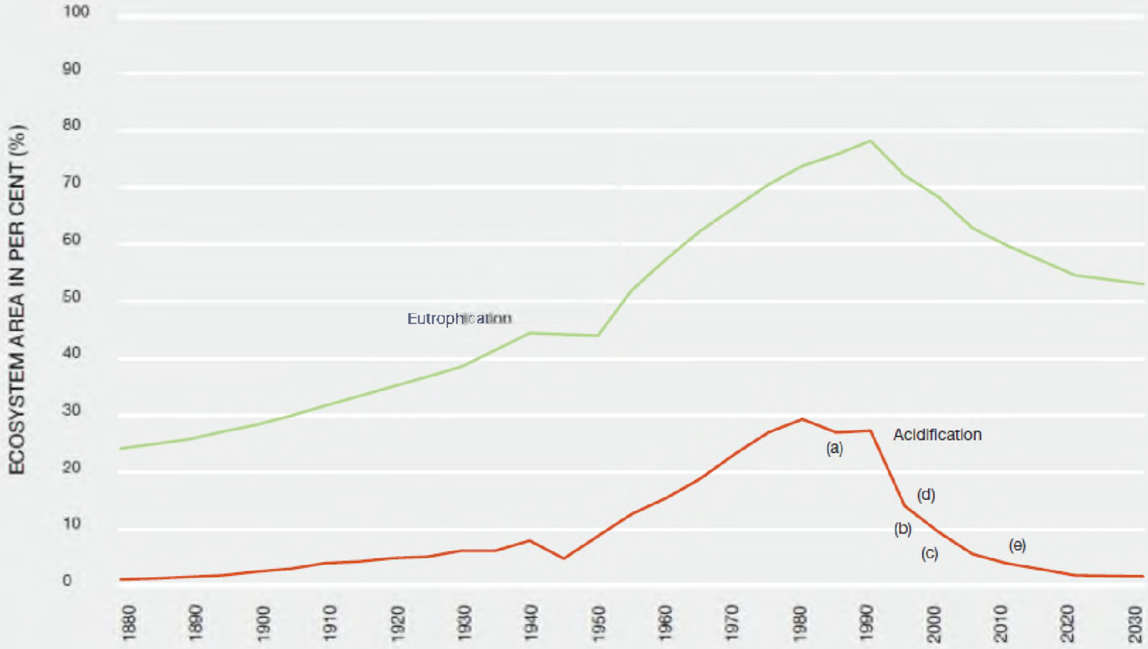
4.6.1 Nutrient pollution

Nutrient pollution arises when the concentrations of nutrients that are naturally found in low concentrations, such as phosphorus (P) and nitrogen (N), increase to excessive levels. This also causes eutrophication in freshwater and marine ecosystems. In Europe and Central Asia phosphorus is often the main problem (nutrient which constrains eutrophication) in freshwater while nitrogen is most often the limiting nutrient in terrestrial and marine environments.

4.6.1.1 Effects of nutrient pollution on biodiversity and nature's contributions to people

Increased nitrogen concentrations enhance productivity through fertilization and they decrease biodiversity through eutrophication and acidification (**Figure 4.44**). The deposition of reactive nitrogen (nitrogen oxides (NO_X) and ammonia (NH_3)) reduces biodiversity in terrestrial ecosystems by favouring plant species well adapted to nitrogenous or acidic conditions at the cost of less tolerant species (Bobbink *et al.*, 2010). Susceptibility to stress, such as frost damage or disease, may also be enhanced (Dise *et al.*, 2011). An annual deposition of 5–10 kg nitrogen per hectare has been estimated as a general threshold for such adverse effects (Bobbink *et al.*, 2010). The species richness of understory vegetation of Western and Central European forests also decreased with increasing nitrogen deposition rates and oligotrophic species were replaced by eutrophic ones (Dirnböck *et al.*, 2014).

Figure 4 44 Acidification and eutrophication of terrestrial ecosystem in Western and Central Europe peaked in the 1980s and early 1990s and have been reduced in extent (area) since then. Source: EEA (2014a, 2014b).

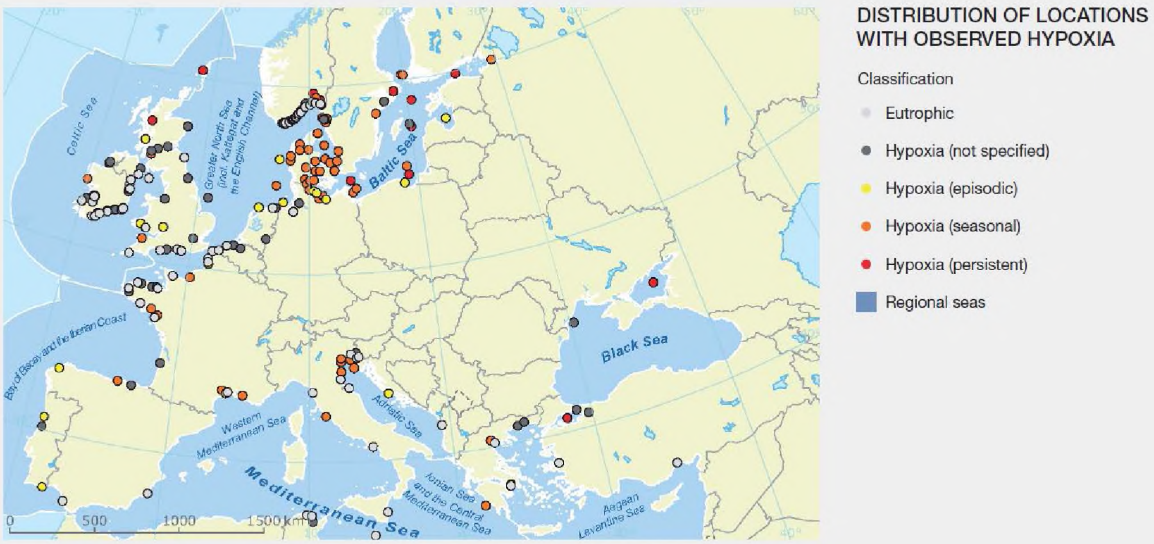


(a) First Sulphur Protocol (1985);
(b) Second Sulphur Protocol (1994);
(c) Gothenburg Protocol (1999);
(d) NEC Directive (2001);
(e) Amended Gothenburg Protocol (2012).

The (a) to (e) show the point in time when protocols under the LRTAP Convention or the EU's NEC Directive were signed or adopted. The area covered is the so-called EMEP domain, here the geographic area between 30°N-82°N latitude and 30°W–90°E longitude. This includes all EU-28 countries as well as the EEA member and cooperating countries, other non-EU eastern European countries, parts of the Russian Federation and parts of Turkey (EMEP, 2014a and 2014b). The percentage (%) results are based on emission trends since 1880 (Schöpp *et al.*, 2003), with deposition patterns following different versions of the EMEP model (e.g. Hettelingh *et al.*, 2013), and the most recent critical load database (Posch *et al.*, 2012) in combination with the current legislation (CLE) scenario developed for the Gothenburg Protocol amendment for the period 2010 to 2030 (Amann *et al.*, 2011).

Eutrophication of marine ecosystems is perhaps more worrying than freshwater eutrophication since, although recent studies have shown a decrease in marine and coastal eutrophication, the number of marine dead zones due to hypoxia (oxygen depletion due to organic pollutants) fuelled by eutrophication has increased markedly (EEA, 2014a, 2014b) (**Figure 4.45**).

Figure 4 45 Eutrophication and hypoxia are very frequent in coastal areas in Europe and Central Asia. Source: EEA (2014d).



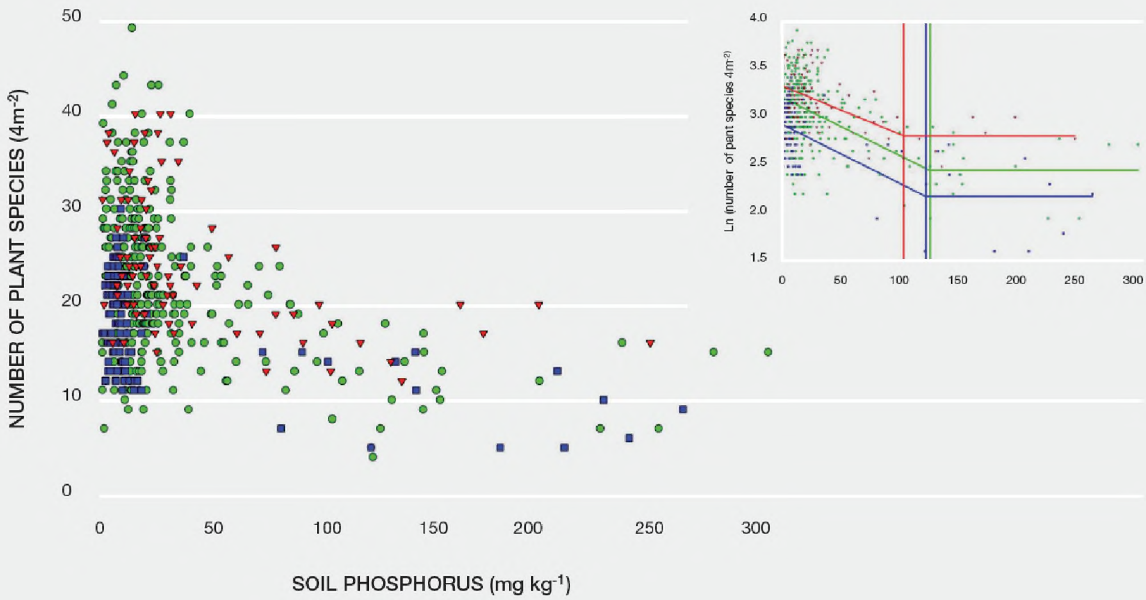
Emissions of nitrogen have contrasting implications on nature's contributions to people. There are clear and well-established negative impacts of nitrogen, derived from anthropogenic reactive nitrogen (NO_X and NH_3) on eutrophication, soil acidification, drinking water quality (Villanueva *et al.*, 2014) and human health (WHO, 2013). Besides, nitrous oxide (N_2O, a potent greenhouse gas, produced in soils with excess nitrogen, is increasingly emitted into the atmosphere, where it contributes to climate warming and, in the stratosphere, to the decomposition of ozone (Ravishankara *et al.*, 2009).

Increased nitrogen deposition, however, can positively influence other contributions of nature to people like crop, timber and livestock production (Wang *et al.*, 2015). Carbon sequestration is higher in nitrogen-limited systems if nitrogen deposition increases (Erisman *et al.*, 2014). In an evaluation of these opposing effects on nature's contributions to people, a reduction in nitrogen deposition was estimated to have net benefits to society by reducing the need for greenhouse gas regulation measures and by increasing non-material contributions, such as recreation. These benefits exceeded the total cost of material contributions (Jones *et al.*, 2014).

Phosphorous has long been regarded as the main driver of eutrophication in freshwater ecosystems. Excessive levels of phosphorous and soil erosion (organic P) cause an overgrowth of plants and algae that in turn increases the level of activity of decomposers and decreases the dissolved oxygen levels (hypoxia). This affects biodiversity negatively, mainly invertebrates and higher plants (Lepori & Keck, 2012; Lyons *et al.*, 2014; Noges *et al.*, 2016). The internal loading of phosphorous from sediments in lakes can keep them in a state of eutrophication even when external inputs are reduced, a process that is further promoted by increased temperatures (Moss *et al.*, 2011). Such legacy effects, i.e. phosphorous accumulation in sediments, have recently been observed in the River Thames (UK) where algal blooms still occur in most years, controlled by light and water temperature (Bowes *et al.*, 2016).

A meta-analysis found that phosphorous limitation of primary production is as strong as nitrogen limitation and is not confined to freshwater ecosystems and tropical forests as previously believed (Elser *et al.*, 2007). A study of more than 500 unfertilized grasslands in five countries in Western Europe found a significant negative effect of soil phosphorous on plant species richness, mainly in acidic grasslands (Ceulemans *et al.*, 2014). Species richness decreased until a threshold value (104-130 mg P/kg soil depending on grassland type), indicating that species loss is fastest at low phosphorous concentrations (**Figure 4.46**).

Figure 4 46 High soil (plant-available) phosphorous concentration negatively affects terrestrial plant species diversity. Source: Ceulemans *et al.* (2014).



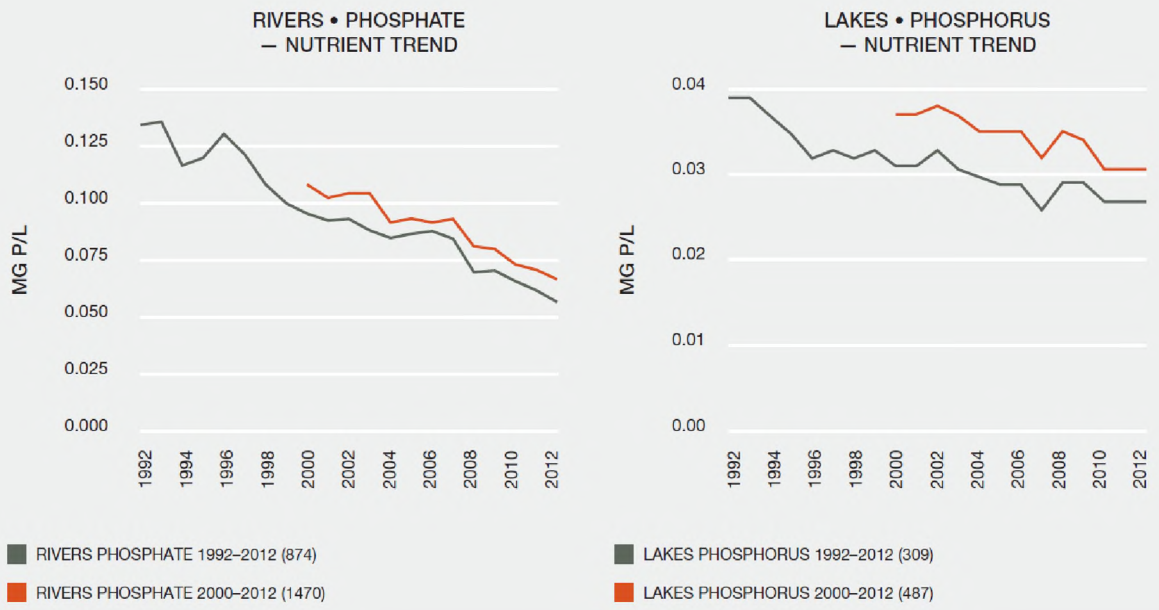
4.6.1.2 *Trends in nutrient pollution*

Between 1980 and 2011, NO_X and NH_3 emissions in the European Union declined by 49% and 18%, respectively (EEA, 2014b). 94 % of NH_3 emissions come from agriculture (EEA, 2016a). However, while NO_X continues to decrease, NH_3 emissions in Western Europe have stabilized with even slight increases in recent years (EEA, 2016a) (see also **Figure 4.49** under Acidification). For Western and Central Europe (EEA-39), NO_X emissions are projected to further decrease in future years while NH_3 emissions will stay approximately constant until 2020 (EEA, 2016a). Some uncertainty prevails, for example Turkey reported a doubling of NH_3 emissions between 2012 and 2013; the level was kept for 2014 (EEA, 2016a).

The average nitrogen deposition rate in the region is about 5 kg/ha/yr, in contrast to a background rate of 0.5 kg/ha/yr or less (BIP, 2016).

The anthropogenic input of phosphorous increased from <0.3 Tg/yr before the industrial revolution to 16 Tg/yr currently (Peñuelas *et al.*, 2012). Over 50% of the soils studied in Belgium, Netherlands and Sweden had higher phosphorous levels than recommended (Ceulemans *et al.*, 2014). In contrast, the levels of total-phosphorous decreased markedly in rivers and lakes, mainly due to advances in wastewater treatment (**Figure 4.47**).

Figure 4 47 Changes in phosphorous concentrations in Western and Central Europe. Source: EEA (2015c).



There is little information on changes expected until 2050. However, phosphorous-limited terrestrial ecosystems have lately increased in extent and will continue to do so due to climate change (Peñuelas *et al.*, 2012).

4.6.1.3 Drivers of nutrient pollution

In Western and Central Europe, nutrient pollution is driven by agricultural land-use change (intensification by increasing manure, fertilization and soil erosion), wastewater (sewage and septic systems), storm water, fossil fuel combustion in transportation and energy production (increasing NO_X), and households (gardens, detergents) (EEA, 2015c).

Regulations and technological innovation have been effective in reducing NO_X and, except in recent years, NH_3. These decreases are mainly due to policies that enforced measures in transportation (catalytic converters and fuel switching), plant improvement (e.g., flue-gas abatement techniques) in the energy and production industries, and the Nitrate Directive in agriculture reducing the use of fertilizer. The emissions of N_2O decreased by 38% mainly due to the measures of the European Union Nitrate Directive, the Common Agriculture Policy (CAP), and the Land-fill Waste Directive (EEA, 2014c).

Vegetarianism is a cultural driver with a high potential: a 50% reduction in the consumption of animal products would lead to at least a 10%-reduction in nitrogen pollution in the EU-27 (Van Grinsven *et al.*, 2015).

Climate change is expected to have adverse effects by increasing erosion and nutrient run-off in agricultural areas, frequency of wastewater overflow, water temperature, and the duration of the growing season (Dokulil & Teubner, 2010).

4.6.2 Organic pollution

Organic pollution refers to large emissions to water of organic compounds that can be oxidized by naturally occurring micro-organisms. Organic pollution is most often point-source, i.e., released directly into the water, although diffuse loss from catchments can also yield large amounts of organic compounds. The most important sink of organic pollutants is decomposition by bacteria and fungi by enzymatic catalysis. These decomposers grow rapidly and use a great deal of oxygen during their growth. When they die, they are broken down by other decomposers, which causes further depletion of the oxygen levels (hypoxia and eventually anoxia).

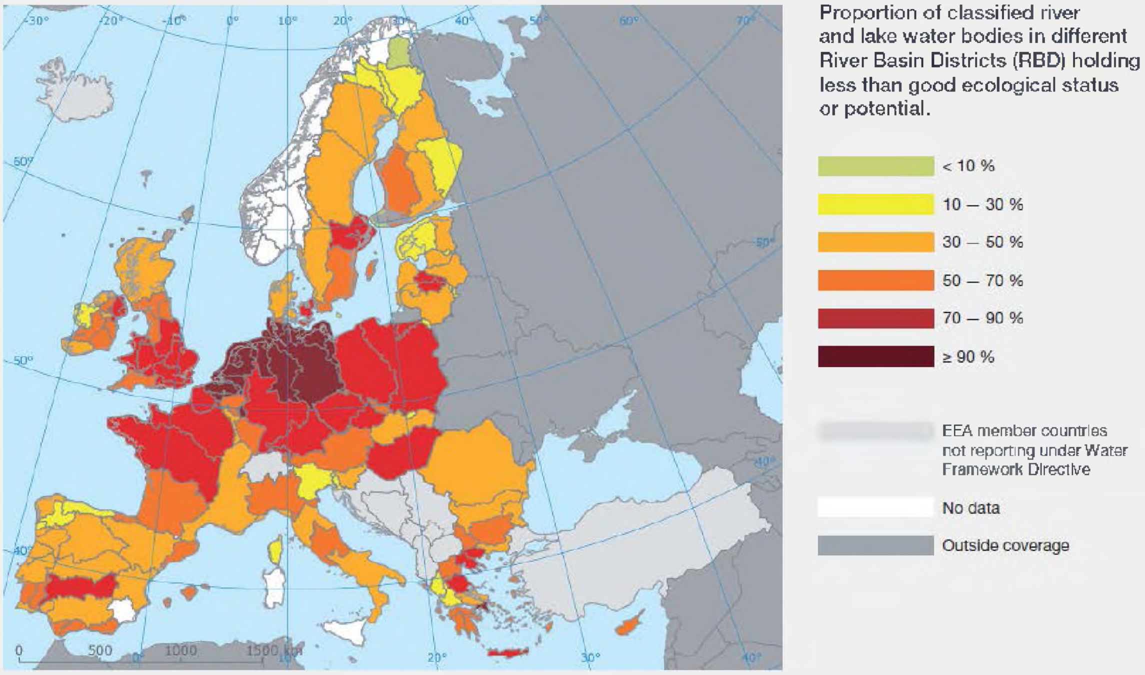
4.6.2.1 Effects of organic pollution on biodiversity and nature's contributions to people

In freshwater, increased levels of easily degradable organic compounds reduce primary production and degrade habitats for aquatic life (Couture *et al.*, 2015). Easily degradable organic compounds have a well-documented, strong negative impact on riverine biodiversity by depleting oxygen to critically low levels for benthic macroinvertebrates and fish (Connolly *et al.*, 2004; Hering *et al.*, 2006; Sand-Jensen & Pedersen, 2005). Organic compounds also increase light attenuation in the water column ("browning" if the source is humic compounds) and the epiphytic growth of biofilm (Burns & Ryder, 2001; Richardson *et al.*, 1983).

Organic compounds strongly bind various toxins, thereby somewhat reducing their bioavailability (Ravichandran, 2004). They also serve as transport vectors for heavy metals and organic pollutants (Kopáček *et al.*, 2003), which are toxic for aquatic life (Ravichandran, 2004; Teien *et al.*, 2006).

The ecological status of Western and Central European rivers and lakes is strongly linked to pollution by nitrogen, phosphorous and organic compounds (EEA, 2015c). Rivers with high concentrations of nitrogen, phosphorous and organic compounds are more likely to be in a poorer ecological state. Lakes with high nutrient loads will have high chlorophyll concentration and low water clarity due to abundant phytoplankton growth. As a result, a large proportion of lakes and rivers in Western and Central Europe do not reach a satisfying ecological status (**Figure 4.48**).

Figure 4 48 Proportion of rivers and lakes of less than "good" ecological status or potential in the European Union. Source: EEA (2012e).



4.6.2.2 Trends in organic pollution

Emissions of easily degradable organic compounds is decreasing in Western and Central Europe thanks to improved sewage treatment and better storage of animal manure in agriculture, for example, as a result of effective regulations during the past 30 years (European Commission, 2012). However, several monitoring programmes have detected significant increases in the concentration of dissolved organic carbon since 1990 in Western Europe (Monteith *et al.*, 2007). Water colour, an easily observable consequence of organic matter in the water, has changed markedly in lakes and rivers across the boreal zone in the past decades and this trend is likely to continue (De Wit *et al.*, 2016). Currently, surface waters in northern waters are browning as a result of reduced acid deposition (Garmo *et al.*, 2014; Monteith *et al.*, 2007) and increased precipitation (De Wit *et al.*, 2016).

Although the causal relationships are not straightforward, a combination of climate change induced increases in run-off and temperature, and indirect changes in terrestrial vegetation (Meyer-Jacob *et al.*, 2015) are projected to increase organic matter loads in future (Hejzlar *et al.*, 2003). These increases will be strongest in the boreal zone of the region and in the Arctic, where thawing of permafrost is a further source of organic matter (Abbott *et al.*, 2014).

4.6.2.3 Drivers of organic pollution

Demographic and economic drivers have increased organic pollution from sewage, agriculture (livestock manure), aquaculture (fishponds and farms), and certain types of industries (such as dairy, or sugar refinery). Except for land-use change, the source of organic pollution is mainly point sources and therefore regulations and technological innovations have managed to reduce emissions (EEA, 2012c). Sewage overflows in connection with high precipitation events, however, remains a problem (Rauch & Harremoës, 1996).

4.6.3 Acidification

Acidifying substances such as sulphur dioxide (SO_2), ammonia (NH_3) and nitrogen oxides (NO_X) undergo chemical transformation into acids as they are dispersed in the atmosphere. Their subsequent downwind deposition leads to acidification of the soil and surface water.

4.6.3.1 Effects of acidification on biodiversity and nature's contributions to people

Historically, SO_2 was the dominant pollutant causing acidification, but today NO_X are increasingly important. Effects of terrestrial acidification from nitrogen were briefly assessed in Section 4.6.1.1. Anthropogenic acidification has profound, well-documented ecological impacts, including the loss of many acid-sensitive species from all trophic levels (e.g. Hildrew & Ormerod, 1995; Likens & Bormann, 1974; Schindler, 1988). In catchments with an insufficient supply of base cations to buffer acidity, runoff to freshwater ecosystems becomes strongly acidic and at a pH of 5.5, alkalinity falls to zero and inorganic aluminium concentration rises to become toxic to many forms of life, including almost all fish (Sutcliffe & Hildrew, 1989).

Despite reduced emissions there is still a legacy effect on biodiversity. Evidence for biological recovery from anthropogenic acidification has therefore been much less obvious than changes in, for example, water chemistry (Battarbee *et al.*, 2014). Soil and surface water acidification remains an issue in the most sensitive areas of Nordic countries, the United Kingdom and Central Europe (EEA, 2017). Kernan *et al.* (2010) found that invertebrate assemblages showed signs of partial recovery at around half of sites in the UK acid water monitoring network that had recovered in terms of water chemistry and even less showed any evidence of recovery of salmonid populations (Malcolm *et al.*, 2014; Murphy *et al.*, 2014).

4.6.3.2 Trends in acidification

In Western and Central Europe (EEA-33) emissions of SO_2 decreased by 74% between 1990 and 2011 (EEA, 2016a). **Figure 4.49** illustrates this trend, which is projected to continue (**Figure 4.50**). Within the European Union, NO_X emissions decreased by 49% between 1980 and 2011 (EEA, 2014b), or by 40% between 2000 and 2010 (EEA, 2016a). Data is limited from other regions and contingent on economic activity; as an example, NO_X emissions in Montenegro dropped after 1990 but increased to the same level in 2009 (EEA, 2015b).

Figure 4 49 Emissions of air pollutants, EU-28, 1990–2015, Index 1990=100. Note: PM2.5 time series start in 2000. Source: Eurostat (2016).

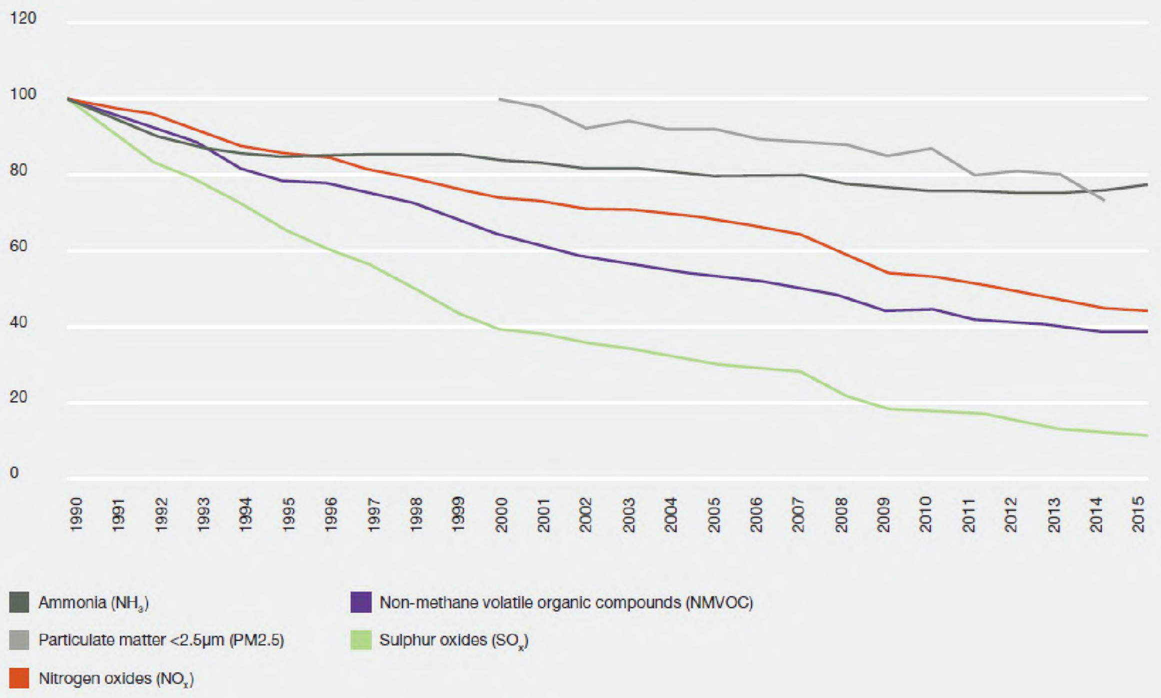
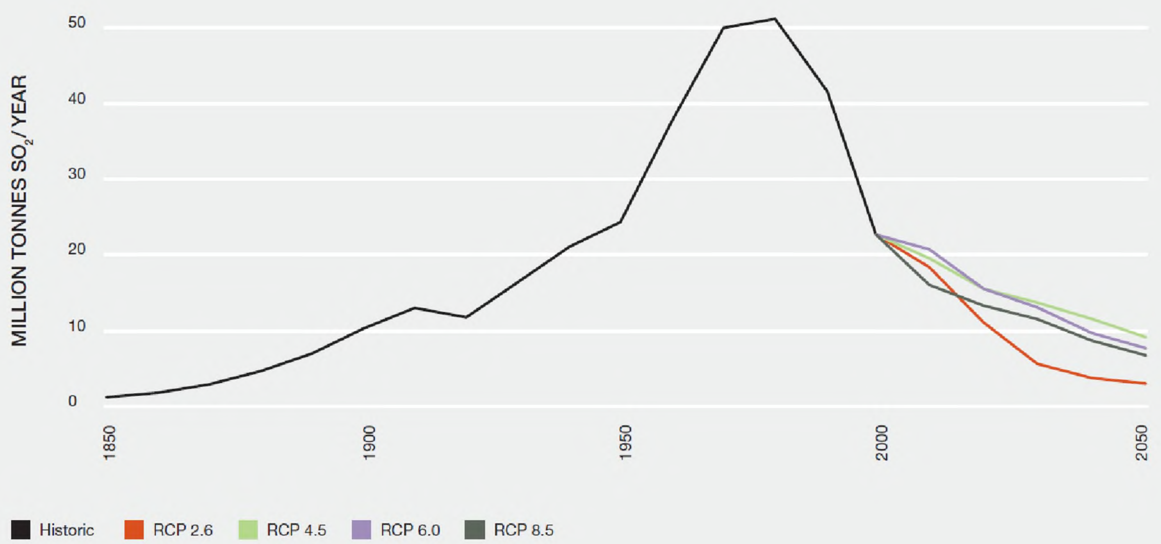


Figure 4 50 The emission of Western and Central Europe and this trend is projected to continue in all IPCC scenarios. Source: EEA (2014a). Representative concentration pathways (RCPs) are four greenhouse gas concentration (not emissions) trajectories adopted by the IPCC for its fifth assessment report (IPCC, 2013b).



