University of Debrecen
Juhász-Nagy Pál Doctoral School
Debrecen 2018

Theses of Doctoral (PhD) Dissertation

Biodiversity patterns and conservation priorities: case studies on the herpetofauna of Albania and the freshwater biodiversity of Europe

Márton Szabolcs

Supervisor: Szabolcs Lengyel

DE TTK

1949

BIODIVERSITY PATTERNS AND CONSERVATION PRIORITIES: CASE STUDIES ON THE HERPETOFAUNA OF ALBANIA AND THE FRESHWATER BIODIVERSITY OF EUROPE

BIODIVERZITÁSI MINTÁZATOK ÉS TERMÉSZETVÉDELMI PRIORITÁSOK: ESETTANULMÁNYOK ALBÁNIA HERPETOFAUNÁJÁRÓL ÉS EURÓPA ÉDESVÍZI BIODIVERZITÁSÁRÓL

Egyetemi doktori (PhD) értekezés

SZABOLCS MÁRTON

Témavezető
Dr. Lengyel Szabolcs
tudományos tanácsadó

DEBRECENI EGYETEM
Természettudományi Doktori Tanács
Juhász Nagy Pál Doktori Iskola
Debrecen, 2018

A doktori értekezés betétlapja

Ezen értekezést a Debreceni Egyetem Természettudományi Doktori Tanács a **Juhász Nagy Pál Doktori Iskola Kvantitatív és Terresztris Ökológia** doktori programja keretében készítettem a Debreceni Egyetem természettudományi doktori (PhD) fokozatának elnyerése céljából.
Debrecen, évszám.

…………………………
a jelölt aláírása

Tanúsítom, hogy **Doktorjelölt neve** doktorjelölt **2013-2016** között a fent megnevezett Doktori Iskola **Kvantitatív és Terresztris Ökológia** doktori programjának keretében irányításommal végezte munkáját. Az értekezésben foglalt eredményekhez a jelölt önálló alkotó tevékenységével meghatározóan hozzájárult. Az értekezés elfogadását javasolom.
Debrecen, évszám.

…………………………
a témavezető aláírása

A doktori értekezés betétlapja

BIODIVERZITÁSI MINTÁZATOK ÉS TERMÉSZETVÉDELMI PRIORITÁSOK: ESETTANULMÁNYOK ALBÁNIA HERPETOFAUNÁJÁRÓL ÉS EURÓPA ÉDESVÍZI BIODIVERZITÁSÁRÓL

BIODIVERSITY PATTERNS AND CONSERVATION PRIORITIES: CASE STUDIES ON THE HERPETOFAUNA OF ALBANIA AND THE FRESHWATER BIODIVERSITY OF EUROPE

Értekezés a doktori (Ph.D.) fokozat megszerzése érdekében
a Környezettudomány tudományágban
Írta: **Szabolcs Márton** okleveles Biológus

Készült a Debreceni Egyetem **Juhász-Nagy Pál Doktori Iskolája**
(**Kvantitatív és Terresztis Ökológia** doktori program) keretében

Témavezető:
Dr. Lengyel Szabolcs …………………………

A doktori szigorlati bizottság:
elnök: ………………………… …………………………
tagok: ………………………… …………………………
………………………… …………………………
A doktori szigorlat időpontja: …………………………
Az értekezés bírálói:
………………………… …………………………
………………………… …………………………

A bírálóbizottság:
elnök: ………………………… …………………………
tagok: ………………………… …………………………
………………………… …………………………
………………………… …………………………
………………………… …………………………

Az értekezés védésének időpontja: 2019. ……………… … .

Table of contents

General introduction and aims of the thesis

After more than a hundred years of research, there is still an unabated interest in the drivers of species distributions and species richness in geographic areas and on planet Earth. One of the longest recognized patterns in ecology is the decrease of biological richness from tropical areas to the poles (Willig et al. 2003). Still, this pattern is heavily modified by the unevenness of climatic and geographic conditions and the complexity of ecosystems. For example, terrestrial biodiversity hotspots are not necessarily close to the Equator, their locations are better explained by long-term climatic stability (Fjeldså & Lovett 1997; Myers et al. 2000; Zachos & Habel 2011). Biodiversity is also currently being redisitributed by anthropogenic climate change, habitat modification and the synergies between them (Brodie 2016; Frishkoff et al. 2016; Pecl et al. 2017).

