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Abstract

We review observed and projected future changes in processes that affect biodiversity in Europe, concentrating on terrestrial ecosystems. Large numbers of species from a variety of organism groups have shown an earlier onset of their annual life cycle (*phenology*) in the order of several days per decade. The recorded changes have been more pronounced in northern Europe, partly because of the larger temperature increases compared with southern Europe, but also because fewer studies are available along the southern trailing edges of species ranges. Most species studied have recently expanded their *ranges* towards the north by several km, but species with limited dispersal capability have responded much less and amphibians and reptiles, in particular, show range contractions. Some species have also expanded uphill whilst others have contracted at low altitudes. *Extinctions* primarily caused by climate change have been reported for amphibians and butterflies, but attributing extinctions to a single factor is difficult, partly because of poorly known time lags between climate change and extinctions. *Community and ecosystem* structure and the spatial distribution of different vegetation types have also changed in response to recent climate change. In particular, tree lines have moved up in altitude, and northern, temperature-limited ecosystems have become more productive, primarily because of longer growing seasons. *Indirect changes*, in particular in land use resulting in the loss and increased fragmentation of habitats, and *sea level rise* further strengthen the pressure from climate change on biodiversity.

The observed trends are generally projected to continue in the future, but responses will likely be more pronounced because climate is projected to change faster than in the recent past. Climate zones in Europe are expected to move towards the north-east with rates likely to exceed the species' capacity to track suitable climates. As current techniques for projecting changes in biodiversity do not account for all factors influencing species, it is difficult to predict accurate extinction rates for species. However, available estimates, based on different modelling approaches, indicate that globally 20-30% of assessed plant and animal species are likely to face substantially increased risk of extinction within the 21st century. Species with limited dispersal, small ranges, or habitat specialists will be substantially more affected than good dispersers, wide ranging, or generalist species. While recent climate changes have been primarily characterized by changes in temperature and precipitation patterns, changes in water availability might play an important role in southern and central Europe. Pronounced drought could have strong effects in particular in the south-west of the Iberian Peninsula, potentially leading to ecosystem-level changes such as forest die-back. Even though projections for individual species are highly uncertain, a number of robust patterns emerge from the available studies and should be considered in conservation planning.

1. Introduction

Temperatures in Europe have on average increased by 0.9°C from 1901 to 2005, together with changes in precipitation regimes, wind patterns and an increase in extreme events (Alcamo et al. 2007). The rate of increase in the combined radiative effect of the atmospheric greenhouse gases CO_2, CH_4 and N_2O over this period is very likely to be unique with respect to at least the last 16,000 years (Jansen et al. 2007), and adapting to this high rate of change is a challenge for human society as well as for natural ecosystems (Reid 2006). Indeed, there is solid evidence that anthropogenic climate and land-use changes have already altered phenology and distributions of several species, with 21st century impacts projected to be even greater (e.g., Kerr 2007).

Projections for the 21st century indicate that the European mean annual temperature is likely to increase by 2.2°C to 5.3°C under the IPCC[1] A1B scenario (Nakićenović and Swart 2000), which is more than the projected global average increase (Christensen et al. 2007). The warming is likely to be largest in winter for northern Europe, but in summer for the Mediterranean region (Christensen et al. 2007). Annual precipitation is very likely to increase in northern Europe, while decreasing in most of the Mediterranean region (Christensen et al. 2007). In northern Europe, precipitation increases will likely be most pronounced in winter, while in the Mediterranean rainfall will likely decrease mostly in the summer, when water is already highly limiting in today's climate (Schröter et al. 2005, Christensen and Christensen 2007). Furthermore, the frequency of extreme climatic events is expected to increase in Europe, with heavier precipitation events and longer dry spells (Christensen and Christensen 2003).

Biodiversity is frequently defined as biological variability at genetic, species, community and ecosystem levels (Sala et al. 2000). The diversity of species supplies our society with a variety of ecosystem services that are crucial for our well-being and survival, e.g., food and timber production, freshwater and medicine (Millennium Ecosystem Assessment 2005, Schröter et al. 2005). The combined economic value of these services is considerably larger than the world economy (Costanza et al. 1997). In addition, diverse ecosystems are thought to be more resistant and resilient to environmental change, for example in their continuing ability to provide ecosystem services (Secretariat of the Convention on Biological Diversity 2006, Tilman et al. 2006), and often have a high aesthetic and recreational value (Costanza et al. 1997).

Several recent reviews support the view that climate change is starting to affect biodiversity and will become a major driver of biodiversity losses (IUCN 2005, Millennium Ecosystem Assessment 2005, Fischlin et al. 2007, Rosenzweig et al. 2007). However, there is increasing concern that the interaction between land use and climate change will magnify these negative effects on biodiversity (Sala et al. 2000), with larger climate impacts at high latitudes and larger habitat loss from land use in the tropics and subtropics (Jetz et al. 2007, Lee and Jetz 2008). Human-driven land-use changes result in the loss and increased fragmentation of habitats (e.g., Metzger et al. 2006, Rounsevell et al. 2006), whereas climate change causes suitable habitats to shift from their current locations (Theurillat and Guisan 2001). The complex interactions between land use and climate change make it difficult to attribute biodiversity changes to either one of these drivers (Sala et al. 2000, Araújo et al. 2008), and this has a consequence that current

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[1] IPCC Intergovernmental Panel on Climate Change: http://www.ipcc.ch

estimates of biodiversity loss might miss synergistic effects arising from such complex interactions.

When climate conditions change beyond species' breadth of tolerance, species may be forced to respond by shifting the timing of life-cycle events (*phenology*), shifting their geographical boundaries (*range shift*), changing morphology, behaviour or their genetic make up; when neither adaptation nor shifting range is possible, *extinction* is a likely scenario (Rosenzweig et al. 2007). The cause behind the response can be either *plastic* (changes within individuals during their lifetimes) or *genetic* (changes in genotypes between generations and among populations) (Theurillat and Guisan 2001, Parmesan and Matthews 2006). Several recent reviews have presented and discussed the evidence of responses of species and ecosystems to recent climate changes, the so-called ecological 'fingerprints' of climate change (e.g., Walther et al. 2002, Parmesan and Yohe 2003, Root et al. 2003, Walther et al. 2005a, Menzel et al. 2006, Parmesan 2007) as well as recent reports from the EEA (2007) and the IPCC (Alcamo et al. 2007, Rosenzweig et al. 2007).

Projections of changes in biodiversity in response to future climate changes are frequently based on modelling techniques such as bioclimate envelope (or climatic niche) modelling (e.g., Sykes et al. 1996, Pearson and Dawson 2003, Guisan and Thuiller 2005, Araújo and Guisan 2006, Heikkinen et al. 2006) and dynamic vegetation modelling (e.g., Prentice et al. 1993, Smith et al. 2001, Sitch et al. 2003, Hickler et al. 2004, Malcolm et al. 2006, Thuiller et al. 2006a, Prentice et al. 2007). For a review of this work, see Thuiller et al. (2008). Results from these modelling tools can be used to identify regions and taxa (e.g., Berry et al. 2002, Thuiller et al. 2005, Araújo et al. 2006, Huntley et al. 2008) or ecosystems (Schröter et al. 2005) that are most threatened by climate change. Results can then be applied for conservation planning procedures (e.g., Araújo et al. 2004, Williams et al. 2005, Hannah et al. 2007), though model projections are still subject to considerable uncertainty (Thuiller et al. 2004, Araújo et al. 2005a, Heikkinen et al. 2006, Pearson et al. 2006, Fischlin et al. 2007, Thuiller 2007).

