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**Climate Change Impacts
on European Amphibians and Reptiles**

Report prepared by

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EXECUTIVE SUMMARY

There is mounting empirical evidence that climate change is already having various impacts on different aspects of the ecology of organisms, including amphibians and reptiles. Long-term studies on European amphibians and reptiles show already a tendency to earlier breeding in many species. Also, the decline of some species has been linked to changed climatic conditions.

Most aspects of the life of amphibians and reptiles critically depend on temperature and water. While reptiles have developed adaptations to cope with water scarcity, all European amphibians require moist habitats and, with few exceptions, open water for reproduction. Species will become threatened by climate change particularly in regions where water and humid habitats are already scarce and expected to become even drier. As wetland habitats disappear, aquatic and semi-aquatic species will suffer declines.

Warm winter conditions may deplete the energy resources of hibernating species in the Temperate Zone and individuals may be killed by late frosts. In the Mediterranean, excessive heat and prolonged droughts can have negative effects on amphibians and reptiles and species adapted to cool environments may experience lethal temperatures.

The principal response of species to climate change is either a range shift or in-situ adaptation by evolutionary change. Apart from marine turtles, reptiles and amphibians have a too low dispersal capacity to follow the expected rapid changes, especially in the highly fragmented European landscapes. In-situ adaptation requires large populations – beyond the size of most amphibian and reptile populations in modern landscapes.

Climate envelope modelling and the assessment of the climate sensitivity of amphibians and reptiles clearly show that climate change impacts will considerably differ among species and regions. Overall, amphibians are expected to suffer more than reptiles which are better adapted to dry environments.

Proposed actions:

1. Take early action on the species listed in Table 4 of the attached report, including through species-specific climate change mitigation plans. Species in Table 4 are expected to be the most affected ones. They comprise primarily
 - Amphibians from dry Mediterranean regions (especially in Spain, Western France, and Italy);
 - Amphibians requiring cool environments
 - For reptiles, projected losses are also highest in areas with high temperatures and major reductions in precipitation (Spain, Italy, the Balkans, and Greece);
 - Island endemics, such as *Alytes muletensis* (Balearic Midwife Toad), the lizards *Algyroides fitzingeri* (Pygmy Algyroides), *Lacerta bedriagae* (Bedriaga's Rock Lizard), *Podarcis tiliguerta* (Tyrrhenian Wall Lizard), and *Gallotia simonyi* (El Hierro Giant Lizard), and the snake *Macrovipera schweizeri* (Cyclade Blunt-nosed Viper) are predicted to become the most affected species, together with *Phyllodactylus europaeus* (European Leaf-toed gecko);
 - In Central and Northern Europe, early breeding amphibian, i.e., primarily brown frogs (*Rana arvalis*, *Rana dalmatina*, *Rana temporaria*) and the common toad (*Bufo bufo*) may be placed at increasing risk due to late frosts, less snow cover, and warmer winter temperatures.
2. Highly sensitive species should be monitored as indicators of climate change.
3. Facilitate in-situ adaptation and natural range shifts by redoubling efforts to maintain or restore large intact habitats and large-scale connectivity.
4. Countries with breeding populations of sea turtles and endemic island taxa potentially threatened by sea level rise should gather data and undertake studies to improve knowledge on climate change impacts on endemic island species.
5. Mediterranean countries should assess the reduction of permanent wetlands and rivers by the combined effects of land use and climate change to better understand impacts on amphibian species.

6. Further research should be undertaken on the potential impacts of climate change on amphibian and reptile species.

1. INTRODUCTION

In the 1970s, environmental biologists and atmospheric scientists predicted that human-induced global warming could potentially affect the biology of plants, animals, and microorganisms (Wyman 1992). An increasing body of evidence provides a picture of a warming world accompanied by other significant climate changes (IPCC 2007). The projected rate of global mean warming exceeds by at least one order of magnitude the rate of global warming during the most rapid large magnitude events of the Quaternary period. During the Quaternary geologic period, the difference in global mean temperature between glacial and interglacial stages is comparable in magnitude to the upper end of the range of possible global mean warming whilst at the lower end of this range the warming results in conditions at least as warm as any during the past two million years. Regional climate modelling studies (e.g. Räisänen et al. 2004) indicate that Europe can be expected to experience warming of a greater magnitude than the global mean warming.

The principle response of species to climate change is a spatial response. Their geographical range changes as the area shifts in which their climatic tolerances and/or requirements are met (e.g., Walther et al. 2002). This spatial response may be complemented by adaptive genetic responses of the population at any given locality as the climatic conditions at that locality change (Bradshaw & McNeilly 1991). Species unable to achieve a sufficient spatial and/or adaptive response will suffer extinction, at least regionally and in some cases globally. Thus species with low dispersal potential and species that have narrow climatic requirements are likely to suffer most strongly from climate change. More subtle responses, such as changes in the phenology of important biological traits (Blaustein et al. 2003) or changes in community composition due to altered hydroperiods (Semlitsch 2003), may precede evolutionary adaptation or shifts in geographic distribution.

There is mounting empirical evidence that climate change is already having various impacts on different aspects of the ecology of organisms (e.g., Walther et al. 2002, Parmeson & Yohe 2003), including amphibians and reptiles (Boone et al. 2003). Likewise, modelling the climatic requirements of species and matching these 'climate envelopes' with projected future climatic conditions show that many species face major range shifts and a substantial threat of eventual extinction in Europe, and in some cases of global extinction (Thuiller 2003, Araujo et al. 2006).

Amphibians and reptiles are particularly suitable model groups to compare in terms of climate change sensitivity. Because temperature and moisture affect multiple aspects of amphibian and reptile biology (Heatwole 1976, Duellman & Trueb 1986, Bradshaw 1987, Angilletta et al. 2002), they should be extremely vulnerable to the effects of climatic change and thus excellent indicators for impacts on biodiversity (Blaustein et al. 2001, Carey & Alexander 2003). With few exceptions, they share very low dispersal capacity (Fog 1993, Blaustein et al. 1994, Settele et al. 1996, Semlitsch 2003, Smith & Green 2005, Jehle & Sinsch 2007) and thus have limited scope to track climate change by dispersal. However, in response to drying climates in the Carbon (290-333 Mio. years ago) reptiles evolved from their amphibian progenitors a range of adaptations for coping with water scarcity, especially eggs with eggshells, scaled skin with lipids in the epidermal layer, and the excretion of urea as an adaptation against evaporation (Lillywhite & Maderson 1982, Bradshaw 1987, Packard & Packard 1988). In contrast, all amphibians critically depend on water availability and humid environments for most of their life (Duellman & Trueb 1986). As a consequence, amphibians show their greatest diversity in wet tropical regions (Duellman 1999) while hot and dry regions are particularly rich in terms of reptile species diversity (Pianka 1986). One may thus predict that reptiles in general should be less sensitive to climate change than amphibians and, in Mediterranean and Temperate regions, may even benefit from it.

However a simple analysis of the direct impacts of climate change on a species may be misleading and it is important to recognise the complexity of effects of climate change on different aspects of the biology of the animal and its environment. Some of these effects may counteract each other, others may act synergistically. These factors may include (Gent SEH-CC, comm. by lett.):

- Behavioural changes and other such adaptations to climate change,

- Effects of predator, prey, and competitor species,
- Opportunities for non-native species,
- Disease: prevalence, impact, and spread, and impacts on other species that may help carry, spread or act as a reservoir for disease,
- Effects on the habitats – these may be expected to alter,
- Micro-climatic effects: though the ‘gross climate’ changes will affect the ‘climate space’, impacts may largely occur at the micro-climate level. Changes in use of habitat and landscape features (e.g., moving from southern to northern aspects) or difference in vegetation height or structure, perhaps due to change in plant communities or differing management regimes (e.g., removal of trees, change in grazing regimes) may exacerbate or mitigate the impacts of climate change,
- The underlying geology, or other such factors, may impact on the ability or scope for animals to change environment (different pH, different water retention abilities, etc); physical features may alter the effects of climate (proximity to the coast, change in the Gulf Stream, features that shelter from prevailing winds), effect of coastal erosion or sea level rise, and the presence of physical barriers may affect the ability to disperse and the availability of space to move into. The physical structure of the land will therefore provide many and varied impacts and effects.

While these aspects are important, even the likely direct effects have not yet been reviewed in depth and it is impossible to model all of these aspects for species at a large scale and as such the analyses in this paper are simplified. Nonetheless, a wide range of factors would need to be considered when considering the impacts on species at different geographic scales and may implicate conservation policies and management practices.

In this report, we first assess the relationships between climatic factors and phenology, demographic changes, and habitat dependency of European amphibians and reptiles that ultimately translate into different viability of populations and shifts in species range. In this assessment we focus on temperature and water availability. We further evaluate the evidence that the current ranges of amphibians and reptiles in Europe are limited by climatic factors and assess the knowledge about adaptive responses to climatic factors. We review evidence that European amphibians and reptiles are already affected by climate change. We then develop and evaluate new climate envelope models for all European species with a sufficiently broad distribution. Based on the combined evidence from these assessments, we finally derive recommendations for the conservation of European amphibians and reptiles in the face of climate change, accepting the uncertainty of the modelling and the need for a simplified analysis to address the issues at a high level.

2 SENSITIVITY OF AMPHIBIANS AND REPTILES TO CLIMATIC FACTORS

Because many facets of the biology and ecology of amphibians and reptiles are tightly related to temperature and moisture, their phenology, demography, and habitat choice is sensitive to climatic factors (Duellman & Trueb 1986, Bradshaw 1987, Angilletta et al. 2002). This dependency should ultimately translate into a latitudinal and altitudinal distribution that is strongly influenced by and will change with climatic conditions – an assumption that is generally made in models that predict climate change impacts on organisms (Davis et al. 1998) but which is rarely assessed when more than a few species are modelled at the same time. In this section, we evaluate the differential sensitivity of amphibians and reptiles to climate change in respect to these aspects of their life.

2.1 Phenology

Shifts in the timing of species’ life cycles have been noted in locations from around the globe for all major groups of animals and plants (Parmesan & Yohe 2003, Root et al. 2003). Changes in phenology are likely to impact the reproductive success of individuals and hence the dynamics and persistence of populations, the structure and diversity of communities, and the functioning of entire ecosystems.

In reptiles and amphibians the timing of the seasonal activities hibernation, aestivation, and breeding are tightly related to climatic conditions (e.g., Reading 1998, Tryjanowski et al. 2003, Chadwick et

al. 2006, Malkmus 2008). This is particularly pronounced at higher latitudes with their distinct seasonal difference between summer and winter temperatures. In Europe, amphibians and reptiles hibernate during winter except in the Mediterranean zone; for reptiles the exceptions are only thermally favourable habitats (Malkmus 2004) and the warmer parts of the Mediterranean (see Böhme et al. 1981-1999, Fritz (2001, 2005), and Joger & Stümpel (2005) for species specific data). Increased risks associated with climate change are posed by fixed endogenous rhythms of hibernation and feeding that can lead to depletion of energy stores in mild winters and concomitantly to reduced survival and/or reproduction and as a consequence to a decline in population size as suggested by Reading (2007) for the common toad (*Bufo bufo*). In the laboratory, Jørgensen (1986) showed such an effect for *B. bufo*. Other studies also showed that spermatogenesis and ovulation are governed by endogenous rhythms in amphibians (e.g., van Oordt 1960, Juszczak & Zamachowski 1965). However, these endogenous rhythms usually are not fixed, rather they are modified by environmental factors, including in the common toad (e.g., Reading 1998). Of the 17 amphibian species, for which Blaustein et al. (2003) evaluated long-term studies, approx. 50% showed trends of earlier breeding as a response to increased temperatures. In contrast, almost all long-term studies of European species found such a trend (see section 3 for details). The reason for the variation among species still remains unclear but may be related to the degree of climate change that took place in the study locations and statistical power of the analyses.

A further risk that increases with climate change is reduced snow cover in winter that could lead to higher mortality during cold spells. In amphibians, the critical thermal limits decrease with the latitude of the most northern occurrence (Brattstrom 1968) but only few species of reptiles and amphibians are freeze-tolerant, e.g. the viviparous lizard (*Zootoca vivipara*) (Voituron et al. 2002), the Siberian newt (*Salamandrella keyserlingii*) (Kuzmin & Maslova 2003), eggs of some North American turtles (Kenneth et al. 1988), and the North American wood frog (*Rana sylvatica*) (Layne & First 1991) – see Layne & Lee (1995) for adaptations of frogs to survive freezing and Spellerberg (1973, 1976) for critical thermal limits in reptiles (including European species). Utsch et al. (1999) reviewed critical thermal minimum temperatures for tadpoles, but his table contains no European species. While it is known that cold winters can affect negatively the survival of adult frogs (e.g., edible frog – *R. esculenta*, pool frog – *R. lessonae*: Anholt et al. 2003) and mass mortality in hibernacula followed by population declines have been observed, e.g. for Siberian wood frog (*R. amurensis*) (Kuzmin & Maslova 2003) and Dybowski's frog (*R. dybowskii*) (Maslova 2000), data on the effects of reduced snow cover are largely lacking due to study design difficulties. However, an unusual sharp decline of the adder (*Vipera berus*) occurred throughout Germany in winter 2002/2003, presumably caused by an unusually warm but extremely wet winter forcing the snakes to hibernate at shallow depth and late deep frosts (Podloucky et al. 2005).

As an additional risk, climate change increases the likeliness of late frosts in Central Europe (Hänninen 1991, Walther et al. 2002), which can have fatal consequences for early breeding amphibians (Costanzo & Lee 1997) and hibernating reptiles. Spawn killed by late frost have been observed for agile frog (*R. dalmatina*), the grass frog (*R. temporaria*) (Henle, unpubl., Gamer SEH-CC, comm. by letter, Podloucky SHE-CC, pers. comm.), the moor frog (*R. arvalis*), and the palmate newt (*Triturus helveticus*) (Podloucky, pers. comm.). Likewise, reptiles that were active early in the year in fine warm weather but found dead during a sudden period of cold weather, or specimens trapped in late snow have been repeatedly discovered, e.g., the sand lizard (*Lacerta agilis*), *Z. vivipara* (Spelelberg 1976, Henle 1998), smooth snake (*Coronella austriaca*) (Spellerberg 1976), and *V. berus* (Bischoff 1972), and amphibians, e.g. *R. amurensis* (Kuzmin & Maslova 2003) and *R. dalmatina* (Henle unpubl.), trapped in late snow or have been recorded but no data are available to quantify this risk.

Like hibernation aestivation may be affected by climate change with extended droughts forcing prolonged aestivation. Aestivation occurs in amphibians and reptiles living in regions characterized by hot and dry summers, primarily in arid and semiarid zones (Heatwole 1976, Duellman & Trueb 1986, Bradshaw 1987). European species do not fully aestivate but adult and subadult amphibians may be inactive during stressful xeric summer conditions (Andreone et al. 1990, Jakob et al. 2003b, see also Grossenbacher & Thiesmeier 1999, 2003 for European urodeles). Most reptile species in Mediterranean climates only shift their daily activity to the cooler parts of the day and snakes may even become nocturnal (Henle 1989; snub-nosed viper – *V. latastei* Brito 2003; this also applies to most other Euro-

pean species: Böhme 1981-1999, Fritz 2001, 2005, Joger & Stümpel 2005). Moreover, amphibians and reptiles are well adapted to survive long periods of aestivation (Duellman & Trueb 1986, Bradshaw 1987, Ultsch 1989, Warburg 1997).

The phenology of breeding also is usually tightly related to climatic factors. Egg deposition in reptiles of the Temperate Zone may start a month earlier in years with a warm compared to a cold spring (e.g., *L. agilis*: Olsson & Shine 1997). Even in Mediterranean species, weather conditions may shift hatching time by a month (e.g., Schreiber's green lizard – *L. schreiberi*: Marco & Perez-Mellado 1998). Rainfall is especially important for the initiation of breeding in amphibians (Blaustein et al. 2001, Carey et al. 2003). In Mediterranean habitats, adult amphibians enter temporary ponds predominantly in autumn after the onset of rains (Diaz-Paniagua 1992, Cvetkovic et al. 1996, Kyriakopoulou-Sklavounou 2000, Jakob et al. 2003a) but in Temperate regions both rainfall and temperature are important (reviewed for Central European species in Günther 1996) and low temperatures may inhibit clutch deposition (e.g., yellow-bellied toad – *Bombina variegata*: Niekisch 1995). For montane species it is debatable whether initiation of breeding is driven more by the timing of snowmelt than by temperature (see Corn 2003 versus Blaustein et al. 2001). Rainfall can also synchronize egg deposition in reptiles in arid and semiarid regions (Henle 1990a) and lead to an increased activity (Henle 1990b).

Amphibian species differ in their flexibility regarding onset and duration of larval development (e.g., Diaz-Paniagua 1988, 1992, Jakob et al. 2003; see also Table 1). In Mediterranean regions, the earliest breeders show the largest interannual variability in the onset and duration of the larval season (Diaz-Paniagua 1992) but later-breeding species may still show considerable variability. In the study of Jakob et al. (2003), for example, not only the two early breeding species, the Iberian spadefoot toad (*Pelobates cultripes*) and the common parsley frog (*Pelodytes punctatus*), but also the late breeding Mediterranean tree frog (*Hyla meridionalis*) and Perez' frog (*R. perezii*) were flexible in the onset and duration of larval occurrence. In contrast, the marbled newt (*T. marmoratus*), *T. helveticus*, and the natterjack toad (*B. calamita*), all late breeding species, showed low inter-annual variability (Jakob et al. 2003a). Concomitantly, the former group had a more constant breeding success. For carnivorous larvae (in Europe only newts), Kuzmin & Meschersky (1987) and Jakob et al. (2003a) argued that the synchronisation with prey availability or diversity makes a change of breeding time nearly impossible. However, long-term studies in the UK show an earlier breeding of newts in response to increased temperatures (Beebee 1995).

Flexibility may also differ intraspecifically (c.f. Table 1). Although some of these opposing observations may be due to different degrees of temperature increases, phenological flexibility may also differ strikingly within the same region as observed by Kuhn (1994, 2001) for *B. bufo*. In comparison to populations breeding in stable permanent ponds, which showed the typically low flexibility in breeding time, populations from a dynamic floodplain system exhibited a very pronounced intra- and interannual variability.

In Mediterranean regions an inter-annually fixed but early onset of reproduction in temporary ponds might be a compromise between the environmental constraints affecting breeding adults (fat storage after the xeric summer period before egg-laying) and the survival of larvae that is mainly affected by the length of the hydroperiod (Pechman et al. 1989, Jakob et al. 2003a). In drought years, the hydroperiod may be too short to allow completion of metamorphosis and no reproduction may occur when flooding is delayed (Jakob et al. 2003a). Malkmus (2006) reported that during the extreme drought of 2004/2005 in Portugal, which was the most extreme one within the last 50 years, most species of the Alentejo region were unable to complete larval development in temporary water bodies. Complete failure of metamorphosis due to early drying of ponds in extreme drought years (1991-1993 in Poland) has even been observed in pristine forest of Central Europe (Jedrzejska et al. 2003). As a consequence population size may exhibit dramatic fluctuations (Pechman et al. 1989, Semlitsch 2003). Longevity and iteroparity, which can differ considerably among and within amphibian species (e.g. Moravec 1993, Kuhn 1994; reviewed in Grossenbacher & Thiesmeier 1999, 2003 for most European urodeles; see Günther 1996 for Central European amphibians), should lower the demographic consequences of a year with total reproductive failure in harsh environments (Stearns 1976).