A number of new research methods have become available recently for the analysis of patterns in the distribution and number of species and the factors determining these patterns. These methods rely on extensive databases organized from occurrence records of many species over large spatial scales. However, such databases often contain gaps or biases in space and time (Bird et al. 2014; Sillero et al. 2014a; 2014b). For example, areas with large conservation or aesthetic value, and proximity to roads and towns are more frequently surveyed than remote places or areas with a degree of degradation (Tulloch et al. 2012).

These deficiencies can lead to biases in species distribution modelling and differences in model performance in ecological analyses (Rocchini et al. 2011). For example, Tsianou et al. (2016) demonstrated that patterns in species richness of selected vertebrates were better explained by landscape features if atlas-based range maps were used, whereas climate was more important if expert-based maps were considered. The optimal outcome of spatial conservation planning can also vary based on different quality of data, which can lead to wasted efforts (Carvalho et al. 2010).

Systematic conservation planning (SCP) is a stakeholder-focused process based on geographic information systems (GIS), which aims to maximize conservation benefits from protected area networks at the lowest possible cost (Moilanen et al. 2009). Although protected areas are the backbone of the conservation of biodiversity, the designation of such areas has traditionally been guided by socioeconomic or aesthetic criteria rather than by scientifically sound principles (Pressey et al. 1993). This situation is blissfully changing towards a more science-based selection of sites and surely this will be the standard in the future (Moilanen et al. 2008; Ardron et al. 2010).

Although SCP is increasingly applied in marine and terrestrial ecosystems (Saunders et al. 2002; Abell et al. 2007), freshwater biodiversity is rarely in the focus of such efforts (Linkeet al. 2011) and terrestrial protected areas do not match areas of high freshwater biodiversity (Herbert et al. 2010). The few prioritisations for freshwater biodiversity were usually based on river sections or grids as planning units, typically on modelled ranges of species of one taxonomic group (most frequently fish) and on the regional spatial scale (Abellán et al. 2007; Thieme et al. 2007; Esselman & Allan 2011; Dolezsai et al. 2015). The uptake of these studies in conservation policy is slow as they are limited by scale and are rarely based on water management units (catchments) (Collares-Pereira & Cowx 2004). There are few published exercises at larger scales and where catchments are used as planning units (Lawrence et al. 2011; Holland et al. 2012). Thus, for effective conservation, large-scale (preferably continental) conservation prioritisations based on data on the occurrence of multiple freshwater taxon groups in catchments as planning units are urgently needed, which simultaneously consider the threat status and range size of species and the uniqueness of species assemblages.

The general aim of my work is to contribute to the knowledge of the ecological properties of rarely studied areas and taxonomic groups and to their more efficient conservation. The specific aim of my dissertation was to demonstrate a more-or-less complete arch of studies in conservation ecology, ranging from the discovery of species occurrences and the organizing of species occurrence records into large-scale spatial databases, through the study of biogeographic patterns and analysis of factors responsible for these patterns, to the application of large-scale databases in the spatial prioritization of conservation effort.

The aim of the work described in Chapter 1 was to obtain information on the occurrence, distribution and diversity patterns of species and the factors influencing these patterns in Albania, a hardly explored but extraordinarily species-rich country. The concrete aims of the study were (i) to compile a complete checklist of the amphibian and reptile species of Albania, (ii) to prepare up-to-date range maps well supported by observations, and (iii) to analyze patterns in species diversity in the light of the most important climatic and environmental factors.

Chapter 2 describes how we confirmed the occurrence of the Italian wall lizard (*Podarcis siculus*) and the Syrian spadefoot toad (*Pelobates syriacus*) in Albania. This is significant because the occurrence of these species in the country had not been confirmed before and because it demonstrates what elements (observations) make up the large-scale databases, such as the one studied in Chapter 1.

The aim of the study described in Chapter 3 was to identify critical catchments, i.e., areas of key importance for the conservation of freshwater biodiversity in Europe. The analysis was based on distribution data on freshwater fish, mollusc, dragonfly and plant species and considered the threat status and range size of species, and the proportion of endemic species in the catchments.