The most popular approaches so far for projecting climate change impacts on biodiversity relies on the use of bioclimate envelope models, also referred to as niche-based models or habitat models. These models relate current species' distributions (either presence/absence or abundance) with current climate variables and thereby define the climatic "envelope" of each species (Guisan and Thuiller 2005). By doing so, they are strictly empirical, include all biotic interactions constraining a species' distribution and are thus based on the realised climatic niche of species (according to Hutchinson 1957, see Araújo and Guisan 2006). By applying changing climate variables to the model, the species potential future climate space can be projected (Thuiller et al. 2008). This technique has been used to quantify potential changes for a large number of species and for a variety of organismal groups (e.g., Berry et al. 2002, Peterson et al. 2002, Thomas et al. 2004, Thuiller et al. 2005, Araújo et al. 2006, Thuiller et al. 2006b, Berry et al. 2007, Huntley et al. 2008).

The accuracy of projections from such models is sometimes questioned on the grounds that these models are intrinsically based on the realized rather than on the fundamental niche and as such they neglect a number of processes that affect species distributions, such as population dynamics and competition, land use, dispersal and the direct physiological effects of CO_2, for many of which assessing the fundamental niche would be necessary (e.g., Pearson and Dawson 2003, Hampe 2004, Guisan and Thuiller 2005, Araújo and Guisan 2006, Dormann 2007, Thuiller et al. 2008). Dispersal capabilities of species is typically addressed with two extreme options, allowing either unlimited or no migration at all for the species (e.g., Thuiller et al. 2006c, Fig. 1). Current developments attempt to provide more realistic projections by using more sophisticated dispersal models (Pearson et al. 2005, Broennimann et al. 2006, Midgley et al. 2006, Albert et al. 2008, Engler and Guisan in review). However, the few empirical tests of the validity of the approach have shown a remarkable predictive accuracy at large spatial scales demonstrating that the strong underlying hypotheses of climate envelope models are not strongly limiting at large spatial scales (Martínez-Meyer et al. 2004, Araújo et al. 2005b).

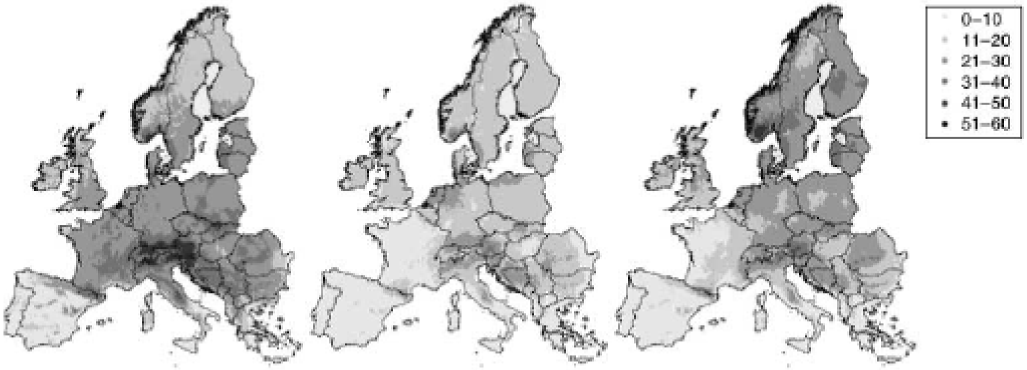

Figure 1. Illustration of the importance of dispersal ability in modelling studies. The spatial pattern for broadleaved deciduous tress: current species richness (a), species richness by 2080 assuming null migration (b), and species richness by 2080 assuming unlimited migration (c). The grey scale legend corresponds to the number of species per class. (From Thuiller et al. 2006c, Fig. 2a.)

A complementary approach relies on mechanistic models such as process-based dynamic vegetation models (DVMs). These models explicitly simulate the population dynamics of shrubs and trees, but they tend to be either parameterized for particular study sites (Badeck et al. 2001) or vegetation is represented by a few plant function types (PFTs, such as broad-leaved deciduous tree), not specifically accounting for changes in species richness and implicitly assuming unlimited dispersal of species or PFTs (Prentice et al. 2007). Furthermore, this approach is generally only applied to woody plants, summarizing all herbaceous species in one or two types; and the generalized, PFT-based models have been developed focusing on the modelling of ecosystem processes, such as net primary productivity, terrestrial ecosystem water cycling and carbon sequestration (Prentice et al. 2007).

We have performed a detailed search based on published literature and access to unpublished sources from EU research projects such as ATEAM[2], ALARM[3], MACIS[4] and BIOASSESS[5], and compiled current known impacts of climate change on terrestrial

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[2] ATEAM Advanced Terrestrial Ecosystem Analysis and Modelling: http://www.pik-potsdam.de/ateam/
[3] ALARM Assessing LArge scale Risks for biodiversity with tested Methods: http://www.alarmproject.net/alarm/ (Settele et al. 2005)
[4] MACIS Minimisation of and Adaptation to Climate change Impacts on biodiverSity: http://www.macis-project.net/
[5] BIOASSESS The biodiversity assessment tools project: http://www.nbu.ac.uk/bioassess/

biodiversity, including studies on relevant processes such as so-called “fingerprints” and phenological changes. Furthermore, we have completed a review of existing projections of climatic impacts on phenology, range shifts, extinctions, ecosystem changes and indirect changes through land use for terrestrial life forms, covering model outputs currently available including those of ALARM and their implications for conservation management and environmental policy in general. Our review is structured along a number of processes (phenology changes, range shifts, extinctions and ecosystem changes) that influence European biodiversity; each section covering evidence of observed changes, projections of future changes and a conclusions section. Indirect changes through land use as well as sea level rise are given one section each, and the main overall conclusions are summarized at the end of this review.

2. Processes

European species and ecosystems are reported to have responded to climate changes in a wide range of ways, such as upward shift of species (e.g., tree line and alpine species), phenological changes (e.g., advanced timing of flowering, breeding and migration), increasing productivity and forest carbon sinks, invasion of evergreen broad-leaved species in Alpine forests, disappearance of wetlands and changes in vegetation composition (Alcamo et al. 2007). An overview of projected global impacts of climate change on ecosystems for different levels of global mean annual temperature rise can be found in the latest IPCC report (Table 4.1 in Fischlin et al. 2007).

2.1 Phenology changes

Most biological events in nature follow a cyclical timing, often on an annual course which is called phenology (Badeck et al. 2004). Several events are triggered by the annual variation in temperature, precipitation and light levels. Changes in climate are thus expected to cause changes in the timing of many of these events as well as interactions with other events such as changing light levels. Moreover, certain phenological events such as first flowering and first arrival of migratory birds have been observed and recorded by scientists as well as ordinary people for in some cases hundreds of years, e.g., blooming dates of cherry trees in Kyoto, Japan are available since the 9th century (Arakawa 1956).

2.1.1 Evidence

Alterations in phenology in response to climate change have been reported in several reviews (e.g., Walther et al. 2002, Parmesan and Yohe 2003, Root et al. 2003, Badeck et al. 2004, Menzel et al. 2006, Parmesan 2006, Parmesan 2007). As a result of long-term observations among ornithologists, changes in phenology for birds have been frequently published across taxonomic and ecological groups (waterbirds, resident insectivores, migrant insectivores, corvids and seed-eaters) (e.g., Crick and Sparks 1999), such as migration changes (Both and Visser, 2001, Lehikoinen et al. 2004, Jonzén et al. 2006, Gordo and Sanz 2006), and changes in nesting time, mismatches in pollination and predator-prey interrelationships (Crick et al. 1997, Visser et al. 1998, Crick and Sparks 1999, Visser and Holleman 2001, Peñuelas et al. 2002, Both et al. 2006). Spring migration typically closely follow weather variations; short-distance migrants, spending the winter within Europe, have been found to respond more strongly to climate change than long-distance migrants (Rubolini et al. 2007), though the latter group have also responded to climate change in their wintering grounds and along their migration routes (Gordo 2007).