While intra- and interspecific differences in phenological shifts of breeding among species are still insufficiently understood, they are likely to differ among species and consequently change competitive larval interactions (reviewed by Alford 1999 and Semlitsch 2003) and predator-prey relation-

ships (reviewed by Alford 1999; see Heusser et al. 2002 for studies on Central European species) and thus most likely will have major consequences for community composition. As a more subtle effect, larval development may proceed more rapidly, with metamorphosis occurring at smaller size (Reading & Clarke 1999, Jakob et al. 2002), which reduces survival to sexual maturity, adult size, and thus fecundity (Semlitsch et al. 1988). Ultimately these climate dependent phenological changes will translate into changed local and regional abundances and persistence of species.

In summary, there is strong evidence that changes in phenology can have major impacts on survival and reproductive success and hence the dynamics and persistence of populations. Positive effects of changes in phenology seem to have not been documented so far for European amphibians or reptiles but this may be due to a lack of attention. With respect to phenology, amphibians and reptiles should be equally at risk in regions with a reduced likelihood of snow cover but longer spells with temperatures well below freezing and in regions with increased likelihood of late severe frosts. High-risk regions for amphibians are further regions, for which the probability of droughts will increase with climate change. In contrast, for reptiles of the latter regions climate change should pose risks only for species that do not aestivate.

2.2 Demographic effects

Effects of climate change may also become manifest on the population and metapopulation level, which ultimately may result in changes in distribution patterns. Predicted increases in temperatures usually should benefit growth in reptiles but may have either positive or negative effects on growth in amphibians. Modified growth rates usually will translate into changes in fecundity, because in many species of reptiles and amphibians fecundity is positively correlated with female body size (e.g., Reading 1986, Henle 1988, Kuhn 1994, King 2000).

In reptiles growth is temperature dependent (Andrews 1982) and is reduced or ceases during the cooler parts of the year. This is the case even in most Mediterranean climates but species-specific differences exist and diurnal species are more likely to continue growth throughout the cooler parts of the year than nocturnal species (Henle 1988, 1990a,b, see Böhme 1981-1999, Joger & Stümpel 2005, and Fritz 2001, 2005 for species specific data on European species). Regions, in which future temperatures will exceed species-specific thermal limits for activity (reviewed by Porter & Gates 1969, Huey 1982), may be an exception. However, many species respond to excessive heat by shifting their activity to the cooler parts of a day or season, and snakes may even become nocturnal (e.g., Brito 2003; see Böhme 1981-1999, Joger & Stümpel 2005, and Fritz 2001, 2005 for species specific data on European species). The extent of these shifts differs among species, and overall activity may not be reduced (Bradshaw 1987, Henle 1989). To predict whether increased temperatures will negatively impact reptiles, these species specific shifts in activity need to be accounted for. Such quantitative analyses still need to be done and summarized for European species.

For amphibians, increased temperatures may have either positive or negative effects on growth. Lillywhite et al. (1973) suggested that growth in postmetamorphic amphibians is temperature dependent (but few studies on natural populations are available). Therefore, they should benefit from predicted increases in temperature during the active season unless temperature surpasses critical thermal limits (see Brattstrom 1963, 1968 and Claussen (1977) for data on critical thermal limits – primarily for North American species). However, there may be negative impacts of increased temperature on hibernating amphibians, since the metabolic rate will be higher in warm winters than in cold winters and consequently the energy reserves will be utilised more rapidly. Moreover, following mild winters, feeding may be reduced by endogenous physiological rhythms. Such an endogenous rhythm for feeding has been shown for *B. bufo* by preventing individuals from hibernating in laboratory studies (Jørgensen 1986). Less energy reserves in body condition of female toads on emergence from hibernation and reduced feeding can act synergistically to cause a decline of female survival rates and body size of females breeding for the first time, which will lead to reduced reproductive output and ultimately population decline. Reading (2007) suggested these pathways as explanatory mechanisms for the decline of a population of *B. bufo* in southern England. To know whether such processes act more generally in amphibians, it is important to balance higher energy needs due to a warmer winter against the gains due to a shorter hibernation period and a seasonally earlier feeding after the earlier reproduction. Such assessments are currently not available. While the same negative processes could operate in rep-

tiles, since hibernation may also be energetically demanding for reptiles and sometimes leads to high mortality (e.g., Gregory 1982; Ultsch 1989, Prior & Shilton 1996), this possibility apparently has not yet been considered in the climate change context.

In any case, such negative effects of increased temperatures should affect only species, in which the length of hibernation and the initiation of feeding are determined primarily by endogenous rhythms. In amphibians, reproduction has an endogenous rhythm but this rhythm usually is modified by external environmental factors (van Oordt 1960). The same applies for the reproductive cycle of male lizards (Fischer 1974). The tendency of many lizard and snake species to be active during mild winter days (e.g. Henle 1998, Malkmus 2004, 2008; see Böhme 1981-1999, Joger & Stümpel 2005, and Fritz 2001, 2005 for species specific data on European species) also argues against endogenously fixed rhythms of hibernation for the majority of reptile species. In addition, shifts in the breeding phenology in response to increased temperatures, as observed in many amphibian and reptile species of the Temperate Zone (see previous section), will counteract these negative effects and could even override them. Finally, these negative effects could operate only in the Temperate Zone (i.e., Central and Northern Europe) and montane regions, because amphibians do not hibernate in lower altitudes in southern Europe (e.g., Diaz-Paniagua 1992, Jakob et al. 2002; see also Grossenbacher & Thiesmeier 1981-2005 for European urodeles).

Temperature may also exert effects upon embryonic/larval development and survival and thus on reproductive success. Temperature strongly accelerates embryonic and larval development in amphibians and reptiles (e.g., Rykena 1988, Ultsch et al. 1999, Andrews et al. 1999) but the effects may differ within and between amphibians and reptiles. In reptiles of more northern latitudes and higher altitudes primarily positive effects of increased temperatures are expected because the risk that the period with temperatures suitable for embryonic development and successful hatching are too short and the risk that hatching takes place too late for juveniles to gain sufficient energy stores for successful hibernation will be reduced (Bauwens 1981, Richards 2002, Arribas & Galán 2005). For example, in coastal Sweden the oviposition dates of *L. agilis* shifted between years depending on weather (basking opportunities). Early clutches were larger than later clutches, had higher hatching success, and tended to have higher post-hatching survival rates (Olsson & Shine 1997). Also offspring from early clutches were larger than later hatchlings, which in reptiles often translates into a survival advantage (e.g., Henle 1990a) and may explain the higher post-hatching survival. Similarly, hatchlings of green lizards (*L. viridis*) that developed in cool summers at the Northern limit of their distribution are smaller, lighter, and less lively and have lower winter survival than those that developed in warmer, sunnier summers (Elbing 2000).

In amphibians, the likely effects of increased temperatures on reproductive success and larval development are more complex than in reptiles and both positive and negative effects may prevail. Temperature determines larval body size, rate of development, and the length of the larval period, which in turn are directly related to survival and reproductive success (e.g., Reques & Tejedo 1995, Jakob et al. 2002; reviewed by Ultsch et al. 1999). At extreme elevations and latitudes amphibians should generally benefit from increased temperatures like reptiles, because temperatures that prevent metamorphosis may be a factor limiting the geographic range of a species (Smith-Gill & Berven 1979) and the risk that larvae fail to metamorphose before winter and die (Pflüger 1883) is reduced. At less extreme latitudes and altitudes an increased temperature may have opposing effects on larval development. Larval period is generally longer and amphibian larvae are generally larger when raised at low temperature (e.g. Reques & Tejedo 1995, Jakob et al. 1992, Ultsch et al. 1999). For example, in years, in which breeding took place earlier in *B. bufo* longer larval developmental times have been observed (Reading & Clark 1999). Slower differentiation rates (Ultsch et al. 1999) and larval food availability may be the reason for the longer developmental times. Algae are the food basis of tadpoles of most species (Duellman & Trueb 1986) and algal growth depends on sunshine and day-length (Reynolds 1984). Besides slower development, there may be increased mortality when breeding earlier, as observed for example in *T. marmoratus* larvae (Martens 1984) but not in *B. bufo* (Reading & Clarke 1999). Even with increased mortality, a longer developmental time may not be a disadvantage because the larger body size at metamorphosis translates into higher survival to sexual maturity and larger adult size and the latter in turn increases fecundity (e.g., Semlitsch et al. 1988, Kuhn 1994, Reading 1986, 2007).

Although increased temperatures due to climate change often will be beneficial for reproductive output and egg development in reptiles, there may also be negative effects. In species from hotter environments, soil temperature may be shifted beyond the upper limit of tolerance for egg development but this should be a problem only in regions, in which temperatures become hot throughout the year (i.e., not in Europe) or in species, in which timing of oviposition is endogenously fixed. Little is known about endogenous circannual rhythms in reptiles (Underwood 1992). Although in a few species studied in the laboratory there is an indication of an endogenous rhythm (Underwood 1992), most species are flexible and the timing of oviposition changes with weather conditions (e.g., *L. agilis*: Olsson & Shine 1997; green lizard – *L. viridis*: Elbing 2001). Whether soils may also become too dry for egg development under decreasing summer rainfall has not yet been assessed. Species with parchment-like eggshells would be those that could be affected by dry soils and this risk will be higher at the southern margin of the range of such species.

Another negative effect may be caused by the temperature-dependency of sex determination, which occurs within some but not all reptile lineages, e.g. turtles (Janzen 1994, Ewert et al. 2005) and geckos (Ciofi & Swingland 1997; *Tarentola*: Hielen 1992). Temperature dependent sex determination is not known in amphibians. In most cases, a preponderance of males is produced at high temperatures and females at low temperatures (Hielen 1992, Janzen 1994, Ewert et al. 2005) and this may have negative effects on the population, if climate change shifts the incubation temperature to male producing levels. Based on a recent model to predict sex ratio Booth (2006) concluded that sex ratios will be fairly robust to moderate global warming as long as eggs experience substantial daily cyclic fluctuations in incubation temperatures, which is usually the case, so that embryos are exposed to temperatures that inhibit embryonic development for part of the day.

Besides temperature rainfall can also be an important determinant of population processes. In northern regions, an increase in rainfall, as expected for Northern Europe (IPCC 2007), may reduce the number of days suitable for activity of reptiles and thus may have negative effects on their growth and/or reproduction. For example, in cool and wet summer, the development of eggs of the pond terrapin (*Emys orbicularis*) is retarded at the northern limit of their range in northern Germany and embryos die latest during winter (Schneeweiss et al. 1998). In xeric regions, reduced rainfall, as predicted for Mediterranean Europe (IPCC 2007), may also reduce growth rates both in amphibians and reptiles, since food availability (arthropods for the majority of species; plants in turtles and some desert lizards) is primarily driven by rainfall in xeric environments (Heatwole 1976, Abts 1987, Bradshaw 1987). Though we know no study on European Mediterranean species, a reduced growth rate and/or reproduction has been observed in the slow worm (*Anguis fragilis*), *L. agilis*, and *Z. vivipara* in western Europe during the dry summer of 1976 (Patterson 1983, Strijbosch & Creemers 1988). Effects on growth should be exacerbated for amphibians by expected fewer rain days, since their activity requires sufficient moisture.

Amphibians breeding in temporary ponds, the main type of breeding sites in many Mediterranean regions (Diaz-Paniagua 1990, Semlitsch 2003, Malkmus 2006), may further be impacted by a shortening of the hydroperiod of breeding sites and this could lead to complete reproductive failure in dry years (e.g. Pechmann et al. 1989, Semlitsch 2003; see Malkmus 2006 for Portuguese species). In line with these ideas, a positive correlation between rainfall in spring and abundance of amphibians has been documented for example in Mediterranean France (Jakob et al. 2003a) and in similar North American habitats (Pechman et al. 1989, Semlitsch et al. 1996). However, results of studies from less xeric regions are inconsistent. For example Jedrzejska et al. (2003) found a positive effect of rainfall during the census and the previous year on amphibian abundance (*B. bufo*, *R. arvalis*, *R. temporaria*) in Poland, whereas Meyer et al. (1998) observed no effect for three populations of *R. temporaria* in Switzerland. For a tree frog (*Hyla arborea*) population growth rate was negatively affected by rainfall with a time lag of two years (Pellet et al. 2006). Pellet et al. (2006) assumed that rainfall reduced larval food availability. These differences show that one must be cautious when predicting likely effects of modified rainfall due to climate change.

Dispersal is a key to understand and predict the potential of species to track shifting climatic space. It is also key to the dynamics of genetic diversity in time and space (Gaggiotti & Couvet 2004) and, therefore, to the potential of species to adapt genetically to environmental challenges (Barton & Whitlock 1997, Hewitt & Nichols 2005). In many animals, dispersal behaviour is plastic and sensitive

to environmental cues (Ferrière et al. 2000, Clobert et al. 2001, Hofmann et al. 2005). However, surprisingly little is known about whether and how animals with plastic dispersal respond to climate change and how climate-related change in dispersal behaviour might cascade to the level of population persistence and species distribution. Massot et al. (2008) discovered in *Z. vivipara* a relationship between temperature and juvenile dispersal. It was strongly negatively correlated with the temperature in June and positively correlated with the temperature in August, i.e., juvenile dispersal decreased with higher prenatal temperature and increased with higher postnatal temperature. Although it is unclear to which extent these positive and negative effects counteract each other, temperature increase may thus interfere with the functional integrity of metapopulations.

Though little is understood about the climatic conditions that favour dispersal due to the difficulties to observe it, studies on breeding migrations in amphibians (see Günther 1996 for Central European species) suggest that they may be triggered by warm and moist conditions (after rainfall). Cursorial observations indicate the same for reptiles in xeric environments (Henle unpubl.). Therefore, the expected reduction in rainfall and rainy days in Mediterranean regions and in summer in Central Europe should have negative effects on reptiles and especially amphibians, since they may desiccate rapidly (e.g., Henle 2006).

Even if dispersal is not negatively affected by climatic conditions, most amphibians and reptiles have limited scope to track climate change by dispersal. Apart from marine species, rafting (e.g. Houle 1998), and passive dispersal facilitated by humans (e.g., Henle & Klaver 1986, Hemmer et al. 1981) amphibians and reptiles have a very low dispersal capacity (Fog 1993, Settele et al. 1996, Smith & Green 2005, Jehle & Sinsch 2007, Semlitsch 2008). As a rule of thumb 400 m to < 2 km are frequently mentioned for amphibians (e.g., Blab 1986, Semlitsch 2003). A recent review of 90 species confirmed that most species migrate < 2 km, though rare cases of migration distances > 10 km have been observed (Smith & Green 2005; see also Jehle & Sinsch 2007 for Central European species). Species commensally with man, prominent in the trait in live animals, or targeted for re-introduction programs are the most likely ones to profit from passive dispersal facilitated by humans (intentional and unintentional). In Europe these are especially the wall gecko (*Tarentola mauritanica*) (e.g., Bruekers 2006), the Italian wall lizard *Podarcis sicula* (Henle & Klaver 1986), the wall lizard (*P. muralis*) (Schulte 2008), water frogs (subgenus *Pelophylax* of the genus *Rana*), and freshwater turtles [*E. orbicularis* and the introduced North American slider turtle (*Trachemys scripta*)] (Fritz 2001, Di Cerbo & Di Tizio 2006).

2.3 Dependency on humid habitats

Whereas reptiles have developed an amazing range of adaptations for coping with water scarcity (Lillywhite & Maderson 1982, Bradshaw 1987, Packard & Packard 1988) and thrive well in hot climates (Pianka 1986), all amphibians critically depend on humid environments for most of their life (Duellman & Trueb 1986, Warburg 1997) and have their highest diversity in the humid tropics (Duellman 1999). Notwithstanding, also a considerable number of reptile species is aquatic or semi-aquatic, noticeably among turtles and snakes. In Europe this applies to all turtles except of the family Testudinidae and all snakes of the genus *Natrix* (Böhme 1999, Fritz 2001, 2005). In addition, the Cyclade blunt-nosed viper (*Macroviperu schweizeri*) feeds predominantly on birds visiting open water for drinking (Nielsen et al. 1999) and *L. schreiberi* lives predominantly along the margin of rivers (Brito et al. 1996) though it does not directly depend on open water. Further species may be found in humid habitats, for example, *Z. vivipara* and *V. berus*, but neither depend on nor prefer them (Günther 1996). While reptiles usually do not require aquatic habitats for reproduction, rain may facilitate egg deposition by softening the soil (e.g., *P. muralis*: Schmidt-Loske 1995) and trigger egg deposition (e.g., in the Australian gecko *Gehyra variegata*: Henle, unpubl.) or hatching of eggs (Kenneth et al. 1993). Also, neither too wet nor too dry soils are a precondition for embryonic development, especially in species with a parchment-like eggshell (Packard & Packard 1988) and soil moisture may also be important for successful releasing of young of viviparous species, such as *Z. vivipara*, *V. berus*, and *V. ursinii* that are born wrapped in the transparent membrane (Cmobrnja-Isailovic SEH-CC, comm. by lett.).

In contrast to reptiles, only few amphibian species have developed physiological or behavioural adaptations that allow them to live in xeric habitats, such as long-term dormancy within impermeable

cocoons or dehydration tolerance combined with the secretion of a body coating having low permeability (Tyler 1989, Hillman 1980, Kobelt & Linsenmair 1992). Another adaptation is the accumulation of urea in the body fluids, which enables some amphibians to tolerate soil with low water potential, e.g. *Scaphiopus* spp. (Jørgensen 1997). None of the European species has evolved such special adaptations to highly xeric environments. They avoid desiccation by staying in water or at the edge of semipermanent water bodies and hide during the day under cover (Gamer SHE-CC, comm. by lett.). A few species, e.g. the green toad (*B. viridis*) and *B. calamita* (see Zavadil & Prikryl 2003), can live in comparably xeric habitats if rehydration opportunities are available. Therefore, amphibians in xeric regions are restricted to moist refugial habitats (Duellman 1999), e.g., golden striped salamanders (*Chioglossa lusitanica*) in Portugal (Alexandrino et al. 2007).

Aquatic habitats are essential for many European amphibians because of their reproduction. Though amphibians have evolved an amazing diversity of reproductive strategies (Crump 1974, Duellman & Trueb 1986), most temperate and Mediterranean species, including most European species, require open sources of water for reproduction: The main strategy is spawning of eggs into water, from which aquatic larvae hatch. The aquatic phase ends with metamorphosis to terrestrial juveniles. Only few European amphibians evolved a reproductive strategy that makes them independent of open water. Exceptions are the direct development of live-bearing salamanders [Alpine salamander (*Salamandra atra*), Lanza's salamander (*S. lanzai*), golden Alpine salamander (*S. aurorae*), Lycian salamander (*Mertensiella luschani*), some populations of *S. salamandra*, and imperial cave salamander (*Speleomantes imperialis sarrabusensis*) (Lanza & Leo 2001, Lanza et al. 2006, Sindaco et al. 2006)] and the deposition of eggs in humid, usually subterranean environments by cave salamanders of the genus *Speleomantes* (probably most taxa except of the subspecies *S. imperialis sarrabusensis*: Lanza et al. 2006, Sindaco et al. 2006). Young cave salamanders develop directly from these eggs without an aquatic phase (Lanza et al. 2006, Sindaco et al. 2006).