In Chapter 4, my aim was to extend the analysis in Chapter 3 to identify areas that have high priority for appropriately ensuring the conservation of freshwater biodiversity. We also studied how these priorities change if we consider the currently protected areas and how the relationship between conservation priority and the current proportion of protection of catchments varies in different areas of Europe. A further aim based on these results was to identify high-priority areas that are not protected appropriately and which are in urgent need of conservation.

Chapter 1

Distribution and diversity of amphibians and reptiles in Albania

Introduction

The distribution of amphibian and reptile species has been in the centre of scientific interest in many European countries for a long time. However, the first atlas on the geographic distribution of these species in Europe was published only in 1997 (Gasc et al. 1997). The first update of this work was published recently and it integrated many new records (Sillero et al. 2014a; 2014b). However, there are still countries and regions with little, poor or no data available. The Balkan Peninsula was such a region (Sillero et al. 2014a), although several comprehensive descriptions on the distribution of amphibians and reptiles have been published recently. Up-to-date distribution maps with species richness patterns were published from Greece (Valakos et al. 2008), Bulgaria (Stojanov et al. 2011), the Former Yugoslav Republic of Macedonia (Sterijovski et al. 2014; Uhrin et al. 2016), Romania (Cogălniceanu et al. 2013a; 2013b), Serbia (Vukov et al. 2013; Tomovič et al. 2014), and Jablonski et al. (2012) published a preliminary study on the herpetofauna of Bosnia and Herzegovina.

Albania is an exception because this country remained largely unexplored, even though it lies in the globally significant Mediterranean biodiversity hotspot (Myers et al. 2000; Griffiths et al. 2004). The earliest works on the herpetofauna of Albania are from the early 20th century by scientists mostly from Austria-Hungary (e.g. Kopstein & Wettstein 1920; Werner 1920) There were only a few studies between 1945 and 1990 (e.g. Frommhold 1962, see Haxhiu 1994; 1998). More recent works provided coarse-scale distribution data (Bruno 1989; Haxhiu 1994; 1998) and a species list (Dhora 2010) for the entire area of the country. Recently an increased number of publications on the distribution, ecology and systematics of particular species and articles on the fauna of some regions provided more insight on the herpetofauna of Albania (e.g. Farkas & Buzás 1997; Haxhiu & Vrenozi 2009; Oruçi 2010a; 2010b; Jablonski 2011).

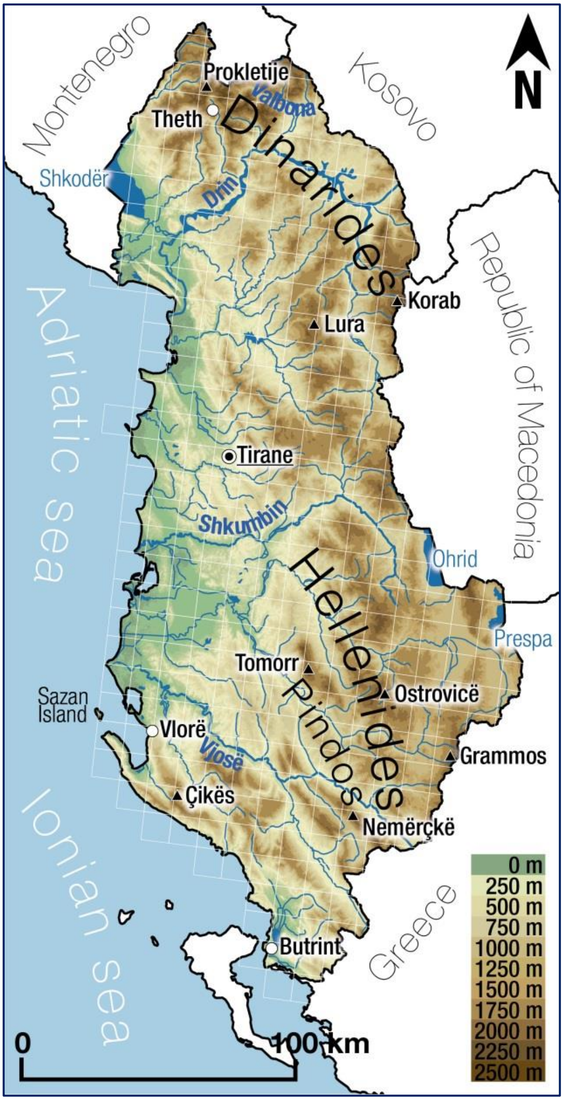


Figure 1.1. Geographic map of Albania indicating toponymics mentioned in the text.