4.6.3.3 Drivers of acidification

Anthopogenic NO_X and SO_2 are mainly caused by fossil fuel combustion. The basic drivers are economic, which Montenegro and Serbia may illustrate: During the period of sanctions on the former Yugoslavia (1990-1995), there was a significant drop in SO_2 and NO_X emissions, due to the overall reduction in economic activities. After 1995, emissions increased steadily with GDP (EEA, 2015b). However, institutional drivers are key to push for technological change which, in the case of acidification, has been relatively simple. Regulations like the Sulphur Protocols (Section 4.6.1.1) have been effective in reducing acidification.

4.6.4 Xenochemical and heavy metal pollution

Xenochemical pollution is the introduction or release of chemical substances into ecosystems where they are not naturally found.

4.6.4.1 Effects of xenochemicals and heavy metals on biodiversity and nature's contributions to people

The polluting impact of many chemicals and heavy metals (e.g. polychlorinated biphenyl [PCB] and lead) are well-known, and their use and emission strictly regulated in most parts of Europe and Central Asia. There are, however, emerging threats to biodiversity and nature's contributions to people, which relate not only to recently introduced compounds but also to inappropriate use of listed toxic compounds, unknown effects of toxic mixtures, unknown effects of chemicals that have not undergone toxicological testing (e.g. hygiene products) and potentially aggravating effects of climate change (Malaj *et al.*, 2014).

Xenochemicals primarily influence ecosystems and biodiversity in close proximity to urban areas, industry and agriculture although a number of studies have shown long range pollution of xenochemicals in air, water and biota. Toxicity of most of the emerging pollutants is unknown, as is knowledge on their persistence in the environment and ability to bioaccumulate. A large amount of literature has documented toxic effects of xenochemicals on both terrestrial and aquatic biota (Beketov *et al.*, 2013; Pereira *et al.*, 2009; Sabater *et al.*, 2007). Impacts of toxic compounds on freshwater macroinvertebrates have been shown for heavy metals and pesticides (Heckmann & Friberg, 2005; Liess & Von Der Ohe, 2005; Rasmussen *et al.*, 2008; Schäfer *et al.*, 2007).

Studies have shown increases of Priority Hazardous Substances like mercury in the aquatic food web, especially fish (Åkerblom *et al.*, 2014), to levels that exceed advised limits for humans and can have negative impacts on wildlife (Scheulhammer *et al.*, 2007). Various synthetic compounds acting as hormone distruptors (e.g. BPA and other bisphenols, phtalates, etc.) have direct negative effects on nature's contributions to people (EEA, 2012c).

Multiple chemicals interact in the environment, producing combined ecotoxic effects that exceed the sum of individual impacts (Kortenkamp *et al.*, 2009). As a result, a substance present in concentrations below the threshold level may still contribute to combined and possibly synergistic effects. In particular, robust evidence exists of combination effects for hormone disrupting chemicals (EEA, 2012c).

4.6.4.2 Trends in xenochemical and heavy metal pollution

In Western and Central Europe more than 100,000 commercially available chemical substances are registered in the European Inventory of Existing Commercial Chemical Substances (EINECS). Global sales from the chemical industry sector doubled between 2000 and 2009, with increases in all world regions (OECD, 2012), which is a development that is predicted to continue. The total sales of pesticides across the European Union increased from 2011 to 2014 by 4 % to just under 400,000 tonnes of active substances, despite the adoption of The Directive on the Sustainable Use of Pesticides in 2009. However, the aim of this Directive was not only to reduce the use of pesticides but to "promote the use of less harmful pesticides and provide incentives to industry to develop pesticides with less hazardous properties" (EEA, 2016c).

4.6.4.3 Drivers of xenochemical and heavy metal pollution

Xenochemical pollution is integrated in all sectors of industrialized countries, driven by market forces in general and globalization in particular. Public awareness has modified institutional drivers, e.g. the European Union “Reach” legislation. However, the globalized characteristics of xenochemicals combined with uncertainty concerning the effects of new substances inhibit effective regulations (OECD, 2011).

4.6.5 Other pollution

4.6.5.1 Ground-level ozone

Ground-level ozone may have significant effects on biodiversity (Wedlich *et al.*, 2012). These effects include changes in species composition of semi-natural vegetation communities (e.g. Ashmore, 2005), reductions in forest net primary productivity (Matyssek *et al.*, 2003), also in combination with nitrogen (Bobbink *et al.*, 2010). Ozone pollution has been linked to the prevalence of damage in mountain forests: the acute effects of O_3 involve visible injuries to leaves and shoots and changes in physiological processes and metabolism.

The emission of O_3 or ozone precursor gases has recently decreased considerably in Western and Central Europe (EEA, 2015c). However, the ground-level concentrations of O_3 have remained stable or even increased due to long-range transport from outside Western and Central Europe (EEA, 2015c). As a result, most types of vegetation and almost all crops (88% of Western and Central Europe’s agricultural area, mainly its southern and eastern parts) are exposed to levels above the critical load, especially near roads with heavy traffic. Drivers of nitrogen concentration and ozone formation interact, which calls for integrated policy responses (e.g. see Table 6.1 in Chapter 6).

4.6.5.2 Light pollution

Light pollution is generated by the use of artificial light at night and affects terrestrial, aquatic and marine ecosystems (Davies *et al.*, 2014; Longcore & Rich, 2004). It is related to material affluence and concerns 23% of global land surface and 88% in Western and Central Europe (Davies *et al.*, 2014). Temperate and Mediterranean ecosystems have experienced the greatest increase in exposure to artificial lighting (Bennie *et al.*, 2015b) and a significant increase in average nighttime lighting has been reported in 32 % of Western and Central European terrestrial protected areas since 1995 (Gaston *et al.*, 2015). This rate is expected to increase in the coming decades because of the replacement of existing lighting infrastructure by broad-spectrum white lighting technologies (such as LEDs), which is expected to double the perceived night sky brightness.

Light pollution also dramatically influences movements and distributions of nocturnal species, which represent 30% of mammals and 60 % of invertebrates worldwide (Hölker *et al.*, 2010). Nocturnal insects present a "flight-to-light behaviour" (Altermatt *et al.*, 2009), which generates insect biomass accumulation in illuminated patches and depletion in surrounding dark areas. Unnatural polarized light sources, e.g. from building materials, can also trigger maladaptive behaviours in polarization-sensitive taxa and alter ecological interactions (Horváth *et al.*, 2009).

Light pollution induces major shifts in biological communities by disrupting the interspecific balance of trophic and competition interactions (Bennie *et al.*, 2015a; Davies *et al.*, 2013; Knop *et al.*, 2017; Rydell *et al.*, 1996). This can have profound impacts on ecosystem functions such as pest control, pollination, and seed dispersal. For example, moths carry less pollen in light-polluted areas than in dark areas (Macgregor *et al.,* 2017), which in turn may impact the fitness of insect-pollinated plant species (Macgregor *et al.*, 2015). Additionally, light pollution induced large-scale phenology changes in UK deciduous tree budburst (Ffrench-Constant *et al.*, 2016). The large spatial scale impacts of light pollution likely interact and accentuates the adverse impacts of both land use and climate changes on biodiversity.

4.6.5.3 Marine and beach plastic debris

Polymers are part of our everyday life. Annual production rates continue to grow and have risen from 1.7 million tonnes in 1960 to 322 million tonnes in 2015, with a current mean annual increase of 4% (Plastics Europe, 2016). Plastic debris and microplastics can affect a wide array of marine organisms, from plankton (Collignon *et al.*, 2012) to filter feeding marine organisms (Fossi *et al.*, 2014; von Moos *et al.*, 2012) and large pelagic fish (Romeo *et al.*, 2015). Size, shape and abundance of plastic debris influence uptake; microfibres are considered most harmful (Wright *et al.*, 2013). Plastics inhibit digestion and can transfer attached chemical pollutants into the animal tissues (Browne *et al.*, 2013). As a result of increasing awareness, developing new polymeric materials today often includes assessment of its durability and its degradation time when exhausted (Hottle *et al.*, 2013).

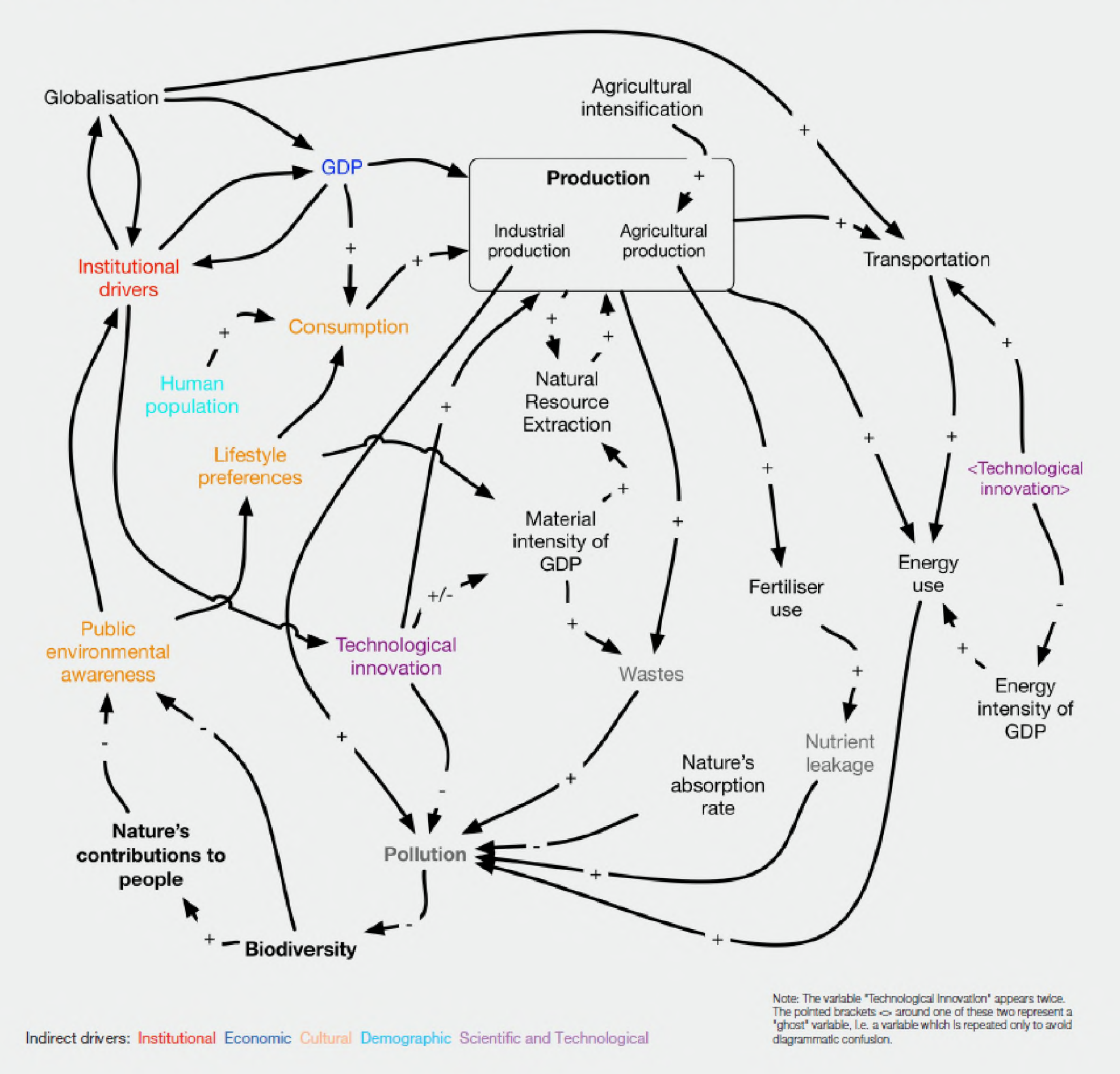
4.6.6 Synthesizing drivers of pollution

The passage of matter through our economy and society, from resource extraction to waste, is the premise for pollution. All factors affecting the size and the quality of the material throughput of our societies (Boulding, 1966) are drivers of pollution. In the 1990s, particularly in Europe, the need for a shift from end-of-the-pipe policies to prevention became evident. Several studies highlighted this (Adriaanse *et al.*, 1997; Matthews *et al.*, 2000; Von Weizsäcker *et al.*, 1997) so that material flow data eventually entered the official statistics of the European Union.

For the above reasons, pollution is driven by the same drivers that are highlighted for natural resource extraction, land-use change, and invasion of alien species, and climate change (which can be seen as a form of pollution). This is also evident from **Table 4.6** (introduction to Section 4.6) that summarizes the main pollution problems and their drivers. As synthesized by **Figure 4.51** below these drivers are mainly economic, i.e. effects of industrialization and globalization and its subsequent increase in transportation. Pollution also increases by population growth, institutional drivers that foster adverse technological development, and the cultural belief that a prosperous life must entail more material consumption (Jackson, 2009). Technological innovation usually increases production and transportation but may also change the material intensity of GDP and production technology to reduce waste and pollution. Recent institutional drivers have succeeded in developing technologies for reducing some pollutants in Europe, especially point sources like air pollutants from industrial effluents (including SO_2, NO_X, lead) and municipal waste water. However, the drivers of xenochemicals and nutrient leakage (NH_3) from agriculture have not successively been reversed.

Figure 4.51 depicts the main causal loops for pollution, emphasising industrial and agricultural production and transportation. There are two feedback loops in **Figure 4.51**. First, the public awareness of pollution influences regulations via political pressure. Second, awareness influences cultural beliefs and consumption patterns, which may alter the material intensity of GDP.

Figure 4 51 Causal loop diagram of drivers of pollution. Source: Own representation.



4.7 Drivers and effects of climate change

4.7.1 Effects of climate change on biodiversity

Climate change is a complex driver of ecosystem change, consisting of changes in precipitation and temperature patterns which lead to changes in drought, flood, and fire risk, ocean-atmosphere interchange, marine circulation and stratification, and the concentrations and distribution of O_2 and CO_2 in the atmosphere and in the ocean (IPCC, 2014a). These impacts affect species and influence and modulate important ecosystem functions and processes that underpin human livelihoods and nature's contributions to people, such as water regulation, food production, and carbon sequestration (CBD, 2016; Gallardo *et al.*, 2015; IPBES, 2016a; IPCC, 2014a; MEA, 2005a).

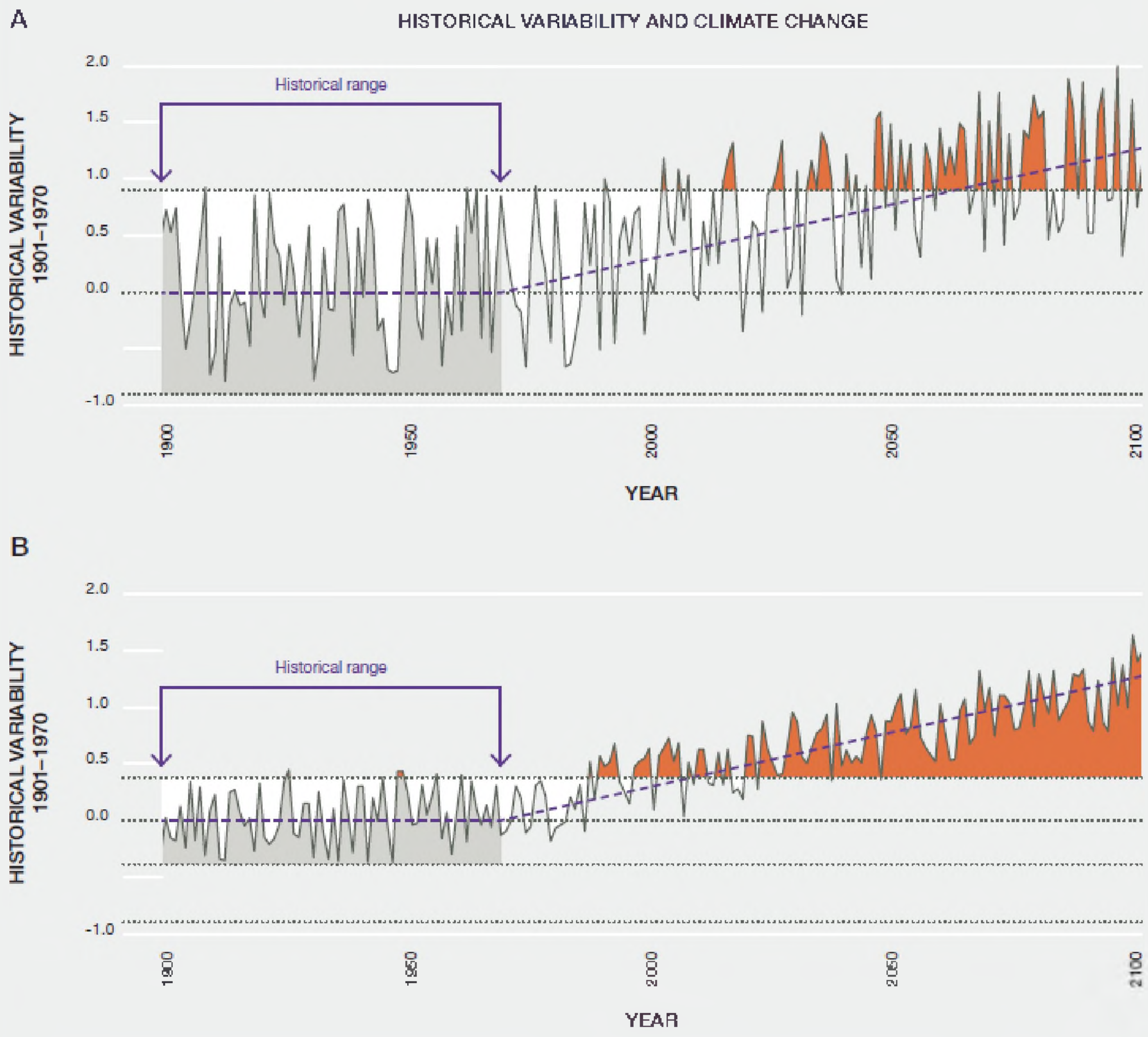
There is strong evidence that climate change affects the biodiversity of Europe and Central Asia through shifts in the timing of species' life-history events, growth, reproduction and population dynamics, and in their ranges and habitat occupancy. The evidence for impacts on the ecological processes underlying range shifts, such as species interactions, is rapidly accumulating. Knowledge gaps remain with respect to changes in physiological processes and evolutionary adaptations to new climatic conditions (Bellard *et al.*, 2012; Merilä & Hendry, 2014).

Climate change impacts are not as strong as we would expect given recent changes in climate. In particular, many species-level responses lag behind the rate of change in climate drivers. These lags are caused in part by dispersal and establishment constraints and by biotic interactions that stabilize extant populations and communities, and in part by the fact that the rates and nature of climate are regionally variable and may only slowly be exceeding the historical range of variability (see **Figure 4.52**; see also Section 4.7.1.2). In Europe and Central Asia, northern areas are projected to experience fast spatial displacement of climate. Therefore, dispersal-related responses may be particularly important, mountainous regions will be subject to local divergences between temperature and precipitation drivers, and novel climates are more likely to appear in the Mediterranean area (Ordonez *et al.*, 2016). Climates outside the current range of variability are a particular threat as populations are unlikely to contain adapted individuals or genes. For a given rate of underlying climate change, novel conditions may be experienced sooner in highly variable systems, though such systems will also occasionally experience historically "normal" years further in the future (**Figure 4.52**).

Complex responses and interplay between direct and indirect effects of multiple climate change drivers challenge our ability to project future trends in nature's contribution to people. Additional interactions with other anthropogenic drivers, such as reinforcement of ocean acidification and land-use change (Cloern & Jassby, 2012; Mantyka-Pringle *et al.*, 2012; Riebesell & Gattuso, 2014) further exacerbate the complexity. As gradual changes in mean climatic conditions can have dramatically different consequences for biodiversity, including ecosystems, compared with changes in the variability of short-term weather, the two are treated separately. Then, important secondary effects of climate change on ecosystems are briefly assessed.

Figure 4 52 Illustration of the link between climate variability and mean trend.

The simulated historical range of climate variability is high in panel A and low in panel B. No trend in climate mean is simulated between 1900–1970 and shows the increasing mean trend from 1070–2100 in A and B. The red zone denotes years in which the historical range of variability is exceeded, which is observed much more rapidly in panel B (compared to A). Source: Illustration follows Beaumont *et al.* (2011).



4.7.1.1 Effects of gradual climate change

4.7.1.1.1 Effects on phenology, growth and fitness

It is well documented that recent climate change has affected phenology, but there is considerable variation across regions, biomes, and taxa (Cleland *et al.*, 2012; Cook *et al.*, 2012; Ma & Zhou, 2012; Parmesan, 2006, 2007; Wolkovich *et al.*, 2012). In Western Europe, standardized assessments have confirmed phenological advancement in terrestrial, freshwater and marine plants and animals (Menzel *et al.*, 2006; Thackeray *et al.*, 2010). Phenological changes are often linked to changes in the onset and duration of the growing season, potentially affecting species and ecosystems. These effects can be both direct, on the survival and population dynamics of individual species (e.g., "developmental traps": prolonged seasons that allow multiple generations of insects but leave the autumnal cohort vulnerable - Van Dyck *et al.*, 2015), and indirect from e.g., phenological mismatches between plants and pollinators (Hegland & Totland, 2008); between predators and prey (Petitgas *et al.*, 2010; Raab *et al.*, 2013; Visser *et al.*, 2006); and between multiple trophic levels (Edwards & Richardson, 2004; Luczak *et al.*, 2012; Möllmann & Diekmann, 2012).

In animals, there are indications of climate change impacts on growth and body size, for example in otters (Yom-Tov *et al.*, 2010; Yom-Tov *et al.*, 2006b) and birds (Yom-Tov *et al.*, 2006a; Yom-Tov, 2001). The strength and direction of the linkage to climate change is not very clear, as effects are mostly indirect through changes in net primary production and thus food availability (Yom-Tov & Geffen, 2011). Body size decreases consistently across freshwater taxa under warming (Daufresne *et al.*, 2009). In marine systems, high temperatures are particularly stressful for vulnerable life stages of coastal zooplankton, especially larvae (Przeslawski *et al.*, 2015). For plants, warming will increase growth and size until reaching a point where other factors limit growth. For example, the largest warming experiment in the region found that climate warming increased alpine plant growth, but only in the first few years, possibly due to onset of nutrient or water limitation later on (Arft *et al.*, 1999).

Climatic factors can also act as forces of selection, driving adaptive differentiation between and within populations at fine spatial scales despite potentially high levels of gene flow (Anderson *et al.*, 2012). Plant populations may adapt in situ via selection on standing genetic variation in response to climate change (Jump *et al.*, 2009). Genetic differentiation in response to temperature or moisture gradients has been observed in plants at both fine spatial scales (e.g. Kelly *et al.*, 2003) and across landscapes (e.g. Jump *et al.*, 2006). Such patterns of genetic structuring are highly indicative of adaptive differentiation in response to environmental selection, which has been confirmed by direct experimental tests of genetic responses to climate change in plant species within intact ecosystems (e.g. Jump *et al.*, 2008; Ravenscroft *et al.*, 2015). However, despite this potential for genetic responses, a number of recent reviews of both terrestrial and marine systems find little direct evidence for adaptive genetic responses to current climate change (Boutin & Lane, 2014; Donnelly *et al.*, 2012; Reusch, 2014; Teplitsky & Millien, 2014). In cases where genetic changes are documented, it is still unclear whether these reflect adaptive responses, whether they are directly caused by climate change, and whether they are sufficient to keep up with future climatic changes (Franks *et al.*, 2014). Even in species with the highest adaptive potential, widespread species with large populations and high fecundity, adaptational lags are likely under future climatic changes (Aitken *et al.*, 2008).

4.7.1.1.2 Effects on biodiversity and community dynamics

Shifts in species ranges in response to climate change are relatively well documented for Europe and Central Asia. Latitudinal and altitudinal shifts in species distributions have been found for many taxa, e.g. 80% of studied taxa in a global meta-analysis by Root *et al.* (2003); for marine systems see Perry *et al.* (2005) and Beaugrand *et al.* (2014). Northwards migrations of warm-adapted species and associated loss of cold-adapted species have been documented for the Barents Sea and North East Atlantic (Beaugrand *et al.*, 2002, 2009; Brander *et al.*, 2003; Fossheim *et al.*, 2015), resulting in increased species richness of zooplankton (Beaugrand *et al.*, 2010) and fish (Hiddink & ter Hofstede, 2008). Northward range shifts (12.5-19 km per decade) are also prevalent among terrestrial species, including arthropods, birds and mammals (Hickling *et al.*, 2006). Similar range shifts are found along altitudinal gradients, but are not ubiquitous across a broad range of taxonomic groups (Benito *et al.*, 2011; Grytnes *et al.*, 2014; Nogués-Bravo *et al.*, 2008), while downslope or no shifts also occur (Lenoir *et al.*, 2010). While evidence for range shifts is mounting, the unequivocal attribution to climatic warming is not always clear, as the magnitude of shifts cannot always be predicted from observed climatic changes (Grytnes *et al.*, 2014).

Climate change does not affect species ranges and biodiversity equally in all regions or for all taxa (e.g. Garrabou *et al.*, 2009; Pairaud *et al.*, 2014; Tunin-Ley *et al.*, 2009). Negative impacts are likely strongest where species' latitudinal and altitudinal shifts are physically limited, for example in the case of mountaintops, northernmost or southernmost areas. The ranges of birds inhabiting northern Fennoscandia are strongly controlled by temperature, and will likely no longer overlap with terrestrial land areas in the future (Virkkala *et al.*, 2008). Strongly negative impacts can also be expected in taxonomic groups with high species turnover along climate gradients and with small range sizes, as for birds in Central Asia (La Sorte *et al.*, 2014), and in biodiversity hotspots, as for the highly diverse reptile fauna of the Central Asian Mountains (Ficetola *et al.*, 2013). Despite individual responses, an overall homogenization of biodiversity has been projected from model experiments for birds in Western and Central Europe (Thuiller *et al.*, 2014), indicating that taxonomic, phylogenetic and functional turnover decrease between regions. Relatively low and slow responses in range dynamics may not imply that climate change does not matter. Rather, it may reflect lagged responses, also known as climatic extinction debts (Devictor *et al.*, 2012; Dullinger *et al.*, 2012), and homogenization of regional species pools (Thuiller *et al.*, 2014).

Species shift their ranges at individual rates and directions (see above), which will result in novel assemblages (Alexander *et al.*, 2015), and may change the intensity of species interactions, such as increased interspecific competition (Olsen *et al.*, 2016), dampened herbivore cycles (Cornulier *et al.*, 2013), and changes in predator-prey dynamics (Schmidt *et al.*, 2012; Terraube *et al.*, 2011; Winder & Schindler, 2004). Such indirect impacts may be particularly important at the warmer-climate distributional edge of species ranges, where the intensity of interactions may be higher, and could lead to loss of specialized interactions (pollination, predator-prey, dispersal, consumer, trophic, etc.) to be replaced by generalists (Lurgi *et al.*, 2012).

Some of this context-dependency in species' ability to withstand climatic change can be predicted by species traits. For example, a global analysis indicates that thick leaves, high below-ground biomass, and tall growth are key traits for montane grassland species' ability to withstand climatic warming (Willis *et al.*, 2017). This is empirically confirmed for the Norwegian mountain flora (Guittar *et al.*, 2016) and plants in the Caucasus Mountains (Soudzilovskaia *et al.*, 2013), where the losers under climate change are plants lacking these traits.

A warmer climate will not only have negative impacts on species richness. As can be observed for many taxa, the biodiversity of algae in the south-Tajik depression (Barinova *et al.*, 2015), and zoobenthos in the Onega Bay of the White Sea (Denisenko, 2010) both increase towards warmer regions. Functional shifts have also been observed. In Georgia, a shift towards a higher species richness of ants is expected, at the expense of a decreased species richness of spiders (Chaladze, 2012; Chaladze *et al.*, 2014). However, the predictive ability of climate change responses based on such spatial gradients will be modified by nonlinearities (Nagorskaya & Keyser, 2005).

4.7.1.1.3 Effects on ecological processes and ecosystem functioning

It is well documented that climate change impacts vegetation and ecosystem functioning in Europe and Central Asia, but strength and direction depend on region, unit of analysis, and on the nature of the climatic changes. The relative importance of climate change and other concurrent drivers on ecosystem functioning are hard to disentangle.