Cool microhabitats, in addition to moist environments, are essential for several amphibian species, throughout their life because they suffocate under higher temperatures (Boone et al. 2003). For example, 24 °C is close to the critical maximum temperature for *Speleomantes* spp. (reviewed by Lanza 1999, Lanza et al. 2006). Similarly, larvae of the olm (*Proteus anguinus*) require temperatures between 8-18°C for successful development and have their optimum at approx. 11°C (Durand & Delay 1980).

As wetland habitats disappear under climate change, amphibians and aquatic and semi-aquatic reptile species will suffer declines (Gibbons et al. 2000). Therefore, species should become threatened by climate change particularly in regions, in which water and humid habitats are already scarce and in which temporary ponds are a major breeding habitat for amphibians. In Europe, this will be especially the case in countries bordering the Mediterranean Sea (e.g., Diaz-Paniagua 1990, Blondel & Aronson 1999, Rabou et al. 2007) that are expected to become drier. This conforms to model predictions for anurans of the Iberian Peninsula (Araujo et al. 2006; see also section 3).

2.4 Latitudinal and altitudinal distribution

Climate envelope models to predict future likely distributions under different climate change scenarios assume that the altitudinal and latitudinal distribution of species is primarily limited by climatic factors. Evidence in support of such assumptions comes from four different sources: a) documented historical changes in distribution, b) shifts in altitudinal distribution with latitude, c) shifts in habitat use along latitudinal distributional gradients, and d) a limited ability of exothermal organisms to become independent of environmental temperatures. While substantial evidence has been amassed of the geographical expansion of the northern and uphill margins of populations of many different taxa in Britain and elsewhere (e.g., Hickling et al. 2006) and the thermal tolerance of amphibians for low temperatures increases with latitude (Brattstrom 1968), evidence of retreating southern and downhill limits has proved more elusive (Thomas et al. 2006). This may be because evidence has been sought at inappropriate scales; where populations are sparsely dispersed, substantial losses of fine scale and habitat occupancy may occur before any change is evident at coarse resolution.

For European amphibians or reptiles recent range shifts that clearly can be attributed to global warming have not yet been documented, which may be explained their limited active dispersal potential – see above) and several species may not be directly temperature limited at the southern margin of

their distribution (see below). However, there is strong evidence for warm-adapted Central European reptile species that they historically extended further north during the maximum post-glacial warming (during the Middle Ages) than their current northern limit (Böhme 1979).

Shifts in altitudinal distribution with latitude occur in many organisms and vegetation types (e.g., Rosenzweig 1995, Cabrera 1996, Morison & Morecroft 2006) and are a traditional indicator for inferring a climatic limitation of the range of species. Such shifts are also clearly evident in many European amphibians and reptile species (e.g., Meliadou & Troumbis 1997, Araujo et al. 2008). While most species are restricted to lower altitudes in the northern part of their distribution and reach higher altitudes with decreasing latitude, in Europe many species still occur at low altitudes in the southern part of their distribution (see Böhme 1981-1999, Fritz 2001, 2005, Grossenbacher & Thiesmeier 2001, 2005, and Joger & Stümpel 2005 for species specific descriptive summaries). This suggests that a general distributional limitation by hot temperatures cannot be assumed for them. For example, the lower altitudinal limit increases for the lizard *Z. vivipara* towards the south but in the Pannonian region it is found again at low altitudes - though this is a climatically differently adapted evolutionary significant unit (Böhme 1978). In contrast, the lizard *P. sicula* occurs at sea level even at the southern margin of its distribution (Henle & Klaver 1986). At the eastern Adriatic coast it is still spreading south and it has been successfully introduced to North Africa. Thus, it is more likely limited by geographic barriers (particularly the Mediterranean Sea) than by high temperatures. This renders climate envelope model predictions that the species will suffer range decline even under unlimited dispersal (Araujo et al. 2006) questionable. A careful analysis, for which species there may be such a limitation and for which not, is urgently required to better assess the reliability of climate envelope models and to better predict the likely impacts of climate change at the southern margin of distribution.

For many species, it is well-known that their habitat niche shifts latitudinally with narrower niches at the northern limit of the distribution and broader niches at the centre of their distribution (Kühnelt 1965), a relationship that became known as Kühnelt's principle (Böhme 1978). Central European lizard species also follow Kühnelt's principle (Böhme 1978) and require more open, warmer, and drier habitats in the northern part of their distribution but thrive well or even prefer dense vegetation further south [e.g., *L. agilis*: Märtens et al. 1997; see Böhme et al. 1981-1986 for descriptive species specific data that indicate similar relationships for other widespread species). This further supports the hypothesis that their northern distribution is climatically limited.

Apart from the marine leatherback turtle (*Dermochelys coriacea*), reptiles and amphibians primarily depend on external heat sources for activity and physiological processes. Lethal lower thermal limits have been identified for several species, including some European species (e.g., Rühmekorf 1958, Spellerberg 1973, 1976, Voituron et al. 2002, Kuzmin & Maslova 2003). In general, the latitudinal distribution seems to be mainly limited by embryonic thermal tolerance (Wells 2007) respectively the availability of thermally favoured micro-sites for egg development in reptiles (*L. agilis*: Strijbosch 1987, green lizards (*Lacerta sensu stricto*): Rykena 1987, *E. orbicularis*: Meeske 1997, Schneewis et al. 1998) but extinctions of established populations at northern latitudes may also occur in harsh winters (e.g., *P. sicula*: Henle & Klaver, 1986, Henle & Fritz 2007; *V. benus*: Podlocky et al. 2005). Similarly, the altitudinal distribution of amphibians is often limited by the availability of temporarily ice-free ponds, with an altitudinal limit of European species at about 2500 m (Wells 2007). In reptiles, the altitudinal limits are probably determined by temperature and, in humid soils, oxygen constraints (Navas 2002).

Lethal maximum temperatures also have been determined for many amphibian and reptile species (reviewed by Brattstrom 1968, Duellman & Trueb 1986, Pianka 1986, Ultsch et al. 1999; no overview for European species is known to us). Anuran tadpoles generally have high critical thermal maximum temperatures (> 36°C, even in species from northern latitudes) and rapidly acclimatize (Ultsch et al. 1999). Thus, it is unlikely that the distribution of anurans is limited by too hot water temperatures, even under strong climate change. However, early breeding amphibian species (e.g., *R. sylvatica*) are less tolerant to high temperatures than later breeding ones (Bachmann 1969) and this may contribute to the southern limit of their distribution. Also, some urodeles have low critical thermal maxima (e.g., cave salamanders, genus *Speleomantes*: Lanza et al. 2006) and their distribution may well be limited by the availability of cool microhabitats. However, the availability of cool microhabitats may coincide more with the availability of extensive subterranean habitats rather than latitude

itself. Moreover, many reptiles and amphibians shift their activity to the cooler parts of a day or season and snakes and amphibians are able to shift their activity to the night (Pianka 1986, Bradshaw 1987; see Böhme 1981-1999, Grossenbacher & Thiesmeier 1999, 2003, Fritz 2001, 2005, and Joger & Stümpel 2005 for European species). Thus, it largely remains unclear to which extent high temperatures limit amphibian and reptile species at the southern limit of their distribution. In fact, we know of no study that systematically analysed this for any Northern Hemisphere species.

In conclusion, while there is very strong evidence that most European amphibians and reptile species are climatically limited at the cooler end of their distribution, this is less clear for the warmer end, with the best evidence available for urodeles. For predicting the likely impacts of climate change on amphibians and reptiles with climate envelope models, this means that the range extension predicted for many species at the northern distributional limit are more reliable – if realistic dispersal is assumed – than range contraction at the southern limit (see section III). Notwithstanding, abiotic changes or different community composition caused by climate change may render habitats in the north unsuitable even if suitable temperatures occur (Garner SEH-CC, comm. by lett.).

2.5 Evolutionary response

Empirical evidence suggests that evolution is responsive to climate variation (Bone & Farres 2001) and for a wide range of taxa, including amphibians and reptiles, thermal performance varies within species' geographic ranges, suggesting both genetic variation in critical traits and localized evolution in response to climate factors (Conover & Schultz 1995, Gilchrist et al. 2004). For example, the frog *R. temporaria* shows temperature adaptation of larval, but not of embryonic, development to latitude in Sweden (Laugen et al 2003) and there is evidence that populations from higher altitudes develop faster than lowland populations (Angelier & Angelier 1968). Likewise, the lizard *Z. vivipara* shows different thermal adaptations and accompanying shifts in altitudinal distribution the Boreal and Alpine compared to the Pannonian biogeographic region (Böhme 1978).

Adaptive evolution in response to climate change depends on genetic variation. While numerous studies on the genetic variability of presumably neutral genetic markers have been undertaken in many organisms, including reptiles and amphibians, and used for inferences on adaptive potential, conservation based inferences from such studies are insufficient and even can be misleading (Lynch 1996). To evaluate the scope for adaptive responses knowledge about heritability of quantitative genetic traits is essential (Falconer 1989). Heritability measures the genetic component in trait variability. The heritability of quantitative traits that are responsive to climate change remains poorly understood for natural populations of amphibians and reptiles. In one of the few studies, Laugen et al. (2003) demonstrated heritability of embryonic survival, growth, and development, and hatching size in *R. temporaria* and differential adaptation to the temperature regime of the parent populations. Also, populations of the North American frog *R. sylvatica* have undergone localized evolution in thermal tolerance and development rate in response to altered temperature in their wetlands (Skelly & Freidenburg 2000, Freidenburg & Skelly 2004).

Adaptation to climate change may be rapid. Laboratory studies of insects show changes in thermal tolerance after as few as 10 generations (Good 1993). Preliminary assessments for a population of the Australian gecko *Gehyra variegata* that underwent major adaptive change in growth rate, body size, and sexual maturity in response to anthropogenically driven changes in food availability within a few decades (i.e. a few generations) (Henle, 1990 and unpubl.) indicates that in reptiles localized evolution can proceed fast as well.

The speed of adaptive change depends on the selection differential and heritability of the traits. Variability of quantitative genetic traits is in a dynamic equilibrium between drift and mutation on one site and selection as an opposing force. At low effective population sizes, drift and mutation may be the dominating evolutionary factors and populations may become fixed for inferior variants of a trait, whereas at high effective population sizes a selection mutation balance will develop (Lynch 1996).

Extrapolating from *Drosophila*, the only species for which the different processes of selection, mutation, and drift have been quantified, Lynch (1996) suggested that the speed of adaptive change reaches a maximum at an effective population size of 1000 individuals. The effective population (N_e) size usually is smaller than the census population size (N) because not all individuals contribute

equally to the number of offspring. For natural populations mean empirical estimates of the ratio of N_e/N are around 0.1, although they can reach 0.5 after correction for fluctuation of N (Frankham 1995, Vucetich et al. 1997). In amphibians, empirical N_e/N ratios range from 0.03 to 0.65 for frogs (*Rana spp.*), toads (*Bufo spp.*), and newts (*Triturus spp.*) (Hoffman et al. 2004, Jehle et al. 2005, Brede & Beebee 2006, Schmeller & Merilä 2007). To date, an astonishingly small number of studies on the effective population sizes of reptiles have been conducted (Hranitz & Braid 2000). Estimates range from 0.42-0.81 with a significant negative correlation with population size. These figures suggest that population sizes of 1,750-20,000 should have the highest potential to adapt to climate change for amphibians and of 1,250-2,500 for reptiles, i.e., figures that may be reached only in large intact high-quality habitats (compare e.g., Günther 1996).

2.6 Interaction of climate change with disease

Pounds et al. (2006), Wake (2007), and Bosch et al. (2007) strongly argued for a link between climate change, recent epidemic outbreaks of the chytrid fungus *Batrachochytrium dendrobatidis*, and global extinctions of amphibian populations. *B. dendrobatidis* caused mass mortality and dramatic declines in various amphibians in many parts of the world (Berger et al. 1999). It is now also present in at least nine European countries and made responsible for amphibian declines in Spain and Switzerland (Bosch et al. 2007, Gamer et al. 2005). Pounds et al. (2006), Wake (2007), and Bosch (2007) suggested that increasing temperatures since the last years may have benefited the epidemic outbreaks of *B. dendrobatidis* in two ways, by stressing individuals leading to dysfunctioning of the immune system and by reaching the temperature optimum of the fungus (17-25°C: Berger et al. 2004). Whereas in the Temperate Zone and in montane areas water temperatures in many breeding ponds are likely to get closer to this optimum, in Mediterranean Europe temperatures in temporary water bodies, i.e. the preferred breeding sites for the majority of species, will often exceed not only this optimum but will also exceed the lethal maximum for *B. dendrobatidis* at 30°C, which should clear amphibian larvae of their infections (Berger et al. 2004). However, mortality caused by *B. dendrobatidis* recently has been observed in Mediterranean locations at which the proposed thermal limit is exceeded (Gamer SEH-CC, comm. by lett.). This suggests that the interaction between climate change and the prevalence of *B. dendrobatidis* epidemics is a potentially very serious threat to European amphibians, especially from montane areas and the Temperate Zone. Surprisingly, there can be even positive indirect effects, as suggested by Bosch & Rincón (2008). More research is urgently needed to better understand the interactions between climate change in Europe and the differential susceptibility of amphibians to *B. dendrobatidis*.

Other pathogens threatening Europe's amphibians may benefit from climate change. For example, Ranavirus continues to extend its reach in the UK (Cunningham et al 1996) and mortality usually occurs in the summer months and is associated with warming temperatures (Garner SEH-CC, comm. by lett.).

3. EVIDENCE FOR EXISTING IMPACTS OF CLIMATE CHANGE ON EUROPEAN AMPHIBIANS AND REPTILES

The main evidence for existing impacts of climate change stems from long-term phenological monitoring studies. For European amphibians only few long-term phenological studies exist and we found no published one for reptiles, though some exist (eumon.ckff.si). We include in our assessment only studies that span more than 15 years, with the longest bridging the last 150 years (Terhivuo 1988). Almost all of these studies show a trend towards earlier breeding (Table 1), concomitantly with warmer temperatures in winter and early spring and appear to be associated with increased temperatures in winter and early spring (e.g. Tryjanowski et al 2003). Reading (1998) reported that *B. bufo* did not show a change towards earlier breeding. However, the relationship was almost significant and significantly more early breeding years occurred in the period 1989-1998 compared to 1980-1988 ($\chi^2=5.8$; $\alpha < 0.05$). Also, more recent data showed a statistically significant trend (Reading 2007). Though *R. temporaria* did not show a trend towards earlier breeding in the study of Beebee (1995), this can be explained because breeding time was strongly correlated with winter maximum temperature but this parameter did not change significantly during the 17 years of study. Notably, in warmer parts of the UK, the species has been observed to oviposit eggs already in December and January. These clutches

are either unfertilized or do not hatch, as temperatures are usually too cold for embryonic development, or sudden cold snaps kill the clutch (Garner, pers. comm.).

Table 1. Trends towards earlier breeding times of European amphibians within recent decades. 1: see text for discussion.

Country	Species	Breeding earlier?	Temperature increase?	References
UK	<i>Triturus cristatus</i>	Yes	Springtime	Beebee 1995
	<i>Triturus helveticus</i>	Yes	Springtime	Beebee 1995
	<i>Triturus vulgaris</i>	Yes	Springtime	Beebee 1995
	<i>Bufo bufo</i>	No ¹		Reading 1998
	<i>Bufo bufo</i>	Yes	Annual, winter, spring	Reading 2007
	<i>Bufo calamita</i>	Yes	Springtime	Beebee 1995
	<i>Rana esculenta</i>	No	No, for winter	Beebee 1995
	<i>Rana temporaria</i>	Yes	Springtime	Beebee 1995
	<i>Rana temporaria</i>	Yes		Scott et al. 2008
Finland	<i>Rana temporaria</i>	Yes	Spring	Terhivuo 1988
Poland	<i>Bufo bufo</i>	Yes	Winter & early spring	Tryjanowski et al. 2003
	<i>Rana temporaria</i>	Yes		

Very few long-term studies are available that could document responses of demography to climate change in European amphibians or reptiles. Reading (2007) observed that increased annual temperatures in southern England since 1983 have been paralleled by a decline in the body condition of female common toads (*B. bufo*), annual survival, and population size. He further discovered a relationship between the occurrence of mild winters and a reduction in female body size, resulting in fewer eggs being laid annually. He explained these observations with depletion of energy resources due to an endogenously fixed rhythm of hibernation and feeding as observed by Jørgensen (1986) under laboratory conditions. However, under natural conditions, endogenous rhythms are modified by climatic variables (van Oordt 1960, Reading 1998). Reading & Clarke (1999) further reported a significant positive correlation between the duration of the tadpole stage and tadpole mortality with spring temperature.

In mountain populations of *Z. vivipara* in Southern France Chamaillé et al. (2006) observed an increase in daily maximum temperatures in August during the last 18 years and in yearling snout-vent-length. As a result, adult female body size increased markedly, and, as fecundity is strongly dependent on female body size, clutch size and total reproductive output also increased. For one population where capture-recapture data were available, adult survival was positively related to May temperature. All fitness components investigated therefore responded positively to the increase in temperature, such that it might be concluded that the common lizard has been advantaged by the shift in temperature. Chamaillé et al. (2006) contrasted these short-term results with the long-term habitat-based prediction that these populations located close to mountain tops on the southern margin of the species range should be unable to cope with the alteration of their habitat. To achieve a better prediction of a species' persistence, one will probably need to combine both habitat and individual-based approaches.

Evidence for impacts of increased frequency of climate extremes was provided by Jedrzejewska et al. (2003) in Poland. They compared amphibian abundance from 1992-1994 with surveys for the same region between 1955 and 1998 and observed a dramatic decrease in amphibian abundance in 1993. The years 1991-1993 were the three driest years in recent decades for their study area, with spring-summer precipitation 35% lower than the long-term average. Similarly, Piha et al. (2007) observed a severe decline in the number of egg clutches deposited by *R. temporaria* in a Finnish study site after a major summer drought in 2002.

4. MODELLING EXPECTED RANGE SHIFTS

4.1 Predicted climatic change in Europe

The IPCC (2007) summarizes the expected climatic changes for Europe: Annual mean temperatures in Europe are likely to increase more than the global mean. The warming in Northern Europe is likely to be largest in winter and that in the Mediterranean area largest in summer. The lowest winter temperatures are likely to increase more than average winter temperature in Northern Europe, and the highest summer temperatures are likely to increase more than average summer temperature in Southern and Central Europe. Annual precipitation is very likely to increase in most of Northern Europe and decrease in most of the Mediterranean area. In Central Europe, precipitation is likely to increase in winter but decrease in summer. Extremes of daily precipitation are very likely to increase in Northern Europe. The annual number of precipitation days is very likely to decrease in the Mediterranean area. The risk of summer drought is likely to increase in Central Europe and in the Mediterranean area. Changes in precipitation may vary substantially on relatively small horizontal scales, particularly in areas of complex topography. Details of these variations are sensitive to changes in the atmospheric circulation with respect to the driving global model. In one, an increase in westerly flow from the Atlantic Ocean (caused by a large increase in the north-south pressure gradient) is accompanied by increases of up to 70% in annual precipitation over the Scandinavian mountains. In the other, with little change in the average pressure pattern, the increase is in the range of 0-20%. The duration of the snow season is very likely to shorten in all of Europe, and snow depth is likely to decrease in at least most of Europe.