The territory of Albania has a great importance in the understanding of Mediterranean biogeography and is the least known in Europe, thus studies of the country's biodiversity are highly warranted (Sillero et al. 2014a). The territory of Albania (28,748 km^2) is mostly mountainous (70% of the territory, up to 2746 m above sea level). (Fig. 1.1). Orogenic processes induced by the Adriatic microplate colliding with the Eurasian plate largely shaped the topography of the country's territory (Aliaj et al. 2001). The geological and topographic complexity is reflected by the enormous diversity of plants and vegetation types (Barina et al. 2017). In addition, the territory of Albania was a speciation centre in the Miocene-Pliocene (Médail & Diadema 2009; Pabijan et al. 2015). Moreover, glacial periods in the Pleistocene little influenced the country's territory, which also held refugial areas. Finally, the country's territory is roughly divided into two halves by the Hellenide mountain range that runs north to south, a significant barrier for the dispersal of many species (Gvoždík et al. 2010; Psonis et al. 2017). These factors led to increased allopatric speciation and diversification in several lineages The Western Balkans and Albania are home to several endemic amphibians such as *Triturus macedonicus*, *Lissotriton graecus*, *Pelophylax epeiroticus*, *P. shqipericus*, *Rana graeca* and reptiles such as *Anguis graeca*, *Podarcis ionicus*, *Dinarolacerta* sp., *Dalmatolacerta oxycephala*, and *Vipera graeca*. Patterns of hidden genetic diversity have also been detected (e.g. for *Lacerta viridis* complex, *Natrix tessellata*, *Vipera ammodytes*, Ursenbacher et al. 2008; Guicking et al. 2009; Marzahn et al. 2016). Despite its relatively small size, the country is home to half of the amphibian and two-thirds of the reptile species of the Balkan Peninsula (Sillero et al. 2014a). These previous findings suggest that the biogeographic explanation and effective conservation of the current high diversity of amphibian and reptile species of the Western Balkans hotspot is not possible without a thorough and up-to-date understanding of the distribution and diversity patterns of amphibian and reptile species in Albania.

The objectives of our work were to assemble a new and complete checklist of the amphibian and reptile species of Albania, to prepare up-to-date range maps for every species, and to analyze the patterns of diversity as a function of climatic and environmental factors. Such an update and synthesis of previous and new knowledge is justified because of the country's great importance in the western

Balkan biodiversity hotspot and the scarcity of knowledge on the distribution of its species.

Material and methods

Study groups

Amphibians and reptiles are often treated together and the majority (but not all) of herpetologists study both of these groups. During data collection, we handled records of amphibians and reptiles together, but we performed analyses separately for the two groups for two main reasons. One reason is that the two groups are ecologically very different. The life cycle of most amphibians has an obligate aquatic stage, and amphibians are usually associated with wet/moist areas and show nocturnal activity. In contrast, reptiles often live under dry conditions and are diurnal. Therefore, it was highly likely that the environmental variables influencing species distributions would differ between the two groups. The other reason is that extremely little had been known on the herpetofauna of Albania and we wanted to raise attention to and provide detailed description for both the amphibian and the reptile fauna of the country. In summary, results are given separately for the two groups in my thesis, which corresponds to our two separate publications (Mizsei et al. 2017b; Szabolcs et al. 2017). Because the methods of data collection and statistical analysis were similar for the two groups, these are given jointly below.