Under increased temperatures, soil respiration, microbial activity (Sowerby *et al.*, 2005) and decomposition of lignified materials (Zell *et al.*, 2009) increase, but only if there is adequate moisture (Poll *et al.*, 2013). In the Mediterranean, temperatures are already close to the optimum for photosynthesis, so warming mainly increases plant water loss, whereas in temperate areas a warming of 1°C can increase biomass production by as much as 15% (Peñuelas *et al.*, 2004). In the UK, experimentally-increased temperatures led to a decrease in soil nitrogen leaching, probably due to increased nitrogen uptake because of increased plant growth (Ineson *et al.*, 1998a, 1998b). Warming also has a negative effect on soil biota abundance at all trophic levels, especially in cold dry regions, affecting their ecosystem functions (Blankinship *et al.*, 2011; Briones *et al.*, 2007).

Changing precipitation jointly impacts plants and biogeochemical cycles, a phenomenon well studied in Western and Central Europe with >70 experimental sites manipulating precipitation. Global meta-analyses reveal that plant biomass, productivity, respiration, ecosystem photosynthesis, and net carbon uptake are generally stimulated by increased precipitation and supressed by decreased precipitation (Vicca & Bahn, 2014; Wu *et al.*, 2011). Ecosystems are generally more sensitive to increased, than to reduced precipitation. Precipitation also affects decomposition, with coarse woody debris decay rate peaking at around 1,250 mm annual precipitation in temperate Western Europe (Zell *et al.*, 2009). Microbial soil communities in the northern parts of Western Europe may be more sensitive to changes in rainfall patterns than more moisture-limited soils in the southern parts of Western European (Sowerby *et al.*, 2005). Winter precipitation change also affect ecosystems, and snow depth manipulation experiments find that decreasing snow depth may reduce soil CO_2 efflux, increase N_2O efflux, and increase mobile nitrogen concentration (Blankinship & Hart, 2012).

Gradual warming favours harmful cyanobacterial blooms in freshwater systems, particularly in combination with eutrophication (O'Neil *et al.*, 2012). Warming will increase the spread of invasive fish in freshwater ecosystems, as cold seasons currently limit the spread of many freshwater invasive species (Rahel & Olden, 2008). Anadromous fish important for recreational fishing (salmonids) will shift their ranges northwards and suffer negative effects of warming in dry areas due to reduced river flows (Jonsson & Jonsson, 2009). Reduced precipitation will directly reduce water supply but considerable uncertainties remain regarding the impact of changing temperature and precipitation regimes on water quality. A review focussed on the UK found that there is insufficient evidence to link observed decreases in water quality to climate change (Watts *et al.*, 2015).

In oceans, recent temperature-driven changes in species ranges have strongly affected the trophodynamics of North East Atlantic ecosystems (Goberville *et al.*, 2014; Luczak *et al.*, 2011) as well as benthic-pelagic coupling (Albouy *et al.*, 2013; Kirby *et al.*, 2007). Increased vertical stability (strengthening of water stratification) leads to decreasing nutrient replenishment, which leads to changes in phytoplankton bloom phenology (Herrmann *et al.*, 2014), biomass and community structure (Bosc *et al.*, 2004; Goffart *et al.*, 2002; Tunin-Ley *et al.*, 2009). Reduced nutrient availability and phytoplankton biomass strengthens the microbial pathway in the plankton ecosystem (Bosc *et al.*, 2004; Goffart *et al.*, 2002; Tunin-Ley *et al.*, 2009). A reduction in primary production and reduced upwelling intensity will also have negative impacts on fisheries (Chassot *et al.*, 2010). In Mediterranean systems, warming leads to a shift in plankton communities towards smaller species, and a decrease in diatoms (Durrieu de Madron *et al.*, 2011). Temperature increase will, however, increase the metabolic activity of the surviving species, and modelling suggests that this could compensate for the species loss, resulting in similar net primary production by 2100 (Lazzari *et al.*, 2014).

4.7.1.2 Effects of extreme events on biodiversity

Climate change leads to more extreme and less predictable weather events (heat waves, droughts, floods, heavy precipitation, windstorms) that impact biodiversity across ecosystems. Ecosystem response to climate extremes depend upon the ecosystem itself, in particular on whether productivity is precipitation-, radiation- or temperature-limited (Seddon *et al.*, 2016). The spatial distribution of Central European forest trees is partly explained by climatic extremes, in addition to average climate, suggesting such extreme events have long-term distribution-wide impacts (Zimmermann *et al.*, 2009).

Observations of extreme weather events are important sources of information on ecosystem responses. For example, the unusually hot and dry summer of 2003 in Western and Central Europe resulted in decreased primary productivity and increased net carbon flux to the atmosphere (Ciais *et al.*, 2005; Reichstein *et al.*, 2007). Trees growing at high elevations in the Alps benefitted due to release from snow cover while there was decreased growth of lower-elevation trees due to increased evapotranspiration (Jolly *et al.*, 2005). The decrease was greatest in grasslands and croplands (Reichstein *et al.*, 2007), while among forests beech and Mediterranean broadleaved forests were the most susceptible (Granier *et al.*, 2007). Species richness also decreased in several heathlands across Western and Central Europe except for cool, damp heathlands in the UK (Peñuelas *et al.*, 2007). Similarly, other droughts have been shown to reduce carbon flux from roots to the soil compartment in north-western Europe (Gorissen *et al.*, 2004), to reduce the number of flowering shoots (Peñuelas *et al.*, 2004), and to cause forest dieback in the Arkangelsk region (Aakala & Kuuluvainen, 2011) and southern Siberia (Kharuk *et al.*, 2013), altering forest vulnerability to damaging agents and pathogens (Jactel *et al.*, 2012; Morley & Lewis, 2014). Across Central Asia, drought had affected grasslands, shrublands and areas of sparse vegetation, and the desertification risk in the Kakheti Region, the most drought-sensitive part of Georgia, is driven by increased drought frequency (Basialashvili *et al.*, 2015). Experimental evidence is now emerging to complement the observational evidence on extreme events. However, published studies largely focus on Western and Central European grasslands, for which there is a broad range of experimental evidence that growth and biomass accumulation recover rapidly from drought (Geels *et al.*, 2015).

The impact of an extreme event on biodiversity and nature's contributions to people is highly contingent on the timing of the event. In Arctic and alpine regions, short-term heat waves in winter have the greatest negative impact on productivity in the following summer (Bokhorst *et al.*, 2011; Bokhorst *et al.*, 2009). Ice forming on vegetation, when winter precipitation falls as rain instead of snow, decreases the availability of vegetation to herbivores and in turn has negative consequences for top predators (Hansen *et al.*, 2013, 2014).

Freshwater systems are highly sensitive to temperature extremes, leading to habitat and species loss under drought events (Matthews & Marsh-Matthews, 2003; Woodward *et al.*, 2016). Heatwaves lead to the proliferation of toxic algal blooms, reducing both biodiversity and the provisioning of drinking water (Gallina *et al.*, 2011; Jöhnk *et al.*, 2008; Paerl & Paul, 2012). Floods, on the other hand, directly impact fresh water provisioning by increasing water turbidity and eutrophication (Khan *et al.*, 2015). The protection from flooding and erosion provided by coastal and intertidal vegetation is reduced by increased storm activity (Cardoso *et al.*, 2008; Gedan *et al.*, 2011; Kinsella & Crowe, 2015). Drought has affected nutrient leaching into lakes, with knock-on effects on aquatic communities, with rotifers and cladocerans dominating during dry periods and copepods dominating during wet periods (Krylov *et al.*, 2013). In coastal systems, benthic macroinvertebrates suffered high mortality in the middle of the 2003 heat wave, exacerbated by nutrient stress (Garrabou *et al.*, 2009).

Variability has always been part of natural systems. Climate extremes denote events that depart clearly from the past range of variability of a given unit of analysis or region. Systems with low natural variability are usually at risk of rapidly exceeding their natural range, while systems of high variability may depart from this range frequently, but return to historical conditions for longer into the future than systems with low variability (**Figure 4.52**). High latitude units of analysis are much less prone to completely depart from the historical range of variability than more equatorial units, due to the higher variability in the former and lower variability in the latter systems (Beaumont *et al.*, 2011). On the other hand, the absolute departure from the historical range of variability is higher in systems with high variability, and strong extremes (strong departures) may build more rapidly in such highly variable systems, with devastating effects from single events.

4.7.1.3 Secondary climate effects

While air temperature and precipitation changes may be seen as primary climate changes, other warming effects are also important for biodiversity and ecosystem function. Here we consider permafrost melting, atmospheric CO_2, ocean acidification and stratification, and sea level rise. Decreased precipitation and increased temperature also lead to an increased risk of fire (see Section 4.7.2).

As permafrost is the second largest terrestrial carbon pool (after soil), there is the potential for release of large amounts of carbon and methane when it thaws, thereby intensifying global warming and its earlier-mentioned effects on biodiversity; including ecosystems (Zhang *et al.*, 2017). Thawing permafrost has been shown to result in shrinking lakes due to drainage (Smith *et al.*, 2005), rapidly eroding river banks, the disappearance of wildlife including fish and migratory birds, altered migration routes, and shifted distributions of birds, reindeer, and caribou, and has been suggested to increase the danger of forest fires. Further, it may change plant species composition and productivity, as a result of warmer temperatures and associated changes in soil hydrology (Schuur *et al.*, 2007; Turetsky *et al.*, 2007).

An increase in atmospheric CO_2 concentration, a major cause of climate change, directly impacts plant functioning. The increase in total plant biomass under elevated CO_2 is contingent on adequate other resources such as water and nutrients (Zheng & Peng, 2001). A recent global analysis combining remotely sensed leaf area index time-series with biogeochemical modelling revealed a significant increase in leaf biomass of land vegetation (termed “greening”) in many regions of the world, and thus a significantly altered biogeochemistry (Zhu *et al.*, 2016). This was primarily driven by CO_2 fertilization effects, which accounted for 70% of the greening – the remainder by climate change, nitrogen deposition and land-cover change. The greening also varied regionally. It was statistically significant in many parts of Western and Central Europe, and prominent in Eastern Europe, but not observed in Central Asia. Elevated CO_2 can increase root activity, the abundance of microflora and microfauna, and particularly detritivores (Blankinship *et al.*, 2011). These effects result in changes in ecosystem biogeochemistry and altered vegetation-atmosphere feedbacks.

Ocean acidification results from increased atmospheric CO_2 and affects organism physiology (e.g. calcification, dissolution), biology (e.g. reproduction, skeletogenesis) with potential consequences for ecosystem structure and functioning (e.g. resistance to disease, unbalance of predator-prey interaction) and global carbon cycle (Hofmann *et al.*, 2010; Kroeker *et al.*, 2010). In the Atlantic and Arctic oceans, all calcifying plankton organisms exhibited simultaneously abrupt shifts in abundance during the mid- to the late-1990s (Beaugrand *et al.*, 2015). However, these large-scale ecological shifts appeared more correlated to changes in northern hemisphere temperature than to ocean acidification.

Changes in the UV-B radiation in oceans, caused by changes in the mixed-layer depth, impair photosynthesis, growth and reproduction (Llabrés *et al.*, 2013; Helbling *et al.*, 2003). Increased stratification is expected in the Mediterranean Sea during the 21st century (Somot *et al.*, 2006), and this may modify the exposure of organisms and organic compounds to solar radiation and favour photochemical oxidation reactions. The extratropical North Atlantic Ocean and its adjacent seas may be an important carbon sink (Sarmiento *et al.*, 2004). The carbon sink may become less efficient in a warmer world because of changes in phytoplanktonic types (floristic turnover) but also because upward mixing of nutrients will diminish due to increased stratification of the oceans (Bopp *et al.*, 2005; Thomas *et al.*, 2004). Deepening of the nutrient gradient would favour coccolithophorids against the diatoms, which are the major sink agents of carbon (Cermeño *et al.*, 2008). Indeed, coccolithophorids have increased in the North Sea during recent decades (Beaugrand *et al.*, 2013).

A sea-level rise of 1 m, a realistic maximum projected by 2100, will affect primarily the heavily populated regions in Western Europe (mostly The Netherlands, but also Germany, Denmark and UK). Such drastic shifts will have a strong effect on coastal ecosystems, on sessile and migrating animals (birds - Iwamura *et al.*, 2013), and on the structuring of the coastal biomes. A population of 21.7 million is calculated to be at risk in the inundation area (Rowley *et al.*, 2007).

4.7.2 Trends in climate change

4.7.2.1 Temperature change

There is strong agreement that temperature has increased in Europe and Central Asia over the last sixty years (**Figure 4.53**), especially after 1980, both for summer and winter average temperatures. The increase for 1950-2016 is significant for almost all of Europe and Central Asia, and was generally higher in winter (specifically in the Arctic Ocean and in Eastern Europe) than in summer, but was higher in summer in the south of Western Europe and Central Asia (see **Figure 4.54**).

The increase in temperatures was significant for all units across Europe and Central Asia, with positive trends of 0.15-0.30°C per decade for summer, and 0.10-0.45°C per decade for winter (see **Figure 4.55**). Increases in temperature were larger in winter than in summer for most units of analysis, except for southern biomes (Mediterranean, subtropical forests, temperate grasslands and deserts in Western Europe and Central Europe. All increases were significant except for winter temperatures of southern units in Western Europe, Central Europe and Eastern Europe.

Figure 4 53 Observed annual mean temperature (°C) trends from 1950 to 2016, relative to mean temperatures for 1986–2005, for summer (June to August) and winter (December to February) across Europe and Central Asia, plotted for three datasets: GISTEMP 1200, HadCRUT 4.2.0.0 and NCDC MOST (see Section 4.2.6). Source: Own representation.

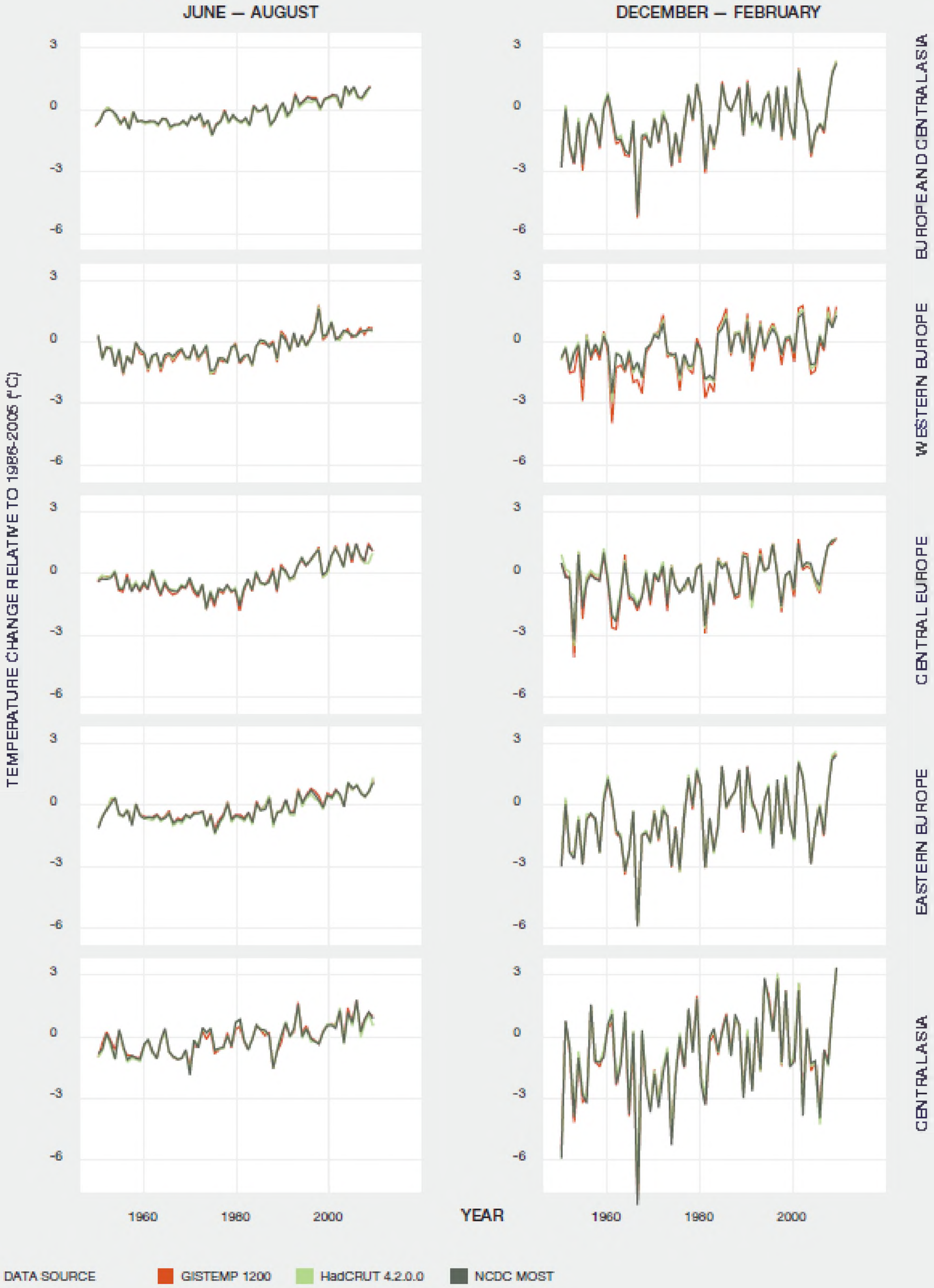
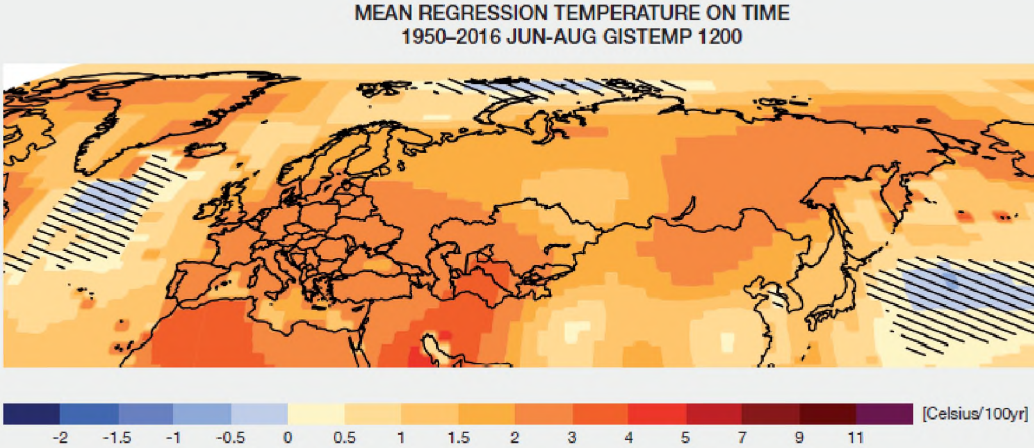


Figure 4 54 **Map of observed (linear) temperature (°C / 100 years) trends between 1950 and 2016, for summer (June to August) and winter (December to February) throughout Europe and Central Asia.**

Only the GISTEMP dataset is presented here (see 4.2.6.1). The hatching represents areas where the estimated trend is less than one standard deviation away from zero (see Hartmann *et al.*, 2013). Source: Own representation.



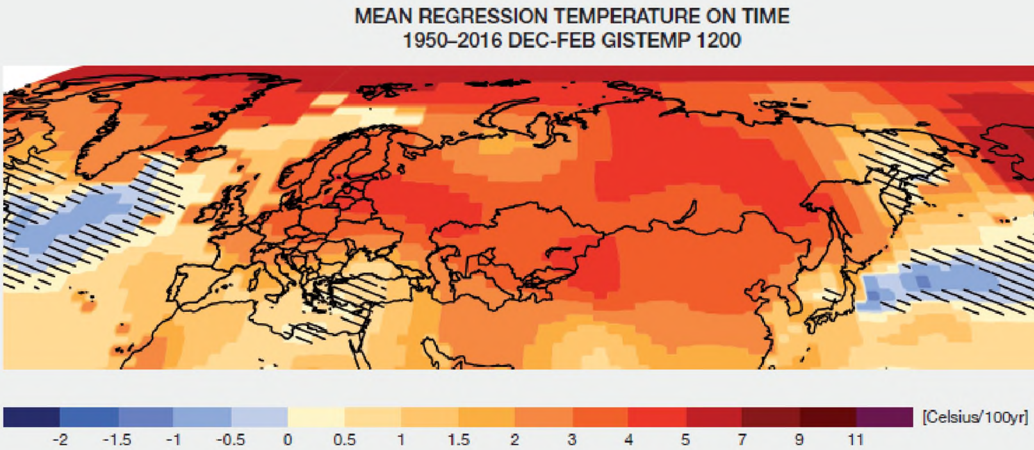
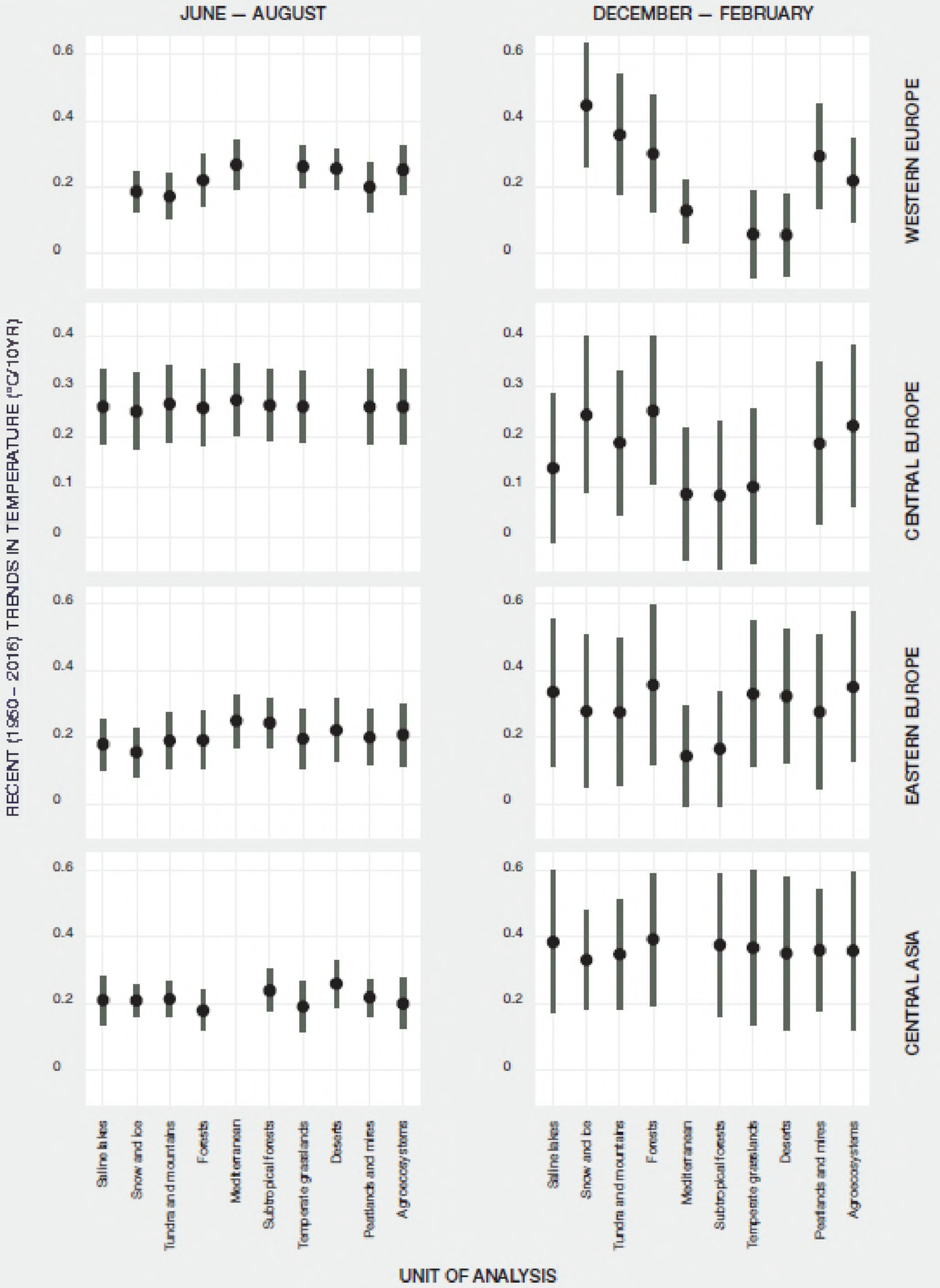


Figure 4 55 **Historical temperature trends (1950–2016) for units of analysis in Europe and Central Asia.**

The calculated linear trend is expressed in °C change/decade (see Hartmann *et al.*, 2013). Colour bars represent 90% confidence intervals around trend estimates. If 90% confidence intervals cross the dashed line of zero, the estimated trend is considered statistically insignificant. Units of analysis are: saline lakes; snow and ice (snow and ice-dominated systems - everything north of or higher than alpine); tundra and mountains (tundra and mountain grasslands - only high elevation grasslands); forests (broad-leaved, mixed and coniferous forests); Mediterranean (Mediterranean forests, woodlands and scrub); subtropical forests (tropical and subtropical dry and humid forests); temperate grasslands; deserts; peatlands and mires; agroecosystems. Source: Own representation.



Temperature is projected to increase across Europe and Central Asia in all RCP scenarios (**Figure 4.56**), with 5 to 95% ranges of projected anomalies for 2041-2060 (relative to 1986-2005) for summer ranging from 0.38 to 3.17°C for RCP 2.6 and from 1.28 to 3.72°C for RCP 8.5 (**Figure 4.56**), and for winter ranging from 0.18 to 3.92°C for RCP 2.6 and from 2.01 to 5.35°C for RCP 8.5 (**Figure 4.56**). Increases in temperatures are projected to continue throughout the 2016-2060 period for RCPs 4.5, 6.0, and 8.5, while a plateau is projected for RCP 2.6 after 2040. Summer temperature increases are projected to be higher for southern parts of Western Europe and Central Europe (see **Figure 4.56**, **Figure 4.57**) than for other subregions. Winter temperature increases are projected to be largest for Central Asia and Eastern Europe, especially at higher latitudes (**Figure 4.56**, **Figure 4.57**).

Temperatures are projected to increase for all units of analysis throughout subregions of Europe and Central Asia (**Figure 4.58**), with increases in summer being projected similarly among all units according to the CMIP5 ensemble ranging from 1 to 3°C depending on representative concentration pathway scenario, and with increases in winter being projected to differ among biomes. Specifically, snow and ice, and tundra and mountain grasslands have larger projected winter temperature increases (2.5 to 5°C) than the other units if analysis (1 to 3.5°C) in both Western and Eastern Europe.

Figure 4 56 Time series of temperature trends relative to 1986–2005 averaged over land grid points for (from top to bottom) Europe and Central Asia, Western Europe, Central Europe, Eastern Europe, and Central Asia in summer (June to August, left panel) and winter (December to February, right panel).

Thin lines denote one ensemble member per model and thick lines the CMIP5 multi-model mean (see **Figure 4.3** for details). On the right-hand side of each graph the 5th, 25th, 50th (median), 75th and 95th percentiles of the distribution of 20-year mean changes are given for 2041–2060 under the four IPCC RCP scenarios. Source: Own representation.

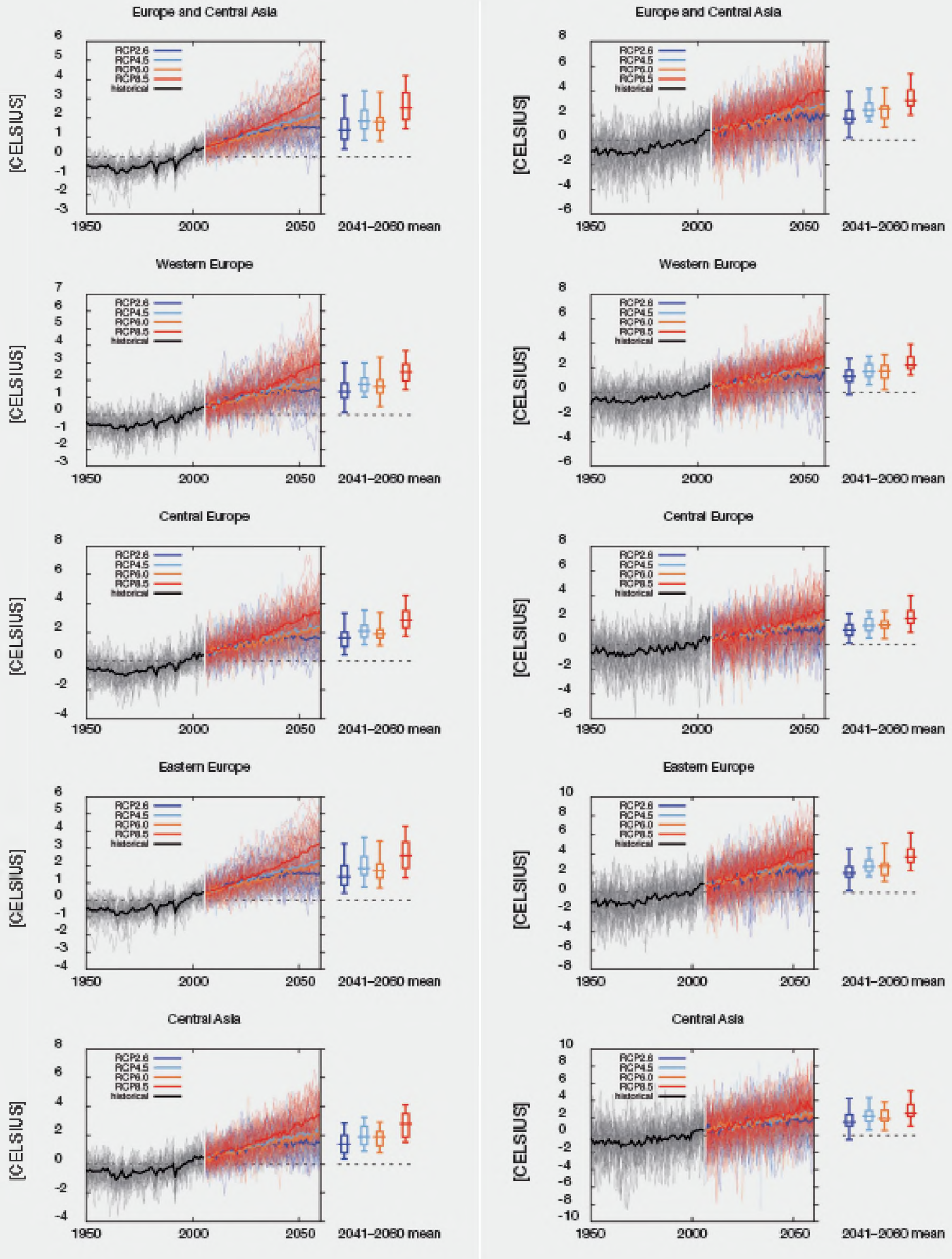


Figure 4 57 **Maps of projected (linear) temperature trends by 2041–2060 relative to 1986–2005 under the RCP4.5 (top two panels) and the RCP 8.5 scenario (bottom two panels) for summer (June to August) and winter (December to February).**

Ensemble means from CMIP5 are presented here, extracted from data of IPCC (2013b). Source: Own representation.