The projected rate of global mean warming (IPCC 2007) exceeds by at least one order of magnitude the rate of global warming during the most rapid large magnitude events of the Quaternary period. Seasonal temperature extremes are projected to increase even more markedly, with annual minimum temperature increases of 6-18°C projected for Central and Eastern Europe and annual maximum temperature increases of 6-12°C projected for areas south of approx. 50° N. Annual precipitation is projected to increase in the north but to decrease in the south, with greatest changes in the summer in the south (June-July-August precipitation projected to decrease by >40% south of 40° N) and in the winter in the north (December-January-February precipitation projected to increase by 20-50% north of 40° N). Such changes can be placed in context by comparing them to the present spatial gradients or to past climatic changes. During the Quaternary geological period, the difference in global mean temperature between glacial and interglacial stages is comparable in magnitude to the upper end of the range of possible global mean warming projected by the IPCC in their Fourth Assessment Report, whilst at the lower end of this range the warming results in conditions at least as warm as any during the past two million years.

4.2 Species distribution modelling

A detailed description of the modelling approach is given in Araújo et al. (2006). Species distribution data were based on the *Atlas of Amphibians and Reptiles of Europe* (Gasc et al. 1997) covering 143 amphibian and reptile species. Because of likely incorrect models when occurrences are too low, 35 species that have less than 20 records were excluded and 42 amphibian and 66 reptile species remained. Species distribution data were based on a 50 km Universal Transverse Mercator (UTM) grid and covered Europe except most of the Eastern European countries where recording effort was less uniform and intensive.

A set of climatic variables was used to relate species distribution data to climatic conditions. Climatic variables were chosen to reflect two primary properties of the climate – temperature and water – that have known roles in imposing constraints upon amphibian species distributions as a result of widely shared physiological limitations (reviewed in section 2). Monthly, interpolated climate data, originally provided at a 10' grid resolution (New et al. 2000, Mitchell et al. 2004), were aggregated to the UTM grid and comprised mean values of five different climate parameters for the period 1961-1991: annual temperature (°C), mean temperature of the coldest month (°C), mean temperature of the warmest month (°C), mean annual summed precipitation (mm), and mean sum of precipitation between July and September (mm).

Species distributions across Europe were modelled using generalised linear models (GLM). Details on model parameterization are provided by Thuiller (2003). Models were calibrated on a 70% random sample of the observed data and predictive accuracy evaluated on the remaining 30% of the data. Agreements between observed species presences and absences and projected distributions were evaluated by the area under curve (AUC) of the receiver operating characteristic (ROC) curve and Cohen's Kappa. Thresholds for calculating presence-absence projections were obtained by a maximizing Kappa approach (Manel et al. 2001).

The resulting ecological niche models were then projected to four different climate-change scenarios for 2050 on a 10' grid. The scenarios were the IPCC SRES climate-change scenarios A1, A2, B1, and B2 (IPCC 2007) based on the coupled Atmosphere-Ocean General Circulation Model HadCM3 (Hadley Centre for Climate Prediction and Research's General Circulation Model) averaged for the period of 2020-2050. These scenarios reflect different assumptions about demographic changes, socio-economic, and technological development (Nakicenovic & Swart 2000) and range from fossil-fuel intensive to alternative futures involving rapid adoption of new technologies while differing in their consequences on future climatic conditions.

It is important to notice that both modelled current and future species distributions data cannot be interpreted as actual distributions. They represent the distributions of suitable climate space for species at a 'macro-level'. Actual distribution will be affected by many factors (e.g. availability of habitat, suitability of habitats following climate change, geology, micro-climatic variation, presence of food, impact of competitors, etc) (Gent, SEH-CC, comm. by lett.). This is also highly related to the topic of dispersal since new suitable climate spaces have to be reached by the species. We initially made two extreme assumptions about dispersal: unlimited dispersal, such that the entire projected climate space denotes the actual future distribution; and no dispersal, in which the future distribution results from the overlap between current and future niche space. Given that for most terrestrial amphibians and reptiles dispersal is low (between years in a range of a few hundred meters to a few kilometers (see section 2.2), the latter assumption is the more realistic one, especially given the relatively short time period for projections and the rather coarse resolution of the 10' grid (ca 10 x 16 km²). In addition, natural barriers (mountains, sea, rivers etc.) and current levels of habitat fragmentation (Amler 1999, Jaeger 2002, Baier et al. 2006) may reduce or totally impede the already low dispersal ability making the assumption of no dispersal realistic. While unlimited dispersal allows the identification of suitable areas for human-aided dispersal, human-aided dispersal is a highly sensitive political issue that, if considered, needs to be carefully evaluated on a case-by-case basis using all available information, not only modeling (see discussion in chapter V). Therefore, we do not consider the "unlimited dispersal" scenarios in this report. Model results will be made available for such case-by-case assessments on request.

To assess the impact of climate change on particular species we assigned them to five classes according to relative changes in their range size: 0 (no reaction or increasing range), 1 (decrease between 0-25%), 2 (decrease between 25-50%), 3 (decrease between 50-75%), 4 (decrease between 75-100%). Category 1 corresponds to just below a 1% annual decrease, a decrease that in several biodiversity policies is used as a threshold between unfavorable-bad conservation status (e.g., the reporting guidelines of the European Habitats Directive). Below, we summarize the results of the climate envelope models. Maps on current and predicted future suitable climate space under the assumptions of no dispersal respectively full dispersal are available from the authors upon request.

4.3 Expected distributional changes

4.3.1 Overall richness – Amphibians

As expected by their limited tolerance to low temperatures, the current richness of amphibian species is lowest in the northern parts of Europe (Scandinavia), the UK, and the mountains (Pyrenees, Alps, Carpathians) while their preference for moist conditions generates hotspots of amphibian richness in Central Europe, Western France, and Northern Italy (Fig. 1a). Projected changes in the course of climate change indicate that even under a minimum change scenario (B1) losses of suitable climate space will be severe (Fig. 2a). Greater losses of amphibian species will occur predominantly in warm and dry areas (see Araújo et al. 2006) in Southern Europe, especially in Spain, Western France, and Italy, which are projected to get disproportionately warmer and dryer during summer (Fig. 2a). The areas of severe losses will increase with higher levels of climate change (Fig. 2b). These changes will

determine future species richness. Besides effects on Southern Europe, also the Central European hot-spots of amphibian richness will be seriously affected (Fig. 1b,c).

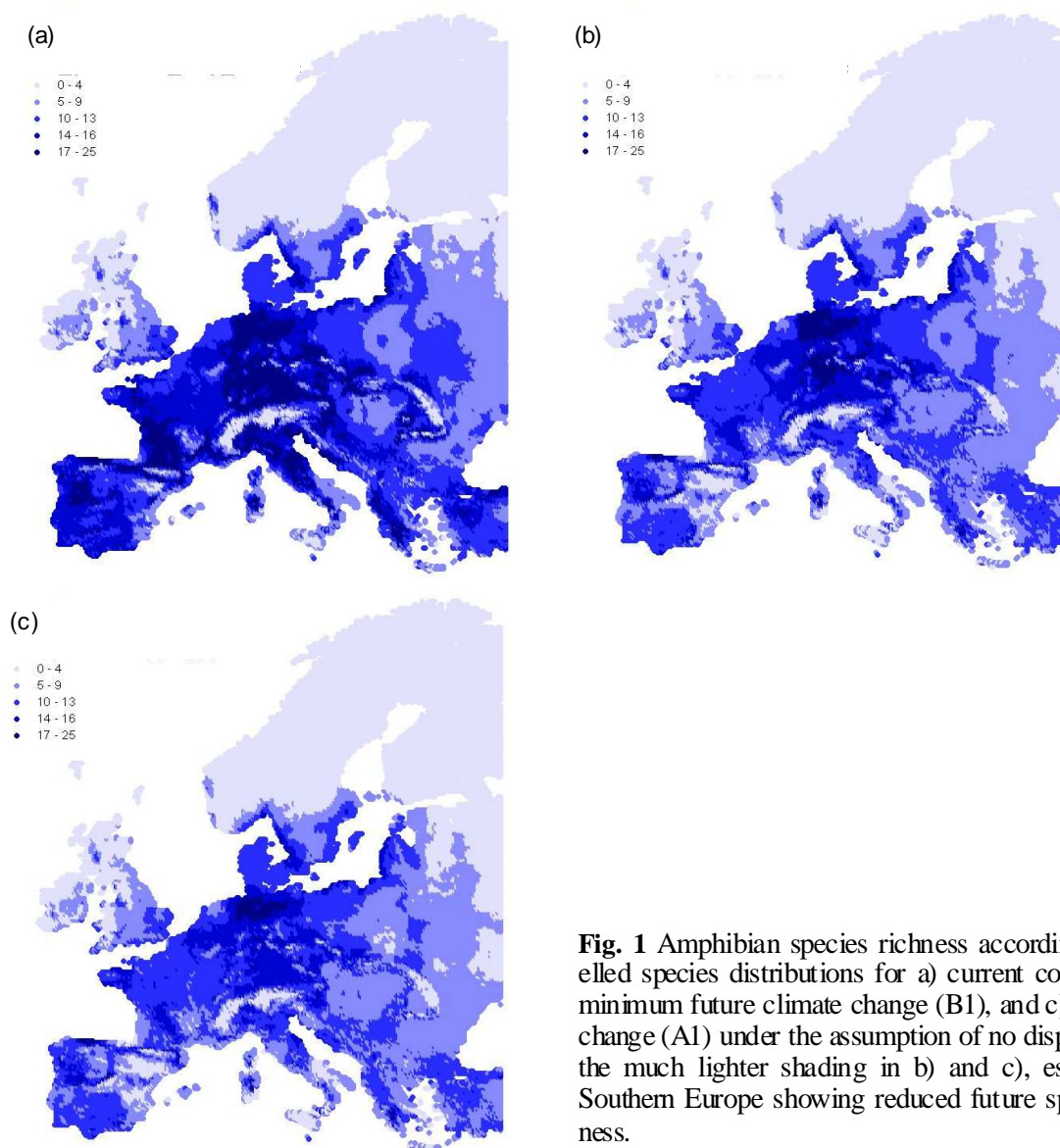


Fig. 1 Amphibian species richness according to modelled species distributions for a) current conditions, b) minimum future climate change (B1), and c) maximum change (A1) under the assumption of no dispersal. Note the much lighter shading in b) and c), especially in Southern Europe showing reduced future species richness.

4.3.2 Overall richness - Reptiles

As expected by their adaptations to hot and dry climates, reptiles have their highest species numbers in Southern Europe, continuously decreasing with higher latitudes (Fig. 3a). Projected losses are also highest in areas with high temperatures and major reductions in precipitation (see Araújo et al. 2006) and predominate in Spain, Italy, the Balkan, and Greece (Fig. 4a,b). Assuming no dispersal ability, climate change is predicted to lead to a general decrease in species richness, which will erode especially severely current hotspots of reptile diversity (Fig 3b, c). However, these results have to be interpreted with caution, since considerations of tolerance to drought of reptiles in Northern Africa render possible that the range contractions at the warm-dry edge of the European gradient are due to incomplete niche descriptions.

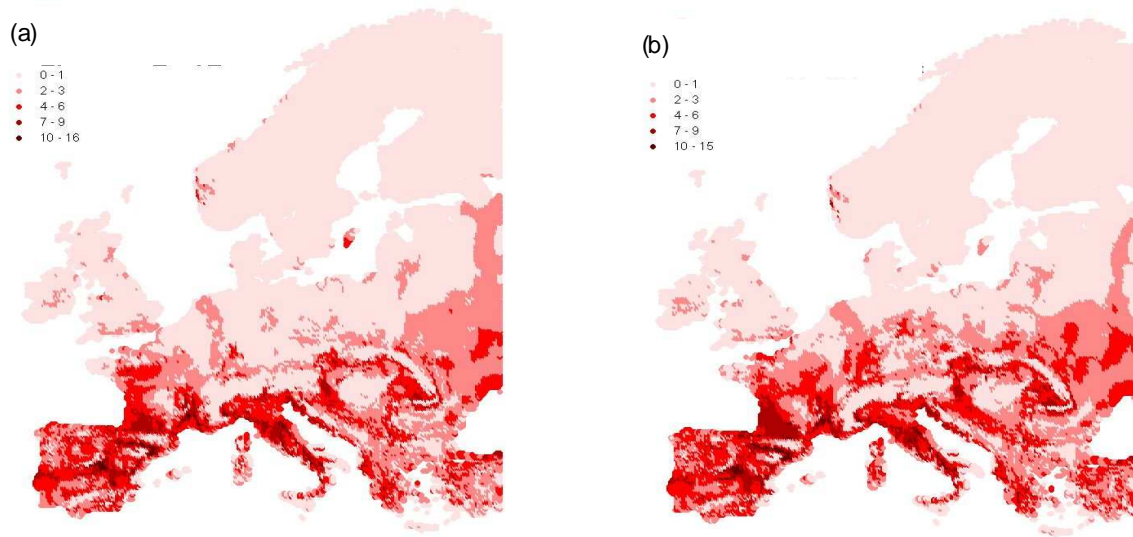


Fig. 2. Projected losses of amphibian species for a) minimum future climate change (scenario B1) and b) maximum change (scenario A1).

4.3.3 Single species

When looking at single species and assuming no dispersal, there are two reptile species that will lose more than 75% of their current range (Fitzinger’s algyroides - *Algyroides fitzingeri*, and Tyrrhenian wall lizard - *Podarcis tiliguerta*; Table 3; see Annex for maps) and this will be the case even when assuming full dispersal. Both species are endemic to Corsica and Sardinia and their future suitable climate space is projected to virtually disappear.

Table 2. Number of amphibian and reptile species per species range loss class for four different climate change scenarios (A1, A2, B2, B1) under the assumptions of no dispersal ability.

Loss	A1 no	A2 no	B2 no	B1 no
<i>Amphibians</i>				
0	1 (2%)	1 (2%)	0 (0%)	1 (2%)
>0-25%	23 (55%)	28 (67%)	26 (62%)	25 (60%)
25-50%	15 (36%)	11 (26%)	14 (33%)	14 (33%)
50-75%	3 (7%)	2 (5%)	2 (5%)	2 (5%)
75-100%	0 (0%)	0 (0%)	0 (0%)	0 (0%)
<i>Reptiles</i>				
0	1 (2%)	1 (2%)	0 (0%)	0 (0%)
>0-25%	35 (53%)	37 (56%)	37 (56%)	41 (62%)
25-50%	20 (30%)	19 (29%)	19 (29%)	15 (23%)
50-75%	7 (11%)	7 (11%)	8 (12%)	8 (12%)
75-100%	3 (5%)	2 (3%)	2 (3%)	2 (3%)

As second most vulnerable species, a group of three amphibians and eight reptiles emerges that will lose between 50 and 75% (Tab. 3) but there are minor differences among scenarios and one of the reptile species will fall even into the highest category under the strongest climate change scenario (A1) (see ANNEX for maps). This group is characterised by species endemic to Spain (including the French Pyrenees) except of the European leaf-toed gecko (*Phyllodactylus europaeus*), which occurs primarily on islands in the Tyrrhenian Sea, and the loggerhead *Caretta caretta*, which is a marine turtle. The results for *C. caretta* must be interpreted with caution since the species ranges well outside the

Mediterranean but these data were not available to determine its climate envelope. Between 11-15 amphibian and 15-20 reptile species will lose 25-50% of their ranges; for *R. arvalis* and the common chameleon (*Chamaelo chamaeleon*) this is the case regardless of the climate change scenario or dispersal ability.

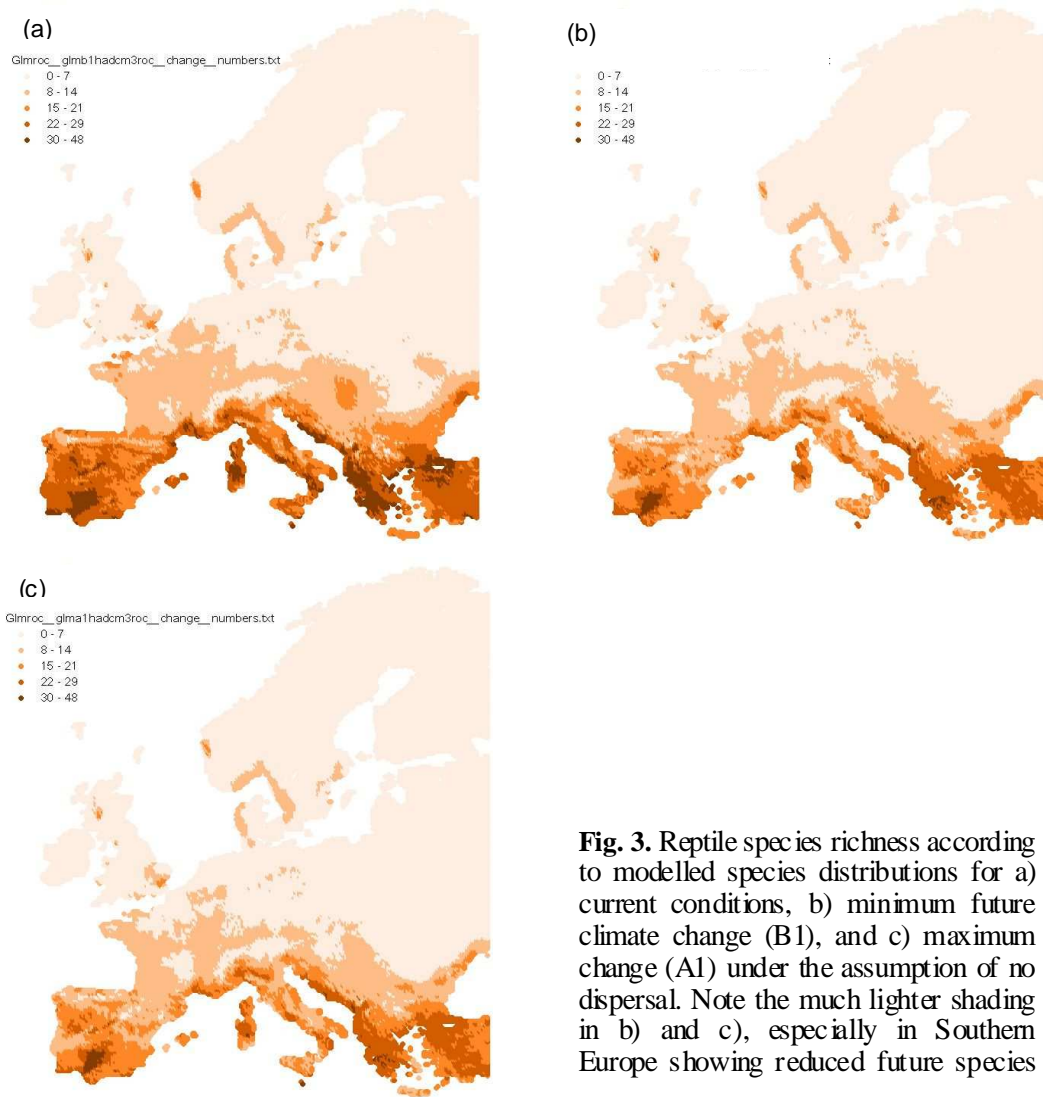


Fig. 3. Reptile species richness according to modelled species distributions for a) current conditions, b) minimum future climate change (B1), and c) maximum change (A1) under the assumption of no dispersal. Note the much lighter shading in b) and c), especially in Southern Europe showing reduced future species

Fig. 4. Projected losses of reptile species for a) minimum future climate change (scenario B1) and b) maximum change (scenario A1).