Study area

We defined our study extent as the terrestrial areas of Albania (Fig. 1.1). The high geomorphological complexity is the result of the Dinarid mountains framing the East and North of the country (e.g. Korab, Koritnik, Prokletije) while the other mountains belong to the Hellenides (e.g. Valamara, Tomorr, and mountain ranges such as Nemërçkë in the Pindos system). The West of Albania is usually lowland with lagoons near the sea. The largest lakes in the Balkans, Shkodra, Ohrid and the Prespas are partially covered by Albania. Ten major river systems (e.g. Drin, Shkumbin, Vjosa) and 150 smaller systems divide the mountain ranges. The climate of lower-lying areas is warm Mediterranean and oceanic whereas mountain areas have a cold Alpine climate. The major type of vegetation of the country is macchia, interspersed with deciduous birch and karst forests at lower elevations and coniferous forests and alpine grasslands at higher elevations.

Data collection

Several sources of information were used to establish and populate the database on the distribution of amphibian and reptile species in Albania. As a first step, we used data available in the literature sources and then the data available in online and museum databases. These datasets were supplemented by species occurrence data that we collected in the field in more than 20 expeditions to the country and by data from other herpetologist experts.

For literature sources, we used both the scientific and the grey literature (e.g. technical reports) for data on species occurrences. To derive spatial information for the localities, we used the original coordinates as they were provided by the authors. In cases when we found maps, we georeferenced species localities in QuantumGIS 1.8 (GDAL plugin). For localities for which only the name of the nearest human settlement was given, we estimated the coordinates using Google Maps, the GeoNames database or other online information sources We estimated localities and marked it in a point shapefile when a location was identified with high certainty, for example, a mountain ridge, a small lake, a river section near a village. For the detailed lists of publications used as data sources see Mizsei et al. 2017b and Szabolcs et al. 2017.

Occurrence records were also obtained from museums (e.g. Hungarian Natural History Museum) and three information sources on the web: the Global Biodiversity Information Facility, iNaturalist, and TrekNature. Finally, we asked our extensive network of fellow field herpetologists with experience in Albania through personal communication for occurrence records via the internet forum Fieldherping.eu.

To fill gaps in the dataset, we took 21 expeditions to different parts of Albania to collect occurrence records from 2009 to 2017. We specifically targeted rare species with less than 10 records and areas with scarce or no records. Fieldwork was conducted mostly in the summer but also in April and October, and one expedition was in the winter. We surveyed habitats suitable for amphibians and reptiles for 30 minutes using visual and acoustic searches. For each animal found, we recorded the coordinates by GPS and took photographs of the animals and their habitat.

All the records for which coordinates were known or derived and which were collected from either of the above sources were stored in a point shape file. We supplemented these records with various other types of information such as year and date of the observation, source of record among others.

Taxonomic considerations

We used the guide by Arnold & Ovenden (2002) to identify animals in the field. We used standard sources for taxonomy (Sillero et al. 2014a; Speybroeck et al. 2016; Frost 2017). Exceptions were three species of *Pelophylax* (*P*. *kurtmuelleri*, *P*. *epeiroticus*, *P*. *shqipericus*) that were impossible to determine by morphology in the field and because they hybridize with each other. Although *Pelophylax* species can be can be told apart by their calls (Lukanov et al. 2015) we rarely heard them and it was not known for literature observations how the animals were identified in the field, so we merged them into *Pelophylax* spp. (e.g. Mester et al. 2015). Similarly, two *Bufotes* species likely occurring in Albania (Özdemir et al. 2014) are very difficult to tell apart in the field, so these were merged as *B. viridis/variabilis*. Two species of *Anguis*, *A. graeca* and *A. fragilis* and two species of *Podarcis*, *P. tauricus* and *P. ionicus* were also merged for similar reasons. Finally, the name *Pelophylax kurtmuelleri* was used instead of *P. ridibundus*, *Lissotriton graecus* was used instead of *L. vulgaris* and *Vipera graeca* was used instead of *V. ursinii* because these Balkan lineages have been elevated to species level in recent molecular studies (Ferchaud et al. 2012; Pabijan et al. 2017; Dufresnes et al. 2017; Mizsei et al. 2017). Because we primarily focused on terrestrial species, we did not consider the four sea turtles (*Caretta caretta*, *Chelonia mydas*, *Dermochelys coriacea* and *Erethmochelys imbricata*) that are known to occur temporarily along the Albanian coast (Casale & Margaritoulis 2010).