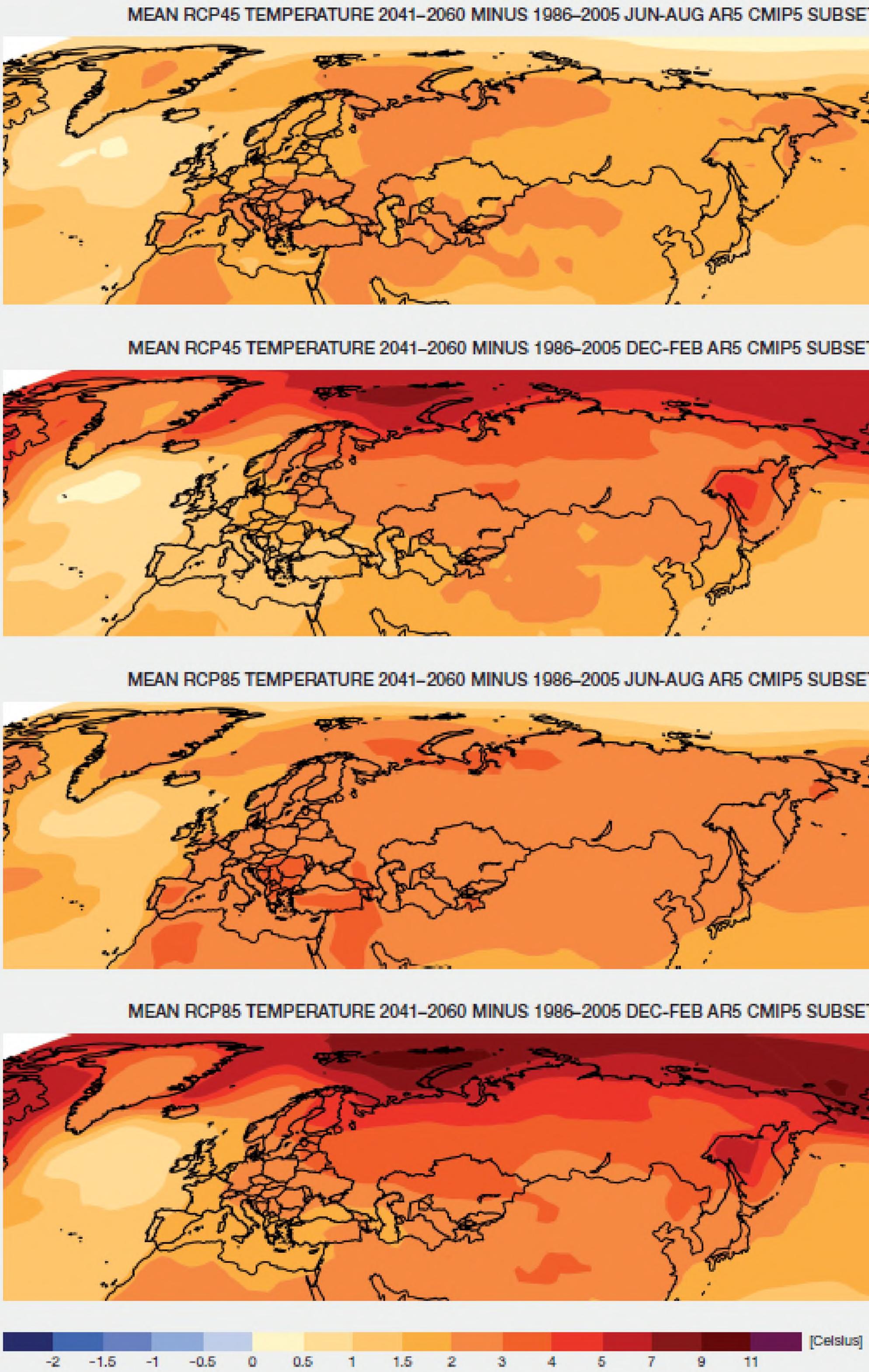
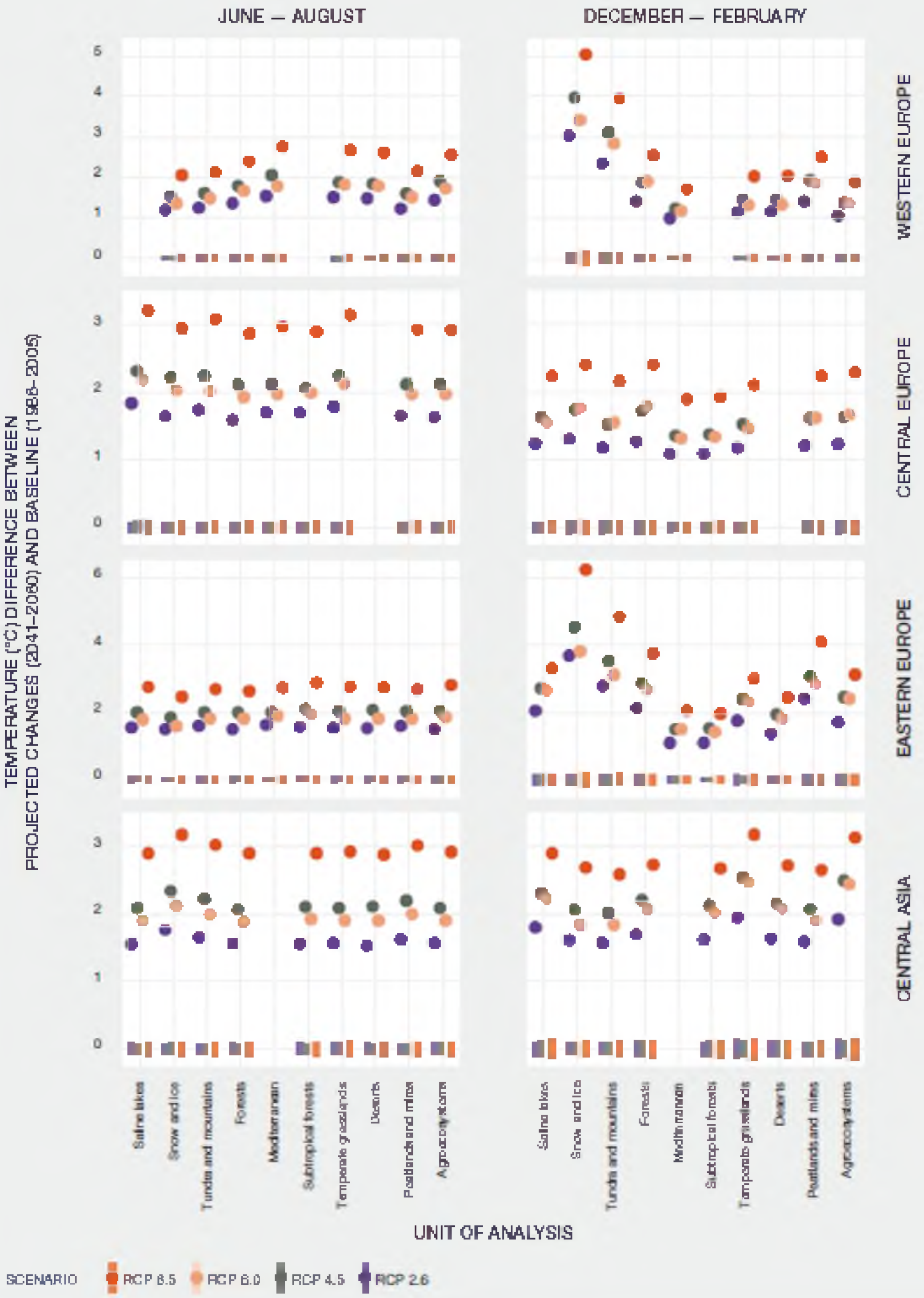


Figure 4 58 **Projected temperature changes (2041–2060) relative to 1986–2005 averaged over land grid points for units of analysis of Western Europe, Central Europe, Eastern Europe, and Central Asia.**

Points represent model ensemble means from CMIP5 for each representative concentration pathway (RCP) scenario. Bars represent one standard deviation of natural variability around 1986–2005 means. Units of analysis are: saline lakes; snow and ice (snow and ice-dominated systems - everything north of or higher than alpine); tundra and mountains (tundra and mountain grasslands - only high elevation grasslands); forests (broad-leaved, mixed and coniferous forests); Mediterranean (Mediterranean forests, woodlands and scrub); subtropical forests (tropical and subtropical dry and humid forests); temperate grasslands; deserts; peatlands and mires; agroecosystems. Source: Own representation.



4.7.2.2 Precipitation change

Precipitation has increased only insignificantly over the last sixty years across most of Europe and Central Asia (see **Figure 4.59**) (Hartmann *et al.*, 2013), with considerable subregional variation (**Figure 4.59**). Significant increases and decreases were only detected for some parts within subregions (**Figure 4.60**). Winter precipitation has decreased for southern Western Europe and Central Europe. In Eastern Europe, summer precipitation has in- and decreased in some areas throughout the subregion, whereas winter precipitation has decreased in eastern, but increased in the western parts of Eastern Europe. In Central Asia summer precipitation has generally decreased but winter precipitation has increased.

Precipitation trends were generally insignificant for the different units across Europe and Central Asia (**Figure 4.61**), and only a few significant changes were detected for tundra and mountain grasslands (winter increase in Western Europe), and saline lakes, temperate grasslands and agroecosystems (winter increase in Central Asia) (**Figure 4.61**).

Figure 4 59 Observed precipitation (mm/day) trends from 1950 to 2016, relative to mean precipitation for 1986–2005, for summer (June to August) and winter (December to February) across Europe and Central Asia, plotted for three datasets: CRU TS 4.00, GPCC V7 and NCDC anomalies. Source: Own representation.

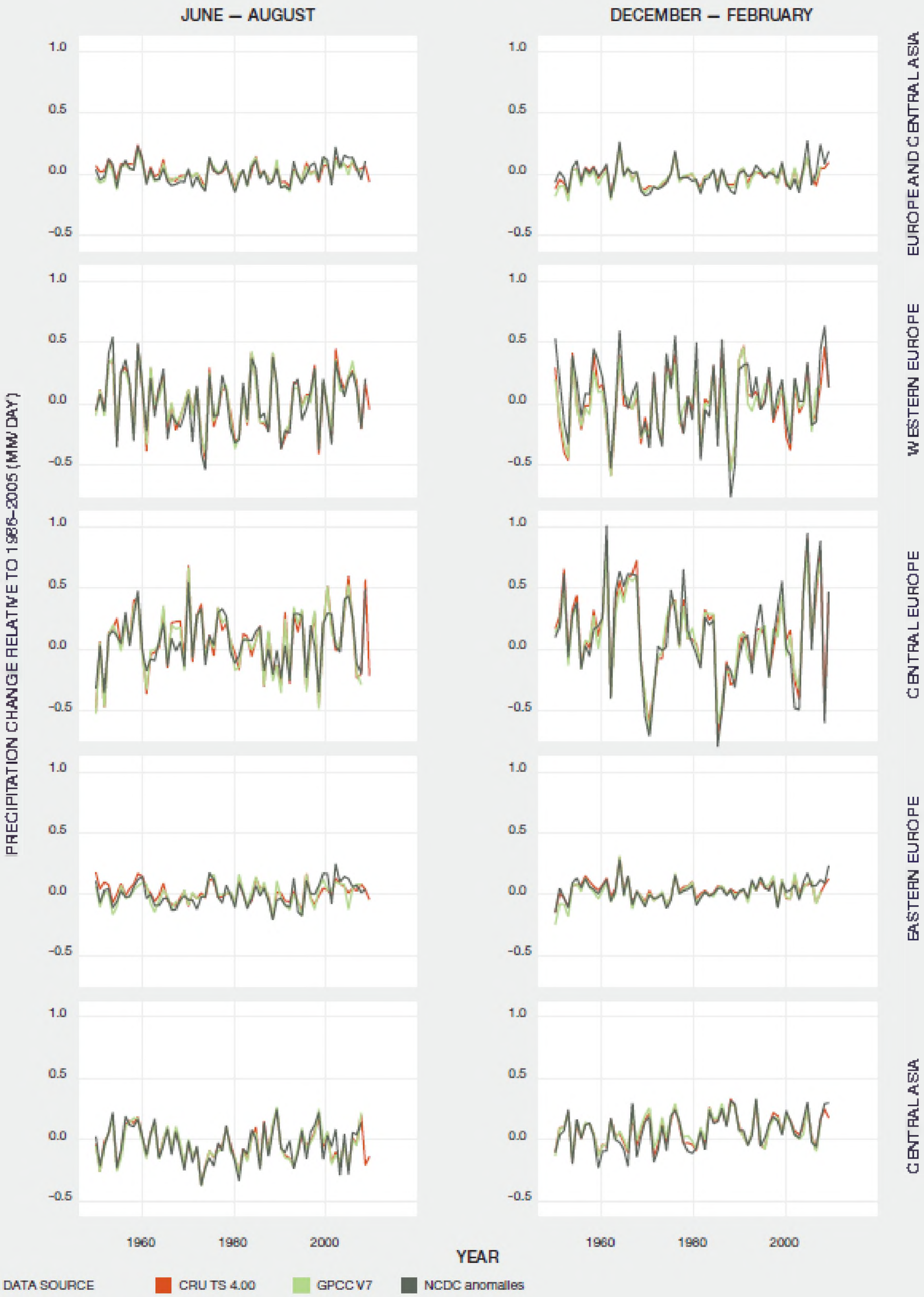


Figure 4 60 Map of observed (linear) precipitation trends (mm/day per 100 year) between 1950 and 2016, for summer (June to August) and winter (December to February) throughout Europe and Central Asia.

The CRU TS 4.00 dataset is presented here. The hatching represents areas where the estimated trend is less than one standard deviation away from zero (see Hartmann *et al.* (2013) for details). Source: Own representation.

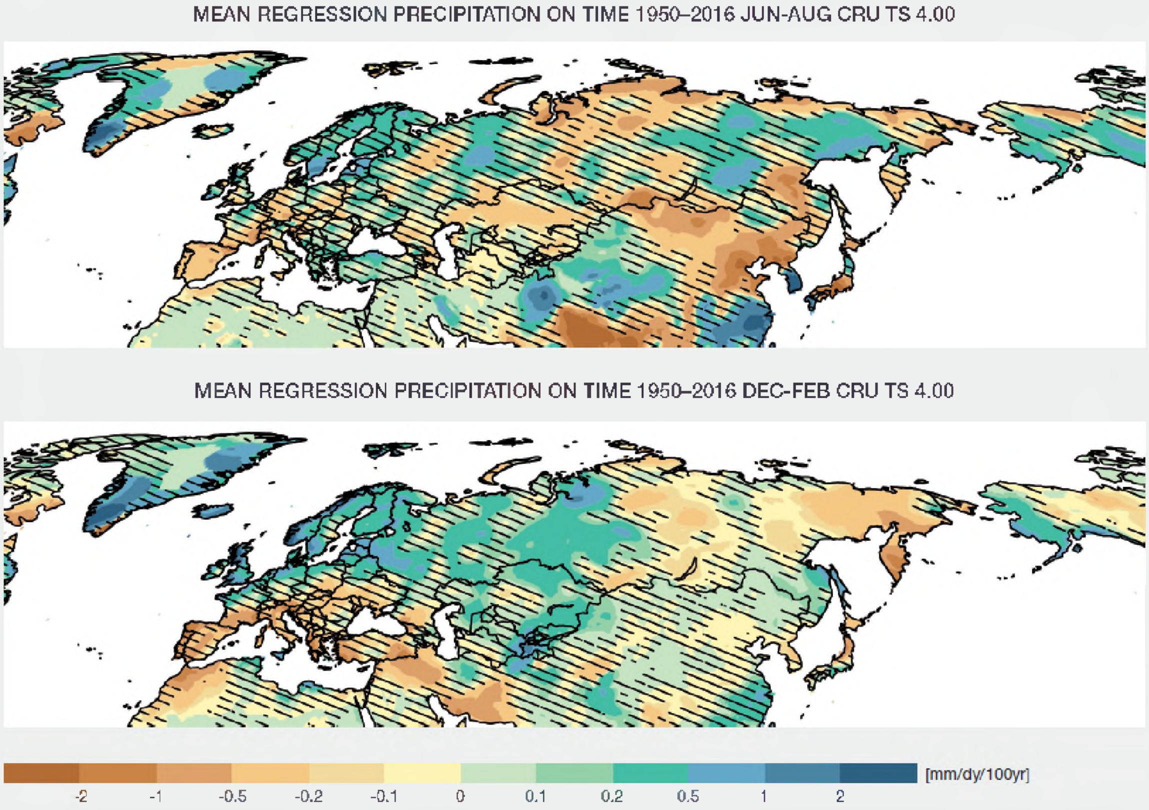
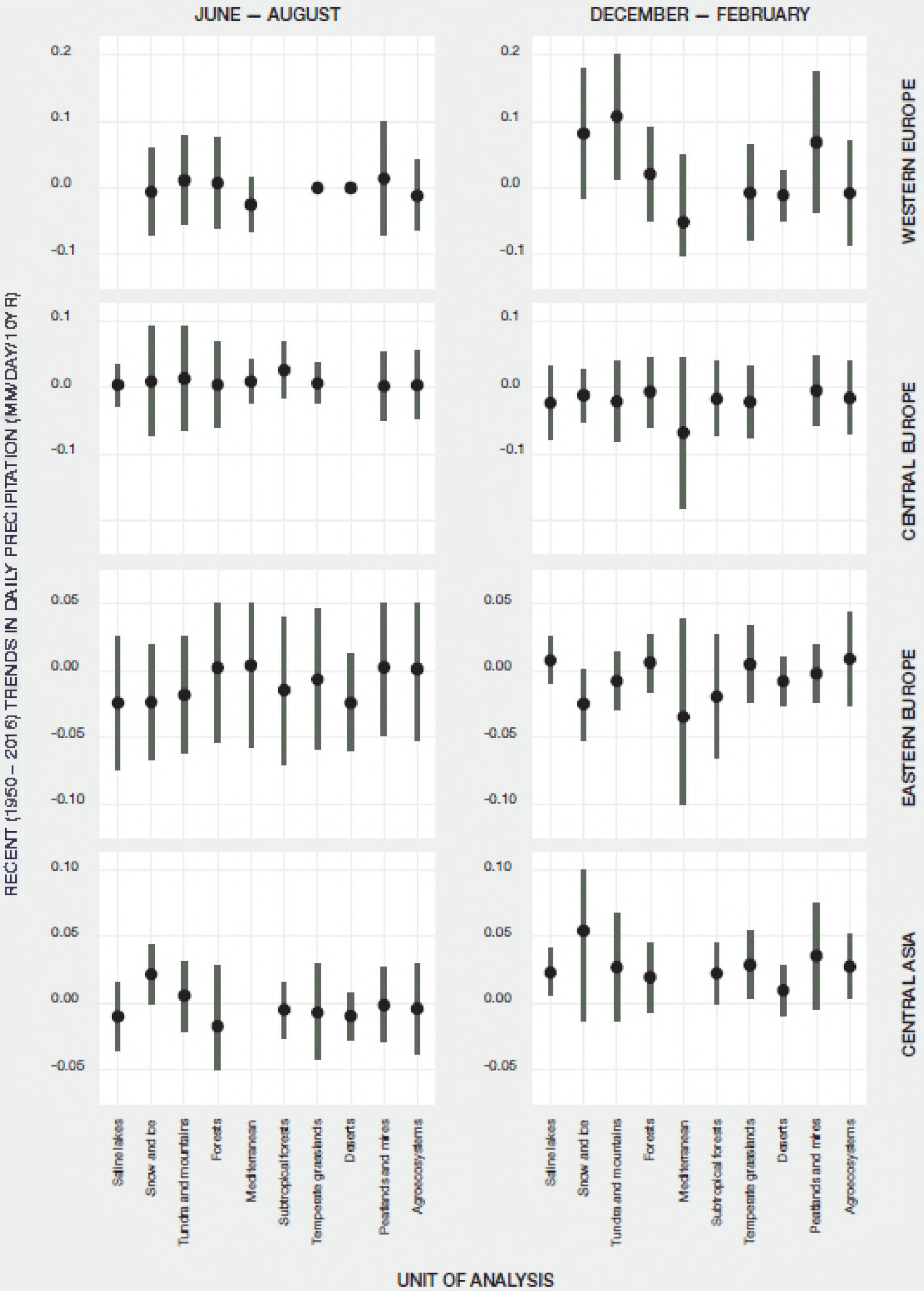


Figure 4 61 **Historical precipitation trends (1950–2016) for units of analysis of Europe and Central Asia.**

The calculated linear trend is expressed in mm/day/decade (Hartmann *et al.*, 2013). Colour bars represent 90% confidence intervals around trend estimates. If 90% confidence intervals cross the dashed line of zero, the estimated trend is considered statistically insignificant. Units of analysis are: saline lakes; snow and ice (snow and ice-dominated systems - everything north of or higher than alpine); tundra and mountains (tundra and mountain grasslands - only high elevation grasslands); forests (broad-leaved, mixed and coniferous forests); Mediterranean (Mediterranean forests, woodlands and scrub); subtropical forests (tropical and subtropical dry and humid forests); temperate grasslands; deserts; peatlands and mires; agroecosystems. Source: Own representation.



Precipitation is projected to increase in future across Europe and Central Asia according to all RCP scenarios (see **Figure 4.62**), yet with important uncertainties. Increases are projected to be larger for winter than for summer. Summer precipitation anomalies for 2041-2060 relative to 1986-2005 for RCP 2.6 range from −0.06 to 0.24mm/day and for RCP 8.5 range from −0.07 to 0.21mm/day (5 to 95%) (see **Figure 4.62**). Projected winter precipitation anomalies for 2041-2060 relative to 1986-2005 for RCP 2.6 range from 0 to 0.21mm/day and for RCP 8.5 range from 0.05 to 0.23mm/day (5 to 95%) (see **Figure 4.62**).

At a subregional scale, most projected precipitation changes fall within one standard deviation of the natural variability over much of Europe and Central Asia (see **Figure 4.63**). For summer, significant increases in precipitation are projected for northern areas of Western Europe and Eastern Europe, and decreases in southern parts of Western Europe (see **Figure 4.63**), in accordance with Kirtman *et al.* (2013). For winter, increases in precipitation are projected over Eastern Europe, Central Asia, and in northern parts of Western Europe (see **Figure 4.63**).

Changes in summer precipitation are projected for most units of analysis throughout Europe and Central Asia except for deserts (**Figure 4.64**). Changes in winter precipitation are projected for almost all units in Western Europe, Eastern Europe and Central Asia, whereas projected changes for Central Europe are less clear with values projected within natural variability range depending on scenarios (**Figure 4.64**). Specifically, in Western Europe, changes (increases and decreases, depending on units) are projected for both summer and winter precipitation. In Central Europe, decreases are projected for all units for summer precipitation, whereas projected changes are variable among units for winter. In Eastern Europe, increases are projected for most units for both summer and winter precipitation, except for summer precipitation in Mediterranean and subtropical forest units, which is projected to decrease. In Central Asia, increases are projected for all biomes for winter precipitation only.

In summary, precipitation will very likely change throughout Europe and Central Asia, will likely increase for most units of analysis in Eastern Europe and Central Asia and for northern units in Western Europe, and likely decrease for southern units in Western Europe and Central Europe.

Figure 4 62 Time series of precipitation trends relative to 1986–2005 averaged over land grid points for (from top to bottom) Europe and Central Asia, Western Europe, Central Europe, Eastern Europe, Central Asia in summer (June to August, left panel) and winter (December to February, right panel).

Thin lines denote one ensemble member per model and thick lines the CMIP5 multi-model mean (see Figure 4.3 for details). On the right-hand side of each graph the 5th, 25th, 50th (median), 75th and 95th percentiles of the distribution of 20-year mean changes are given for 2041–2060 under the four IPCC representative concentration pathway (RCP) scenarios. Source: Own representation.

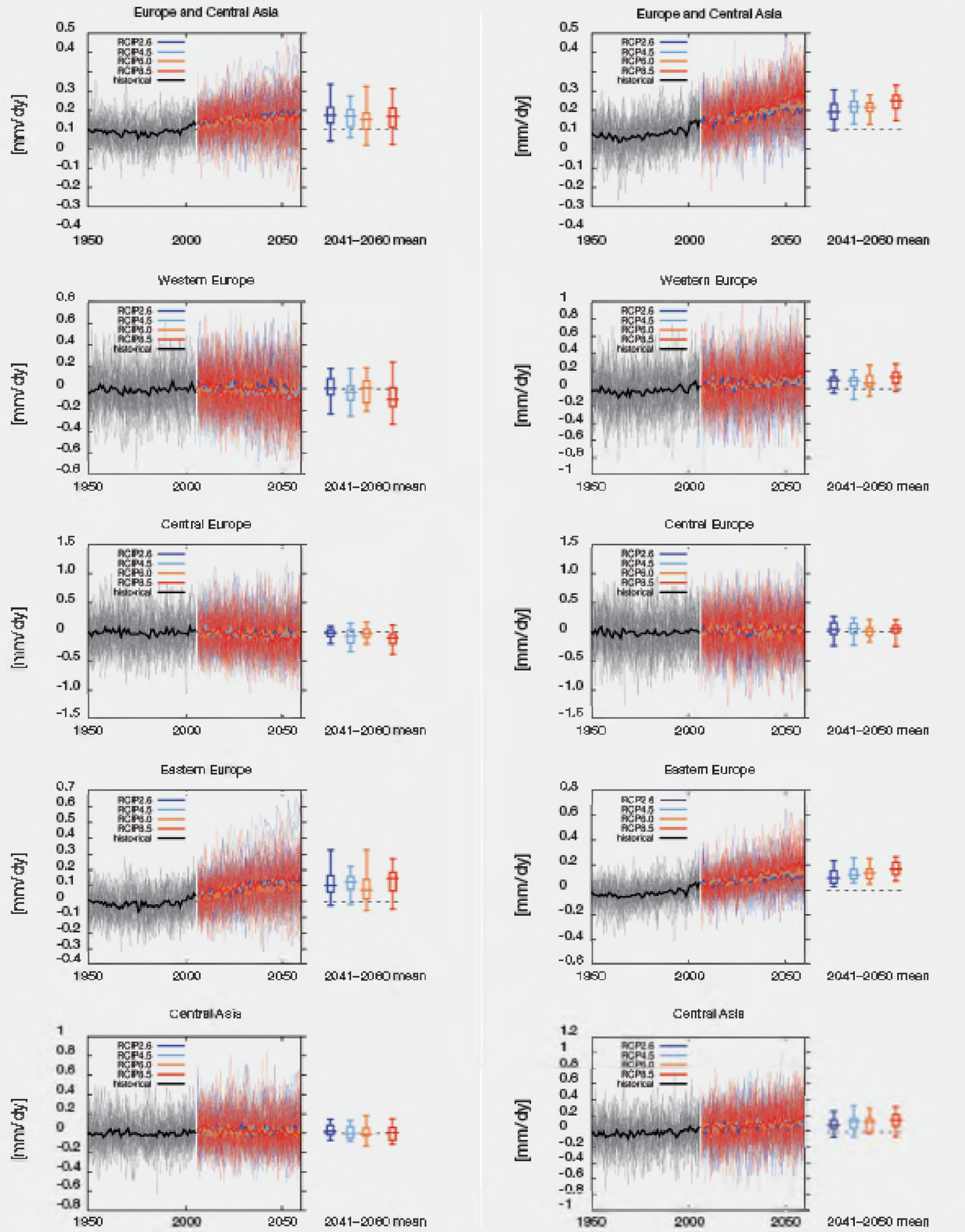


Figure 4 63 **Maps of projected (linear) precipitation trends by 2041–2060 relative to 1986–2005 under the RCP4.5 (top two panels) and the RCP 8.5 scenario (bottom two panels) for summer (June to August) and winter (December to February).**

Ensemble means from CMIP5 are presented here, extracted from data of IPCC (2013b). Source: Own representation.