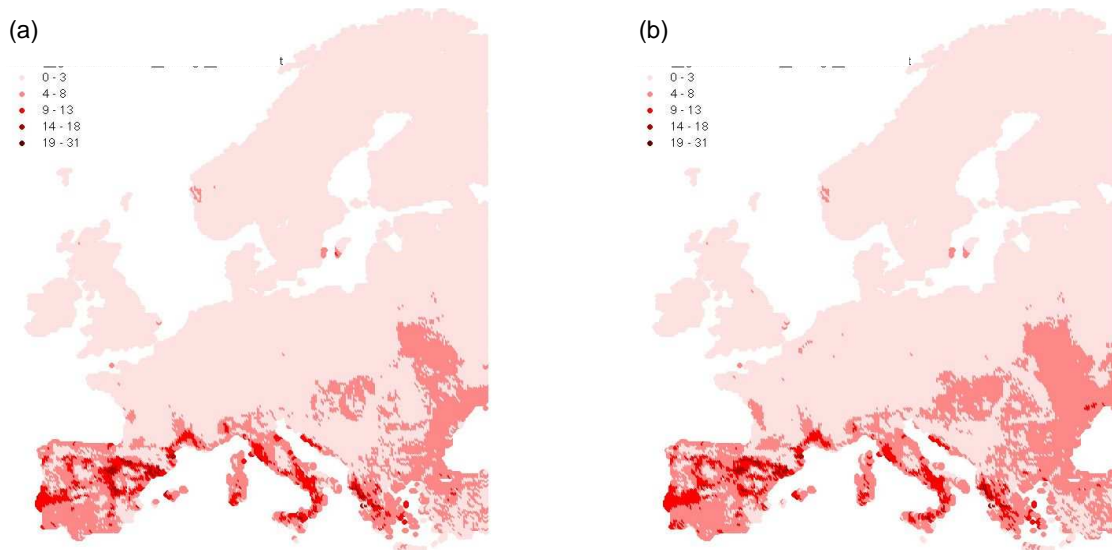


Table 2. Number of amphibian and reptile species per species range loss class for four different climate change scenarios (A1, A2, B2, B1) under the assumptions of no dispersal ability.

Loss	A1 no	A2 no	B2 no	B1 no
<i>Amphibians</i>				
0	1 (2%)	1 (2%)	0 (0%)	1 (2%)
>0-25%	23 (55%)	28 (67%)	26 (62%)	25 (60%)
25-50%	15 (36%)	11 (26%)	14 (33%)	14 (33%)
50-75%	3 (7%)	2 (5%)	2 (5%)	2 (5%)
75-100%	0 (0%)	0 (0%)	0 (0%)	0 (0%)
<i>Reptiles</i>				
0	1 (2%)	1 (2%)	0 (0%)	0 (0%)
>0-25%	35 (53%)	37 (56%)	37 (56%)	41 (62%)
25-50%	20 (30%)	19 (29%)	19 (29%)	15 (23%)
50-75%	7 (11%)	7 (11%)	8 (12%)	8 (12%)
75-100%	3 (5%)	2 (3%)	2 (3%)	2 (3%)

Table 3. Amphibian and reptile species falling into the highest threat categories for four different climate change scenarios (A1, A2, B2, B1) under the assumptions of no dispersal ability.

Loss	A1 no	A2 no	B2 no	B1 no
<i>Amphibians</i>				
	<i>Euproctus asper</i>	<i>Euproctus asper</i>	<i>Euproctus asper</i>	<i>Euproctus asper</i>
	<i>Alytes cisternasii</i>	<i>Alytes cisternasii</i>	<i>Alytes cisternasii</i>	<i>Alytes cisternasii</i>
50-75%	<i>DiscoGLOSSUS galganoi</i>			
75-100%	–	–	–	–
<i>Reptiles</i>				
	<i>Caretta caretta</i>	<i>Caretta caretta</i>	<i>Caretta caretta</i>	<i>Caretta caretta</i>
		<i>Phyllodactylus europaeus</i>	<i>Phyllodactylus europaeus</i>	<i>Phyllodactylus europaeus</i>
	<i>Lacerta schreiberi</i>	<i>Lacerta schreiberi</i>	<i>Lacerta schreiberi</i>	<i>Lacerta schreiberi</i>
	<i>Lacerta monticola</i>	<i>Lacerta monticola</i>	<i>Lacerta monticola</i>	<i>Lacerta monticola</i>
	<i>Podarcis bocagei</i>	<i>Podarcis bocagei</i>	<i>Podarcis bocagei</i>	<i>Podarcis bocagei</i>
	<i>Chalcides ocellatus</i>	<i>Chalcides ocellatus</i>	<i>Chalcides ocellatus</i>	<i>Chalcides ocellatus</i>
	<i>Vipera ursinii</i>	<i>Vipera ursinii</i>	<i>Vipera ursinii</i>	<i>Vipera ursinii</i>
50-75%	<i>Vipera seoanei</i>		<i>Vipera seoanei</i>	<i>Vipera seoanei</i>
75-100%	<i>Phyllodactylus europaeus</i>	<i>Algyroides fitzingeri</i>	<i>Algyroides fitzingeri</i>	<i>Algyroides fitzingeri</i>
	<i>Podarcis tiliguerta</i>	<i>Podarcis tiliguerta</i>	<i>Podarcis tiliguerta</i>	<i>Podarcis tiliguerta</i>

A third group of seemingly little affected species will lose no more than 25% of their ranges under minimum climate change. This corresponds to < 1% range loss per year, which still will lead to a bad conservation status in the prediction period of 40 years. The majority of amphibian (55-67%) and reptile species (53-62%) fall into this category (Tab. 2). This may be a consequence of amphibians and reptiles having limited ability to cope with low temperatures but a much wider reaction range to higher temperatures. Consequently, warming can be expected to be beneficial and cause expansion of the potential range. On the other hand, increasing drought may represent an important threat to species persistence, especially in regions that are already under hydrological stress. This may explain why the species identified as most vulnerable to climate change (i.e. group one and two) come from Spain or Mediterranean islands. These most vulnerable species are not only restricted to areas of increasing hydrological stress but are highly endemic with quite small range sizes, which magnifies negative effects of climate change. Moreover, only one reptile species, the Balkan green lizard (*L. trilineata*), and one amphibian species, the Italian agile frog (*R. latastei*) are predicted to retain its range despite climate

change. In conclusion, it is likely that the most European amphibians and reptiles will be negatively affected and could serve as a suitable indicator of climate change impacts on biodiversity.

5. CONSERVATION IMPLICATIONS

5.1 Sensitive species

Climate envelope modelling (Araujo et al. 2006, this study) and the assessment of the climate sensitivity of amphibians and reptiles clearly show that climate change impacts will considerably differ among species and regions. Although the climate change literature and the different adaptations to hot and dry climates suggest that amphibians should suffer more than reptiles, climate envelope models indicate that both groups are distributed similarly across the sensitivity classes. Here we integrate the results of modelling and the review of biological traits that make amphibians and reptiles sensitive to climate change. We identify those species and regions that are likely to suffer most and compile a list of species that should receive priority because of their sensitivity to climate change.

Modelling predicts that **island endemic** reptile species may become the most seriously impacted ones because their current climate space virtually disappears. Unfortunately, both distribution data and climate data do not have a sufficient resolution to allow the construction of climate envelope models for most endemic island taxa. Additionally, it is particularly difficult to evaluate for endemic island taxa whether they are primarily limited by climatic factors or by the geographic barrier of the sea. Despite these caveats, the following endemic or near endemic island species should be regarded as likely highly sensitive species: the lizards *A. fitzingeri* and *Podarcis tiliguerta*, both endemic to Corsica and Sardinia, and the gecko *Phyllodactylus europaeus*, which is found mainly on these two islands but occurs also on the mainland. Climate envelope models identify them as most sensitive species. The Bedriaga's rock lizard (*Archaeolacerta bedriaga*), which has a too narrow range for modelling on Corsica and Sardinia, needs to be added. We further add the Hierro Giant lizard (*Gallotia simonyi*) from the Canary Islands to this group, because it has an extremely narrow range on steep slopes and is threatened by landslides. With an increased likelihood of extreme events, the risk of landslides will likely also increase. Some island endemic taxa may experience considerable range retraction on flat islands due to sea level rise. Sea level rise is predicted to be approx. 0.5 m in the Mediterranean till the year 2100 (IPCC 2007). This primarily concerns endemic lizard "subspecies", many of which occur on single small island, but currently no overview exists on the height of islands with endemic taxa.

Next to island endemics **amphibians and reptiles from dry Mediterranean regions** are likely to suffer most. This applies especially to species from Spain, but Southern France, Italy, and Greece will also lose relatively high numbers of species. The threats probably will be largest for those amphibian species that depend on temporary water bodies because, in addition to the climatic factors included in modelling, the risk that their breeding sites dry out before they metamorphose will increase and because pressures exerted by climate change are superimposed on the steady decrease of the number of temporary aquatic habitats that occurred due to non-sustainable land use over the last decades in Mediterranean countries (Serrano 1996, Blondel & Aronson 1999, Papayannis & Salathé 1999, Scocianti 2001). In addition to the species listed in the second highest threat category in Table 3, semi-aquatic reptiles (turtles and snakes of the genus *Natrix*) may also become affected by these habitat changes. However, like most water frog taxa (subgenus *Pelophylax* of the genus *Rana*) in Mediterranean Europe, they are able to use permanent water bodies, such as rivers, as reproductive and summer habitats, and the climate envelope models do not classify them into the most affected categories. On the other hand, the allocation of the reptile species into the second highest threat category in Table 3 by climate change models should be regarded as preliminary, because currently few evidence exists that reptiles are indeed limited by hot temperatures or drought at the hot-dry edge of their range (see chapter 2.4). Of the species falling into this category in table 3, the best biological evidence of sensitivity exists for *L. schreiberi* because of its dependence on humid riverine habitats. Cases-by-case assessments (see *Podarcis sicula* in chapter 2.4), which are beyond the scope of this study, are needed to allow separation of those species that probably are from those that probably are not limited by climate factors at the southern margin of their distribution.

One sea turtle species, *C. caretta*, is also included in the second highest threat category by climate envelope models. For this species, it is likely that the result is due to incomplete niche descrip-

tion because neither distribution data nor climate data were available at a sufficient resolution for most of its tropical distribution area. However, inundation of nesting beaches will occur for all sea turtle species by sea level rise. Whether this will become a major threat or not will depend on the geology of current nesting beaches and adjacent regions (availability of adequate coastal habitat above future sea level) and human coastal management (barriers to the access to shifting and new nesting locations). Countries that have breeding populations of sea turtles should make such assessments for their coastlines.

For **montane** species neither assessment of biological traits nor modelling provides a uniform picture. Some species, such as *S. atra*, are predicted to be little affected, whereas others, e.g., the Pyrenean mountain brook salamander (*Euproctus asper*), is likely to suffer considerably. Several **Mediterranean amphibian species** limited to **cool and moist microhabitats** have too narrow distribution ranges for the development of climate envelope models, especially cave salamanders (*Speleomantes* spp.), some mountain brook salamanders (*Euproctus* spp.), and *Proteus anguinus*. Whether they will suffer will primarily depend on the hydrology and associated changes in moisture of their microhabitat. Since hydrology is likely to change in Mediterranean areas and because of the dependence of these species on cool and moist microhabitats, we include them in the list of sensitive species (Tab. 4). One further species, *Alytes muletensis*, is included in Table 4, because it has a very **narrow distribution and depends on aquatic habitats** that are very likely further impacted by human needs for water resources (Podlucky, pers. comm.).

In **Central and Northern Europe**, early breeding amphibian, i.e., primarily brown frogs (*Rana arvalis*, *R. dalmatina*, *R. temporaria*) and *B. bufo*, may be placed at increasing risk due to **late frosts, less snow cover, and warmer winter temperatures** with concomitant energy depletion in winter and the risk of freezing of spawn or adults. Further, reduced snow cover will pose additional risks to all hibernating amphibian and reptile in northern latitudes species (see e.g., Podlucky et al. 2005 for *V. berus*) but in general the magnitude of this risk is still unknown. Since the species of Central and Northern Europe tend to have a wide distribution and because climate envelope models did not classify any of these species into the two highest risk categories, we do not include any of these species in Table 4.

Table 4. Amphibian and reptile species that are likely to suffer most by climate change and should receive priority in management and policy

Species	Comments
Amphibia	
<i>Speleomantes</i> spp.	Temperature and moisture sensitive; some species are island endemics
<i>Proteus anguinus</i>	Dependence on cool habitats and hydrology of cave systems
<i>Euproctus</i> spp.	Dependence on cool and moist microhabitats; climate envelope model for <i>E. asper</i>
<i>Alytes cisternasi</i>	Dependence on temporary water bodies, climate envelope models
<i>Alytes muletensis</i>	Threatened by water scarcity
<i>Discoglossus galganoi</i>	Dependence on temporary water bodies, climate envelope models
Reptilia	
<i>Phyllodactylus europaeus</i>	Second highest threat class in modelling; primarily on Corsica and Sardinia
<i>Algyroides fitzingeri</i>	Endemic to Corsica & Sardinia; highest threat class in modelling
<i>Archaeolacerta bedriagae</i>	Endemic to Corsica & Sardinia; climate space disappears
<i>Lacerta monticola</i>	Climate envelope models
<i>Lacerta schreiberi</i>	Climate envelope models; dependence on humid habitats
<i>Gallotia simonyi</i>	Very narrow range, threatened by increasing risks of land slides
<i>Podarcis bocagei</i>	Climate envelope models
<i>Podarcis tiliguerta</i>	Endemic to Corsica & Sardinia; highest threat class in modelling
<i>Chalcides ocellatus</i>	Climate envelope models
<i>Vipera seoanei</i>	Climate envelope models
<i>Vipera ursinii</i>	Climate envelope models

5.2 Management and policy recommendations

5.2.1 Protective measurements

To be clear about the conservation implications one needs to be clear about the conservation objectives (Gent, SEH-CC, comm. by lett.). Within the current climate window we can start to consider ‘favourable levels’ for species of amphibians and reptiles (e.g., developing the concept of ‘favourable conservation status’ that underpins the EU Habitats Directive). Much of this would focus on ensuring robust populations within the current and recent historic distribution of the species. It is possible to visualize and even to start to quantify such levels using parameters that describe population status, extent of habitat, range, and future prospects. Climate change adds a new level of complexity to this but reflection on ‘favourable levels’ may help provide a framework for assessing targeted outcomes in a changing climate. It may also help guide the degree to which ‘in situ’ conservation in the current geographical range should be attempted versus ‘ex situ’ measures including range changes or conservation of a genetic resource in captivity.

In any case, management needs will differ among species and regions since they will differ in their sensitivity respectively degree of likely impacts of climate change. Habitats too will vary in their response to climate change, and the opportunities for managing these to retain comparable ‘micro-climates’ to allow the persistence of amphibian and reptile in different localities may also vary. Other factors, such as the presence of competitor species and their likely response to climate change, will affect the way in which climate change affects amphibians and reptiles. Management strategies and policy decisions need to account for these differences. We **recommend that species-specific climate change mitigation plans** are developed for those species that are likely to suffer most. These species are listed in Table 4. Although we did not include early breeding amphibians of Central and Northern Europe into Table 4 (see section 5.1), we recommend that countries from these regions also develop climate change mitigation plans for these species. The EU and the countries that share high responsibilities for these species (Schmeller et al. 2008) should share in the development of these mitigation plans. Beyond these species-specific mitigation plans the following general management strategies are recommended.

Species are likely to disappear from some regions. This should not be considered as a justification to reduce the protective level for these species in these regions. Notwithstanding, the only chance to retain these species in these regions is an evolutionary adaptation to the new climatic conditions. Small populations have limited scope for such adaptive shifts but populations with a size of 1250 and 1750 individuals in reptiles and amphibians, respectively, or larger may have the potential for rapid adaptive shifts (see section 2.5). While large populations are generally particularly valuable from a conservation perspective, they are especially important for adaptation to climate change. **Recommendation:** Large populations in large tracts of high quality habitat should receive very stringent protection and the highest attention by management and policy. Additionally, restoration activities should be focussed on those habitats, which could bring populations of sensitive species close to the size indicated above. This strategy will be affective also for regions, which may not become completely unsuitable for the species.

In regions, which will remain climatically suitable, local and regional extinctions nevertheless may occur unless affected species can disperse to new sites. First, this means that species may be lost from protected sites. **Recommendation:** countries should assess whether protected areas remain climatically suitable for target species. Priorities for such assessment should consider the sensitivity of the species to climate change, the national responsibility a country has for the protection of a species (Schmeller et al. 2008), and, in the context of the Berne Convention, whether the species is listed on its Annexes. Gaps in current or likely future representativity of species should be reduced using systematic approaches to reserve site selection (Margules & Pressey 2000). However, species conservation needs to look beyond the constraints of protected areas and, among others, need to consider regional connectivity (Gent SEH-CC, comm. by lett.). This is especially so when addressing conservation measures to mitigate for climate change.

Species will not be able to track geographical shifts in climate space unless they can **disperse to new suitable sites**. A denser network of ponds has been requested for amphibians of Mediterranean climatic zones relative to more temperate regions (e.g., Blaustein et al. 2001, Jakob et al 2003a) be-

cause of the unpredictability of and reduction in rainfalls, coupled with amphibian dependence on humid habitats, superimposed on an increased loss and fragmentation of favourable terrestrial corridors. However, this is misleading because amphibians - and reptiles - of the Temperate Zone also have low dispersal power and therefore also require dense networks of suitable habitats (Fog 1993, Settele et al. 1996, Semlitsch 2003, Smith & Green 2005, Jehle & Sinsch 2007). With an active dispersal in the range of usually < 1 km in urodeles (Smith & Green 2005, Jehle & Sinsch 2007) and lizards (Settele et al. 1996), and usually < 2 km but up to several kilometres in snakes (Settele et al. 1996) and anurans (e.g., Fog 1993, Settele et al. 1996, Smith & Green 2005, Jehle & Sinsch 2007), the density of suitable habitats must remain high in all landscapes and the whole landscape must be made permeable (i.e. suitable) for the dispersal of amphibians and reptiles. **Recommendation:** we have to facilitate natural range shifts by redoubling effort to maintain or restore large-scale connectivity in all affected regions in Europe (Hunter et al. 1988, Hannah et al. 2002) and to avoid carbon management solutions that will reduce the suitability and permeability of the landscape (e.g., biofuel production by the conversion of native vegetation into fuel farms: Cook & Beyea 2000). Otherwise, species with low dispersal capacity, such as most reptiles and amphibians, will not be able to track any geographical change in climate space. Semlitsch (2003) argued that a natural density of at least 0.5- 1 suitable ponds per km² is essential for broad-scale maintenance of pond breeding amphibians.