Phenology changes have also been reported for butterflies (Sparks and Yates 1997, Roy and Sparks 2000), vegetation (Menzel and Fabian 1999, Menzel et al. 2006) and amphibians (Beebee 1995). Earlier spring events are particularly well documented, while a few studies report changes in autumn events, such as delayed leaf fall (Peñuelas et al. 2002, Menzel at al. 2006), earlier fruiting (Peñuelas et al. 2002, Gange et al. 2007) and earlier departure of migratory birds (Cotton 2003) despite an extended growing season (Menzel and Fabian 1999). Both delays and advances in autumn bird migration are reported, and the reasons behind these are less well understood than spring migration changes (Gordo 2007). Another reported climate-related phenology change is that spring flowering plants in urban environments tend to bloom earlier than in the surrounding rural environment (see Neil and Wu 2006 for a review). See Table 1 for a summary of observed phenological changes focusing on the European region.

Table 1. Reported observed phenology changes from the literature.

Study	**Number of species**	**Species**	**Assessed time period**	**Phenology change**
Crick et al. 1997	20 (65)	birds (UK)	1971-1995	earlier egg-laying (3.5 days/decade)
Ahas 1999	8	birds, plants, fish (Estonia)	1916-1996	earlier spring (1.0 day/decade); *delayed* arrival migrating birds (0.4 days/decade)
Menzel and Fabian 1999	616	plants (Europe)	1959-1993	earlier spring (1.8 days/decade)
	178	plants (Europe)	1959-1993	delayed autumn (1.4 days/decade)
Roy and Sparks 2000	35	butterflies (UK)	1976-1998	first appearance (3.7 days/decade)
Both and Visser 2001	1	pied flycatcher (Netherlands)	1980-2000	earlier egg-laying (5.0 days/decade) ; *no* earlier arrival
Fitter and Fitter 2002	385	plants (UK)	1954-2000	first flowering (4.5 days/decade in the 1990s)
Peñuelas et al. 2002	103	plants (Spain)	1952-2000	earlier leafing (3.3 days/decade) earlier flowering (1.2 days/decade) earlier fruiting (3.5 days/decade) delayed leaf fall (2.7 days/decade)
	1	butterfly (Spain)	1952-2000	first appearance (2.3 days/decade)
	6	migratory birds (Spain)	1952-2000	*delayed* arrival (3.1 days/decade)
Cotton 2003	30	migratory birds (UK)	1971-2000	earlier arrival (2.7 days/decade); earlier departure (2.7 days/decade)
Parmesan and Yohe 2003	172	herbs, shrubs, trees, birds, butterflies, amphibians (global)	N.A.	earlier spring (2.3 days/decade)
Root et al. 2003	87	several (global)	1951-2001	earlier spring (5.1 days/decade)
Stefanescu et al. 2003	17	butterflies (Spain)	1988-2002	first appearance (10.4 days/decade)

Table 1 continued

Study	Number of species	Species	Assessed time period	Phenology change
Lehikoinen et al. 2004	983	migratory birds (Europe)	1950-2001	earlier arrival (3.7 days/decade)
Schaber and Badeck 2005	9	plants (Germany)	1880-1999	earlier blooming (up to 21 days/decade), earlier budburst (up to 8 days/decade), longer vegetation period (up to 7.3 days/decade) (all values for 1984-1999)
Stervander et al. 2005	19	long-distance migratory birds (Sweden)	1952-2002	earlier arrival (0.9 days/decade)
Menzel et al. 2006	561	plants, animals (Europe)	1971-2000	earlier leafing, flowering, fruiting (2.5 days/decade); delay leaf colouring/fall (0.2 days/decade)
Zalakevicius et al. 2006	40	migratory birds (Lithuania)	1971-2004	earlier arrival (2.8 days/decade)
Gange et al. 2007	315	fungus (southern England)	1950-2005	increased fruiting period (9.3 days/decade)
Rubolini et al. 2007[6]	672	migratory birds (Europe)	1960-2006	earlier arrival (3.7 days/decade)
Parmesan 2007	203	amphibians, birds, butterflies, herbs/grasses, shrubs, trees, fish, flies, mammals (Northern Hemisphere)	N.A.	earlier spring (2.8 days/decade) (amphibians 7.6, birds 3.7, butterflies 3.7, herbs and grasses 1.1, shrubs 1.1, trees 3.3)
Kauserud et al. 2008	83	mushrooms (Norway)	1980-2006	delayed autumn fruiting (6.4 days/decade)

Note that, in some studies presented in Table 1, phenology changes were calculated based on those species for which significant trends were detected, e.g., 20 of 65 species in Crick et al. (1997). Root et al. (2003) also based their calculations on species with only significant phenology changes, > 1.0 day/decade according to Parmesan (2007), while Parmesan and Yohe (2003) included all studied species in their results, partly explaining the different outcomes in these two comprehensive multi-species studies.

Species respond phenologically very differently; see for example Table 2 in Fitter and Fitter (2002), Fig. 2 in Root et al. (2003), Table 1 and Fig. 2 in Parmesan (2007). Amphibians show significantly stronger shifts towards earlier breeding than all other groups (Parmesan 2007, Table 1), however, other studies report that climate change has not influenced the timing of breeding in amphibians (e.g., Blaustein et al. 2001), and moreover, most of the amphibian response studies have taken place outside Europe. Peñuelas et al. (2002) found a range of changes in flowering date spanning from an advance of 14.6 days/decade[7] for *Lippia triphylla* to a delay of 7.8 days/decade for

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[6] Large overlap with the studies included in Lehikoinen et al. 2004

[7] 70.1 days for the observation period 1952-2000 (Table 2 in Peñuelas et al. 2002)

Fraxinus augustifolia. Even stronger phenological responses are reported for a red squirrel population in southwest Yukon, Canada, which has advanced their breeding by 18 days over the last decade (Réale et al. 2003). Furthermore, the response can vary between sexes of the same species, as shown for two *Triturus* species in Wales where increased spring temperatures have resulted in a greater change in earlier arrivals for males than females (Chadwick et al. 2006).

A few phenological developments that are in opposition to the general trends have been reported, such as delayed spring events for plants in the Balkans probably caused by a different regional climate change (Menzel and Fabian 1999) as well as a later arrival of spring migratory birds in north-eastern Spain, probably resulting from changed climatic conditions along migratory routes or at the over wintering sites in Africa (Peñuelas et al. 2002, see also Gordo 2007). Other delayed spring arrivals of migratory birds have been reported, e.g., white stork in Lithuania (Zalakevicius et al. 2006), skylark and white wagtail in Estonia (Ahas 1999), whinchat in the UK (Cotton 2003), and barn swallow in the Slovak Republic (Sparks and Braslavská 2001). A review study by Lehikoinen et al. (2004) found that 2% of migratory bird species arrived significantly later, while 39% arrived significantly earlier. Migratory pied flycatcher did not arrive earlier on their breeding grounds in the Netherlands though they have advanced their egg-laying by 10 days in two decades (Both and Visser 2001). A species may respond very differently in different parts of its range, as do the great tits in the UK and the Netherlands (Visser and Both 2005).

Unsynchronized phenological changes for different species have resulted in mismatches between trophic levels such as first insect appearance and the arrival of migrant birds (Visser and Both 2005, Menzel et al. 2006, Parmesan 2006, Parmesan 2007), or oak bud burst and winter moth egg hatching (Visser and Holleman 2001). Species with low adaptability to phenology changes, such as plants responding to day length, may suffer greater stress or even extinction risk in extended climate change (Bradley et al. 1999). Mushrooms in the UK have increased their fruiting season from 33 to 75 days between 1950 and 2005, and, moreover, some mushroom species have begun to fruit in spring as well as in autumn (Gange et al. 2007).

Individuals of migrating species have changed their winter locations or even stopped migrating, such as the increasing number of white storks remaining in the Iberian Peninsula instead of migrating to Africa (Gordo and Sanz 2006), indicating the complexity of climate change-related impacts on migration (e.g., Gordo 2007). Where formerly migratory populations become resident they may have large impacts on local ecosystems: these may become exposed to continuous grazing pressure, resulting in ecosystem collapse and, potentially, further local climate changes (Sinclair and Fryxell 1985, in Robinson et al. 2005).