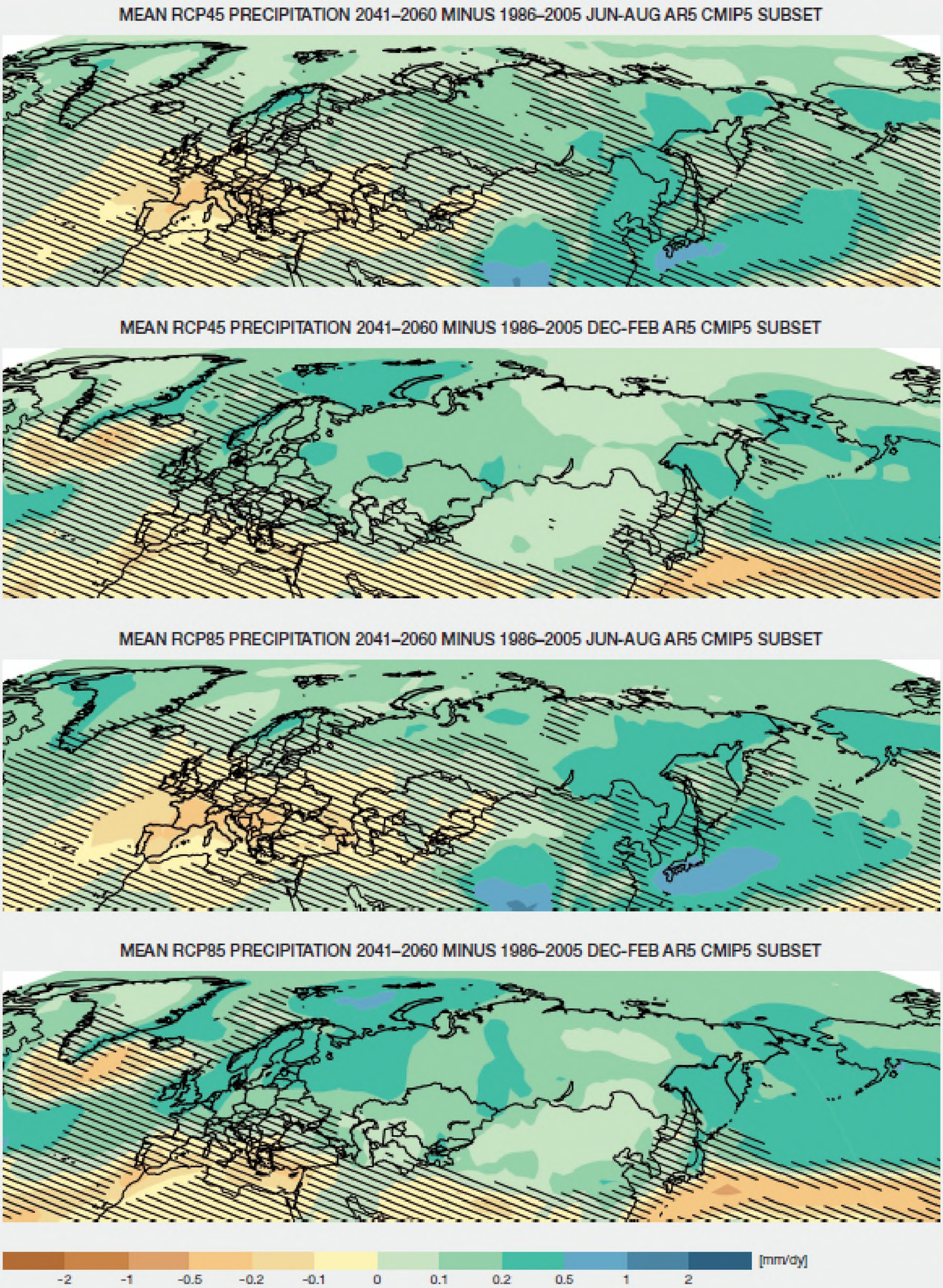
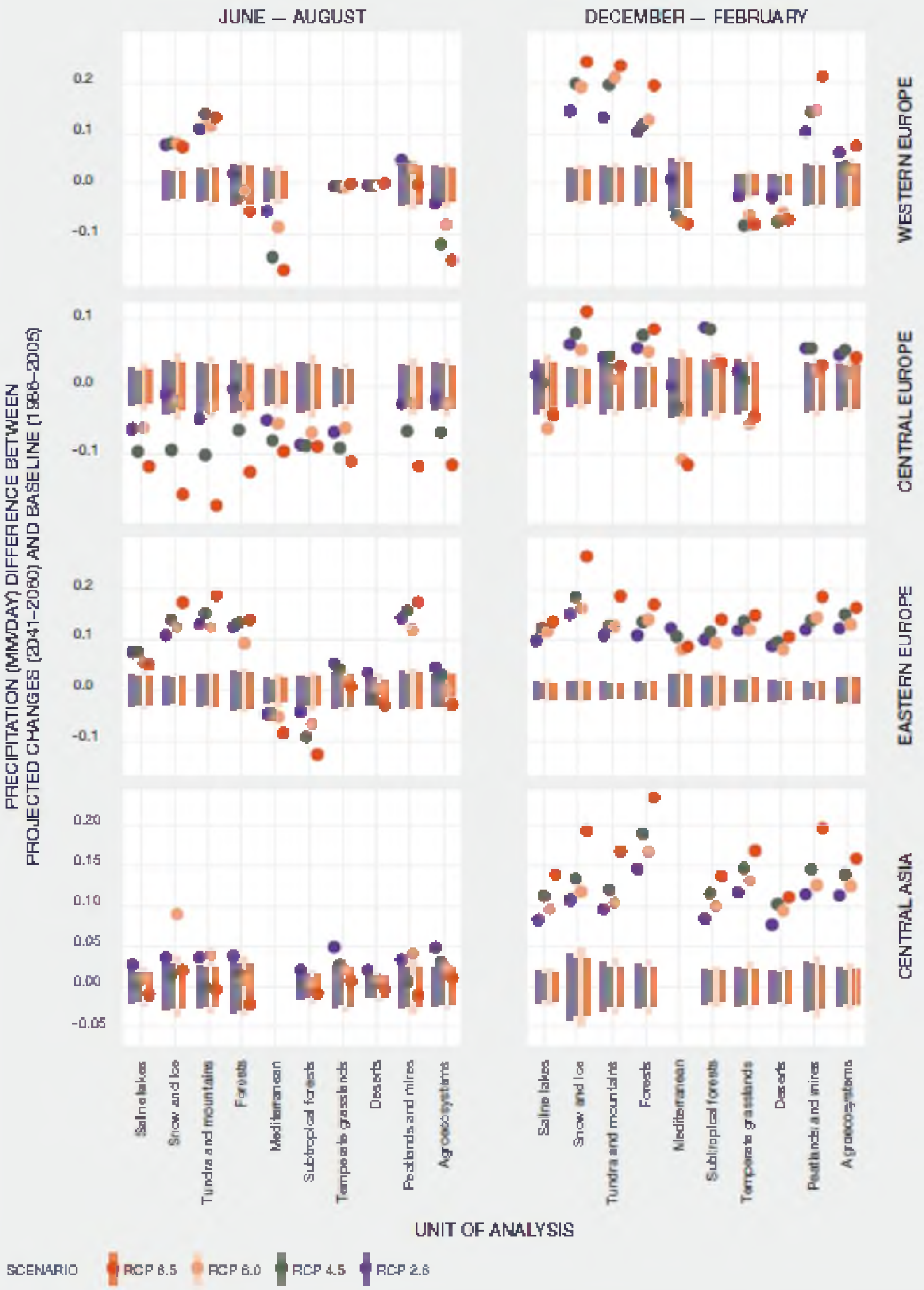


Figure 4 64 **Projected precipitation changes (2041–2060) relative to 1986–2005 averaged over land grid points for units of analysis of Europe and Central Asia, Western Europe, Central Europe, Eastern Europe, Central Asia.**

Points represent model ensemble means from CMIP5 for each representative concentration pathway (RCP) scenario. Bars represent one standard deviation of natural variability around 1986–2005 means. Units of analysis are: saline lakes; snow and ice (snow and ice-dominated systems - everything north of or higher than alpine); tundra and mountains (tundra and mountain grasslands – only high elevation grasslands); forests (broad-leaved, mixed and coniferous forests); Mediterranean (Mediterranean forests, woodlands and scrub); subtropical forests (tropical and subtropical dry and humid forests); temperate grasslands; deserts; peatlands and mires; agroecosystems. Source: Own representation.

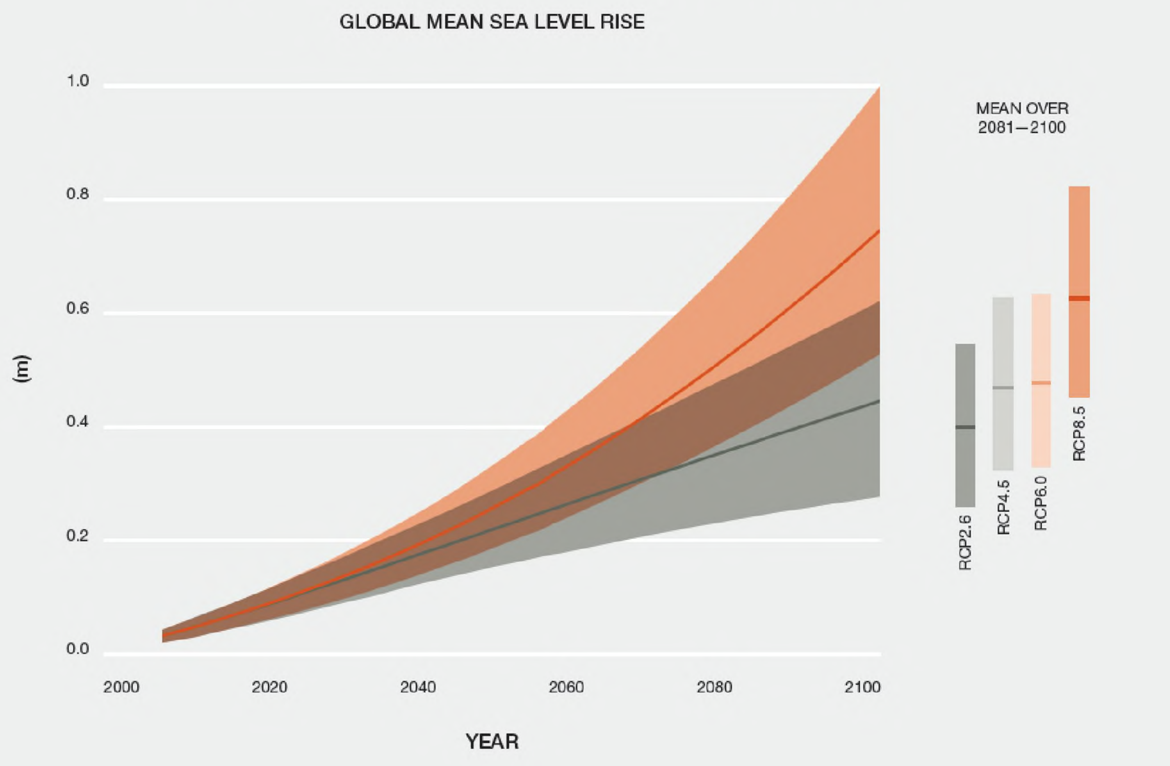


4.7.2.3 Sea-level change

The sea-level has risen ca. 150 mm in the past century. Recent trends in sea-level rise are uniform across the globe. Eastern Europe and Central Asia are exceptional because of the Caspian Sea, where sea-level change is decoupled from global sea-level rise and trends are uncertain. The Caspian Sea level decreased throughout the first half of the previous century, and subsequently increased to almost re-reach its historical level (Arpe & Leroy, 2007; Leroy *et al*, 2006 and references therein).

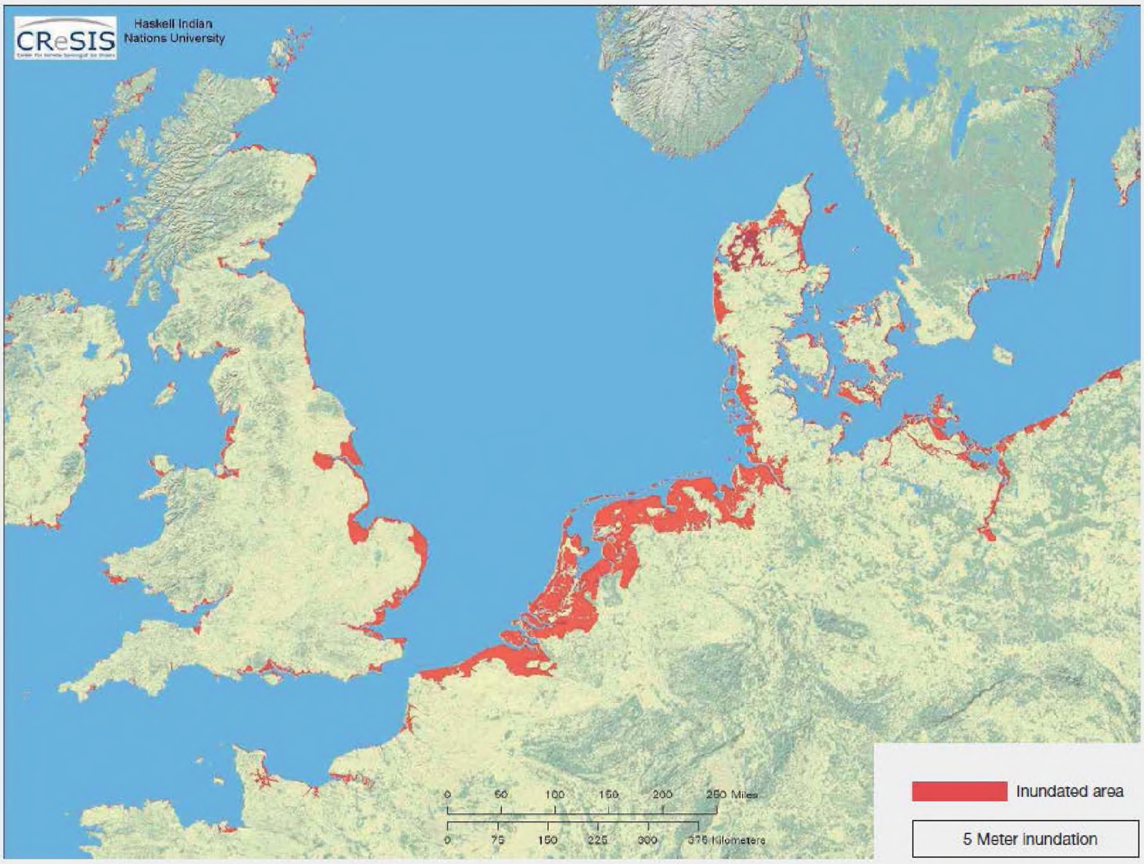
In the future sea-level is projected to continuously increase globally (**Figure 4.65**) due to various reasons, including temperature induced swelling and melting land ice, to reach a total increase by 2100 of 0.3-1 m (IPCC, 2013b). Sea level changes in the Caspian Sea will most likely depend on the projected precipitation regimes of its watershed (Arpe & Leroy, 2007), which are least certain but potentially increase, while fluctuations may be extreme (Roshan *et al.*, 2012).

Figure 4 65 Projected sea level rise according to the different representative concentration pathways (RCP) used in the 5th assessment report of the Intergovernmental Panel on Climate Change. Source: Figure SPM.9 from IPCC (2013b).



While coastal habitats and estuaries experience strong effects from rise in sea-level, benthic habitats are less concerned (only near-shore) and pelagic habitats are least affected. Within the region, a sea-level rise of at least 1m would affect primarily the heavily populated regions in Western Europe (mostly The Netherlands, but also Germany, Denmark and UK), where a rise of 1-5 m (**Figure 4.66**) would affect up to 22 million inhabitants. A realistic sea level rise of just 1m would affect almost the same amount of land, biomes and people as a 5m rise. Such drastic shifts will have a strong effect on coastal ecosystems, on sessile and migrating animals (birds), and on the structuring of the coastal biomes.

Figure 4 66 Projected impact from a sea level rise (5 m) for north-western Europe. A population of 21.7 million is calculated to be at risk in the inundation area (Rowley *et al.* 2007).



4.7.2.4 Trends in glaciers and permafrost

4.7.2.4.1 Glacier melting

There is high general confidence that current glacier extents are out of balance due to increased recent temperatures, indicating that glaciers will continue to shrink in the future even without further temperature increase (Hagen *et al.*, 1993; IPCC, 2013b). The average rate of ice loss from glaciers around the world (including both Alpine and Arctic glaciers), excluding glaciers on the periphery of the ice sheets, was very likely 226 [91 to 361] Gt yr^{-1} over the period 1971 to 2009, 275 [140 to 410] Gt yr^{-1} over the period 1993 to 2009 and 301 [166 to 436] Gt yr^{-1} between 2005 and 2009 (IPCC, 2013b). Most glaciers around the globe have been shrinking since the end of the Little Ice Age (ca. 1300-1850), with increasing rates of ice loss since the early 1980s.

There is, however, regional variation and also wide variation within regions related to precipitation patterns, altitudinal range, area distribution and dynamic responses. For instance, in the Jotunheimen region, the highest mountain massif in Norway, the general trend (based on Landsat TM/ETM+ data from 2003) is glacier recession, while some glaciers in that region increased their size or remained nearly unchanged over these decades (Andreassen *et al.*, 2008). Another example is the Svartisen region in Norway, where the overall glacier area changed from 1968 to 1999 was close to zero, but where there was a stronger relative area loss towards the wetter coast (Paul & Andreassen, 2009). Generally, the investigated glaciers in the Jotunheimen region shrank since the 1930s, with an overall area reduction of about 23% for 38 glaciers. Since the 1960s the area reduction for 164 glaciers in that region was 12% (*c.* 3.2% per decade) and since 1980 3% per decade. The 3.2% per decade reduction in glacier area since 1965 and 3% since 1980 in Jotunheim is comparable to other parts of the region with mountain and valley glaciers. In the Swiss Alps, the area change was −2.2% per decade for the period 1850-1973 and -6.4% per decade for the period 1973-1999 (Paul *et al.*, 2004). In the Jostedalsbreen region, Norway, there was an area loss of 2.3% per decade in the period 1966-2006 (Paul *et al.*, 2011). Inventory results from the Austrian Alps show a net reduction of glacier area of 17% between 1969 and 1998 (Lambrecht & Kuhn, 2007), or −6% per decade. In southern Spitsbergen, most glaciers - whether tidewater or land-terminating, large or small, debris-covered or comparatively clean ice types – have undergone retreat, both over the period 1936-1990 (832.5 km^2) and 1990-2008 (243.1 km^2). In the latter period, the glacier area change was on average around −3% per decade (König *et al.*, 2014). Also in other parts of Svalbard, glacier area has been decreasing substantially during the past 50 years (Hagen *et al.*, 1993). In the Russian High Arctic, the archipelagos have lost ice at a rate of −9.1 ± 2.0 Gt per year, which corresponds to a sea level contribution of 0.025 mm per year. Approximately 80% of the ice loss came from Novaya Zemlya with the remaining 20% coming from Franz Josef Land and Severnaya Zemlya (Moholdt *et al.*, 2012). In the Tien Shan (in the border region of Kazakhstan, Kyrgyzstan and north-western China) the area reduction was 32% between 1955 and 1999 (Bolch, 2007), or – 9% per decade.

4.7.2.4.2 Permafrost thawing

There is agreement that near-surface permafrost extent at high northern latitudes will be reduced as global mean surface temperature increases. By the end of the 21st century, the area of permafrost near the surface (upper 3.5 m) is projected to decrease by between 37% (RCP2.6) and 81% (RCP8.5) for the model average (IPCC, 2013b). Permafrost temperatures have increased, and the depth of seasonally frozen ground has become reduced, in most regions since the early 1980s, although the rate of increase has varied regionally. Also, the temperature increase for colder permafrost was generally greater than for warmer permafrost. Significant permafrost degradation has occurred in the Russian European north, where observed warming was up to 2 °C in the period 1971 - 2010 (Malkova, 2008; Oberman, 2008, 2012; Romanovsky *et al.*, 2010). In the latter region, a considerable reduction in permafrost thickness (up to 15 m) and areal extent (poleward shift up to 80 km for discontinuous and up to 50 m for continuous permafrost extent) has been observed over the period 1975 to 2005 (IPCC, 2013b).

In northern Yakutia (Russia), permafrost temperatures have warmed by 0.5-1.5 °C between the early 1950s and 2009 (Romanovsky *et al.*, 2010), and in the Trans-Baykal region (Russia) by 0.5-0.8 °C between the late 1980s and 2009 (Romanovsky *et al.*, 2010). In Tian Shan, permafrost temperature has increased by 0.3-0.9 °C during 1974-2009 (Marchenko *et al.*, 2007; Zhao *et al.*, 2010). In the Alps, permafrost temperatures have increased by 0.0 – 0.4 °C in the period 1990-2010 (Christiansen *et al.*, 2012; Haeberli *et al.*, 2010; Noetzli & Mühll, 2010), and in the Nordic countries by 0.0-1.0 °C during 1999-2009 (Christiansen *et al.*, 2010; Isaksen *et al.*, 2011). The thickness of the seasonally frozen ground in some non-permafrost parts of the Eurasian continent likely decreased, in places by more than 30 cm from 1930 to 2000.

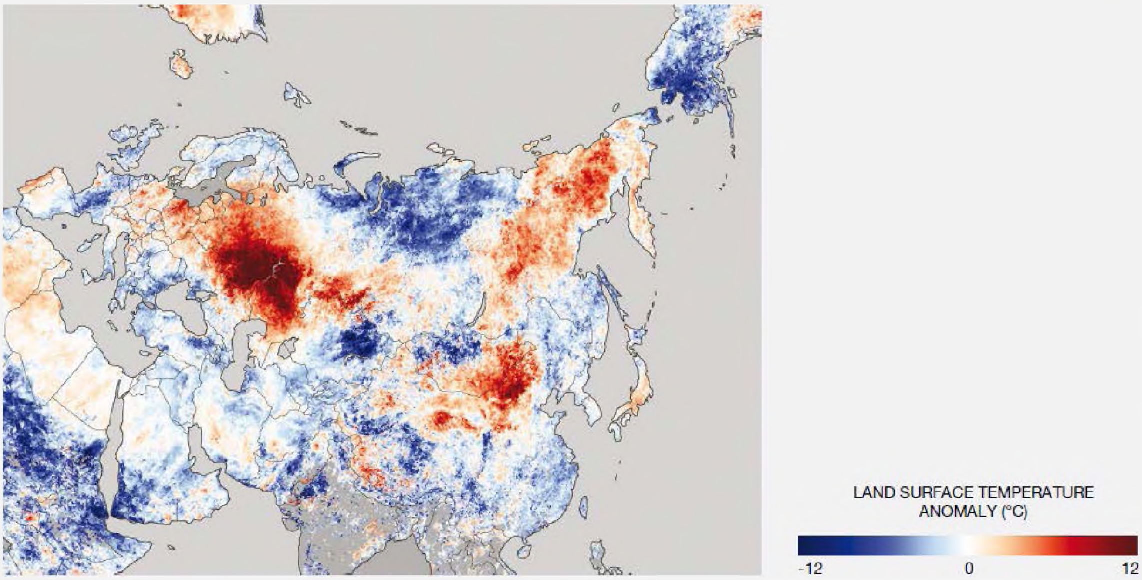
4.7.2.5 Trends in extreme events

4.7.2.5.1 Drought and temperature extremes

In recent decades, drought and heat waves have increased in Western and Central Europe, while showing a north-south gradient in both subregions (drier in the south, no change or moister in the north, Alexander *et al.*, 2006; Kiktev *et al.*, 2003; Sheffield & Wood, 2008a). These recent trends for Western, Central, and Eastern Europe are considered likely, while trends in Central Asia are as likely as not (IPCC, 2012). Drought is often associated with extreme heat waves, which can stretch over large regions, and which may result in punctuated drought events (**Figure 4.67**).

Figure 4 67 **Russian heat wave in 2010, which has stretched over large parts of Eastern Europe and affected also Central Europe and Central Asia.**

Image derived from NASA. Colours indicate the degree of positive (red, hotter) or negative (blue, colder) deviations from July 20-27, 2010 compared to the average temperature of the same dates over the measurement period 2000–2008, as obtained by the MODIS Earth Observation Satellite. Source: https://earthobservatory.nasa.gov/images/45069.



Projected trends in drought are considered very likely in Western and Central Europe since projections are in high agreement (drier in the south, no change or moister in the north, Alexander *et al.*, 2006; Kiktev *et al.*, 2003; Sheffield & Wood, 2008b), while for Eastern Europe and Central Asia the projected trends are about as likely as not. Generally, the largest increase in the duration and intensity of drought periods is projected for Mediterranean climate zones (Beniston *et al.*, 2007; May, 2008), while for northern parts of Western Europe only moderate or no increase in drought is expected (IPCC, 2012), so this trend is as likely as not. Future trends for Eastern Europe vary between projections, but also spatially, with potentially less drought in northern parts of Eastern Europe (Dai, 2011; Sillmann & Roeckner, 2008). Seasonality in drought events is also expected to change throughout Europe and Central Asia (Orlowsky & Seneviratne, 2012).

4.7.2.5.2 Floods

Recent trends in floods are very difficult to assess because of a lack of long time-series of gauge-stations and because floods are rare. Therefore, flood assessments of the recent past are least certain and without a directional trend throughout Europe and Central Asia (IPCC, 2012).

Projections of floods are to a large degree based on projections of heavy precipitation events which, in turn, are based on physical reasoning, but changes in snow accumulation and the timing of snow-melt potentially also contribute to flood-risk projections. However, the magnitude of this contribution is uncertain (IPCC, 2012). Heavy precipitation is expected to increase in Western and Central Europe, with highest certainty and magnitude in the north while Mediterranean Europe may not experience the associated increase in flood risk (Beniston *et al.*, 2007; Frei *et al.*, 2006; Kendon *et al.*, 2008). While the trend is consistent between summer and winter, the magnitude may vary between seasons, but also spatially (Frei *et al.*, 2006; Kendon *et al.*, 2008). Coastal regions of Western Europe may be exposed to north-shifted extra-tropical cyclones (IPCC, 2012) and thus be under increased flooding risk. Increased frequency of heavy precipitation events is highly certain in Eastern Europe (IPCC, 2012)[35], while for Central Asia projections are partly contradictory resulting overall in no projected increase of heavy precipitation, but with least certainty (IPCC, 2012).

4.7.2.5.3 Fire

An observed global increase in fire frequency and burnt area is most likely driven by climate (Marlon *et al.*, 2008). Fires have generally increased in recent decades in the Mediterranean area (EEA, 2012a; Pausas, 2004). Forest fires have also generally increased in Europe and Central Asia (Schelhaas *et al.*, 2003), with highest increases in the southern parts of Western, Central and Eastern Europe (EEA, 2012a), while the boreal forest in the north of the Europe and Central Asia region does not show increased fire frequency (EEA, 2012a; Lehtonen *et al.*, 2014). Trends in frequency of fires in Central Asia are less certain, but fires have also generally increased (Goldammer *et al.*, 2004). Risk and spread of fire is often a direct consequence of multiple other direct drivers, notably of drought, heat, and tree mortality due to insect disturbance (Bigler *et al.*, 2005; Clark *et al.*, 2016; Gouveia *et al.*, 2016).

Fire danger is projected to increase, especially for the Mediterranean areas of Western and Central Europe (Karali *et al.*, 2014; Khabarov *et al.*, 2014), and potentially for large parts of the Alpine Arc (EEA, 2012a) and boreal forests of Western Europe (EEA, 2012a; Lehtonen *et al.*, 2014). Fire danger projections for Eastern Europe and Central Asia are less certain, but increase in fire risk is potentially low in the north and moderate in the south of Eastern Europe (Mokhov *et al.*, 2006; Tchebakova *et al.*, 2009), and generally increased in Central Asia (Goldammer *et al.*, 2004).

4.7.2.5.4 Windthrow

Trends in windthrow are difficult to assess because they are rare, thus confidence in emerging trends is low. Studies consistently report an increase in storms or storminess from 1960 to 1990, yet no long-term trend reaching further back in time is available (Allan *et al.*, 2009; Bärring & von Storch, 2004; Matulla *et al.*, 2008; Schelhaas *et al.*, 2003; Wang *et al.*, 2009). The number of available studies and the certainty of trend assessments are highest for Western Europe and lower for Central Europe, Eastern Europe, and Central Asia.

Future projections of extreme winds are highly uncertain (IPCC, 2012). A north-south gradient with more extreme winds in the north and less extreme winds in the south is projected for Western and Central Europe (Beniston *et al.*, 2007; Mcinnes *et al.*, 2011), but the expected poleward shift of extra-tropical storm tracks (IPCC, 2012) indicates increased extreme winds for Western Europe in general, and therefore an increase is very likely for coastal habitats. Eastern Europe is projected to be under increased wind throw risk across most of its range, while Central Asia will likely experience less extreme winds (Beniston *et al.*, 2007; Mcinnes *et al.*, 2011).