5.2.2 Assisted dispersal

With global climate change looming large in the public psyche, a debate on assisted dispersal has started (Hunter 2007, McLachlan et al. 2007) and the discussion is particularly relevant for reptiles and amphibians with their low dispersal power. Assisted dispersal usually is defined as the intentional introduction of species to climatically favourable areas outside their 20th century geographic range as a proactive measure to combat biodiversity loss by climate change (Mueller & Hellmann 2008).

Deciding whether undertaking a particular assisted-colonization project should be undertaken is patently complex. Although various guidelines for the translocation of species exist (e.g., ANL 1982, <http://www.kew.org/conservation/RSGguidelines.html>), in the face of climate change new problems surfaced that need addressing. Hunter (2007) suggested that three major issues must be carefully considered:

1) Candidate species: Assisted dispersal should be considered only for species with low dispersal (almost all reptiles and amphibians) and there must be confidence that climate change is the primary threat (Thomas et al. 2004). For European amphibians and reptiles, currently habitat destruction is still the main threat (e.g., Henle & Streit 1990). We recommend that climate change should not only be the primary threat but the predicted loss of range must be substantial and that a normative policy discussion of acceptable thresholds for substantial" is started.

2) Candidate sites: The amount of disturbance at a candidate site must be considered. The discussion is still on-going but may probably settle at sites with intermediate disturbance (Hunter 2007). Also, connected sites that have experienced species range shifts in the past are more acceptable than islands. Only sites within the long-term biogeographic range should be considered. Research on invasive species clearly shows that the worst case invasive species are those that have been brought to areas far outside this range (see also Mueller & Hellmann 2008). Species rich systems are often regarded as less likely to be disrupted by a translocation than a species poor ecosystem but this issue is still controversial.

Beyond direct interactions with the species present in the translocation site and resulting ecosystem effects, indirect effects due to inadvertent introduction of diseases have to be carefully considered. This is particularly relevant for anurans, with a major concern of an inadvertent spreading of the chytrid fungal parasite *Batrachochytrium dendrobatidis* with translocations of anurans. *B. dendrobatidis* can cause high mortality in some anurans so that its introduction to novel sites could be a major catastrophe for resident species (Berger et al. 1999). For reptiles, spread of disease has not yet received the same concern except for translocation programs of gopher tortoises (*Gopherus polyphemus*) because of the spreading of a respiratory disease and a herpesvirus (Boarman 2002). However, other diseases, such as parapox virus in lizards of the genus *Lacerta* and *Podarcis*, need also be considered (Gent SEH-CC, comm. by lett.). Nevertheless, treatments, taking all possible pathogens into account, quarantine, and pre-lease monitoring are essential for translocations of any species. These should follow

standards for captive and re-introduction programs that deal with pathogens and parasites (Garner SEH-CC, comm. by lett.).

3) Feasibility: Costs and translocation techniques must be considered. The translocation technology is still far from being perfect (Fischer & Lindenmayer 2000) and a range of research questions still need to be answered (McLachlan et al. 2007).

Island endemics need additional considerations. For endemic subspecies, translocations are inadequate because subspecies evolved in response to the unique biotic and abiotic environment on each island. These conditions cannot be recreated at any other place so the subspecies will be lost if it cannot cope or adapt to climate change. However, other populations of the species may become isolated on land masses that become islands due to sea level rise and can form the basis for evolutionary pathways to lead to new subspecies. Therefore, we recommend that policy and management should secure that these evolutionary processes can continue on newly formed islands. For endemic species assisted dispersal also will not be a real solution because they also evolved in response to the unique conditions of their Island(s). Thus, they also will be definitely lost if they cannot survive climate change on their current island(s). However, in contrast to subspecies, an ethical evaluation should be made whether assisted translocation may be justified even so the translocated population will soon be no more the same as the original species.

In conclusion, since no assisted dispersal program will be free of risk and because robust predictors of invasion risk have proven elusive (Levine et al. 2003), great care must be taken before and when adopting an assisted dispersal program. Currently, there are no specific regulations to govern the actions envisioned in assisted dispersal but some groups are already beginning to move species (McLachlan et al. 2007). **Recommendation:** Policy should limit unsupervised translocations and require thorough risk analyses and impact assessments on a case-by-case basis. Such regulations should adapt existing guidelines on translocations (e.g., of the IUCN: <http://www.kew.org/conservation/RSGguidelines.html>) that were developed before climate change came to the forefront.

5.2.3 Monitoring

Since differences among species and probably even within species exist in their phenological response to climate change and this can have considerable influence on the sensitivity of a species to climate change, we recommend an intensification of phenological monitoring. For amphibians and reptiles, the EuMon database (eumon.ckff.si) provides an overview of programs in Europe that monitor the phenology of amphibians and reptiles. The EuMon database contains several but not all monitoring programs started nationally or subnationally for fulfilment of the reporting requirements for the EU Habitats Directive. All these activities could serve as a starting base for the development of a European wide program. Such a monitoring program should include the monitoring of pressures, such as climate change and habitat alteration, that may impact the conservation status of species (Henry et al. 2008). **Recommendation:** We recommend that a workshop is organized to invite representatives of these monitoring programs to discuss and design such joint efforts and that start-up money is made available for the coordination and integration of existing programs.

5.2.4 Research

There are still a number of major knowledge gaps on the likely effects of climate change on reptiles and amphibians (and on other organisms). This hampers the design of better policies and management strategies. Therefore, we recommend that further research is initiated on the potential impacts of climate change and on suitable adaptation and mitigation strategies. The following research topics should receive high priority:

- potential impact of sea level rise on endemic island taxa and nesting populations of sea turtles potentially to allow a better understanding of which endemic island taxa are likely to suffer from climate change and how these impacts can be mitigated;
- assessment of the likely degree of reduction of permanent wetlands and rivers in regions predicted to become drier by the combined effects of climate change, land use change, and changing demands on water resources, how these pressures affect amphibians and reptiles, and how these effects can be mitigated;

- hydrological changes in Mediterranean regions that have the potential to affect amphibians depending on cool moist microhabitats and to use such studies as a basis for the design of mitigation strategies;
- relationships of the biology of species with a too narrow range for the development of climate change models with climatic factors as a basis to evaluate likely impacts of climate change;
- improved understanding of the extent to which species are (or are not) limited by temperature (or other climatic factors) at the southern margin of their distribution;
- development of more complex but still generic models of impacts of climate change that address a wide range of factors (including indirect effects) at given localities so that conservation measures can be adapted;
- identification of a set of amphibian and reptile species as indicator for climate change and the development of long-term monitoring schemes for these species.
- better capacity to improve the permeability of the landscape for reptiles and amphibians (and other organisms with low dispersal capacity) under the constraints of conflicting land use demands;
- evolutionary adaptation of amphibians and reptiles to changing climates and how the potential of such adaptations can be realized.

Further important knowledge gaps that require more research are identified in sections 2 and 5.2.2. Finally, our **recommendation of overriding importance** is research that fosters the willingness and ability of politicians, institutions, and society to reduce the emission of green house gases without compromising the conservation of amphibians and reptiles and their habitats. Without that, we will not be able to make major progress in the conservation of biodiversity in a changing world.

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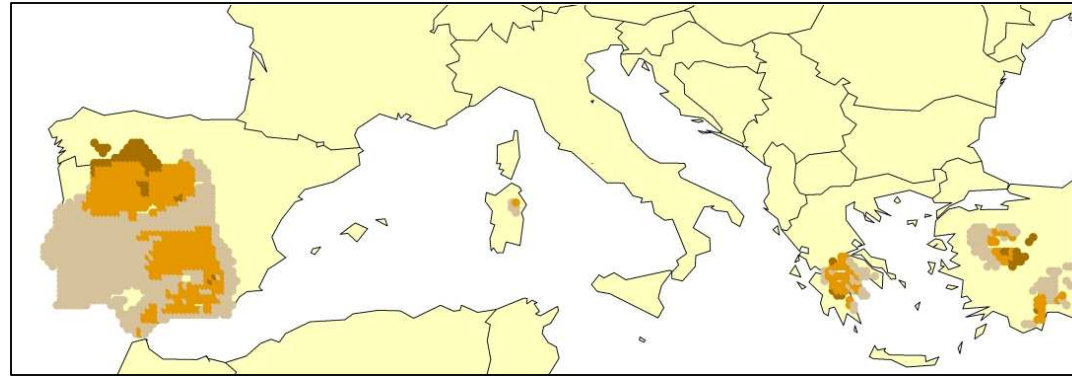
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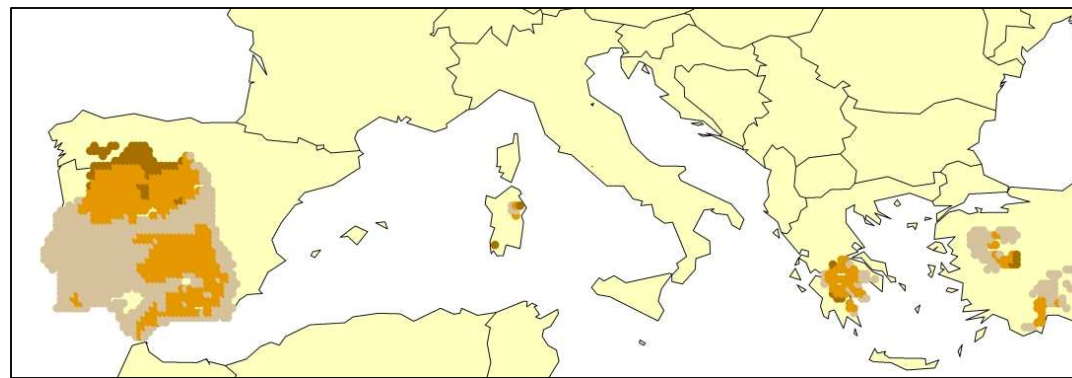
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ANNEX: Changes in suitable climatic niche for 2050 for two groups of most vulnerable species

Alytes cisternasii



A1



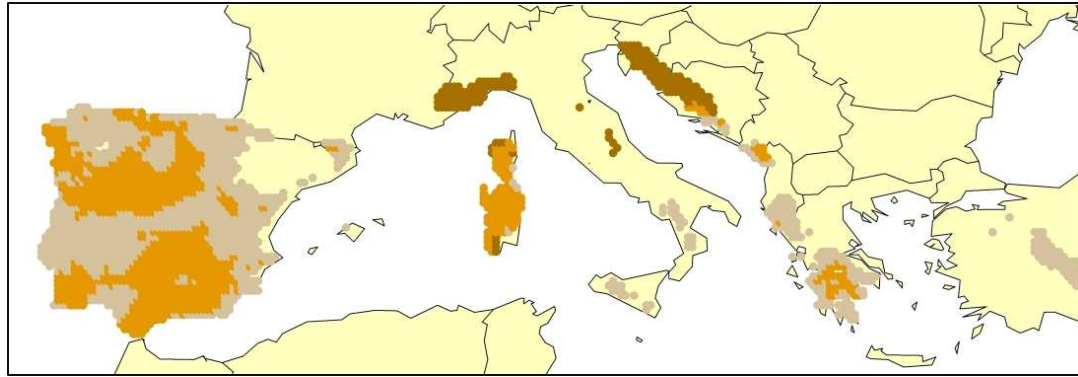
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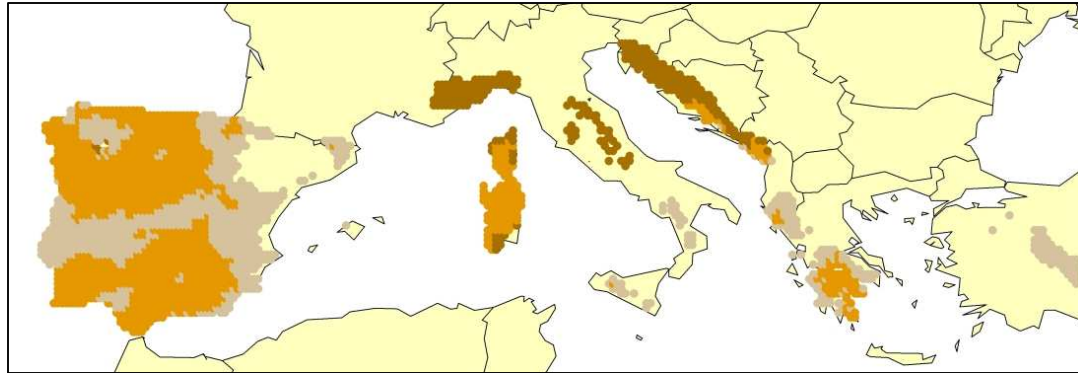
B1

- gain
- loss
- remain

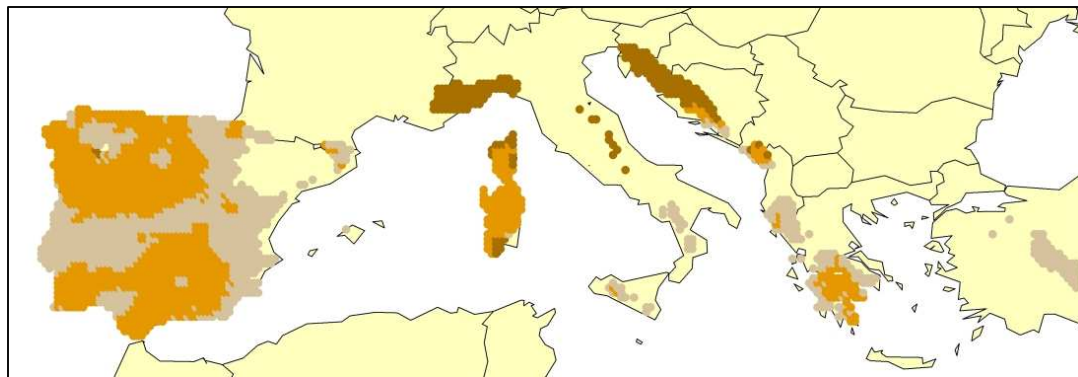
Discoglossus galganoi



A1



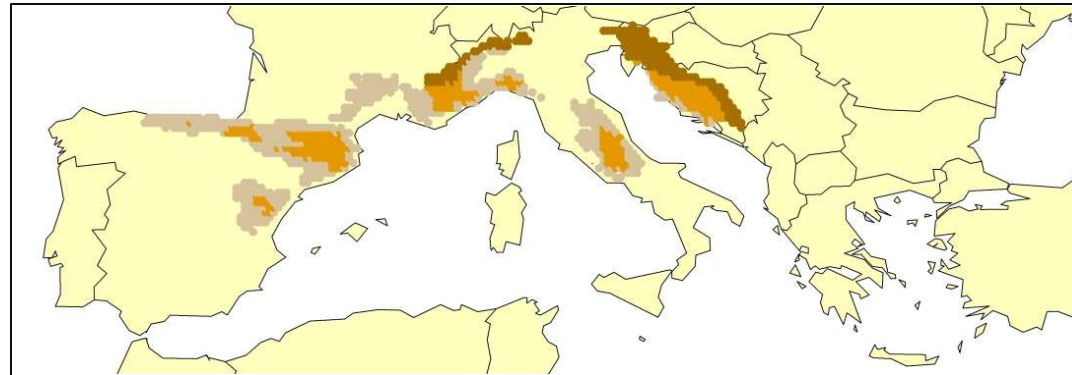
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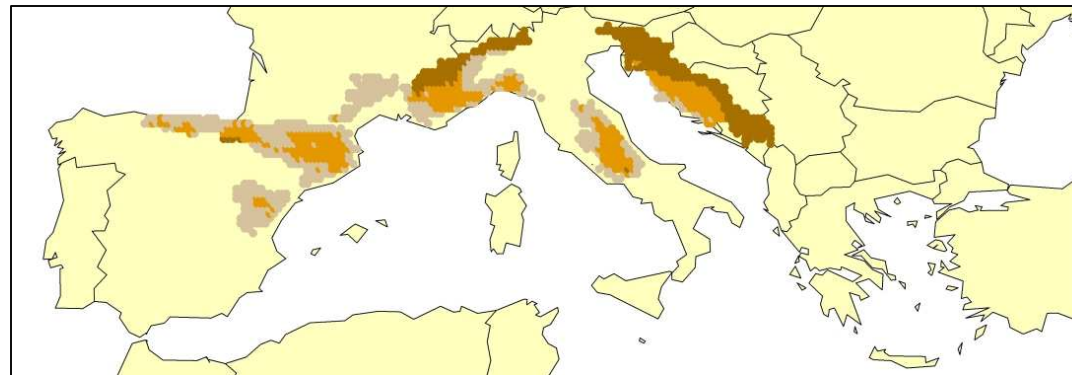
B1

- gain
- loss
- remain

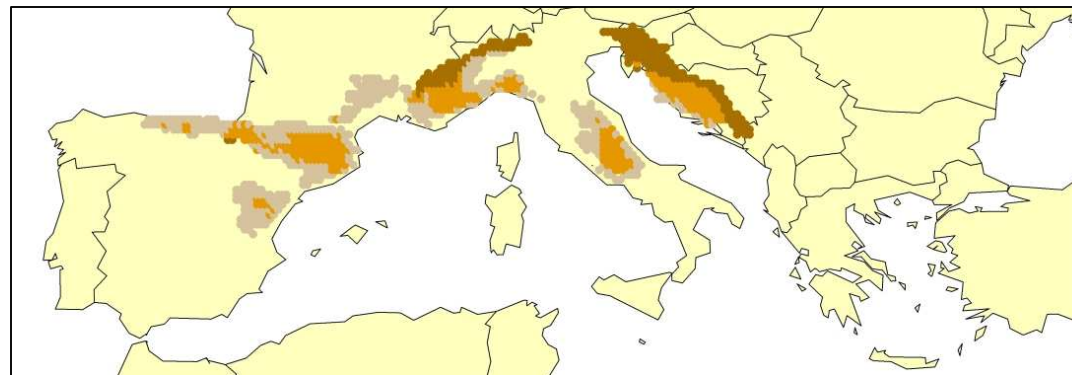
Euproctus asper



A1



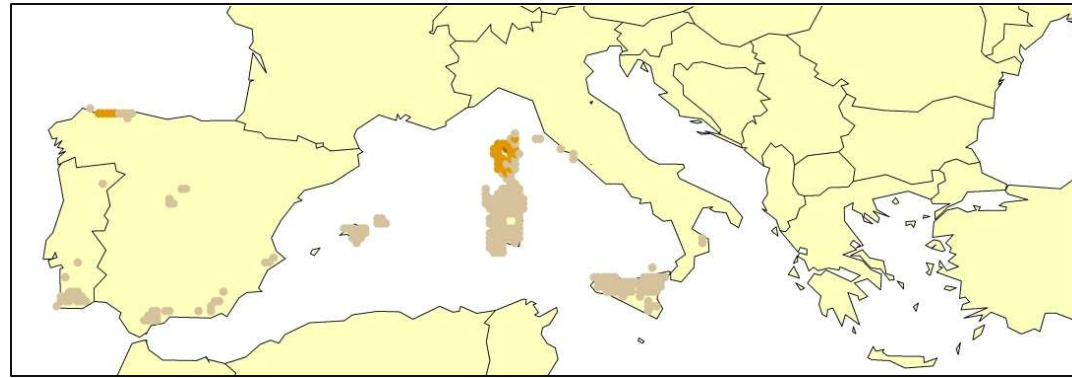
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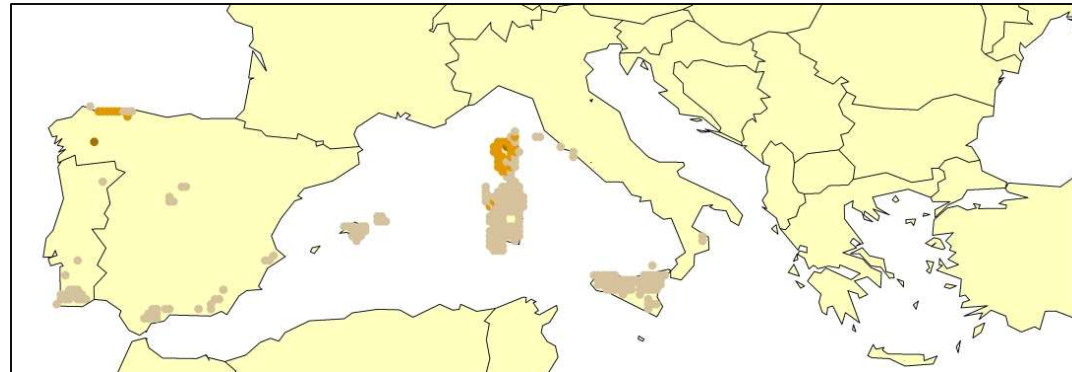
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- gain
- loss
- remain