Additionally, phenology changes seem to depend on the latitude - this is probably caused by stronger recent climate warming in the north. Root et al. (2003) found spring to be earlier by 5.5 days/decade for the 50° to 72° N latitude band, with a smaller change of 4.2 days/decade for the 32° to 49.9° N latitude band. A significant stronger response at higher northern latitudes is also reported by Parmesan (2007).

The average European advance of spring and summer during the last three decades of the 20th century has been 2.5 days per degree Celsius, while autumn has been delayed by on average 1.0 day per degree Celsius of warming temperatures (Menzel et al. 2006).

2.1.2 Projections

By 2080 birds in the UK are projected to start laying eggs on average 8, ranging up to 18, days earlier than today (Crick and Sparks 1999). Warming by 1°C could advance first and peak appearances of most UK butterflies by 2-10 days (Roy and Sparks 2000), and advance the date of olive flowering in the western Mediterranean region by 6 days (Osborne et al. 2000). The advancement of spring is estimated at between 2.5 and 6 days per degree Celsius (Rosenzweig et al. 2007); see Table 2 for examples of projected phenological changes focusing on the European region.

Table 2. Examples of projected phenology changes from the literature.

Study	Number of species	Species	Assessed time period	Phenology change
Crick and Sparks 1999	27 (36)	birds (UK)	2080	earlier egg-laying (8 days, up to 18 days)
Roy and Sparks 2000	N.A.	butterflies (UK)	warming by 1°C	first and peak appearance advance by 2-10 days
Osborne et al. 2000	1	olives (western Mediterranean)	warming by 1°C	earlier flowering by 6 days
Rosenzweig et al. 2007	N.A.	N.A.	warming by 1°C	earlier spring by 2.5-6 days
Visser et al. 2006	2	caterpillar and great tits (Netherlands)	2005-2010	earlier peak day by 18 days; earlier laying date by 15 days

2.1.3 Conclusions

- There is strong evidence that many, though not all, species show phenological responses to climate change. Responses have been stronger in the north, probably as warming has been more pronounced at northern latitudes.
- Species respond differently, resulting in mismatches between trophic levels as well as altered relative competition between species. The responses have also been shown to vary within species (e.g., males vs. females), further emphasizing the complexity of mismatches and thus unpredictable consequences.
- Migrating birds show complex response patterns because their behaviour is influenced by changes in climate and the phenology of prey species in their summer and winter habitats, as well as along their migratory routes.
- However, few groups have been studied and studies do not report in a standard manner (e.g., calculating means across species responding or means across all species studied), thus complicating both evidence and predictions of future phenology changes.

2.2 Range shifts (latitude, altitude)

Climatic conditions, particularly temperature and moisture, determine suitable regions for certain species (Woodward 1987, Guisan and Thuiller 2005). Changes in these climatic conditions are therefore likely to change the geographical extent of suitable areas, and these changes in latitude and/or altitude are called range shifts. A warming by 1°C in Europe corresponds approximately to a 150 km shift northwards in temperature isotherms (Watson et al. 1998), or a shift upwards in altitude by 150[8]-180[9] meters. In temperate and colder areas of Europe, range shifts have thus often involved the northwards or upslope extension of climate space - and thus of geographic range - for many species which may or may not be accompanied by a contraction of climate space at the southern or lower altitude range limit (Woodward 1987; Parmesan et al. 1999). These patterns are also modulated by change in precipitation regimes, but in a complex way due to this variable varying less linearly with latitude or altitude.

2.2.1 Evidence

Evidence of range shifts caused by recent climate change have been reported in several reviews (e.g., Walther et al. 2002, Parmesan and Yohe 2003, Root et al. 2003, Lavergne et al. 2006, Parmesan 2006, Table 3). Shifts in polar, boreal and temperate species (e.g., Parmesan 2006, Callaghan et al. 2007) are frequently reported. Range shifts upward in elevation have been mostly reported for plant species (e.g., Grabherr et al. 1994, Kullman 2001, Dobbertin et al. 2005, Vittoz et al. 2008; see Grabherr 2003 and Walther et al. 2005b for reviews) but also for butterflies (Konvicka et al. 2003, Thomas et al. 2006, Wilson et al. 2007). Range shifts northwards in latitude are often reported for well-studied species groups such as birds (e.g., Thomas and Lennon 1999), butterflies (e.g., Hill et al. 1999, Parmesan et al. 1999), but also recently for plants (e.g. Walther et al. 2005a). See Table 3 for a summary of observed range shifts focusing on the European region.

Table 3. Reported observed range shifts from the literature.

Study	**Number of species**	**Species**	**Assessed time period**	**Range shift**
Latitude				
Thomas and Lennon 1999	59	birds (UK)	1970-1990	9.4 km/decade
Parmesan and Yohe 2003	99	birds, butterflies, alpine herbs (global)	N.A.	6.1 km/decade
Brommer 2004	116	birds (Finland)	1976-1988	16 km/decade
Franco et al. 2006	2	butterflies (UK)	1985-2004	45 km/decade
Hickling et al. 2006	329	16 taxonomic groups (UK)	1970-1995	18 km/decade (range: dragonflies 42 km/decade to herptiles – 33 km/decade)
Mittika et al. 2008	1	Map butterfly (*Araschnia levana*) (Finland)	1983-1991 1992-1998 1999-2004	13 km/decade 15 km/decade 75 km/decade

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[8] Based on an average lapse rate of 0.65°C per 100 meter (Met Office 2008).

[9] Based on the annual mean lapse rate of 0.55°C per 100 meters for the Alps (Ozenda and Borel 1991).

Table 3 continued

Study	**Number of species**	**Species**	**Assessed time period**	**Range shift**
Altitude				
Grabherr et al. 1994	9	Alpine plants (the Alps)	1913-1992	up to 4 m/decade
Kullman 2001	3	tree line (Sweden)	1900-2000	10-16 m/decade
Konvicka et al. 2003	15 (117)	butterflies (Czech Republic)	1965-1998	18 m/decade
Parmesan and Yohe 2003	99	birds, butterflies, alpine herbs (global)	N.A.	6.1 m/decade
Peñuelas and Boada 2003	1	*Fagus sylvatica* (Spain)	1945-2000	13 m/decade
Tryjanowski et al. 2005	1	white stork (Poland)	1974-1999	48 m/decade
Walther et al. 2005b	18	Alpine plants (Switzerland)	1905-2003	28 m/decade
Dobbertin et al. 2005	1	*Viscum album* (Switzerland)	1910-2003	~ 20m/decade
Franco et al. 2006	1	butterflies (UK)	1985-2004	74 m/decade
Hickling et al. 2006	329	16 taxonomic groups (UK)	1970-1995	10 m/decade (range: soldier beetles 25 m/decade to herptiles -13 m/decade)
Gehrig-Fasel et al. 2007	N.A.	tree line (Switzerland)	1985-1997	23 m/decade
Wilson et al. 2007	107	butterflies (central Spain)	1970-2004	85 m/decade (increase in lower elevational limit)

Range restricted species show more severe range contractions in response to climate change, where amphibians and reptiles are most negatively affected (Parmesan 2006). An important note is that, as for phenology, not all species respond in the same way (Hickling et al. 2006, Table 3). Hickling et al. (2006) reported that most studied species (275 out of 329) expanded northwards; however, amphibians and reptiles were found to actually retreat *southwards* (33 km/decade), and to *lower* altitudes (13 m/decade), which might be caused by reduced dispersal ability in fragmented habitats.

Climate changes resulting in range shifts also lead to changes in species community composition (Walther et al. 2002, Vittoz et al. 2008). For instance, Walther et al. (2005b) found that the upward shift of alpine plants in synchrony with climate warming from 1985 to 2003 resulted in an increase in species richness by 3.7 species per decade, an accelerated trend compared to the observed increase of 1.3 species per decade from 1905 to 1985. Vittoz et al. (2008) observed a species enrichment with more thermophilous plants from lower elevation on an isolated nunatak surrounded by glaciers in the Swiss Alps, evidencing a shift in species composition and community structure. The same observation of change in community structure was observed by Vittoz et al. (in review) at lower elevations in subalpine plant communities.