[35] See table 3-3 therein

4.7.2.5.5 Trends in marine circulation and deoxygenation

Among all marine waters in Europe and Central Asia, the Mediterranean Sea could be particularly vulnerable to climate variations (Turley, 1999) and was identified as a hot spot for climate change (Giorgi, 2006). It is indeed characterized by very short ventilation and water residence times (70 years) compared to other oceanic zones (Durrieu de Madron *et al.*, 2011). This specificity makes it a marine area where climate variations may strongly and rapidly impact hydrodynamics and marine ecosystems.

4.7.2.5.6 Ocean warming

From the 1980s to the late 2000s, the surface temperature of the North Atlantic has warmed faster than the overall northern hemisphere, as is depicted in the Atlantic multi-decadal oscillation index (www.esrl.noaa.gov/psd/data/timeseries/AMO/). This is also seen in an enhanced warming of the upper ocean integrated to 700m, with particularly large changes in the eastern Atlantic inter-gyre region, as well as on the outskirts of the North Atlantic sub-polar gyre (but not in the Nordic Seas). Hydrological observations showed that the temperature and salinity of the western Mediterranean deep-water masses have increased by 0.0034C/year and 0.0011psu/year between 1959 and 1997 (Bethoux *et al.*, 1998). Numerical studies confirmed that the increase of net atmospheric heat flux to the sea surface associated to climate change could induce a warming and a salinization of Mediterranean water masses, in particular at the surface, as well as an intensification of the water column stratification and a weakening of the thermohaline circulation and winter deep convection (Adloff *et al.*, 2015; Bozec, 2006; Herrmann *et al.*, 2008; Somot *et al.*, 2006; Thorpe & Bigg, 2000).

4.7.2.5.7 Water masses and horizontal circulation

Climate change has also been identified as cause of weakening and shrinking the North Atlantic sub-polar gyre and a shift of the sub-polar front (Hatun, 2005). This westward shift of the sub-polar front implies that the waters in the eastern North Atlantic part of the inter-gyre gyre seem to originate in recent decades from further south (and get warmer and saltier) than in the 1950-60s. This represents a shift to more subtropical origin and implies an increased northward flow along Western Europe (Lozier & Stewart, 2008).

In addition, there is clear evidence from upper temperature and salinity measures of a near decadal variability in the North Atlantic sub-polar gyre, propagating in a few years all the way through the Nordic Seas towards the Barents Sea (Yashayaev & Seidov, 2015). Atmospheric forcing or input of cold and fresh water from the Arctic could contribute to these signals that have been observed since regular observations began at least 60 years ago. The origin of this ocean variability is debated, and could be in part natural but also anthropogenically enhanced (for example more North Atlantic oscillation-related atmospheric variability, or variable fresh water exports from the Arctic Ocean). It is also possible that some of these changes in the warming and vertical structure will reverse, as the Atlantic multidecadal oscillations (AMO) shifts to another phase, as has been witnessed twice in the past 120 years.

4.7.2.5.8 Vertical circulation and mixing

There is debate as to whether the meridional circulation component of the ocean circulation (Atlantic Meridional Overturning Circulation) has or has not become stronger during the last century. There is accumulating evidence that a slow-down of this circulation is outruled by its large interannual to decadal variability, for example induced by wind variability (Rahmstorf *et al.*, 2015). This seems logical, based on the expectation that Atlantic Meridional Overturning Circulation intensity is influenced by changes in surface water density. Thus, an observed reduction in surface density could result in a decrease in Atlantic Meridional Overturning Circulation. A decrease in surface density seems to have happened in those areas since 1996 until the early 2010s, as the change due to surface warming was not fully compensated by the change due to salinity increase. This is, however, difficult to accurately estimate from observations over the relevant time scales.

It is likely that some of these observations and regional patterns of variability might be dependent on natural multi-decennial variability such as the North Atlantic oscillation, but there are fewer observations to support this. However, these long time-scale trends can be interrupted as the result of intense vertical mixing in individual years, such as in 2005 in the Bay of Biscay (Somavilla *et al.*, 2016). Clearly, there is also a very large year-to-year variability as the result of surface forcing in the eastern Atlantic north of 35-40°N.

4.7.2.5.9 Ocean acidification

The uptake of increased anthropogenic CO_2 is causing profound changes in seawater chemistry resulting from increased hydrogen ion concentration (decrease in pH) referred to as ocean acidification (IPCC, 2013b). Repeated hydrographic sections provide an understanding of these changes in the basin-wide seawater CO_2 chemistry over multi-decadal timescales. The formation of "North Atlantic deep water" makes the Atlantic unique with regards to the depths to which anthropogenic CO_2 can penetrate over these time scales, reaching the bottom at about 3,000 m in the far north of the North Atlantic (Wanninkhof *et al.*, 2010). Here, the ocean acidification signal adds to that of temperature. As a consequence, the lysocline, i.e. the depth in the ocean below which the rate of dissolution of calcite increases dramatically, could be shallower in polar regions while the decreased rate of pH is expected to be similar to the other latitudes.

A recent estimate suggests that all water masses in the Mediterranean Sea are also already acidified (-0.14 to -0.05 pH units; Touratier & Goyet, 2011). Considering the highest values of this range, the Mediterranean Sea appears to be one of the most acidified marine basins in the world. Yet, as far as we know, no estimates of future acidification rates in the Mediterranean Sea have been carried out.

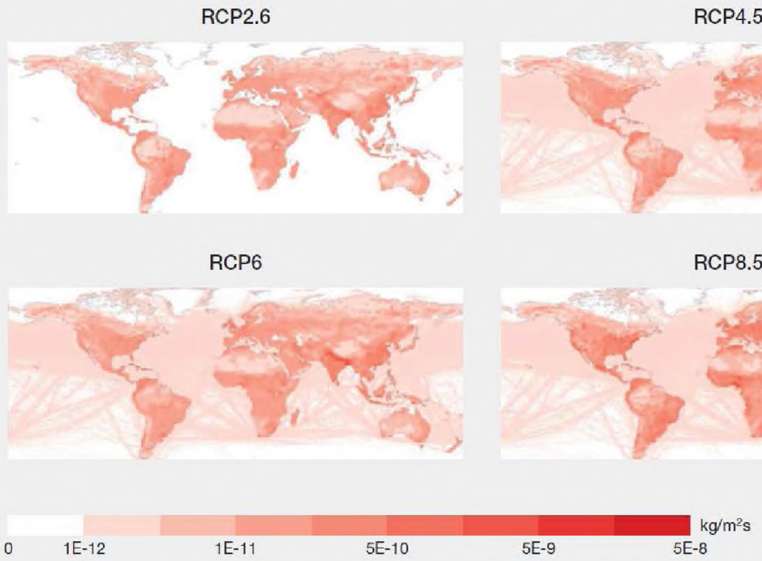
4.7.2.6 Trends in atmospheric CO_2 concentration

The atmospheric concentration of CO_2 is rising with the well-known seasonal pattern, and this trend is accelerating steadily (IPCC, 2013b). While from 1965-1974 the atmospheric CO_2 concentration increased by 1.06 ppm/yr, this increase has reached 2.11 ppm/yr for the decade 2005-2014, and in 2013 the level of 400 ppm of atmospheric CO_2 concentration was reached (IPCC, 2013b) for the first time sinceabout 23 million years ago (Pearson & Palmer, 2000). This steady increase in atmospheric CO_2 affects the ocean surface partial CO_2 pressure, and the increase of this partial pressure reduces the ocean pH, leading to steady acidification (**Figure 4.68**), and the trend is expected to continue over the 21st century (**Figure 4.68**).

Figure 4 68 A Partial pressure of dissolved CO_2 at the ocean surface (violet, upper curves) and in situ pH (grey curves), a measure of the acidity of ocean water recorded in three different locations in the Atlantic and Pacific Ocean. Source: Figure SPM.4 from IPCC (2013b). B Trends in concentrations of greenhouse gases. Grey area indicates the 98th and 90th percentiles (light/dark grey) of the recent EMF-22 study (Clarke *et al.*, 2009). Source: van Vuuren *et al.* (2011).

The steady increase in atmospheric CO_2 concentrations originates from spatially variable emission patterns, and these emissions are projected to depend on socio-economic determinants reflecting human decisions (indirect drivers) regarding greenhouse gas emissions. **Figure 4.69** illustrates the CH_4 (methane) emission patterns for 2100 according to the four representative concentration pathway (RCP) scenarios that represent different levels of radiative forcing.

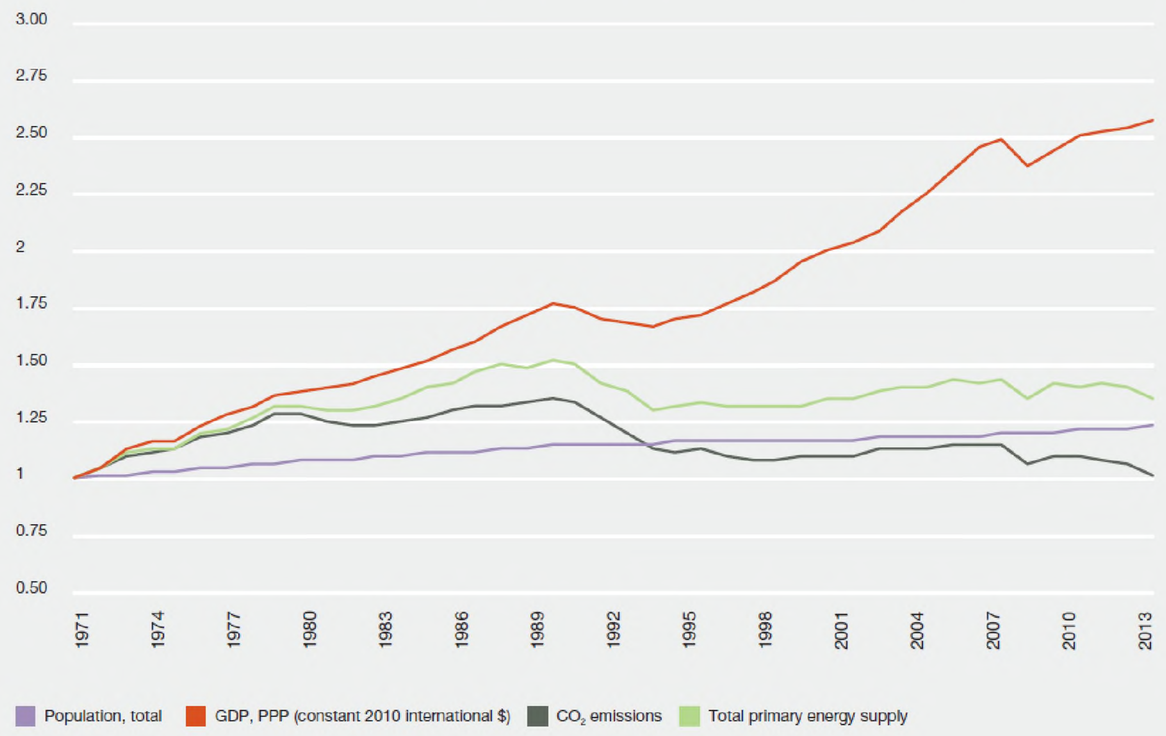
Figure 4 69 **Emission pattern for 2100, for CH_4 across the four representative concentration pathways (RCPs). Source: van Vuuren *et al.* (2011).**



4.7.3 Indirect drivers influencing climate change

Drivers of climate change are the same regardless of whether we are ultimatelyminterested in effects on biodiversity, nature's contributions to people, or other effects. Drivers of emissions from land-use change have been discussed in Section 4.5. Drivers of greenhouse gas emissions have been assessed in the fifth assessment report of the Intergovernmental Panel on Climate Change (Blanco *et al.*, 2014) and will not be repeated here. However, in contrast to the global level, both primary energy and CO_2 emissions in Europe and Central Asia have been reduced since 1990 (**Figure 4.70**).

Figure 4 70 Territorial CO_2 emissions in relation to other factors in Europe and Central Asia, 1971–2014. Source: Own representation based on OECD (2016).



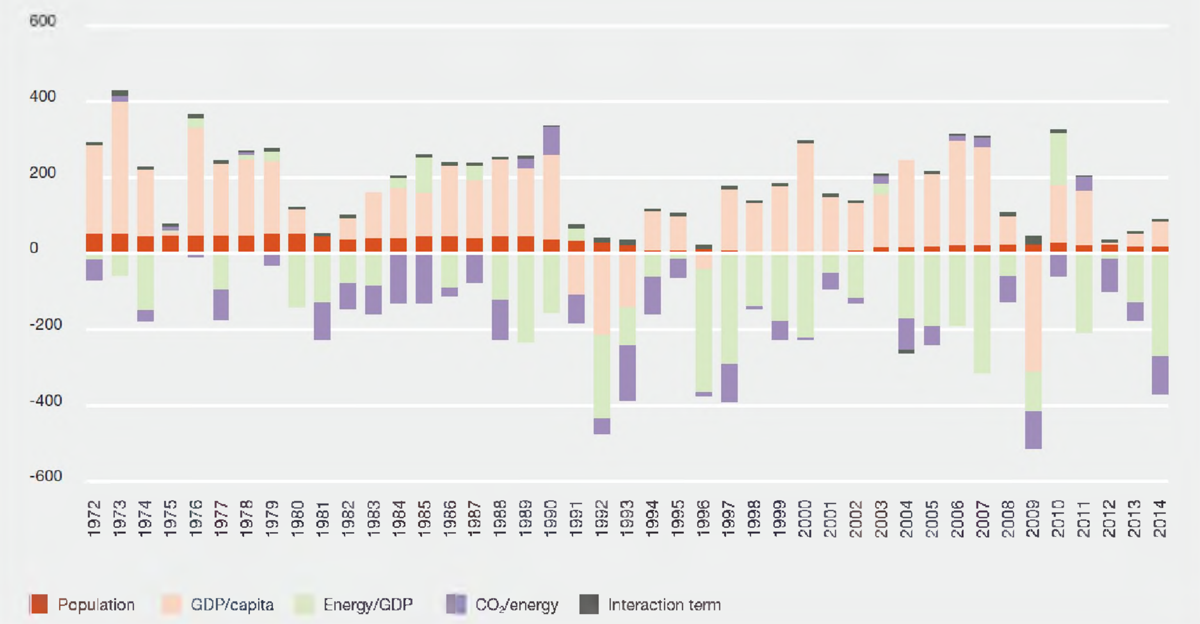
To analyse the reasons for this decreasing mission, we use the "Kaya identity" (Kaya, 1990) for territorial CO_2 emissions:

CO_2 emissions = $population \times \frac{GDP}{population} \times \frac{energy}{GDP} \times \frac{CO_2}{energy}$

The two last parts of the Kaya identity, energy intensity of GDP and CO_2 content of energy production, have declined since 1990 in all developed and large developing countries mainly due to technology, changes in economic structure, the mix of energy sources, and changes in the participation of inputs such as capital and labour used (Blanco *et al.*, 2014).

Figure 4.71 shows the relative contribution of each term of the Kaya identity to the annual change in CO_2 emissions in Europe and Central Asia. By comparing the size of the different bars one notices that the growth of GDP per capita (second term of the Kaya identity) is the main driver of CO_2 emission increase, which is well-established (Blanco *et al.*, 2014) and that its effect has for most years only been partially offset by improvements in the energy intensity of GDP (third term) and the CO_2 emissions intensity of energy production (fourth term).

Figure 4 71 Kaya decomposition of CO_2 emissions change in Europe and Central Asia. Source: Own representation based on OECD (2016). "Energy" is total primary energy supply, while "GDP" is taken in purchasing power parity values at 2010 international US dollars.



The time series in **Figure 4.71** shows two structural breaks, the first after the dissolution of the Soviet Union and the second following the great recession. These years with negative GDP growth (1991-1993 and 2009) were also the years with highest reduction in CO_2 emissions.

As reported in **Table 4.7**, between 1995 and 2008 energy and emissions increased at a lower rate than before 1990, while average GDP continued increasing at more or less the same rate (relative decoupling). In the last four years of available data (2011-2014), there is evidence of small increases in GDP growth, but decreasing paths in both energy and CO_2 emissions (absolute decoupling).

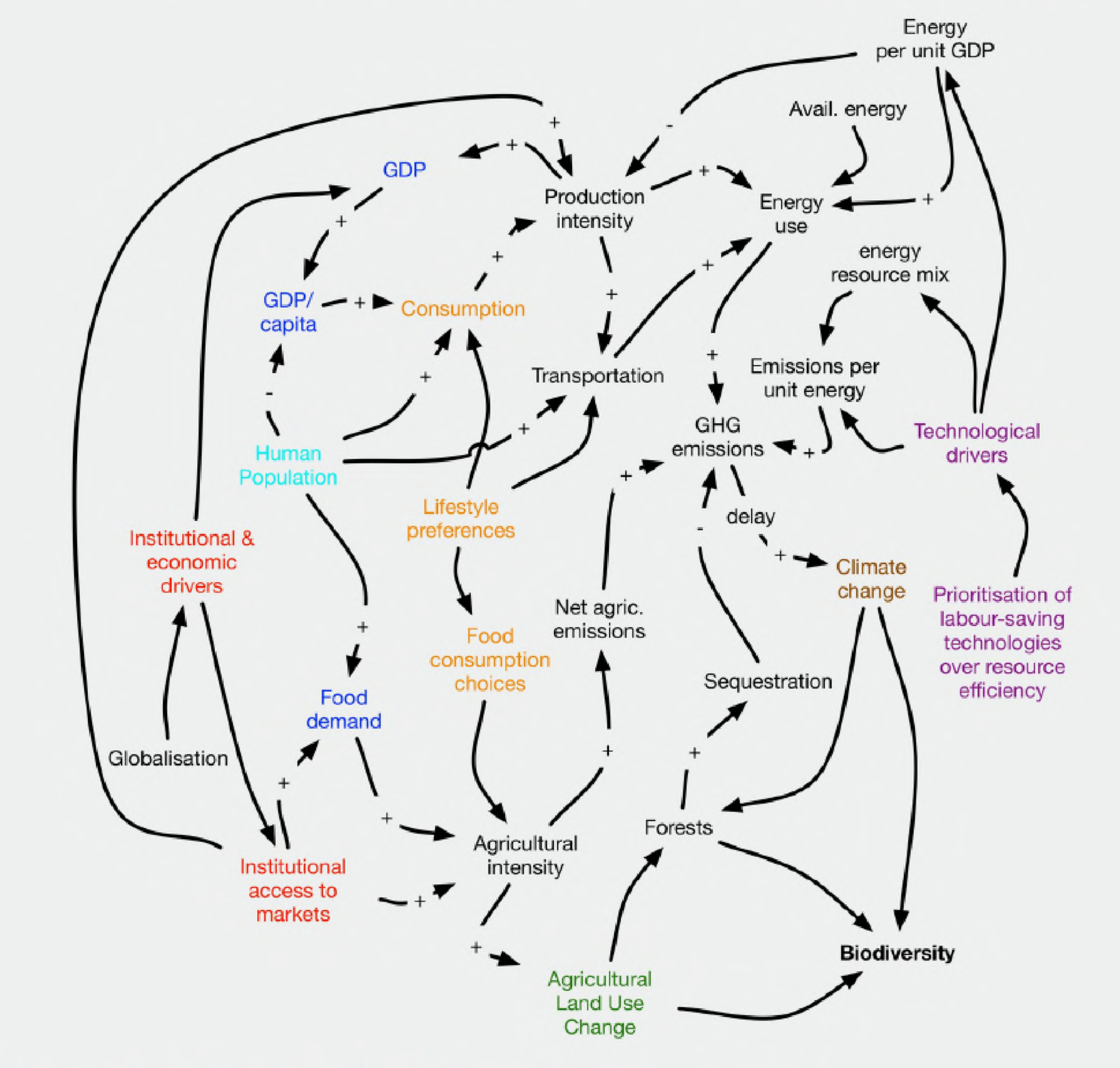
Table 4.7: Rates of change of population, GDP, CO_2 emissions, and energy in Europe and Central Asia. Source: Own elaboration based on OECD (2016). PPP denotes purchasing power parity.

Rates of growth of	1971-1990	1995-2008	2011-14
Population	0.7%	0.2%	0.4%
GDP, PPP (constant 2010 international US $)	3.0%	2.9%	1.4%
CO_2 emissions	1.6%	0.1%	-2.0%
Total primary energy supply	2.2%	0.7%	-1.2%

Two caveats are to be considered. First, the data refer to the aggregation of all countries in Europe and Central Asia. Within this region different patterns are observable and, as noted by the Intergovernmental Panel on Climate Change, the increase of emissions for an additional person varies widely, depending on geographical location, income, lifestyle, and the available energy resources and technologies (Blanco *et al.*, 2014). Second, there is no clear evidence whether (and to what extent) the relative decoupling of CO_2 emissions from GDP growth, indicated by much slower growth of CO_2 partial to GDP as observed from 1995, and the absolute decoupling, indicated by a CO_2 decrease at growing GDP from 2011 are the outcome of interregional flows, e.g. de-industrialization in the region caused by economic growth of countries in other regions. According to some researchers, the relevant decoupling is between prosperity and CO_2 emissions, not GDP growth and CO_2 emissions (Jackson, 2009; Raworth, 2017; van den Bergh, 2010).

The drivers of climate change are not limited to the energy and transportation sectors or the Kaya identity. In Section 4.5.1 we assessed intensive agriculture as a carbon source and sequestration by forests as carbon sink. A more comprehensive illustration of major drivers of climate change and their interactions, and impacts on biodiversity, is presented in **Figure 4.72**.

Figure 4 72 Causal loop diagram of drivers of climate change, illustrating the main causes of climate change. Source: Own representation. See text for further clarification.



Indirect drivers: Institutional Economic Cultural Demographic Scientific and Technological

4.8 Drivers and effects of invasive alien species

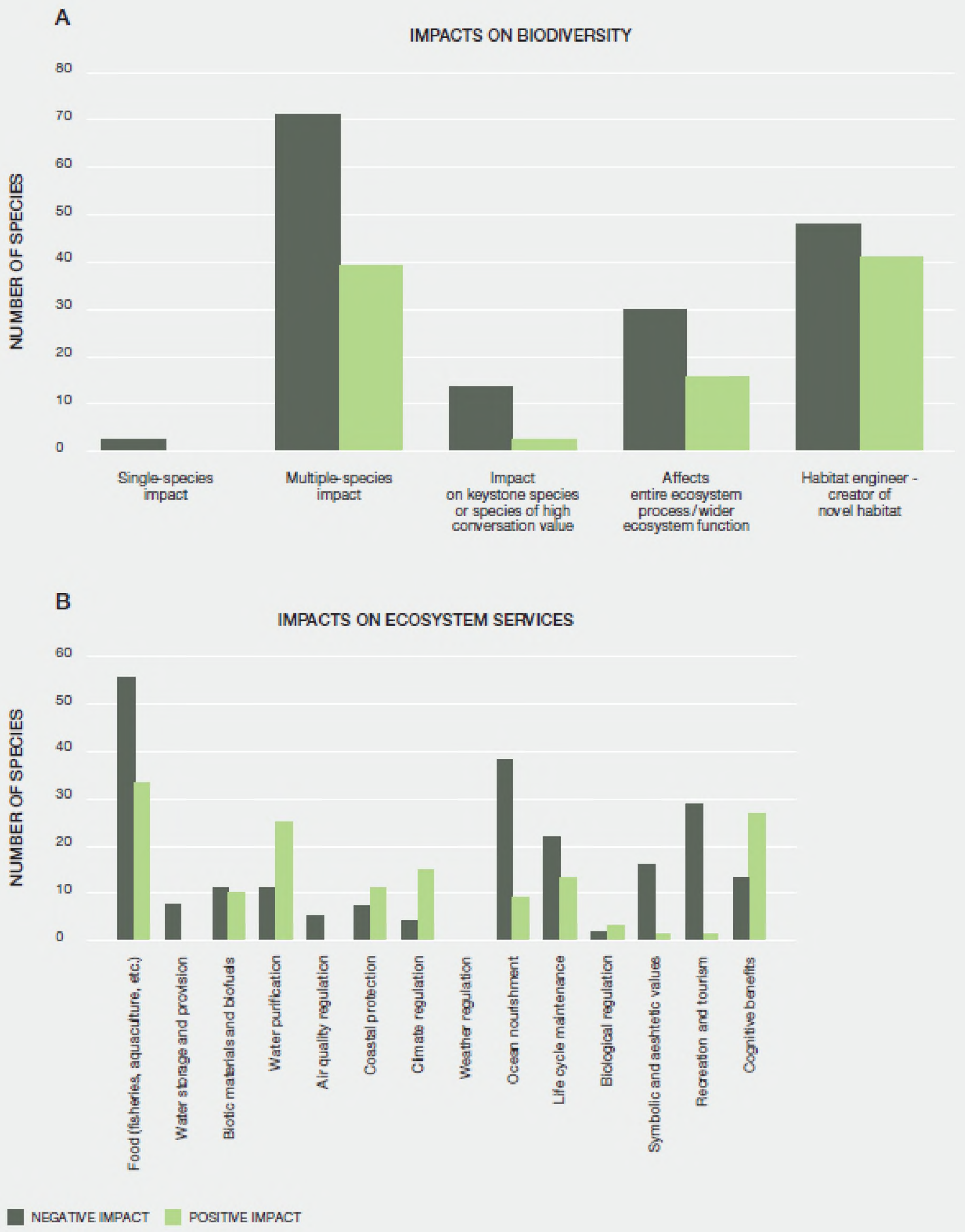
4.8.1 Effects of invasive alien species on biodiversity and nature's contributions to people

Invasive alien species are among the important direct drivers of loss of biodiversity and nature's contributions to people across Europe and Central Asia, especially in combination with other direct drivers (Section 4.9.1) (Anastasopoulou *et al.*, 2007; Clavero *et al.*, 2009; Katsanevakis *et al.*, 2014; MEA, 2005b; Nelson *et al.*, 2005; Sala, 2000). Invasive alien species generally tend to have negative effects on biodiversity (**Figure 4.73**). However, their magnitude and direction vary both within and between types of impact, across taxa and environments (Bradshaw *et al.*, 2016; IPBES, 2016a; Potts *et al.*, 2016; Vilà & Ibáñez, 2011). Negative effects can include displacement and extinction of native species, gene pollution, homogenization of communities, modification of biological interactions, communities, habitats and ecosystem functions, with consequences for human health; and agricultural and economic production (IPBES, 2016a; Katsanevakis *et al.*, 2014; Vilà *et al.*, 2010). Some alien species, and even some invasive alien species, have positive impacts, which include provision of habitat; increasing local species richness and associated ecosystem services, with subsequent economic gains; ecosystem engineering; and aesthetic and cultural value (Goodenough, 2010; IPBES, 2016a; Schlaepfer *et al.*, 2011). Data limitations across the region, particularly in Central Asia (Dinasilov, 2013; Khlyap & Warshavsky, 2010; Mamilov *et al*, 2010; Reshetnikov, 2010) and for pathogens (Roy *et al.*, 2017), impede assessment of trends associated with invasive alien species. Priority should be given to improving the evidence-base for impacts of invasive alien species and thereby capacity to inform future assessments (Gurevitch & Padilla, 2004; Jeschke *et al.*, 2014).

Invasive alien species have considerable economic impacts on forestry (Roy *et al.*, 2014b) and agriculture (Paini *et al.*, 2016). Invasive alien insects alone have been estimated to cost US$2-3.6 billion per year in Western and Central Europe, mostly due to impacts on forestry and agriculture, while invasive alien species can have significant impacts on human health, for example via disease transmission and allergens (Bradshaw *et al.*, 2016; Schindler *et al.*, 2015). However, such impacts are considered to be grossly underestimated because of the limited number of studies available within and across Europe and Central Asia (Bradshaw *et al.*, 2016; Schindler *et al.*, 2015).

Most invasive alien species present in marine habitats in Europe and Central Asia have been reported to affect more than one species (**Figure 4.73**) (Katsanevakis *et al.*, 2014). Invasive alien species within freshwater environments can cause alterations to the physical, chemical and ecological state eliciting cascading effects that modify biodiversity (Martel *et al.*, 2014), and ecosystem structure and function (Kernan, 2015). Freshwater ecosystems are particularly vulnerable to invasions and the impacts of invasive alien species and so the magnitude of the impact and the consequential ecological transformations are often more severe than in terrestrial ecosystems (R. Francis, 2012; Ricciardi, 2015).

Figure 4 73 Overview of the number of marine alien taxa (out of 110 investigated taxa) that have been reported to affect A biodiversity and B ecosystem services (nature's contributions to people) in marine habitats of Europe and Central Asia. Source: Katsanevakis *et al.* (2014).