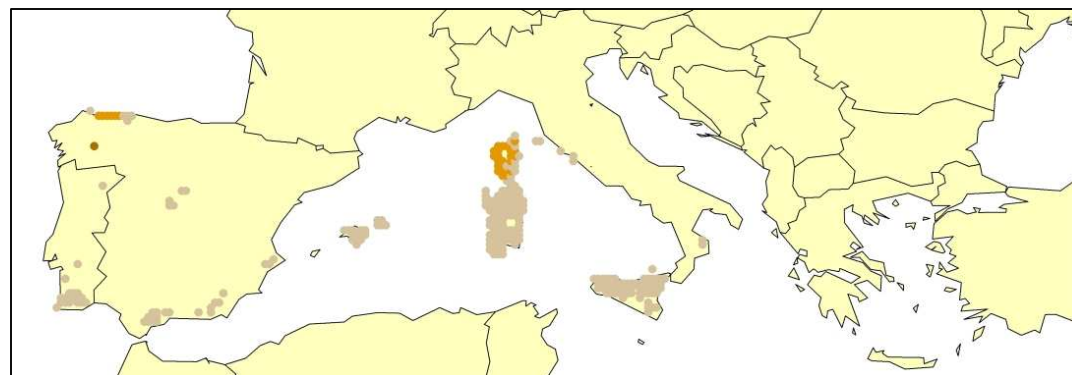
Algyroides fitzingeri



A1



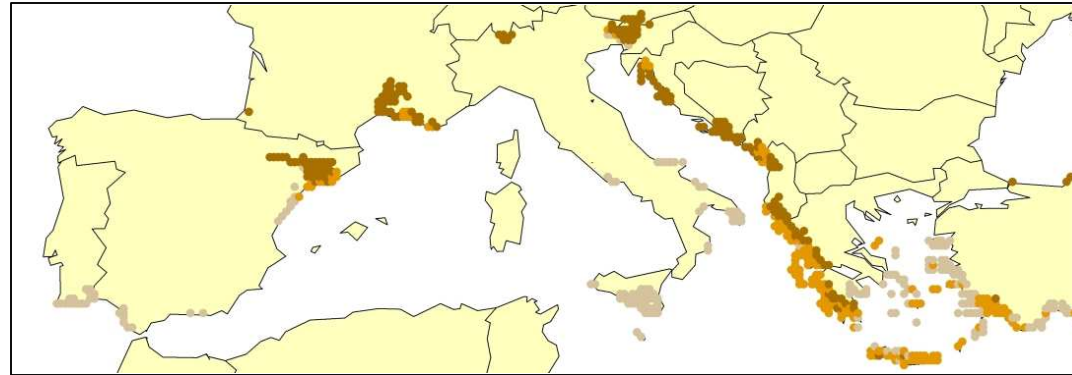
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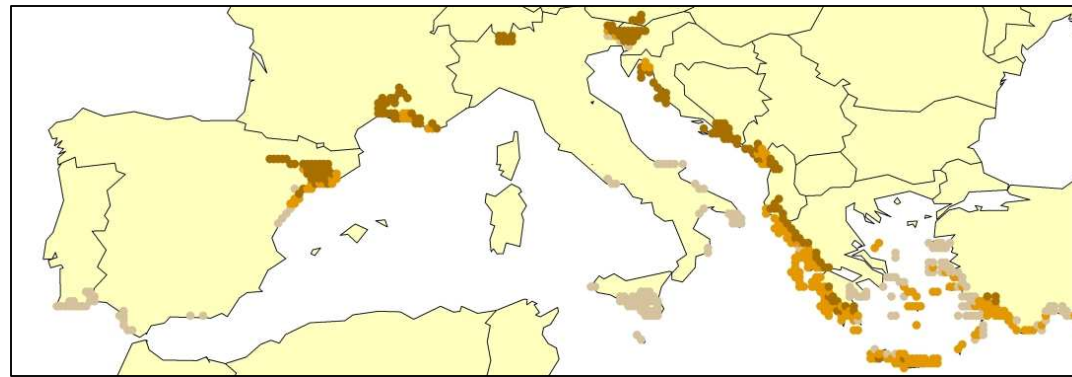
B1

- gain
- loss
- remain

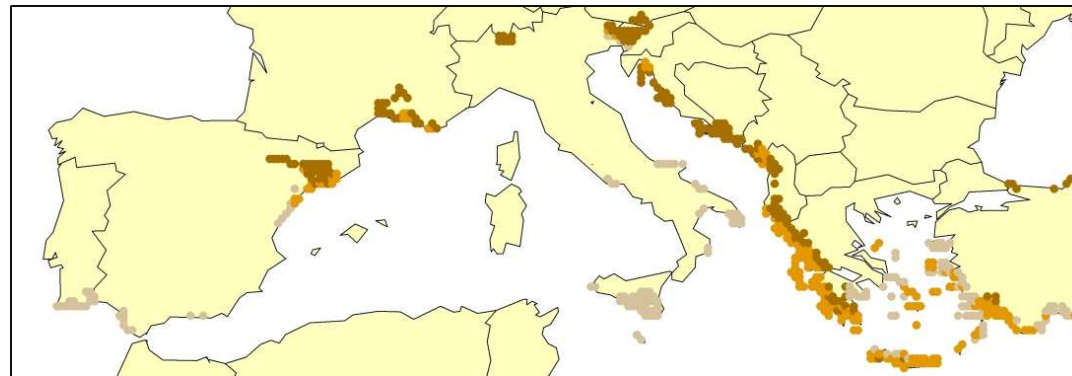
Caretta caretta



A1



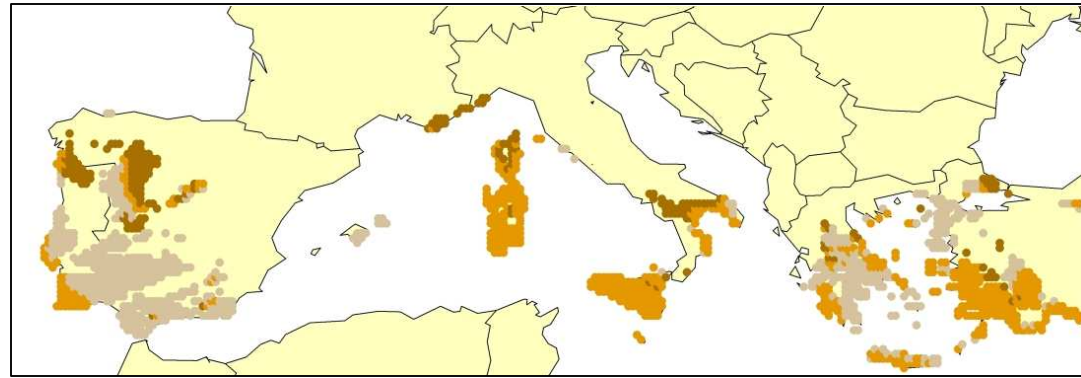
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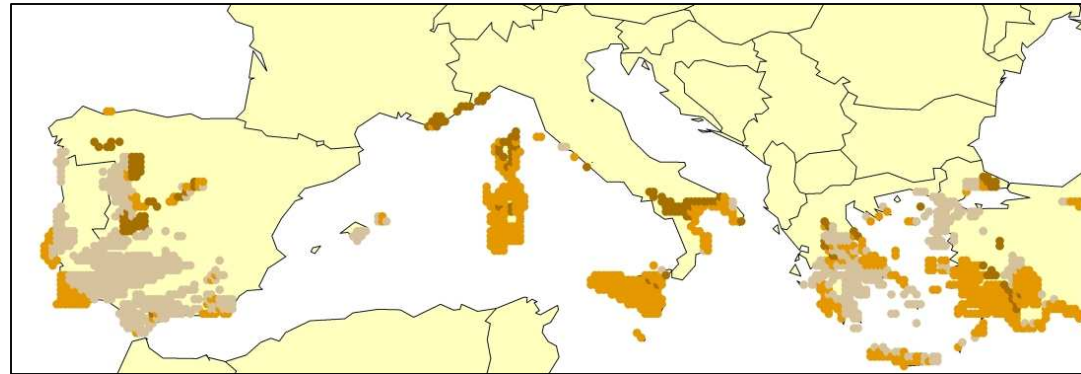
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- gain
- loss
- remain

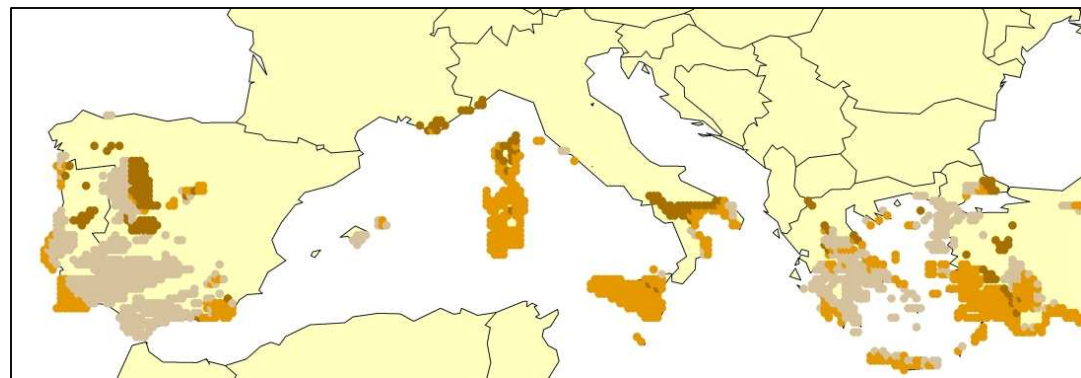
Chalcides ocellatus



A1



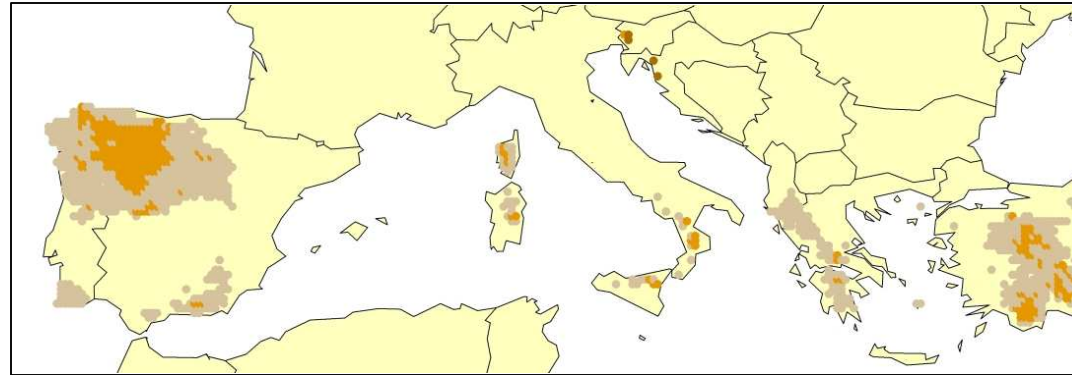
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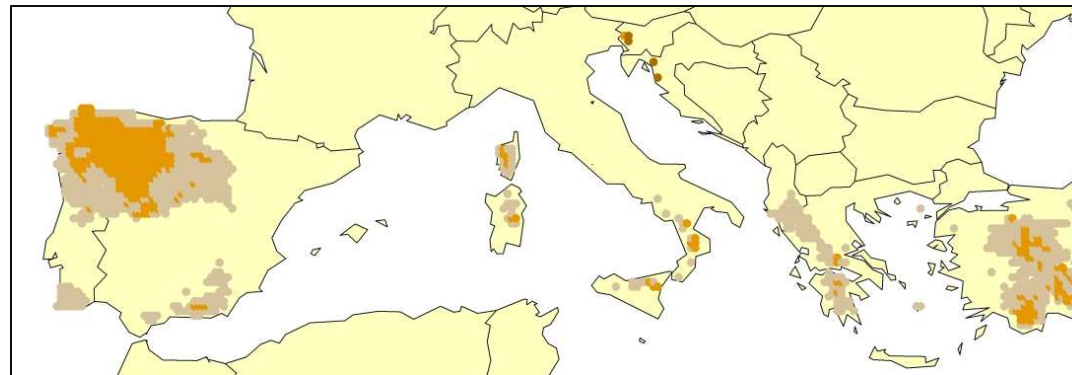
B1

- gain
- loss
- remain

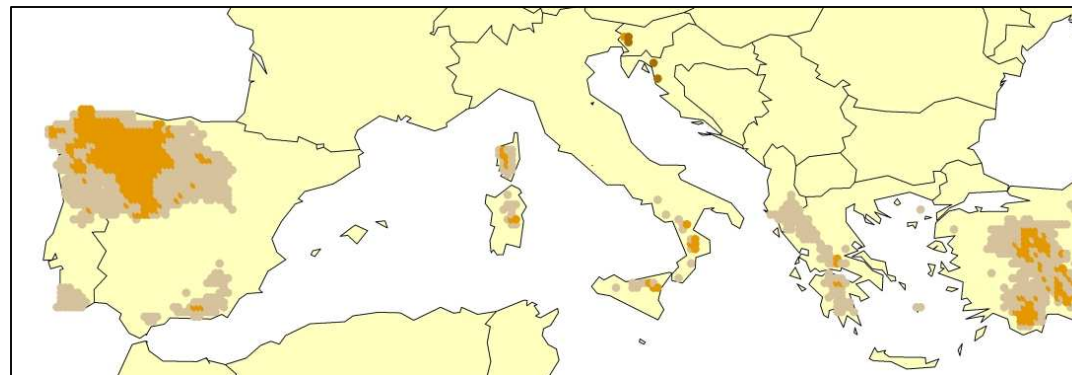
Lacerta monticola



A1



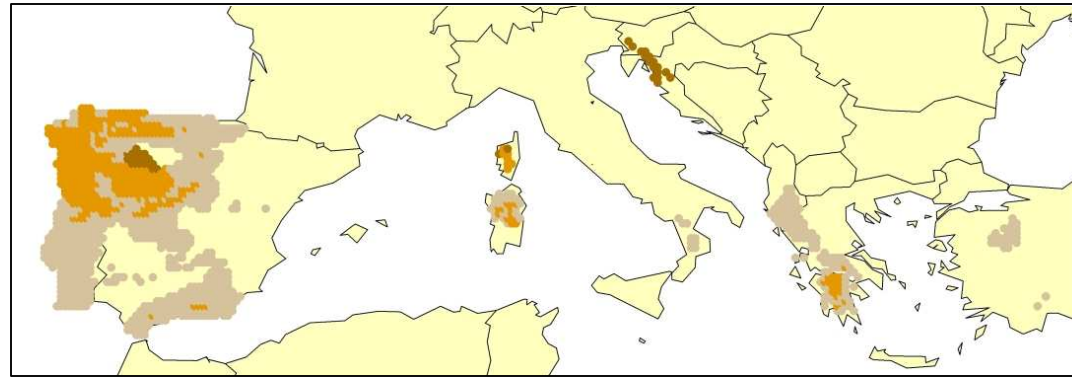
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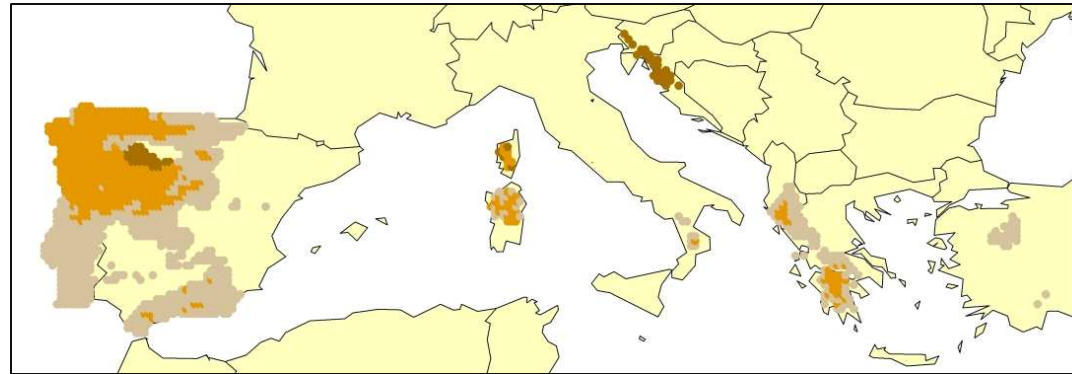
B1