As for the enrichment of the flora of high-elevation summits in the Alps (Grabherr et al. 1994, Grabherr 2003, Walther et al. 2005b, Vittoz et al. 2006, 2008, Pauli et al. 2007), an increased mountain plant species richness is also reported from Norway, corresponding to an average increase of 1.5 species per decade from 1930 to 1998, with climate warming proposed to have been the major driver (Klanderud and Birks 2003). The number of migratory butterfly species recorded in a UK garden has increased by 13 species per decade from 1982 to 2005, together with increasing temperature (Sparks et al. 2007). Climate change is already influencing species richness in European bird communities, resulting in a significant decrease in the proportion of short-distance migratory birds (Lemoine et al. 2007b).

However, for the Mediterranean region, species richness for butterflies was found to be negatively correlated with temperature (Wilson et al. 2007) and positively correlated with precipitation (Stefanescu et al. 2004), indicating a particular response risk to the projected future climate changes for this region.

2.2.2 Projections

European climate zones are generally projected to move from south-west towards north-east (e.g., Berry et al. 2006, Malcolm et al. 2006, Ohlemüller et al. 2006a, Williams et al. 2007). Plants are very likely to expand their ranges northwards while contracting in southern European mountains and in the Mediterranean region (Bakkenes et al. 2002, Thuiller et al. 2005), resulting in a net increase of species richness in northern Europe and a probable decrease in Mediterranean countries (Thuiller et al. 2005, Bakkenes et al. 2006). Similar patterns are expected for amphibians and reptiles, although their limited dispersal abilities might prevent northward expansions (Araújo et al. 2006). For these taxa (in particular amphibians), south-western Europe is the region most likely to be negatively affected, essentially due to increases in aridity. Other European studies including plants, insects, birds and mammals indicate that a general range shift from the south-west to the north-east is projected; however, the effects are likely to differ among species (Harrison et al. 2006, Berry et al. 2007). See Table 4 for examples of projected range shifts focusing on the European region.

Table 4. Examples of projected range shifts from the literature.

Study	Number of species	Species	Assessed time period	Range shift
Latitude				
Skov and Svenning 2004	26	forest herbs (Europe)	2070-2099	21-39 km/decade (required minimum migration rate)
Huntley et al. 2008	431	birds (Europe)	2070-2099	258-882 km
Altitude				
Dullinger et al. 2004	1	*Pinus mugo* (Austria)	1000 years	Increase in area covered, today 10%, future 24-59%

Mediterranean endemic plants and vertebrates are probably mostly influenced as their ranges are projected to become very negatively affected by climate change (Malcolm et al. 2006). The negative correlation between temperature and species richness for

Mediterranean butterflies in combination with the projected warming and drying is a serious threat to biodiversity in this region (Stefanescu et al. 2004).

The Arctic region is projected to experience a stronger change in climate than most other European regions, and the impacts on biodiversity are expected to be strong (Callaghan et al. 2007). The Barents region is expected to be strongly affected with projected changes such as shifting tree lines, shifts of vegetation types as well as shifts, contractions and extinction of several Arctic species (Callaghan et al. 2007).

The observed upward shifts of plant species along the elevation gradient in mountains are expected to continue and be further amplified as a response to the warming climate (e.g., Gottfried et al. 1999, Guisan and Theurillat 2000a, 2000b, Dirnböck et al. 2003, Klanderud and Birks 2003; see also Theurillat and Guisan 2001). As alpine and nival species will shift upward, their lower distribution will be constrained by tree species migrating as well toward higher elevations (Dullinger et al. 2004, Rickebusch et al. 2007). Mountain species are often range restricted with limited dispersal abilities (Vittoz and Engler 2007) and are therefore especially sensitive to climate change (e.g., Gottfried et al. 1998, Guisan and Theurillat 2000b, Berry et al. 2003, Dirnböck et al. 2003, Dullinger et al. 2004, Thuiller et al. 2005, Engler and Guisan (in review)).

The mean potential range shift by the end of the 21st century for European breeding birds is estimated to be 258-882 km, depending on which emission scenario and climate model applied, in a direction NNW to NE (though the range for individual species spanned between 20 and 3578 km), and the mean range shift rate required to follow this boundary extension would be 36-66 km per decade (Huntley et al. 2008). European forest herbs would need to migrate at a rate of 21-39 km per decade in latitude (Skov and Svenning 2004), and 1000m per decade in altitude for low- to mid-elevation grassland herbs depending on the emission scenario applied, to track their potential range shift caused by 21st century projected climate change. (This projection is based on results from a sensitivity modelling analysis by Engler and Guisan (in review)).

In trophically interacting species, Schweiger et al. (in press) were able to show an increasing spatial mismatch. Current niche spaces of the monophagous butterfly *Boloria titania* and its host plant *Polygonum bistorta* already show some degree of spatial mismatch. Under increased climate change, however, the mismatch is expected to be even larger. Assuming restricted dispersal abilities for the host plant, this is projected to result in more than 50% range loss and up to almost 90% if the butterfly is also dispersal limited. In general, already observed changes in composition of species communities are expected to be further emphasised during the 21st century, such as a projected continued change in proportion of short- and long-distance migratory birds (Lemoine et al. 2007b).

2.2.3 Conclusions

- Observed range shifts both in latitude (south-west towards north-east) and altitude (contractions at lower altitudes and expansions at higher altitudes) are evident for several species.
- Species have responded differently and range-restricted species are more negatively affected as their ranges have contracted.

- Mountain and Arctic species and southern European endemic species have higher risks of rapid climatic shifts.
- Spatial mismatch in interacting species can potentially increase range contractions.
- Range shifts are projected to continue with increased climate change, though at a rate that many species will have difficulties to follow.

2.3 Extinctions

Extinction occurs when there has been a global loss of all individuals of a species. The IUCN Red List of threatened species categorises a species as extinct "when there is no reasonable doubt that the last individual has died" (IUCN 2005). In a few cases the extinction moment is well known such as when the very last known individual of a species has died in captivity, such as Martha, the last passenger pigeon (Herman 1948) and the Thylacine, also known as the Tasmanian Tiger (Paddle 2000). Out in nature, a specific extinction event is often more difficult to confirm; a recent example is the likely extinction of river dolphins (Guo 2006). Ecologically, a species can be characterized as being extinct when individuals cannot reproduce as only one of the sexes remains, such as Lonesome George, the last giant tortoise of his kind (Nicholls 2004).

2.3.1 Evidence

Population extinctions caused primarily by climate change leading to range contractions have already been reported for butterflies (Franco et al. 2006) and amphibians (Pounds et al. 1999, Parmesan 2006, Pounds et al. 2006), though it is often very difficult to identify climate change as the only cause for extinctions as several factors tend to interact (Thomas et al. 2006). Furthermore, species extinctions often lag behind climate change, and the lag times have only been quantified for tropical birds (Thomas et al. 2006). In particular long-lived species can survive long time-periods after the habitat has become unsuitable but might not be able to reproduce, and therefore extinctions may be delayed until some event or disturbance finally removes the species from the landscape (Prentice et al. 1993).

At a small scale, however, Both et al. (2006) showed in a comparison of nine populations of pied flycatcher (*Ficedula hypoleuca*) in the Netherlands that populations have declined by about 90% over the past two decades in areas where caterpillars, which are the food for provisioning nestlings, peak early in the season, so that the birds are currently mistimed. In areas where the mistiming between food source and breeding season was minute, the population decline was, at most, weak. Therefore, phenological mismatch between food resource and predator as a result of climate change can lead to population declines.