4.8.2 Trends in invasive alien species

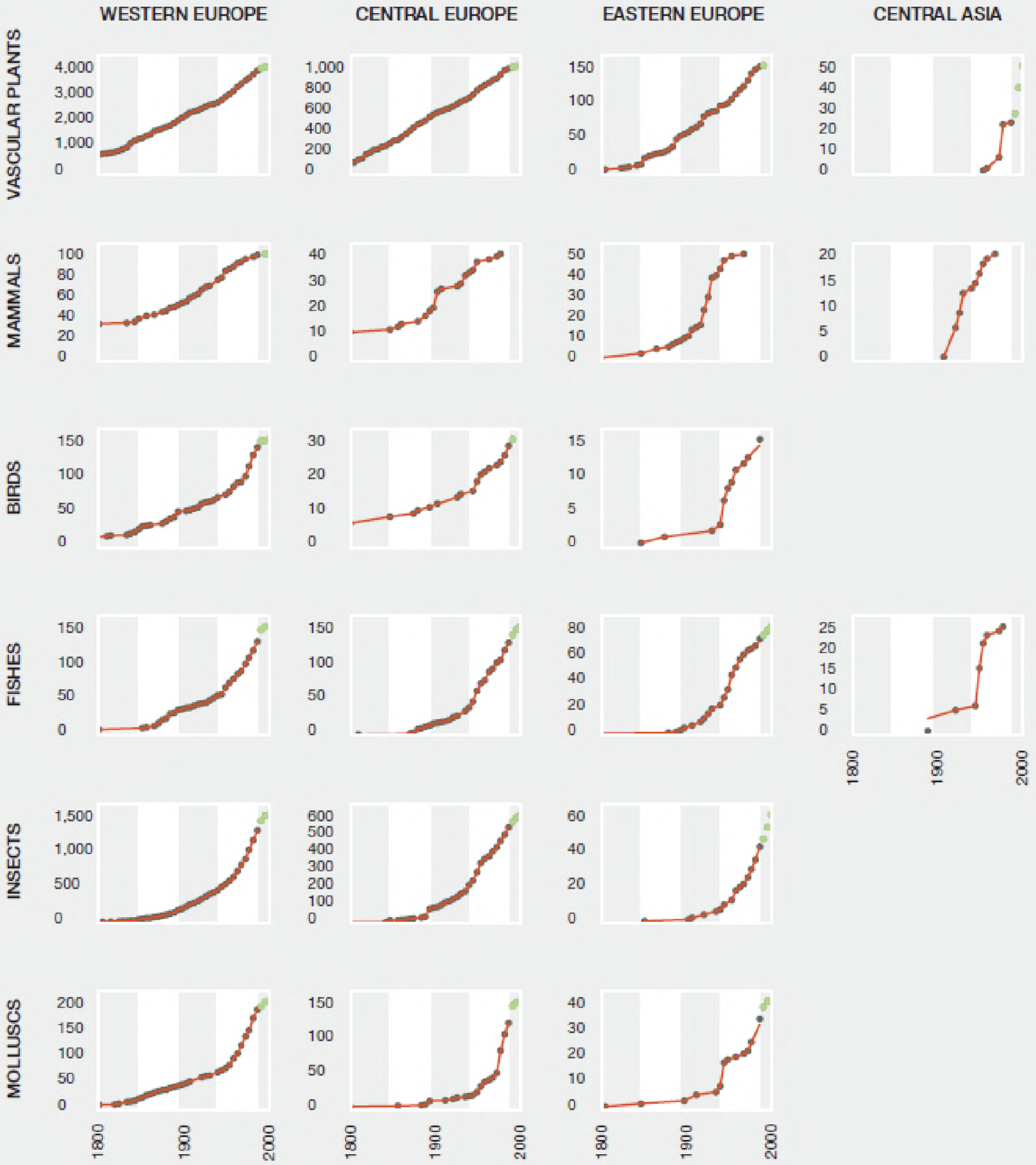
4.8.2.1 Recent trends

Rates of invasions in Europe and Central Asia have increased markedly since the beginning of the 20th century and the scale and impacts are still increasing, despite increased legal and social responses in recent years (Rabitsch *et al.*, 2016). The number of alien species has increased by 76% between 1970 and 2007 (Butchart *et al.*, 2010). This trend is similar across all environments, taxonomic groups (except mammals), and all subregions of Europe (**Figure 4.74**) (Butchart *et al.*, 2010; DAISIE, 2009; Seebens *et al.*, 2017). Even in remote Arctic and sub-Arctic regions in Europe the number of introduced alien species is substantial (Lembrechts *et al.*, 2014; Ware *et al.*, 2012). In Europe and Central Asia, the highest numbers of reported introductions for most species groups have occurred in Western Europe, but this is expected to increase in Central Europe and Eastern Europe. Data for Central Asia is less comprehensive than for the other subregions; but it is likely that the Central Asia trends are similar to other subregions based on comparable economic developments that are a major driver for invasions (Chytrý *et al.*, 2012; Seebens *et al.*, 2015; Vicente *et al.*, 2010).

The number of eradication attempts, and of successful eradications, have been increasing rapidly since the 1990s, but have been mostly confined to Western Europe (DIISE (2015); DAISIE: Database of Island Invasive Species Eradications http://diise.islandconservation.org/, 2017). Eradication of invasive alien species tends to be more successful in offshore island habitats and anthropogenic habitats than in (semi-) natural habitats (DIISE, 2015; Pluess *et al.*, 2012).

Figure 4 74 **Cumulative number of alien species between 1800 and 2000 (dots) for six taxonomic groups in subregions of Europe and Central Asia.**

The trend is indicated by a running median (red line). Data after 2000 (grey dots) are incomplete and not included in the trend analysis. Source: Adapted from Seebens *et al.* (2017).



4.8.2.2 *Projected future trends*

The overall rate of introduction of alien species shows on average no sign of slowing (Chytrý *et al.*, 2012; Seebens *et al.*, 2015) and will most likely remain high or even accelerate due to increasing trade and changing climate (Bellard *et al.*, 2012; Seebens *et al.*, 2017). This high rate is very likely to continue in the short-term, but long-term trends are less clear because they depend on the success of management and policy interventions. Management of invasive alien species is receiving increasing attention but little remains understood about which factors affect the likelihood of successful management (Pluess *et al.*, 2012).

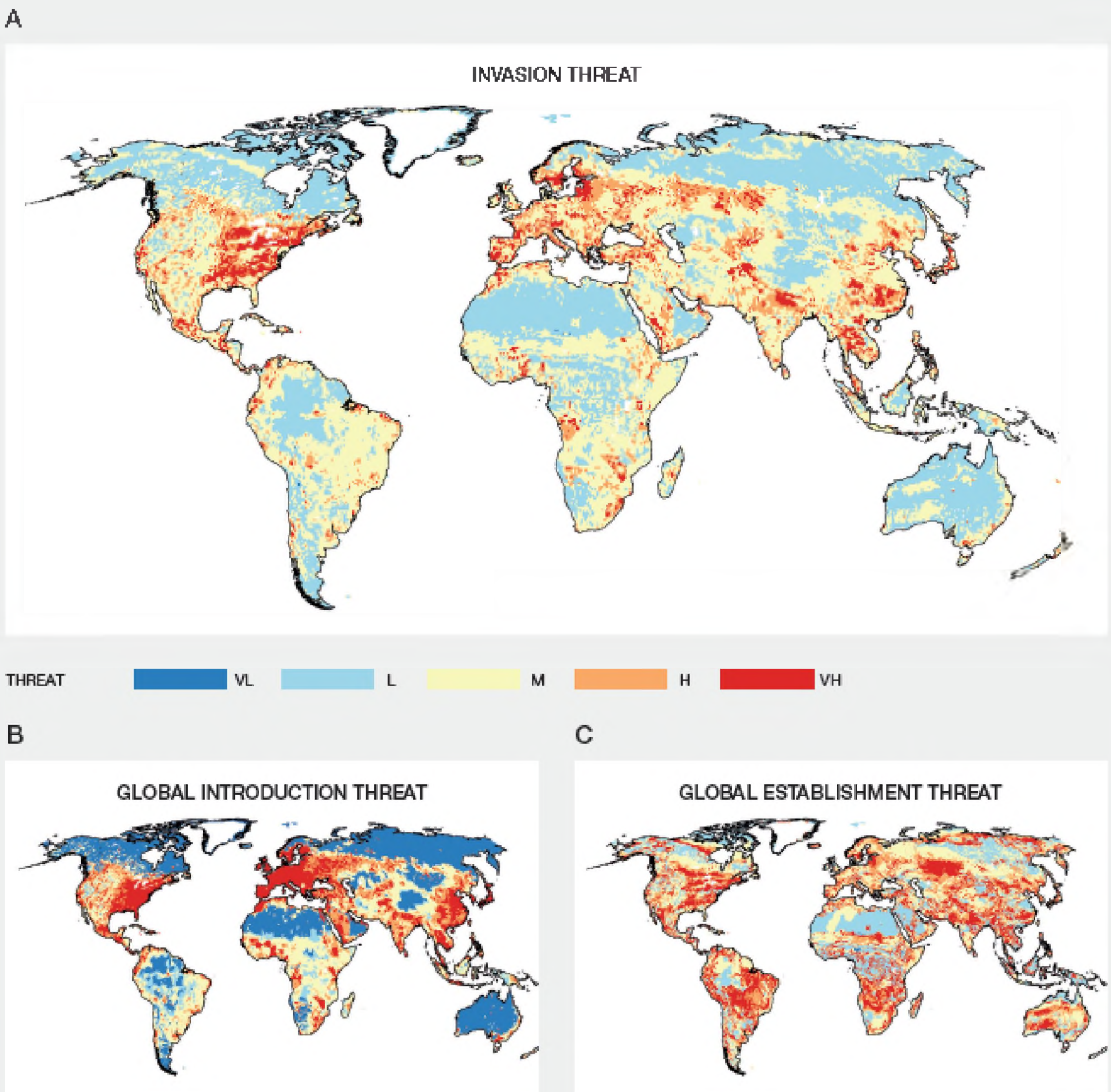
Overall, the invasion threat during the 21st century is expected to be medium to very high in most of the parts of Europe and Central Asia (Early *et al.*, 2016) (**Figure 4.75** A). The exceptions are northern areas of the region, where the threat of invasive alien species is still considered low, although rapidly increasing due to increasing tourism, more human disturbances, and climate warming (Lembrechts *et al.*, 2014, 2016; Pauchard *et al.*, 2016; Ware *et al.*, 2012) (**Figure 4.75** B). The future outcomes of invasions will depend on adoption of effective management and policy measures (Section 4.8.3). For example, plant invasion levels in Western and Central European regions are expected to remain high under "business-as-usual" scenario over the next 60 years (Chytrý *et al.*, 2012). In Eastern and Central European subregions, unprecedented increases in invasive alien species are expected during the 21st century, mostly due to increased transport and indirect effects of socio-economic drivers on other direct drivers (Early *et al.*, 2016). Increasing human population density and increasing national wealth (GDP) are associated with increased risk of alien species introduction and establishment (Chytrý *et al.*, 2008). Lower capacity to apply preventive or mitigation measures, for example in certain Eastern and Central European countries, means that the threats posed by invasive alien species will be greater (Early *et al.*, 2016).

The risk of further invasive alien species establishment is exacerbated because of projected growth in direct (e.g. land-use and climate changes, pollution) and indirect (e.g. trade) drivers facilitating invasions (Bellard *et al.*, 2013; Chytrý *et al.*, 2008, 2012; Early *et al.*, 2016; IPCC, 2014a; Seebens *et al.*, 2015; Vicente *et al.*, 2010) (see **Figure 4.75** C). Some species could increase in abundance in many areas under changing climate conditions, such as grey squirrels that are replacing native red squirrels (Bertolino *et al*, 2014). Other examples include the caterpillar *Thaumetopoea pityocampa* that is threatening Scots pine in locations that were previously too cold (Bernardinelli *et al.*, 2006); the overlap between native crayfish and invasive crayfish plague-transmitting species is also projected to increase in Europe (Capinha *et al.*, 2013). Especially in northern regions, climate warming is expected to affect the number and impact of alien species (Pauchard *et al.*, 2016).

The European Union has recently adopted European Union Regulation 1143/2014 (Section 4.8.3) on invasive alien species. The efficacy of such legislation depends on the commitment of member countries to allocate sufficient resources and ensure adequate enforcement. Furthermore, the ultimate success of regulatory approaches depends on raising public awareness of the threat of invasive alien species leading to changes in lifestyle and consumption preferences (Genovesi *et al.*, 2015). In many countries in the region, awareness, expert knowledge, legislation and allocation for managing threats from invasive alien species is increasing (Early *et al.*, 2016; Turbelin, 2017) but the efficacy of these measures is yet to be assessed. Overall, the analysed literature suggests that neither Target 5 of the European Union Biodiversity Action Plan nor Aichi Biodiversity Target 9 ("protected areas increased and improved") of the Strategic Plan for Biodiversity 2011-2020, will be met for Europe and Central Asia.

Figure 4 75 **Global invasion threat for the 21st century. Source: Modified from Early *et al.* (2016).**

Airport and seaport capacity, as well as animal, plant and total imports between 2000 and 2009, was combined into global introduction risk. Projected biome shifts and increase in agricultural intensity and fire frequency between 2000 and 2100 (emissions scenario A2) were combined into global establishment threat. Introduction and establishment axes were combined into overall invasion threat.
A invasion threat, B introduction threat, C establishment threat. All maps are displayed using the colour scheme from A, which runs from very high (VH; red) to very low (VL; blue). The scale was determined by ranking the threat value in each map grid cell, and binning cells into the following percentiles: 100–90% = very high; 90–80% = high; 80–50% = medium; 50–20% = low; and 20–0% = very low.

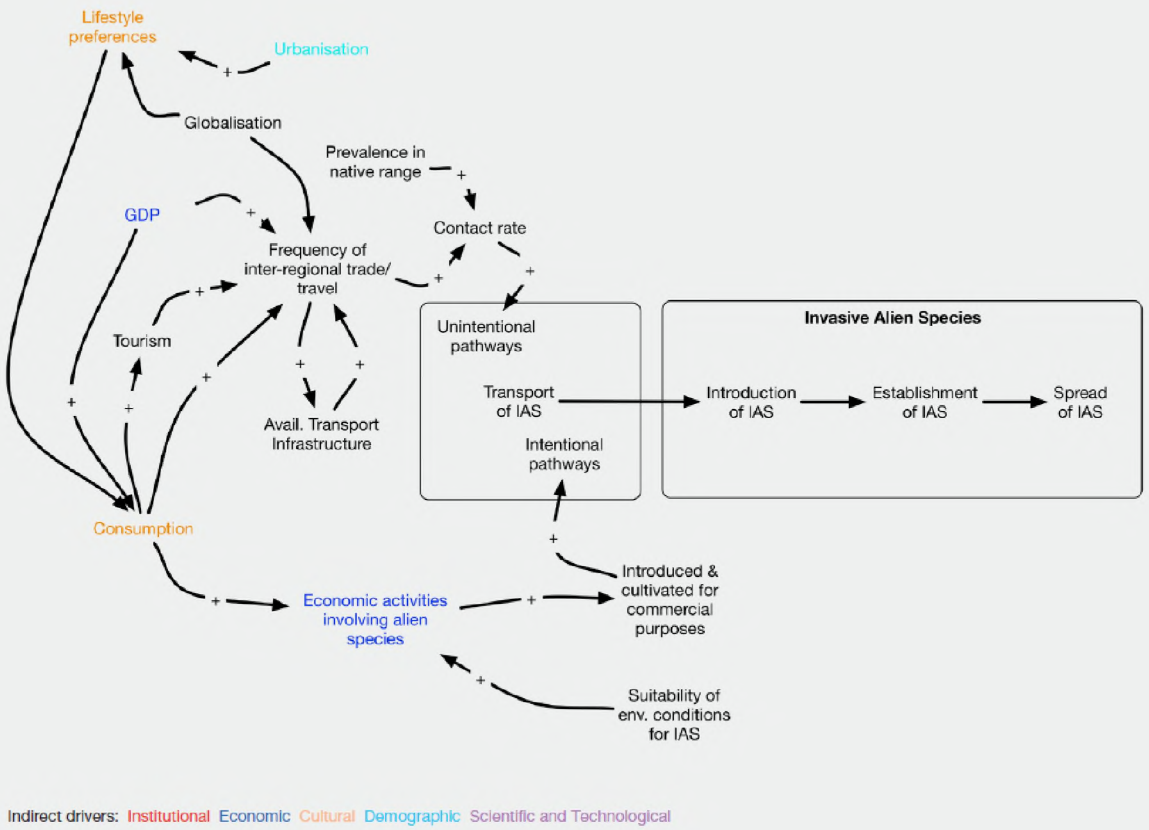


4.8.3 Indirect drivers influencing invasive alien species

The invasion process (i.e. transportation, introduction, establishment and spread) is strongly influenced by economic factors including trade and tourism (see **Figure 4.76**) (Essl *et al.*, 2011; Pyšek *et al.*, 2010; Seebens *et al.*, 2017; Turbelin *et al.*, 2017). Economic activities, either intentional or non-intentional, are the foremost influence on introduction pathways of alien species (see **Figure 4.77**). Most economic drivers have increased in Europe and Central Asia, and they can be associated with an increase in numbers of invasive alien species (Katsanevakis *et al.*, 2013; Turbelin *et al.*, 2017; Zieritz *et* *al.*, 2017). Economic development, especially in emerging economies, will drive future invasions as tourism, trade (including the pet and aquaria trade) and infrastructure projects accelerate the introduction of invasive alien species via the escape, contaminant, stowaway and corridor pathways (Hulme, 2015) (see **Figure 4.80** and **Figure 4.81**).

Figure 4 76 **Causal loop diagram of drivers of invasion by alien species. Source: Own representation.**

The invasion process includes both intentional and unintentional introduction pathways. Interlinked economic, socio-cultural, and demographic drivers account for most introduction of alien species. IAS: Invasive alien species.

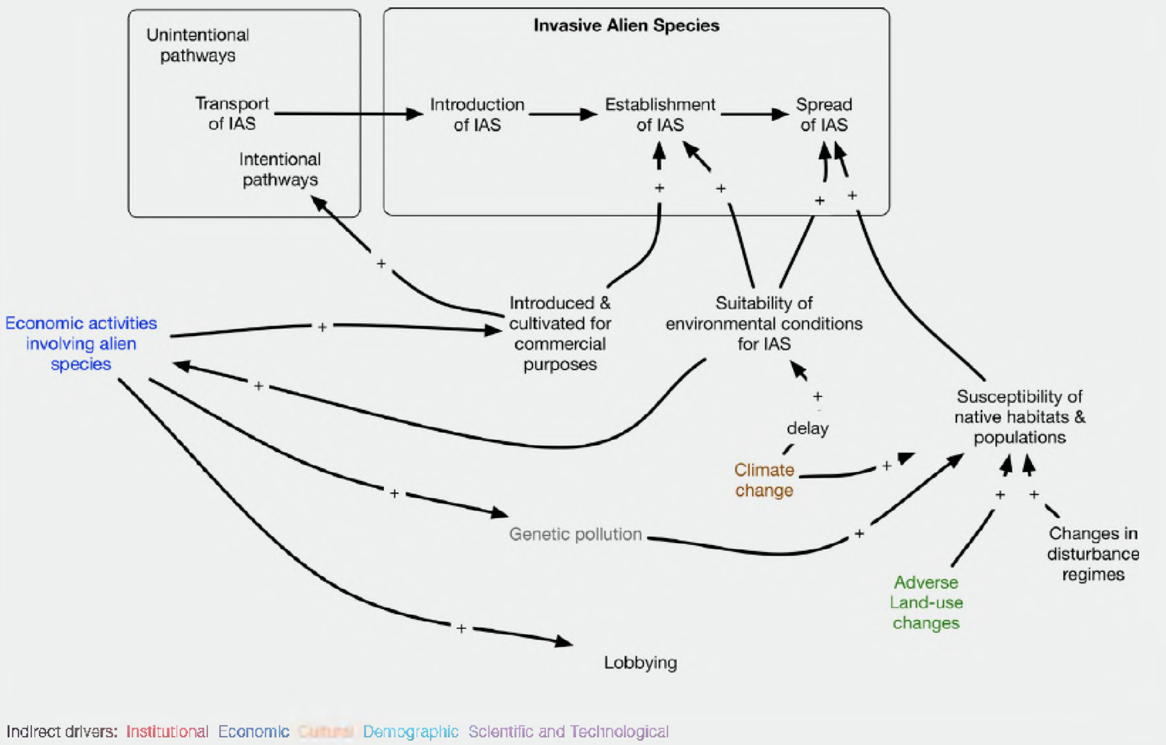


The main pathway for intentional (and to some extent un-intentional) introductions of invasive alien species in all taxonomic groups in Europe and Central Asia is trade of horticultural and ornamental plants (Turbelin *et al.*, 2017; Zieritz *et al.*, 2017). Invasion pathways related to tourism are also likely to increase in importance over the next few decades. Tourists introduce alien species, including potential invasive alien species with a high survivorship, on their clothing, footwear and equipment, for example via transportation of soil containing living organisms (bacteria, fungi, seeds, nematodes, arthropods - McNeill *et al.*, 2011). A greater frequency of contact between tourists and potential invasive alien species is likely because current high levels of international tourist movements are expected to grow up to 2030 and there is increased tourist preference for recreation (e.g. golf, fishing), agrotourism, remote places (e.g., mountains, Arctic), national parks (Hulme, 2015; Pauchard *et al.*, 2016).

The establishment and spread of invasive alien species is ultimately influenced by the suitability of environmental conditions (see **Figure 4.77**) in recipient biomes, or the ability of the invading species to adapt to these conditions or otherwise to self-create suitable conditions. Environmental suitability is a dynamic quality of biomes, subject to influence by climate change, and a key economic factor in determining species viability for import and cultivation (e.g. horticultural and silvicultural trade) and thus represents an important component of establishment processes (Chapman *et al.*, 2017; Early *et al.*, 2016). In addition to environmental suitability, the spread of established invasive alien species is also a factor of the susceptibility of native habitats and populations to further invasion. This susceptibility is influenced by a number of direct drivers of change including climate change, adverse land-use change, genetic pollution, and changes in natural disturbance regimes, which are, in turn, typically directly or indirectly driven by economic development and socio-economic trends (Early *et al.*, 2016).

Figure 4.77 Causal loop diagram of the drivers of establishment and spread of invasive alien species (IAS). Source: Own representation.

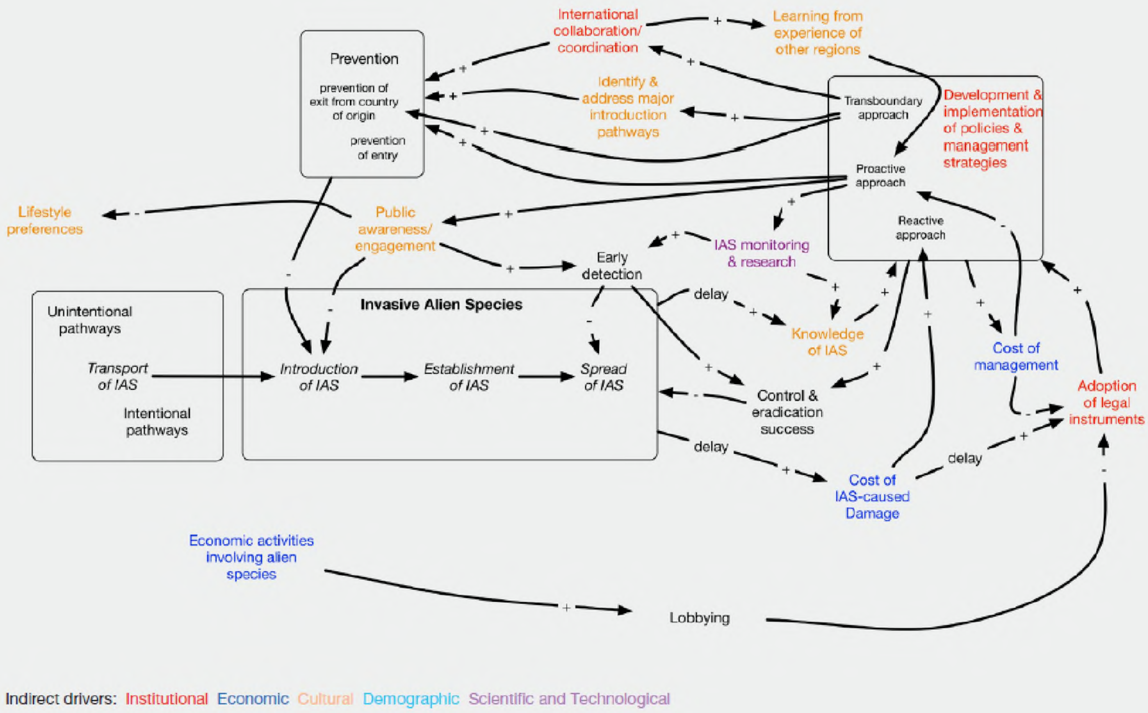
The suitability of environmental conditions has a strong influence on both the economic viability of intentionally introduced species as well as the establishment and spreading of invasive species. A number of indirect drivers including climate change and land-use change also influence the suitability of environmental conditions and the susceptibility of native habitats and populations to invasion.



Diverse impacts of invasive alien species and high eradication costs of already established invasive alien species have necessitated the adoption of legal instruments (see **Figure 4.78**). Countries with greater numbers of recorded invasive alien species have adopted more targeted international treaties (**Figure 4.79** A) and national and subnational regulations and legislation (see **Figure 4.79** B) specifically dealing with invasive alien species (Turbelin *et al.*, 2017). Western European countries have greater numbers of recorded invasive alien species due to trade and colonial histories (Turbelin *et al.*, 2017) and better scientific knowledge of species invasion status and native biodiversity (Lambdon *et al.*, 2008). Consequently, Western European countries have adopted numerous legal instruments targeting alien species; Central European and Eastern European countries have fewer legal instruments, and countries in Central Asia have the fewest legal instruments (**Figure 4.79**). Within the European Union, the regulation on invasive alien species implemented in 2014 includes three types of interventions: prevention, early detection and rapid eradication, and management (European Union, 2014). Globally, the number of international agreements relevant to control of invasive alien species as well as the number of countries that are party to these agreements has consistently increased since the 1950s (McGeoch *et al.*, 2010).

Figure 4 78 **Causal loop diagram illustrating costs from and resulting policies due to invasive alien species (IAS). Source: Own representation.**

Impacts and costs resulting from increased abundance of invasive alien species has driven the adoption of legal instruments in Europe and Central Asia. Most policies are reactive in nature, focusing on control and eradication measures once species have become established.



Information on legal instruments concerning invasive alien species is largely missing from Central Asia, either because of a lack of data or a genuine lack of policy. In the latter case, the development of legislation and regulations in this subregion could (1) prevent the introduction of invasive alien species or (2) help reduce the spread and impact of existing ones. Species introductions as well as spread and impact of existing invasive alien species are likely to be exacerbated (Turbelin *et al.*, 2017) based on trends of other indirect drivers, especially socio-economic drivers such as development of the oil industry and its related infrastructure in Kazakhstan, Turkmenistan and Uzbekistan (Dimeyeva, 2013). From available information, mainly based on eports by the Convention on Biological Diversity, countries in Central Asia currently have little capacity to respond to threats by invasive alien species and impending or future introductions, establishment or spread (Early *et al.*, 2016).

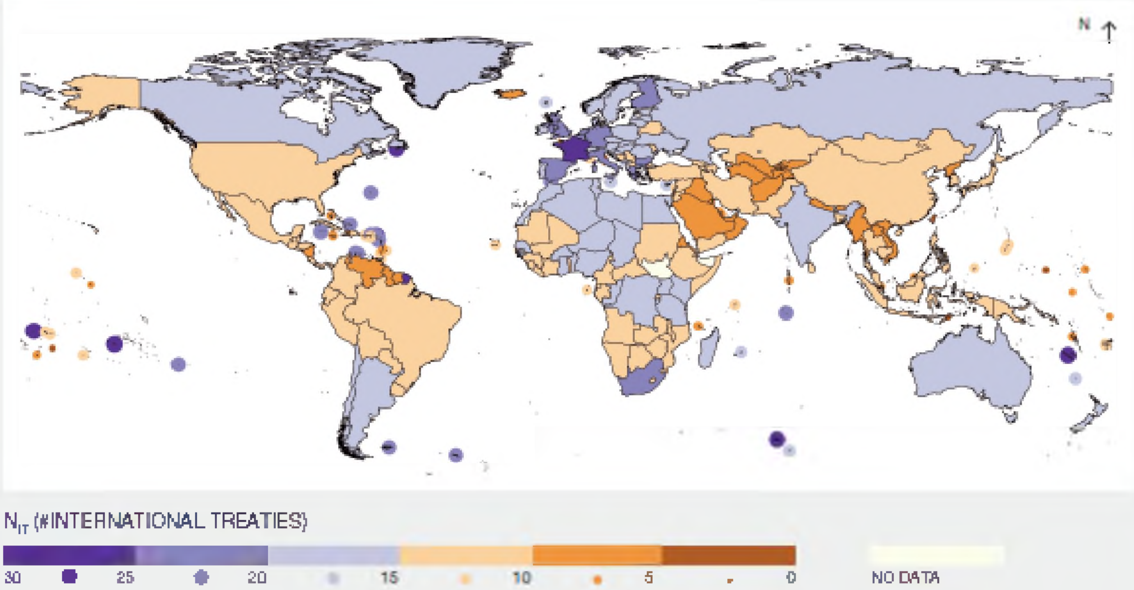
The majority of legal instruments are reactive, targeting introduction and spread of invasive alien species upon arrival within national borders. Very little attention has been given to preventing the arrival of invasive alien species, except for species that have known public health impacts (Turbelin *et al.*, 2017). Comprehensive border controls to prevent introduction of potential invasive alien species are adopted by very few countries in Europe and Central Asia (Early *et al.*, 2016). Current regulations lack a transboundary perspective and insufficiently cover major introduction pathways (Hulme, 2015). For example, most efforts in regulation of transport-related non-intentional introductions of invasive alien species have addressed the role of shipping, while tourism, another major route of stowaway alien species, remains largely neglected (Hulme, 2015).

A general recommendation from studies on invasive alien species regulation and management is to develop educational outreach programmes to raise awareness of the general public and industry (Hulme, 2015; Katsanevakis *et al.*, 2013; Turbelin *et al.*, 2017; Zieritz *et al.*, 2017). Increased public awareness could lead to changes in preferences for alien species as pets or other ornamental purposes, increased vigilance by tourists and the tourist industry, and improved early detection of alien species.