- gain
- loss
- remain

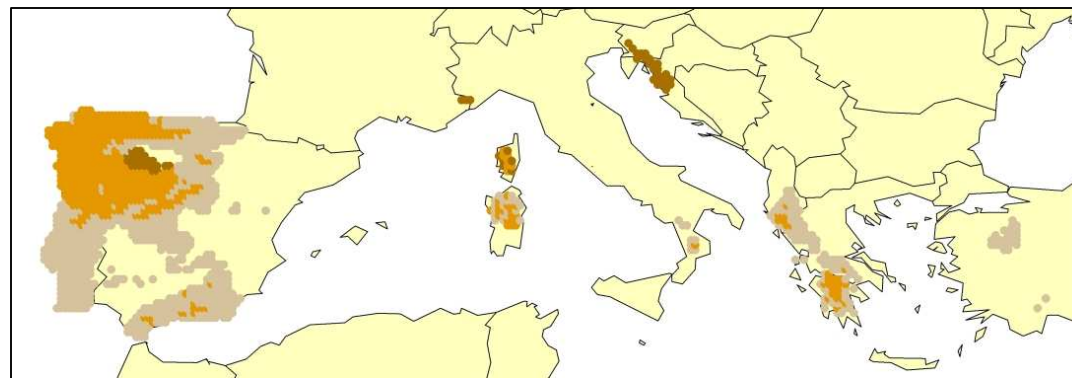
Lacerta schreiberi



A1



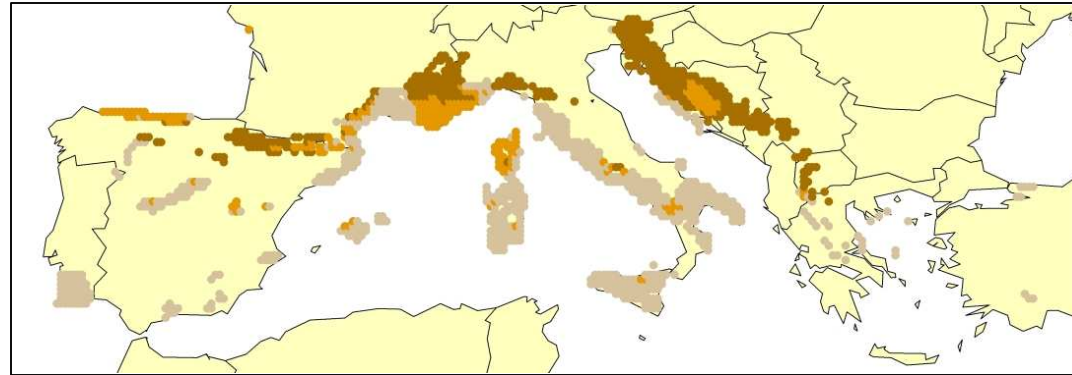
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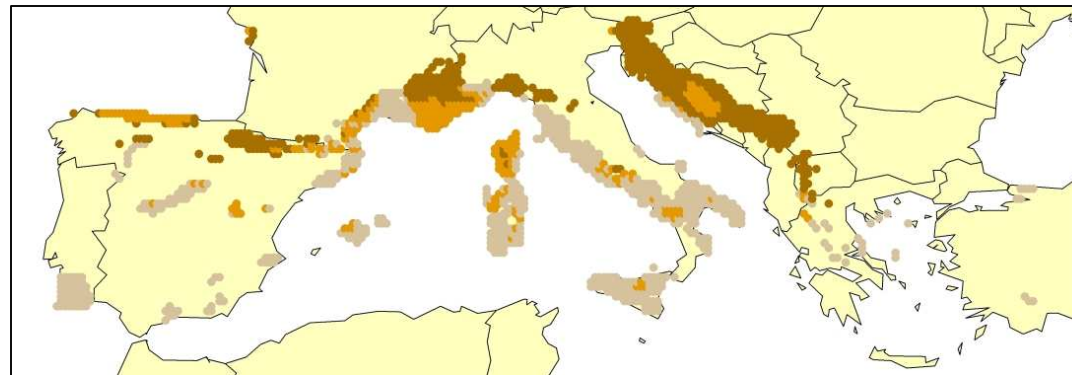
B1

- gain
- loss
- remain

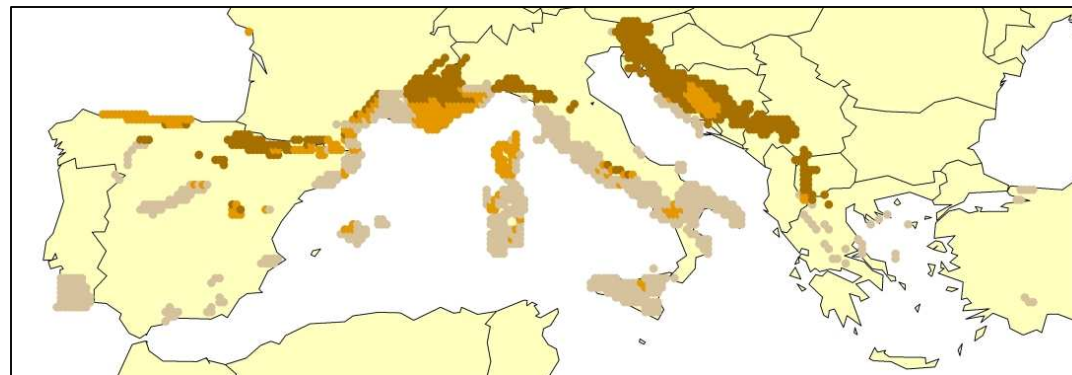
*Phyllodactylus
europaeus*



A1



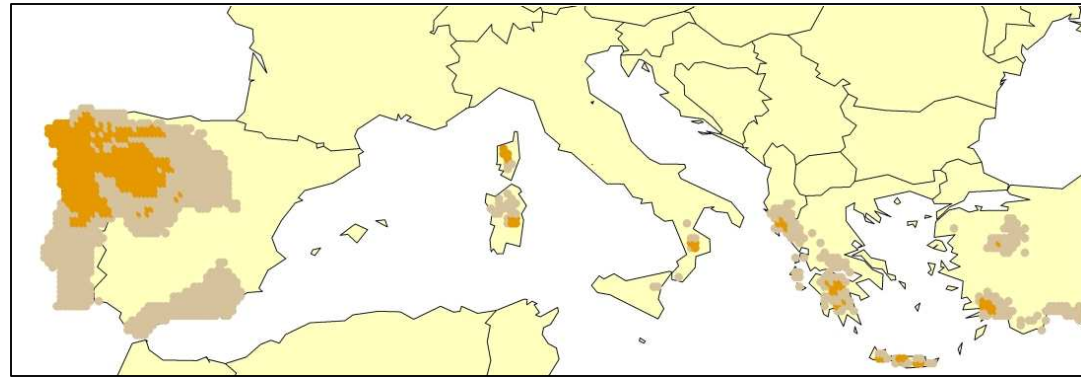
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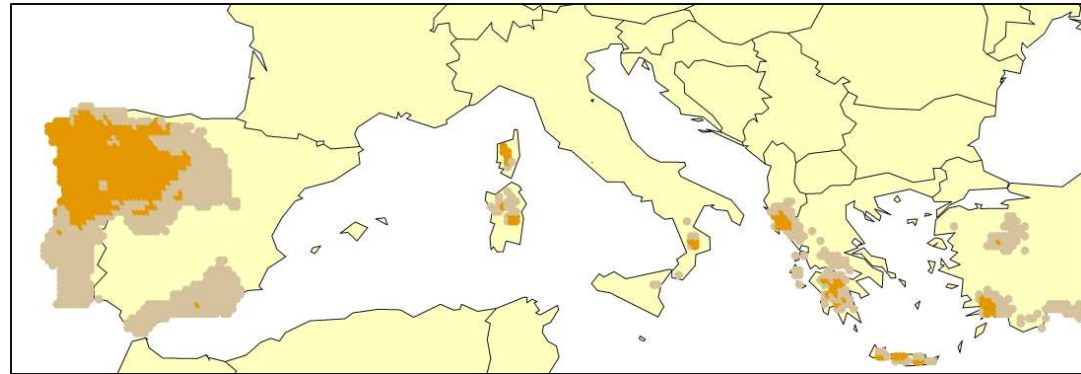
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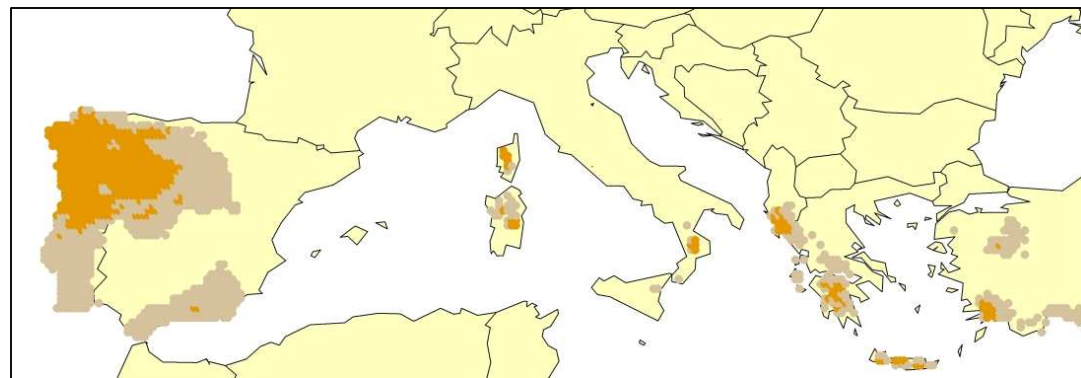
Podarcis bocagei



A1



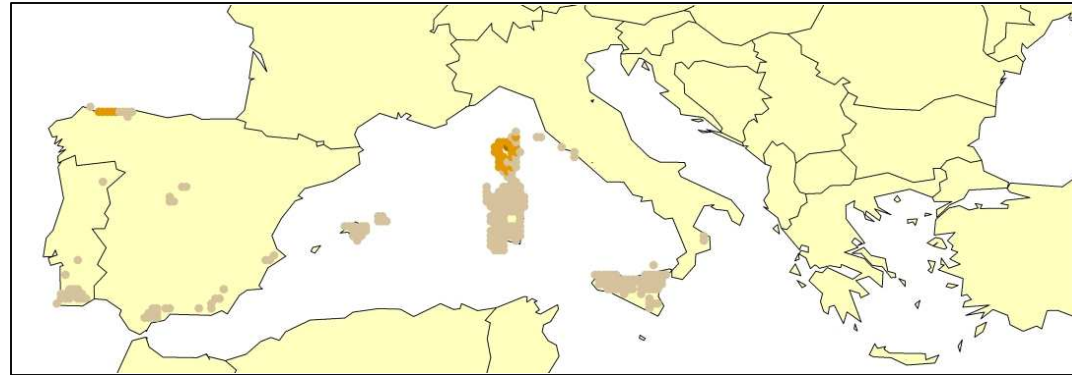
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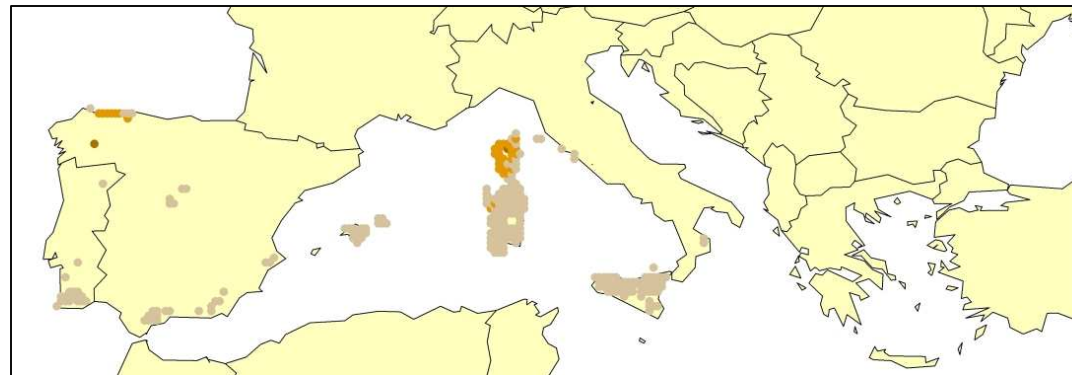
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- gain
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- remain

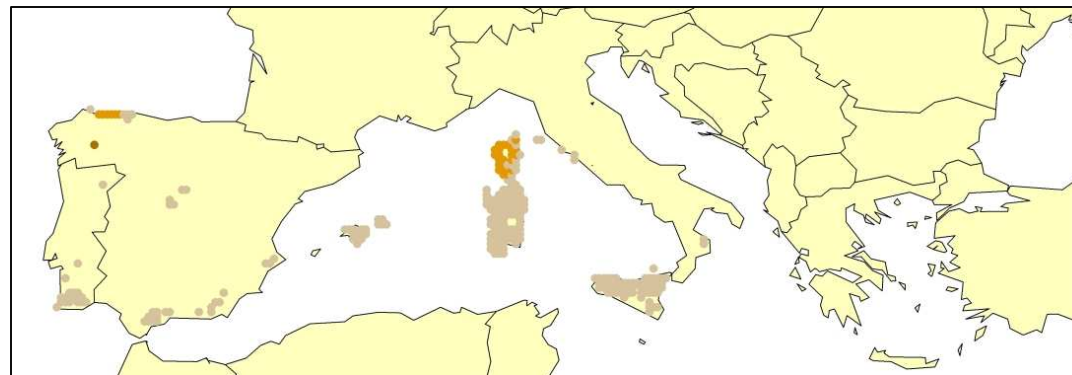
Podarcis tiliguerta



A1



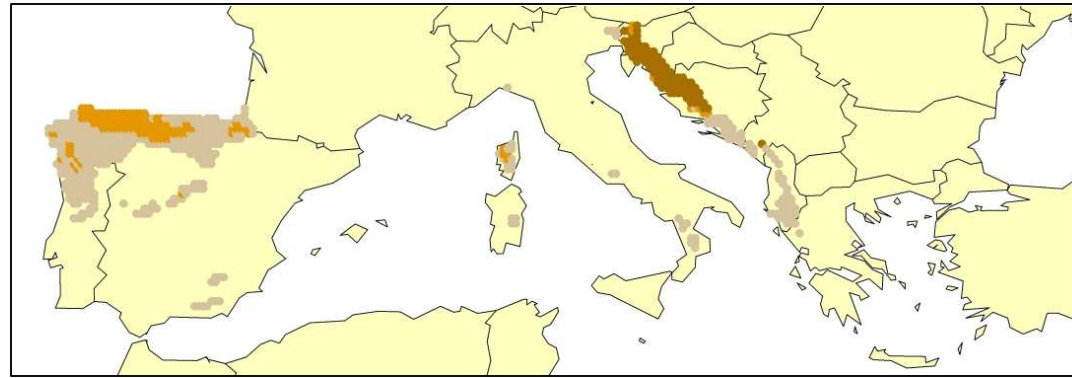
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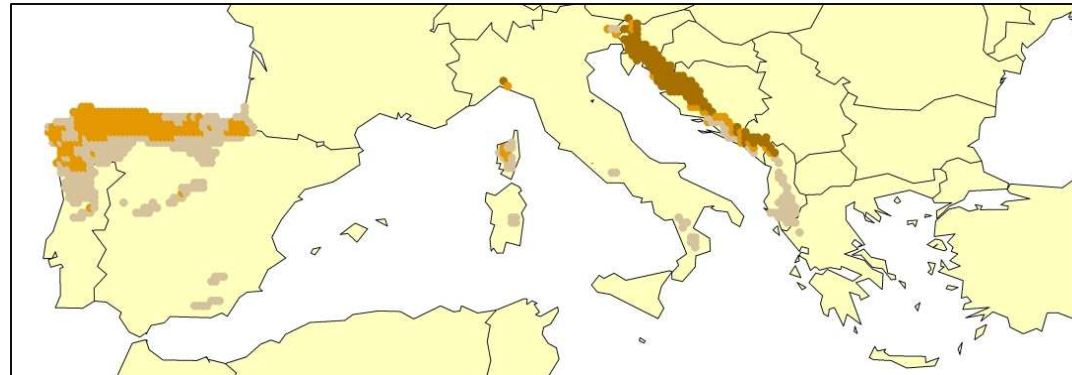
B1

- gain
- loss
- remain

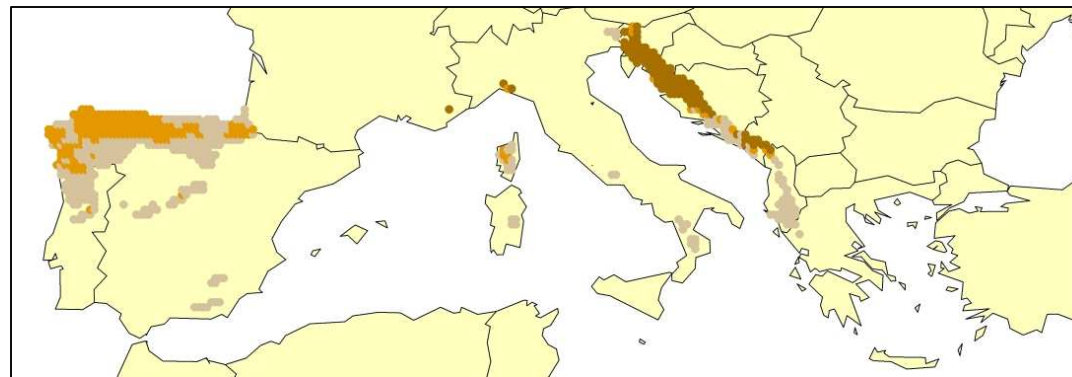
Vipera seoanei



A1



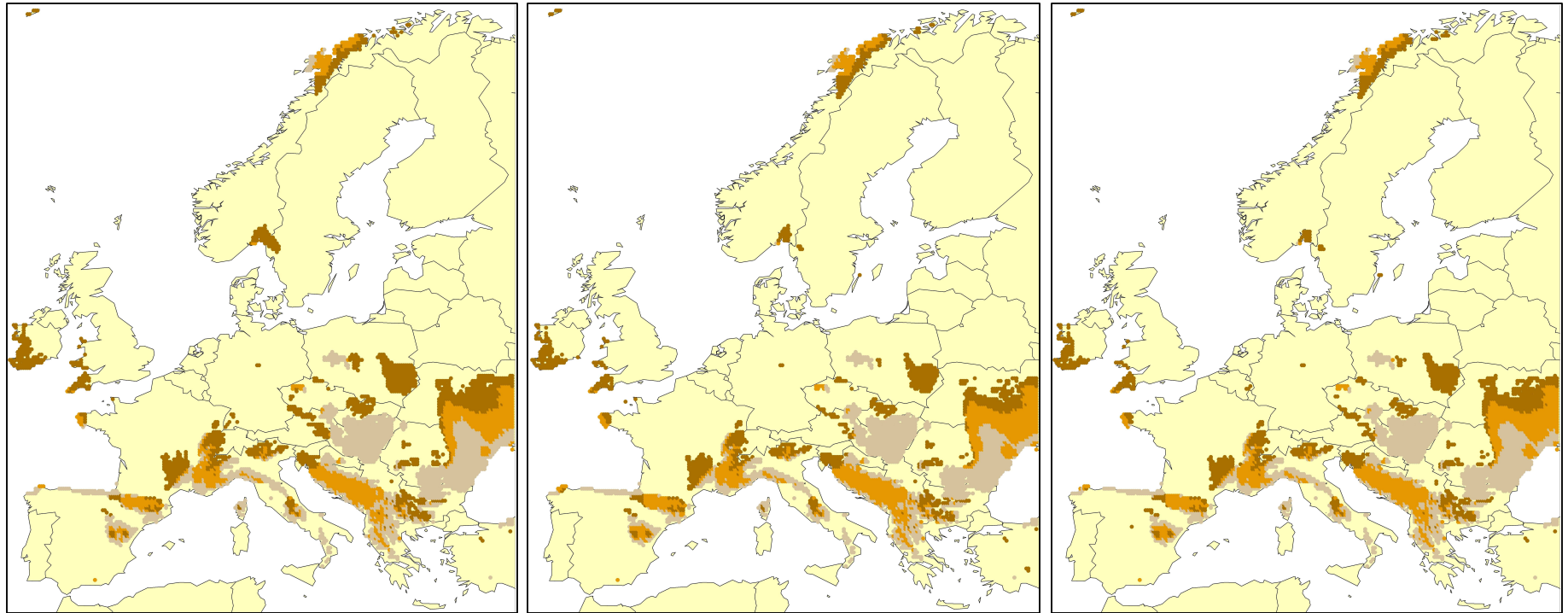
A2



B1

- gain
- loss
- remain

Vipera ursini



● gain ● loss ● remain