2.3.2 Projections

While there is relatively limited evidence of extinctions caused primarily by climate change (but see Pounds et al. 2006), there are a number of projections raising serious future concern for many species. One estimate is that, globally, on average 20% to 30% of the plant and animal species assessed are likely to be at increasingly high risk of extinction caused by climate change within the 21st century (Fischlin et al. 2007). A study by Thuiller et al. (2005) of the future distribution of 1350 European plant species indicates that more than 50% of the modelled species, when assumed to be unable to disperse, might become vulnerable, endangered, critically endangered or committed to extinction by 2080 (Fig. 2). At a local scale, Guisan and Theurillat (2000b) showed that mountain plants at high elevations are particularly susceptible to extinctions, with extinction rates up to 4.8% by 2100 for a 4.5°C warming, and up to 38.7% of species losing more than 90% of their suitable habitat under the same scenario. Similar projections are reported elsewhere (e.g., Dirnböck et al. 2003, Randin et al. in review).

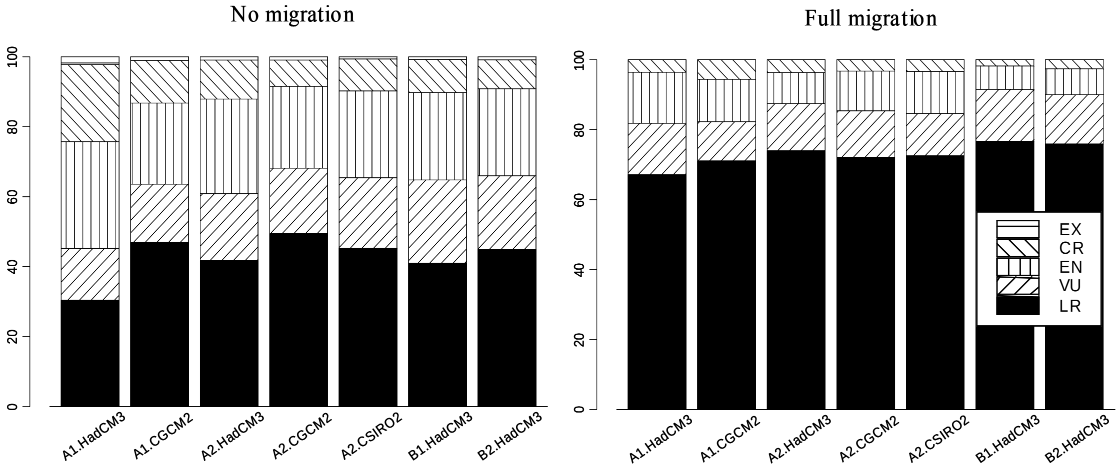

Figure 2. Proportion of European plant species classified according to the IUCN Red List assessment under two extreme assumptions about species migration by 2080. EX, extinct; CR, critically endangered; EN, endangered; VU, vulnerable; LR, lower risk. The different columns correspond to different climate change scenarios. (From Thuiller et al. 2005, Fig. 1)

Most endemic or near-endemic European bird species are projected to have no or only little overlap between present and potential ranges by the end of the 21st century, and are thus at increased risk of extinction (Huntley et al. 2008), with a best guess estimate of 400-550 landbird extinctions worldwide by 2100 (Sekercioglu et al. 2008). Up to 25% of the plant species now present in southern Europe may disappear by 2100 (Bakkenes et al. 2006). The Mediterranean mountain region is one of the most sensitive regions regarding increased vulnerability (Schröter et al. 2005) and predicted species loss (Thuiller et al. 2005). However, these extinction numbers are based only on range shifts, by comparing the areas of suitable climate today and in the future, and the uncertainties are large as different techniques for biodiversity projections can lead to a wide range of results and current models do not account for all factors influencing species (e.g., Araújo et al. 2006, Botkin et al. 2007, Thuiller et al. 2008, Engler and Guisan in review).

Dispersal ability of species is an important parameter in projecting future impacts of climate change (Midgley et al. 2007; see also Fig. 1). Araújo et al. (2006) found that most European amphibians and reptiles would expand their range if dispersal was unlimited, while the range of more than 97% of these species would be reduced if they were not able to disperse, which might be close to the actual situation due to widespread fragmentation of suitable habitats in the highly modified landscapes of Europe. By 2050, 6-11% of 1200 modelled plant species in hypothetical reserves selected by a variety of reserve design methods might be potentially lost, indicating the importance of including climate change in conservation planning (Araújo et al. 2004). Thus, range shifts are a major problem to conservation as species that are already protected might actually shift their distributions outside current reserves as climate changes.

Engler and Guisan (in review) developed a new tool to simulate dynamic dispersal of plant species in mountain landscape and showed that constraining dispersal of species caused extinction rates to dramatically increase (up to 95% decrease in colonized surface compared to unlimited dispersal).

Table 5 summaries projections of extinctions from future climate change.

Table 5. Projections from the literature of extinctions from future climate change.

Study	Number of species	Species	Assessed time period	Projection of extinctions
Guisan and Theurillat 2000b	62	alpine plants (Switzerland)	2100	1.6-4.8% extinctions (scenario dependent)
Thomas et al. 2004	832 (dispersal); 995 (no dispersal)	mammals, birds, reptiles, butterflies, plants (global)	2050	15-37% committed to extinction[10] (dispersal dependent)
Thuiller et al. 2005	1350	plants (Europe)	2080	27-42% species loss per grid cell (however, these species could maintain viable populations in other grid cells)
Lawler et al. 2006	100	mammals (western hemisphere)	2090	0-7% extinction (if unlimited dispersal); 6-14% extinction (if no dispersal)
Malcolm et al. 2006	142794	plants and vertebrates (global)	2100	1-43% (average 11.6%) of endemic species committed to extinction by 2100 (migration/dispersal dependent range)
Ohlemüller et al. 2006b	17	plants woody species (Europe)	2095	Climatic conditions will become less suitable for 76% of the species studied
Berry et al. 2007	389	terrestrial and coastal (Europe)	2080	1.5% of modelled species could lose all suitable climate space; 2.8% could lose >90%
Fischlin et al. 2007	N.A.	plants and animals (global)	2100	20-30% of assessed species likely to be at increasingly high risk of extinction if global average temperature increase more than 2-3ºC relative to pre-industrial levels
Levinsky et al. 2007	120	mammals (Europe)	2099	1% (unlimited dispersal) or 5-9% (no dispersal) committed to extinction; 32-46% or 70-78% severely threatened
Normand et al. 2007	84	plants (Denmark)	2100	4-7% species lost at the Danish scale, 0% at the European scale
Sekercioglu et al. 2008	8459	landbirds (global)	2100	4.7-6.5% extinctions; additionally 25% at risk of extinction (intermediate scenarios)

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[10] Species with zero projected future climate space and therefore committed to eventual extinction (Thomas et al. 2004).

2.3.3 Conclusions

- Observed extinctions from climate change are so far few, though it is often difficult to identify climate change as the only cause since several factors tend to interact.
- Species extinctions often lag behind climate change, and thus the correct extinction risks might be delayed and not yet fully expressed.
- Current modelling techniques do not include all factors affecting biodiversity which make detailed projections difficult and the resulting numbers are therefore uncertain.
- Moreover, there are difficulties in summarizing or comparing projection results due to differences in models, climate change scenario applied, etc.
- However, despite these uncertainties and that only a few extinctions have yet been observed, larger impacts are anticipated for the near future and a number of studies project serious concern and increased extinction risks for many species.

2.4 Community and ecosystem changes

Climate change also influences the species composition and structure of ecosystems. From a species perspective, ecosystem responses, such as shifts in ecosystem or vegetation types, imply changes in habitat quality and distribution (Leemans and Eickhout 2004, Malcolm et al. 2006). Furthermore, climate change is likely to result in increased niche-availability, facilitating the establishment of non-native invasive species and thus changing the ecosystem (e.g., Gritti et al. 2006, Callaghan et al. 2007, Thuiller et al. 2007, Ward and Masters 2007, Wolf et al. 2008).