Figure 4 7B Global map of legal instruments (1933–2015) relevant to invasive alien species (IAS).

Shown are A number of international treaties mentioning invasive alien species to which each country is signatory, including global and regional treaties for 1933–2015 and B map of the maximum relevance score for each country that has national or sub-national regulations of legislation in place, relevant to invasive alien species (1980–2015). Overseas territories have been allocated the same number of international treaties as their sovereign state. Source: From Turbelin *et al.* (2017).

A



B

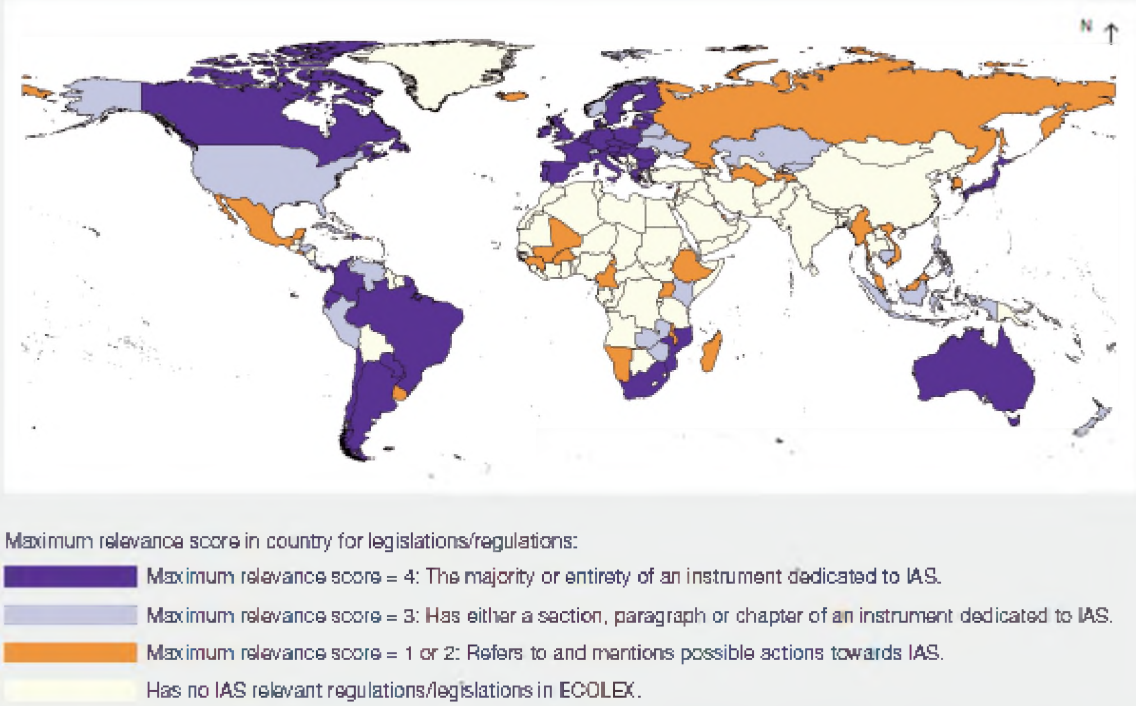


Figure 4 80 Frequencies of introduction pathways of known cases of introduction of over 14,000 invasive alien species profiled in the Delivering Alien Invasive Species Inventories for Europe (DAISIE); level 2 categories. Source: CBD (2014).

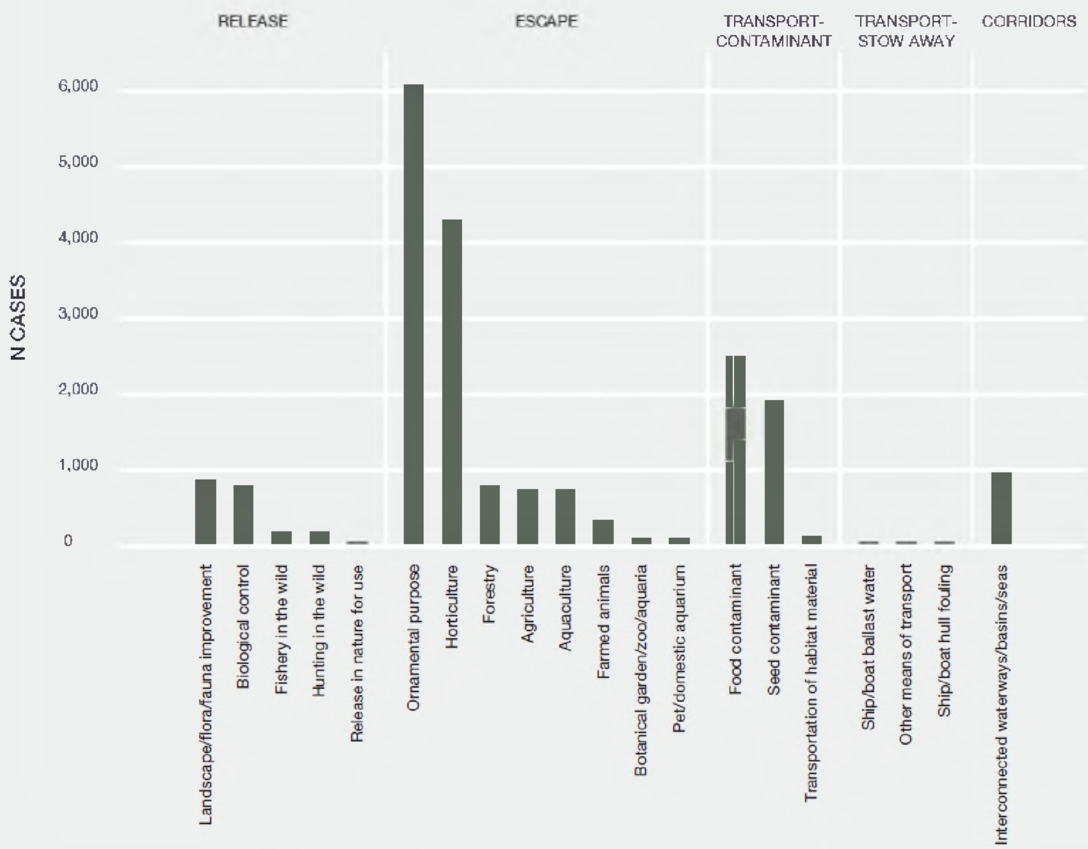
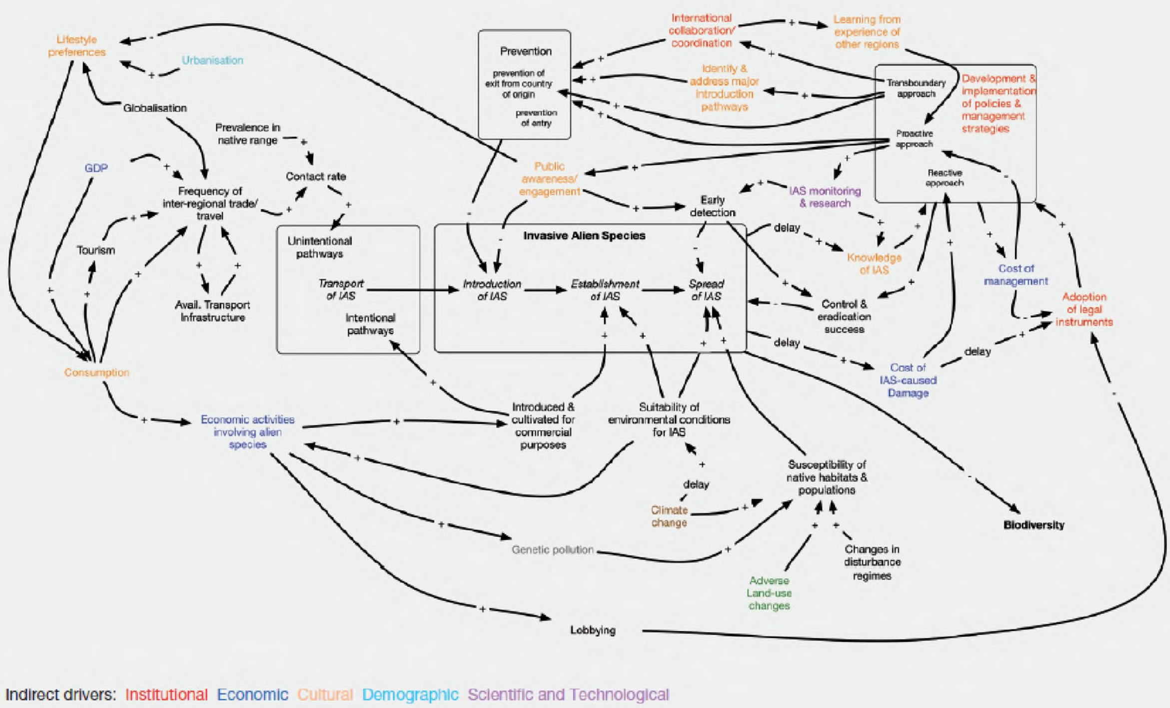


Figure 4 81 **Causal loop diagram of drivers of feedbacks mechanisms between effects of invasive alien species (IAS) and direct and indirect drivers. Source: Own representation.**

Feedbacks are largely limited to rarely implemented policies aimed at raising public awareness. Most current policy responses are reactive, directed towards control of existing invasive alien species populations, and do not address underlying drivers.



4.9 Synthesis of direct driver trends and impacts in Europe and Central Asia

4.9.1 Interaction among direct drivers and time-lagged effects on biodiversity and nature's contributions to people

Drivers, both direct and indirect, rarely act in isolation. In essence, a change in biodiversity and nature's contributions to people is almost always an outcome of several interacting drivers. While it may be possible to determine which drivers are involved, it is not always easy to assess or even quantify the respective contribution of the individual drivers in affecting biodiversity, including ecosystems. In addition, positive feedbacks can influence driver dynamics and amplify their combined effects. For example, land-use change and destruction of habitats can influence climate change (locally) due to the changes in land surface albedo and evapotranspiration (Kalnay & Cai, 2003).

Drivers do not act in isolation with interactions between them affecting driver trends and thus also the effects on biodiversity and nature's contributions to people. In **Box 4.7**, we exemplify the interaction of indirect and direct drivers using three examples of invasive alien species. They illustrate how different drivers – partly indirect and partly direct – jointly affect the driver "invasive alien species". Many other examples of driver interactions exist in the ecological literature. For example, the interplay of climate change, pollution and invasive alien species exacerbates the negative impact of land-use change and management intensity (Collier *et al.*, 2016; Haddad *et al.*, 2015; IPBES, 2016a; Kalnay & Cai, 2003; Mantyka-Pringle *et al.*, 2012; Segan *et al.*, 2016; Vilà & Ibáñez, 2011). Small and isolated populations of organisms are less well buffered against climate change (McInerny *et al.*, 2007), are more susceptible to invasion (Didham *et al.*, 2007; Haddad *et al.*, 2015) and can be more exposed to pollution (Weathers *et al.*, 2001). Declining area of habitats and their increasing isolation also reduces the possibilities for the compensatory migration of species in response to changing climate (Bocedi *et al.*, 2014; Meier *et al.*, 2012; Vanbergen & The Insect Pollinators Initiative, 2013). Furthermore, a modelling study has shown that impact assessments focused on one sector (agriculture, foresty, water use, etc.) alone without considering interactions between these sectors will likely lead to over- or under-estimation of the projected impacts, as direct and indirect drivers affect each other mutually (Harrison *et al.*, 2016).

Individual and combined effects of different direct drivers can have chronic, prolonged and delayed consequences on biodiversity and nature's contributions to people, due to considerable time lags that many species and ecological systems have in response to changes in their environment (Dullinger *et al.*, 2012; Ewers & Didham, 2006; Halley *et al.*, 2016; Hanski & Ovaskainen, 2002; Helm *et al.*, 2006; Kuussaari *et al.*, 2009; Tilman *et al.*, 1994; Urban, 2015). Even if habitat conditions no longer meet the minimum requirements for species persistence (e.g. too small habitat area, too isolated habitats, climatic conditions becoming unsuitable), actual extinctions can take time, creating an extinction debt in many contemporary habitats or ecosystems (Hanski & Ovaskainen, 2002; Kuussaari *et al.*, 2009). Time-lags also characterize species colonizations of new habitats, termed "colonization credit" or "immigration deficit". Delayed immigration characterizes both non-native species invasions as well as natural, climate-driven or land use-driven migrations and colonizations of native species (Jackson & Sax, 2010). By masking the full extent of impacts of direct and indirect drivers on biodiversity and nature's contributions to people, time-delays in species dynamics pose considerable challenges for research and conservation. Extinction debt can last decades or even centuries and, if left unnoticed, can lead to serious overestimation of current biodiversity status and underestimation of the impact of combined and direct effects of direct drivers on biodiversity and nature's contributions to people (Kuussaari *et al.*, 2009). For example, taking extinction debt into account increased projected extinctions threefold from 5% to 15% under currently projected climate change scenarios (Urban, 2015). On the other hand, when recognized in good time, extinction debt and colonization credit can provide opportunity to avoid some of the projected extinctions or undesired colonizations via active and knowledgeable conservation and restoration activities (Halley *et al.*, 2016; Török & Helm, 2017).

Box 4.7: Interaction of direct and indirect drivers in their effects on biodiversity and nature's contributions to people.

Economic and demographic drivers are both highly correlated with invasion of alien species

The number of invasive alien species is strictly correlated with economy and with human population. In particular, the level of wealth, defined as cumulative economic prosperity, has been shown to have a strong influence on the cumulative level of invasions (Pyšek *et al.*, 2010); this correlation has a temporal effect, and the number of invasive alien species reflects historic rather than contemporary economy (Essl *et al.*, 2011).

Climate change, habitat fragmentation and fish invasion

Connectivity is extremely important for freshwater fish migration, and natural and man-made barriers can consequently seriously facilitate or hamper fish dispersal. This has, for instance, been illustrated for pike (*Esox lucius*) in Sweden, where they are currently absent from isolated lakes and lakes upstream from channel slopes steeper than c. 7% (Hein *et al.*, 2011; Spens *et al.*, 2007). At the same time, pike are top predators, able to extirpate cold-adapted salmonid species under warmer conditions in small lakes, whereas those species co-exist under colder conditions and in larger lakes (Hein *et al.*, 2013). Due to human-mediated introductions and climate warming pike are now spreading upstream and to more northern latitudes, while at the same time climate warming improves pike performance, often resulting in local extinctions of cold-adapted specialist fish species (Hein *et al.*, 2013). The strong effect of climate change on these predator fish (both influencing their spread and their competitiveness) provides managers with a difficult challenge regarding the restoration of natural connectivity to improve the free movement of species. Whereas on the one hand connectivity is important to native species that need to track climate change, barriers such as waterfalls, dams and weirs can limit the upstream spread of problematic or very competitive species, thereby creating refuges and protection for threatened species. This example illustrates how climate change may alter the effect of connectivity restoration on fish biodiversity. It also illustrates trade-offs in biodiversity conservation.

Economic and demographic drivers, climate and land-use change, and invasive alien species

Distribution patterns of invasive alien species have been shown to be strongly linked to climate, land use, human demography and socio-economic activities (Bellard *et al.*, 2013, 2016; Gallardo & Aldridge, 2013; Gallardo *et al.*, 2015; Pyšek *et al.*, 2010). Specifically, in Europe and Central Asia, invasive alien species patterns are mostly driven by socio-economic activities (see Section 4.8.3). Climate acts as a broad-scale limiting factor to invasive alien species distributions, whereas land use (also driven by socio-economic activities) affects invasive alien species patterns at the global, regional and local scales (Bellard *et al.*, 2013). Consequently, changes in these drivers alter patterns in invasive alien species distribution and impact (Diez *et al.*, 2012; Dukes & Mooney, 1999; Hellmann *et al.*, 2008; Meyerson & Mooney, 2007; Walther *et al.*, 2009). Climate change (temperature and precipitation changes, CO_2 concentrations, extreme events) has been hypothesized to enhance biological invasions (Bellard *et al.*, 2013; Diez *et al.*, 2012; Dukes & Mooney, 1999; Hellmann *et al.*, 2008). Land-use change is expected to alter invasive alien species patterns depending on habitat types and uses, with the most intensely used and disturbed habitats being the most prone to invasions (Chytrý *et al.*, 2012). Increasing socio-economic activities are expected to increase invasions by increasing propagule pressure, introduction pathways and habitat disturbances (Bellard *et al.*, 2016; Essl *et al.*, 2011; Gallardo & Aldridge, 2013; Gallardo *et al.*, 2015; Pyšek *et al.*, 2010). Projected future patterns in plant invasions in relation to land-use change show strongest increases for the northern parts of Western Europe (all scenarios), and strongest decreases in the southern parts of Western Europe in scenarios of abandonment of agricultural land (Chytrý *et al.*, 2012). Future projections in distribution patterns of 100 of the world's worst invasive alien species in relation to both climate and land-use change project important increases in northern parts of Western Europe, slight increases in Central Asia and western parts of Eastern Europe, but decreases around the Mediterranean basin (Bellard *et al.*, 2013).

End of Box 4.7

4.9.2 Synthesis of direct driver trends and impacts

In this section, we summarize assessed trends and impacts of direct drivers. **Figure 4.82** and **Figure 4.83** illustrate the direction of driver trends and their inferred impact in the different subregions and units of analysis. An increasing trend (upward-pointing arrow) means that the direct driver shows an increasing trend, while downward pointing arrows indicate a decreasing trend. Impacts on biodiversity associated with the direct driver trends are indicated by colours, with red and green indicating negative and positive impacts. An increasing driver trend can have a positive impact on overall biodiversity in an area or unit of analysis. However, we assessed to what degree the trend affects the biodiversity typical for the unit of analysis in that area not simply biodiversity as a whole. For example, climate change can be expected to increase the overall biodiversity in an area, yet it will negatively affect the biodiversity of a given spatial unit, e.g. of temperate forests, if these are converted into other units of analysis, e.g. Mediterranean forests, over time. Hence, overall biodiversity might be higher in Mediterranean forests, but species associated with temperate forests would be lost.

In some cases, trends do not have a clear direction in a subregion and unit of analysis, but rather show both decreasing and increasing trends or impacts. We therefore also allowed trends and impacts to be of "variable" nature. The confidence in the statements on driver trends and impacts is indicated by the thickness of symbols or by the saturation of the colours. The notion of "irrelevant" was assigned to combinations of drivers, subregions and units of analysis that do not exist. For example, there are no "deep marine waters" or "Mediterranean forests" in Central Asia.

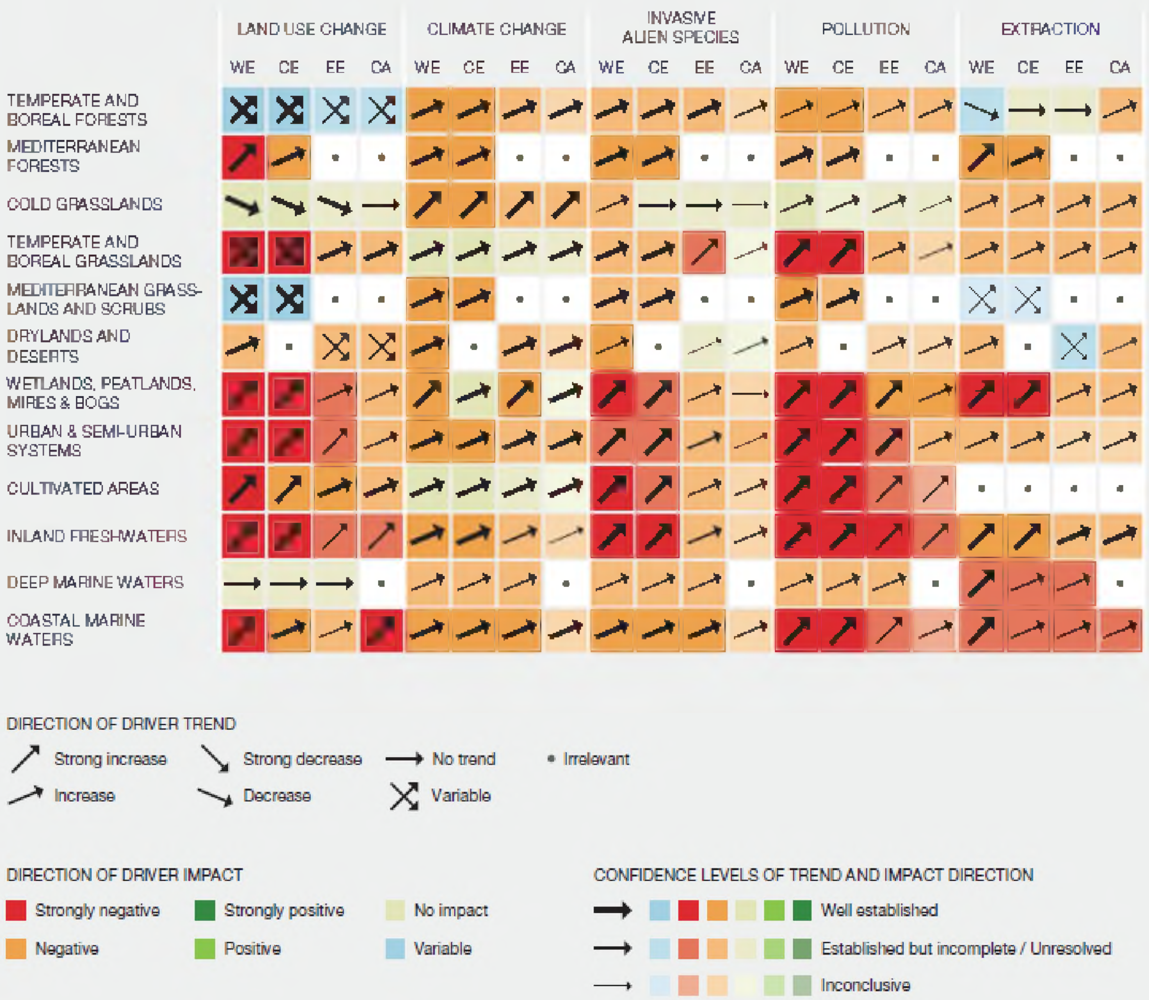
4.9.2.1 Recent trends in direct drivers and their impact

The recent trends in direct drivers and their associated impact on biodiversity and nature's benefits to people are summarized across Europe and Central Asia and major units of analysis within the study region (see **Figure 4.82**). Land-use change, pollution and partly natural resources extraction showed the strongest increase, and had the clearest negative impacts in many units of analysis or subregions. Invasive alien species have steadily increased as a driver of biodiversity alteration and loss, and already showed strong negative impacts in some (wetland types, cultivated areas and inland freshwaters), while in other systems the impact is less severe or still mostly lacking (e.g. cold grasslands). Climate change has also steadily increased as a driver, yet its impact on biodiversity is still marginal and confined to relatively few units of analysis (mostly urban and water systems).

Confidence is generally high in all statements, yet often somewhat decreasing towards Eastern Europe and Central Asia, usually due to a lack of accessible literature and insufficient number of studies analysing trends and impacts. The availability of repeated reports on climate change trends and impacts (e.g. IPCC, 2013b, 2014a, 2014b) results in an overall very high level of confidence in all statements on trends and impacts throughout the study region. Lowest confidence levels were found for trends and impacts for the driver natural resources extraction. This driver is usually more limited in its spatial extent, and thus more difficult to quantify across larger regions, both for trends and impacts.

Figure 4 82 **Combination of recent trends in direct drivers and impacts of driver trends on biodiversity and nature's contributions to people.**

"Increase" stands for accelerating trends in direct drivers, while "decrease" stands for decelerating driver trends. Positive (green) and negative (red) impacts on biodiversity indicate the effects of the associated driver trends. Confidence levels are indicated by the thickness of arrows and saturation of colours. Units of analysis are: temperate and boreal forests; Mediterranean forests; cold grasslands; temperate and boreal grasslands; Mediterranean grasslands and scrubs; drylands and deserts; wetlands, peatlands, mires and bogs; urban and semi-urban systems; cultivated areas; inland freshwaters; deep marine waters; coastal marine waters. WE: Western Europe, CE: Central Europe, EE: Eastern Europe, CA: Central Asia. Source: Own representation.



4.9.2.2 *Projected future trends in direct drivers and their impact*

The projected future trends in direct drivers and their associated impact on biodiversity and nature's contributions to people are summarized across Europe and Central Asia and major units of analysis within the study region (**Figure 4.83**). Land-use change, pollution and partly natural resource extraction showed strongest projected decreases, and had overall some reduction in its impacts on biodiversity in several units of analysis or subregions. On the contrary, climate change and invasive alien species will become the most important threats to biodiversity compared to the other three drivers. The reduction of driver effects on biodiversity is stronger in Western and Central Europe than in Eastern Europe and Central Asia for land-use change, pollution and natural resources extraction. On the other hand, invasive alien species are still projected to be a bigger threat to units of analysis in Western and

Central Europe than in Eastern Europe and Central Asia, reflecting the importance of traffic and economic growth for the trends in invasive alien species.

Figure 4 33 Combination of projected future trends in direct drivers and impacts of projected driver trends on biodiversity and nature's contributions to people.

"Increase" stands for accelerating trends in direct drivers, while "decrease" stands for decelerating driver trends. Positive (green) and negative (red) impacts on biodiversity indicate the effects of the associated driver trends. Confidence levels are indicated by the thickness of arrows and saturation of colours. Units of analysis are: temperate and boreal forests; Mediterranean forests; cold grasslands; temperate and boreal grasslands; Mediterranean grasslands and scrubs; drylands and deserts; wetlands, peatlands, mires and bogs; urban and semi-urban systems; cultivated areas; inland freshwaters; deep marine waters; coastal marine waters. WE: Western Europe, CE: Central Europe, EE: Eastern Europe, CA: Central Asia. Source: Own representation.

Confidence is generally lower for projected future (compared to recent) trends and impacts in all statements, and often even lower towards Eastern Europe and Central Asia. The exception to this rule is the confidence in trends of climate change, for which a wealth of information is available (e.g. IPCC, 2013b, 2014a, 2014b). For some systems only marginal information is available in the Europe and Central Asia region. Again, lowest confidence levels are available for trends and impacts in the driver natural resources extraction.

4.9.3 Synthesis of indirect drivers

Sections 4.4-4.8 presented Causal Loop Diagrams (CLDs) to illustrate the dynamic inter-relationships within and between indirect and direct drivers of change in biodiversity and nature's contributions to people. Our findings show that the general combination of indirect drivers that underpin trends in direct drivers is often similar across Europe and Central Asia. At the same time, causal relationships between individual indirect drivers and their effects on direct drivers are context specific.

Indirect drivers are often triggered by processes in different sectors of society and by the activities of diverse groups of actors and stakeholders. Their cumulative effects provoke dynamics of indirect drivers and as a consequence a specific impact on a direct driver. For example, intensification of conventional agriculture in Europe and Central Asia is influenced by cultural, institutional, economic, technological and demographic drivers. Whilst the consolidation of farms has led to some improvements in economic viability, it is also linked to the erosion of local traditions and of a sense of long-term stewardship and responsibility for the land, which are important for sustaining management practices for biodiversity.

The CLDs were also used to illustrate complex adaptive systems in terms of how the effects on biodiversity and nature's contributions to people feed back to the indirect drivers. Knowledge and awareness, filtered by beliefs and values, are often the primary cultural-religious drivers in this feedback. Environmental NGOs have had an important role in increasing awareness of forest biodiversity, while scientific organizations have been important for issues such as fishing and climate change. Public awareness has not always been necessary, e.g. in the case of invasive alien species, new regulations have been passed in response to scientific knowledge with less political pressure. There are also differences between subregions. For example, for agriculture, the Common Agricultural Policy in the European Union includes institutional and agri-environmental support (payments), which has a positive (green) effect on biodiversity (**Figure 4.84**). However, these payments only reduce the effects of global trade and competition, so the economic drivers work in both ways (grey colour in **Figure 4.84**). Cultural change increases demand for organic food in Western Europe. Technological drivers in the resource extraction sectors generally increase degradation, while "end-of-pipe" technologies have successfully reduced pollution (**Figure 4.84**).

Figure 4 84 Impact of indirect drivers (rows) on direct drivers (columns) of biodiversity loss and nature's contributions to people in Europe and Central Asia. Source: Own representation.

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

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

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

✗ Negative ~ Both ways ✓ Positive Lack of evidence

Institutional drivers have often been used to soften the effects that economic profit-seeking drivers have on technological change and a range of direct drivers. Regulations have reduced some pollution, e.g. acidification and toxicity from heavy metals. Other direct drivers, e.g. pesticides and ammonia pollution from agriculture, have been regulated although not sufficiently to reverse negative trends.

Economic drivers have not changed very much as a result of knowledge and awareness of ecosystem degradation and have generally a negative effect on biodiversity and nature's contributions to people (**Figure 4.84**). Environmental and ecological fiscal reforms have not generally been implemented: environmental taxes have not increased since 2002. On top of this, harmful subsidies to fishing and mining provide market actors with strong incentives to continue externalising environmental costs. Hence, economic drivers still support intensive agriculture and forestry as well as unsustainable natural resource extraction, especially fishing and mining. When economic drivers have been employed to halt biodiversity loss, e.g. through agri-environmental schemes and carbon taxes or trading schemes, this has generally been insufficient to halt habitat fragmentation and degradation or climate change. As long as a good quality of life is associated with GDP growth, the perceived trade-off between a good quality of life and sustainable ecosystem management and governance will continue to be a major obstacle, if sustainable development is to be achieved.

4.10 References

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