2.4.1 Evidence

In high northern latitudes vegetation greenness has increased since the 1980s, primarily driven by increasing temperature and longer growing seasons (Lucht et al. 2002). These changes imply denser canopies, which will affect species differently. Temperate broadleaved trees have started to replace boreal conifers at the southern boundary of the boreal forest, probably in response to milder winters, and tree lines have shifted to higher altitudes; in the Swedish Scandes, for example, by 100-150m since the 1950s, driven by a temperature increase of 0.8ºC between 1901 and 2000 (Kullman 2002). Changes in community composition and structure is already reported for subalpine grasslands in the Swiss Alps (Vittoz et al. in review), and in the Austrian Alps where Pauli et al. (2007) observed an increase in plant species richness and a decline in the most cold-adapted species, a process most likely to continue and thus implying a major threat to biodiversity in the high mountain regions of the European Alps.

While studies on ecosystems in the strict sense are rare, a few analyses exist on changes in composition of functional traits of communities. Matesanz et al. (in press) investigated long-term responses of a range-margin steppic grassland community in central-East Germany to identify temporal trends in cover and species richness, and to assess whether changes in climate might explain these trends. Between 1980 and 2005 annual and April temperatures significantly increased by 1.4°C and 2.3°C, respectively; no significant temporal trend was found for rainfall. Significant temporal trends were found in cover and species richness of the plant community, and these trends were functional group (perennials vs. annuals) and species-dependent. Total cover and perennial cover significantly decreased over time, while no trend was found for the annual cover. The number of perennial species significantly decreased over time, but not the number of annual species. While both total cover and species richness were negatively associated with increasing April temperatures, only perennial species showed a significant decrease with temperatures. Only annual species significantly responded to interannual rainfall fluctuations.

In a study from Lake Constance (which borders Germany, Switzerland and Austria) Lemoine et al. (2007a) analysed changes in regional abundance of 159 coexisting bird species from 1980–1981 to 2000–2002. Farmland birds, species with northerly ranges and long-distance migrants all declined, while wetland birds and species with southerly ranges increased in abundance. A separate analysis of the two decades between 1980–1981 and 1990–1992 and between 1990–1992 and 2000–2002 showed that the impact of climate change increased significantly over time. Latitudinal distribution was not significant in the first decade and became the most significant predictor of abundance changes in the second decade. Considering observed changes in the proportion of long-distance migratory species in 21 sites in Europe between 1972–76 and 1988–92, Lemoine et al. (2007b) showed a slight increase in the proportion of long-distance migratory species and a significant decrease in the proportion of short-distance migratory species. These changes can be explained by the isochronic changes in temperature and precipitation in the 21 sites.

2.4.2 Projections

Because of longer growing seasons and increasing levels of atmospheric CO_2, which has a fertilizing effect (Norby et al. 2005), potential vegetation productivity is likely to increase in most parts of Europe, except in the Mediterranean region, where it might decrease as a result of more pronounced drought (Morales et al. 2007).

Species are predicted to shift their range individually (Huntley 1991) and accordingly tree lines are projected to move to higher altitudes (Gehrig-Fasel et al. 2007, Rickebusch et al. 2007), thus replacing alpine vegetation and the associated species. As a result, the composition of communities is also expected to change drastically, with future communities diverging significantly from those observed today. Guisan and Theurillat (2000a) tested how predicted shift in species range could impact the future structure of plant communities. They found that two out of nine modelled alpine plant communities observed today were no longer predicted in a warmer future. Moen at al. (2004) estimated potential upper tree line shifts in the Swedish mountains of 233-667m, depending on climate change scenario, during the 21st century.

The boundary between temperate and boreal forests might move northwards by several hundred km (Fig. 3) and forest die-back might occur in the Iberian Peninsula (Fig. 3). As it takes hundreds of years for one forest type to replace another one, which is no longer optimally adapted to the local climate, long-term equilibrium changes are substantially larger than the one presented in Fig. 3. Non-adapted forests might suffer from increased susceptibility to environmental stresses, such as drought, pests and pathogens (Bradshaw et al. 2000). Tree line shifts, however, can be counteracted by increased grazing and browsing (Cairns and Moen 2004), and boreal trees, such as spruce (*Picea abies*) can, in managed plantations, to some extent be planted south of its natural distribution. Forest productivity and canopy density are also to a large extent driven by forest management policies, which are difficult to predict.

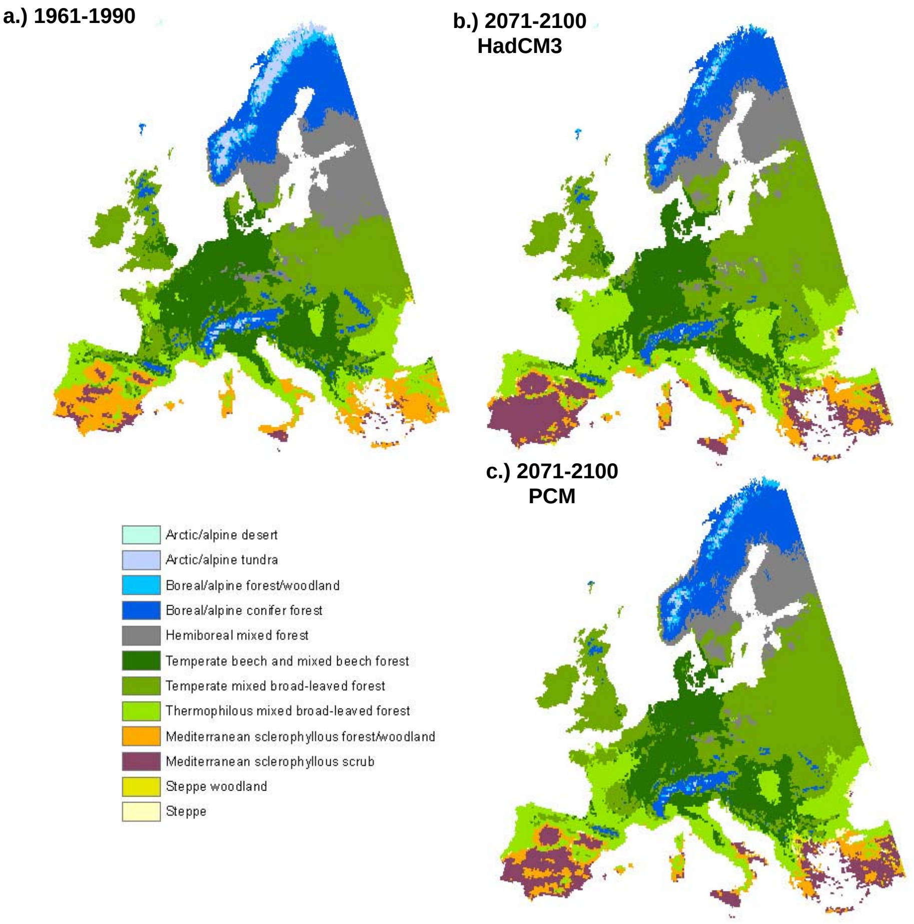

Figure 3. Potential natural vegetation across Europe simulated by the vegetation model LPJ-GUESS (Smith et al. 2001), parameterized for European vegetation: a.) current climate (averaged for 1961-1990); b.) and c.) with two climate models under the IPCC A2 emission scenario. The vegetation classification was based upon Bohn et al. (2003). (Hickler et al., in prep).

Kienast et al. (1998) predict several changes in vegetation composition in Central European mountain forests: Fagus-dominated communities in the colline-submontane belt might eventually be replaced by oak-hornbeam (*Carpinion*) communities. In the montane belt, the dominance of conifers will be seriously threatened by an invasion of deciduous species from the low montane and submontane belt. Furthermore, studies on the vegetation of Castile and Leon in Spain predict the replacement of the evergreen and semi-deciduous forests by deciduous forest because of a trend to a more oceanic climate (del Rio et al. 2005, 2006, del Rio and Penas 2006).

Considering composition of bird assemblages, Schaefer et al. (2008) investigated the relationship between current climate and the proportion of migratory species across bird assemblages in Europe. Their results suggest that increasing winter temperature is expected to lead to declines in the proportion of migratory species, whereas increasing spring temperature and decreasing spring precipitation may lead to increases. Changes in winter and spring temperature are expected to cause mainly adaptation in migratory activity, while changes in spring precipitation may result in both changes in the proportion of potentially migratory species and adaptation of migratory activity. Under current climate change projections, changes in the proportion of migratory species will be modest and the communities of migratory birds in Europe are projected to be altered through adaptation of migratory activity rather than through exchange of species.

In the Arctic and Alpine regions, climate change results in reduced snow cover and changes in frost/thaw periods, which lead to less insulation and frozen resources. Species such as lemming, musk ox and reindeer are affected by these changes, and dramatic population crashes following ice-crusting after freeze-thaw events have been reported, and their frequency appears to have increased in recent decades and is projected to continue increasing (Callaghan et al. 2007). Additionally, mild weather and wet snow spaces destroy burrows for voles and lemmings, while ice-crust formation reduces the insulation properties of the snow pack which is vital for their survival. Vole and lemming cycles are no longer observed in some areas and it is well known that the changes in vole and lemming abundances have cascading effects in the community, especially on the specialist predators such as snowy owls, skuas, ermines and weasels (Callaghan et al. 2007). Frost events often lead to the death of plants, which can have dramatic consequences for herbivores and even cause local extinctions. The influence of climate change on frost events is uncertain, though it is likely that the frequency will change in many regions (Inouye 2000).

2.4.3 Conclusions

- In central and northern Europe, broad vegetation types are likely to move northwards and to higher altitudes, primarily driven by longer and warmer growing seasons.
- Alpine and Arctic vegetation is expected to contract considerably, though the uncertainty is high due to dependence on thresholds such as presence/absence of snow and temperatures below 0°C.
- In southern Europe, drought may cause transformations of ecosystems and the associated habitats.
- Composition of bird assemblages may change according to their migratory behaviour rather than species exchange.

2.5 Indirect changes through land use

Climate change impacts are not the only drivers of change for biodiversity. Other sets of pressures are also leading to impacts upon the environment and upon biodiversity, in both urban and rural environments (reviewed by Mitchell et al. 2007). These pressures may interact amongst themselves and interact with climate change, to produce further indirect effects. Demographic changes are important here; for example, in the urban environment rising human populations may mean either an expanding urban boundary (urban sprawl) or ever-denser construction within the same boundary.

Urban sprawl may encroach upon sites that are important for wildlife directly (protected sites) or which contribute to the quality of wildlife sites elsewhere, e.g., water gathering grounds where precipitation can infiltrate and reach groundwater and river systems. Within cities “intensification” may mean encroachment upon open spaces including brownfield (such as previously industrial sites) which may have developed significant biodiversity value (e.g., abandoned railway lines). Bosher et al. (2007) suggest that intensification may mean fewer opportunities for wildlife, but there is a good deal of debate on the comparative impact of “compact cities” vs. urban sprawl (e.g., Ludlow 2006, McEvoy et al. 2006).

Apart from an increase in populations, other demographic changes (and particularly, a trend towards smaller households living in larger spaces) create similar issues and impacts, especially where these reinforce a tendency towards more soil sealing - i.e. covering land with non-permeable surfaces and channelling rainwater rapidly towards drainage, without opportunity for infiltration (Pauleit 2005). At a smaller scale, changes in building design can provide adaptation and mitigation - green roofs and walls, for example, can provide habitats for biodiversity and contribute to emissions reduction by reducing the need for air-conditioning (Oberndorfer et al. 2007).

Away from urban areas there are also changes and pressures affecting activities, with consequences for biodiversity chiefly as a result of habitat destruction, fragmentation or over-exploitation (Mitchell et al. 2007). For example, within agriculture these include changes to conventional farm practices and crops which result from changes in consumer tastes (e.g., leading to more land under protected environments) or changes in support mechanisms for certain crops or livestock and changes to agri-environment schemes. Policy development in other areas may affect land available to biodiversity, e.g., policy to increase generation of renewable energy (Hossell et al. 2006). Coastal areas in particular demonstrate an array of competing land uses (e.g., housing, tourism, ports and energy), all of which may interact further affecting biodiversity (Mitchell et al. 2007).

Climate change mitigation through intensified forestry and land allocation to biofuel production could also lead to increased pressure on biodiversity. Land use and management clearly fulfill various, often conflicting goals, such as food, timber and biofuel production, carbon sequestration, soil erosion protection, maintenance of biodiversity and recreation. Economic incentives and legislation should be used to balance strategies for fulfilling the different goals. However, a holistic, multi-sector analysis of policy options and their implications has to our knowledge not been carried out.

2.6 Sea level rise

Rises in sea level can pose an important pressure on biodiversity in coastal areas, but future sea levels are uncertain because current ice sheet models do not represent sudden collapse of large ice masses, often referred to as “dynamical ice losses” (Hansen et al. 2007, Meehl et al. 2007, Oppenheimer et al. 2007). Global models of ice melt and sea level rise, which do not account for dynamic ice losses, predict average increases of 18-59cm by the end of the century (Meehl et al. 2007). Local changes are also influenced by factors such as ocean density and circulation, adding between a few to about 20cm in most parts of Europe (Meehl et al. 2007). Reconstructions of sea level changes during glacial-interglacial transitions show that changes of several meters within a century have occurred in the past (Hansen et al. 2007, Jansen et al. 2007), with changes in forcings smaller than the projected climate change (Hansen et al. 2007). During the last interglacial, when temperatures were slightly higher than today, global sea level was likely 4-6m higher than today (Jansen et al. 2007), and one sea level rise reconstruction yielded an average rate of 1.6m per century (Rohling et al. 2006). Rises in global sea level of more than 1m by the end of the century (and according to some authors, of several meters by that time (Hansen et al. 2007)), are therefore possible and would substantially increase various pressures on coastal areas.

3. Overall conclusions

Even though changes in the abundance and distribution of individual species can currently not be predicted with absolute certainty, a number of robust patterns emerge from the available studies and should be considered in conservation planning. These include:

- Accelerating climate change, resulting in increasing pressure on European biodiversity.
- Continued changes towards an earlier onset of lifecycles, especially in spring, for most species. As species respond differently, an increase in mismatches will imply an increasing threat to biodiversity.
- Changing migration patterns including no migration.
- Continued range expansions northwards and to higher altitudes in response to warmer winters and longer growing seasons.
- Potential range contractions in southern and central Europe caused by drought, depending on climate scenario.
- Range contractions also in the alpine and other mountain regions, with important changes in snow cover and frost/thaw periods.
- Rapid movement of climate zones from the south-west to the north-east.
- Movement of ecosystem types and habitats towards the north-east.
- Potential habitat transformation through forest die-back in south-western Europe, depending on climate change scenario.
- Decreasing habitat space for mountain species as tree lines move north and upslope.
- Increasing numbers of species threatened by extinction, in particular species with low dispersal capability and small ranges, the long distant migrants and habitat specialists.
- Mediterranean mountain species under particular pressure.
- Complex interactions between pressures on biodiversity.

Because of these changes, conservation goals will have to be re-defined to account for the “naturally” occurring changes, and to prioritize those ecosystems and species most at risk. Monitoring schemes are needed to evaluate the rate and direction these changes are taking. Dispersal corridors should be created from the south-west to the north-east. Models used for projecting changes in species abundance and distribution should be further developed to account more explicitly for the set of major processes involved, in particular changes in land use, migration processes, and competition between species. New conservation planning tools for selecting, managing and monitoring protected areas should be further developed.

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