Spatial analyses

We first aggregated point records into larger units (grid cells) for analyses. We used the 10 × 10 km grid cells from the system of the European Environmental Agency for compatibility with other databases based on EEA grids. Albania's territory was covered by 349 of such cells. We uniformly determined elevation above sea level for all occurence points from a global database that had a resolution of 90 m (Jarvis et al. 2008).

Spatially uneven sampling can lead to spatial autocorrelation which is a common bias in point occurrence datasets (Rocchini et al. 2011). We studied these biases in two ways. First, we calculated Moran's global I index to quantify spatial autocorrelation in the number of observations in each cell. Values of Moran's I that are significantly greater than 0 indicate that the observations are clustered (i.e., sampling effort is greater than average) and values less than 0 indicate that observations are dispersed (i.e., sampling effort is lower than average). To specifically identify sampling hotspots and coldspots, we also used the Gi* index (Ord and Getis 1995). Significantly high index values indicate sampling hotspots (effort higher than average) and low index values indicate sampling coldspots (effort lower than average). These indices were calculated in ArcGIS 10.0 (ESRI 2010).

As dependent variable in analyses of diversity patterns, we defined "presence/absence" based on whether a grid cell had at least one observation of any species (binary variable, yes/no). We also calculated the Shannon diversity based on relative frequencies of species within each grid cell (R environment, 'vegan' package).

Variables for climate and environment

We chose several climatic and environmental variables that were likely to explain the occurrence and diversity of amphibian and reptile species in Albania (Table 1.1.). To characterize climate, we used 19 variables from the database "WorldClim" (Hijmans et al. 2005). Because some of these variables were highly correlated, we reduced them into four principal components using the 'cluster' R package (Table 1.1.). The principal components explained 99% of the total variance. For environmental variables, we used habitat and altitudinal diversity as well as distance from the sea. To characterize habitat diversity, we calculated the Shannon diversity of land cover types in each grid cell based on the CORINE Land Cover database (European Environmental Agency 2007). Altitudinal diversity was estimated by the standard deviation (S.D.) of elevations above sea level within each grid cell from a 90-m-resolution database (Jarvis et al. 2008). For distance from the sea, we calculated the minimum distance between the sea coast and the centroid of each grid cell. We used QGIS and its appropriate plugins for these calculations.

Table 1.1. Environmental variables used in these studies.

Predictor	Description	Data source
BIO PC1	**"Temperature" principal component** BIO1 = Annual Mean Temperature BIO6 = Min Temperature of Coldest Month BIO11 = Mean Temperature of Coldest Quarter	Hijmans et al. 2005
BIO PC2	**"Precipitation" principal component** BIO12 = Annual Precipitation BIO16 = Precipitation of Wettest Quarter BIO19 = Precipitation of Coldest Quarter	Hijmans et al. 2005
BIO PC3	**"Temperature variation" principal component** BIO2 = Mean Diurnal Range (Mean of monthly (max temp - min temp) BIO4 = Temperature Seasonality (standard deviation *100) BIO7 = Temperature Annual Range (BIO5-BIO6))	Hijmans et al. 2005
BIO PC4	**"Precipitation variation" principal component** BIO9 = Mean Temperature of Driest Quarter BIO10 = Mean Temperature of Warmest Quarter BIO15 = Precipitation Seasonality (Coefficient of Variation)	Hijmans et al. 2005
CORINE DIV	Shannon diversity of CORINE Land cover in 10×10 km cells	European Environment Agency
ALT MEAN	Mean of altitude values in 10×10 km cells, calculated from the SRTM near 90 m data	CGIA-CSI
ALT SD	Standard deviation of altitude values in 10×10 km cells, calculated from the SRTM near 90 m data	CGIA-CSI
SEA DIST	Min distance of 10×10 km cells centroids from sea coast	present study

General linear models

We studied the effects of climatic and environmental variables on the presence/absence and on the diversity of amphibian and reptile species in a model selection process based on generalized linear mixed-effects models (GLMM; ‘lme4’ package in R; Pinheiro & Bates 2000). For presence/absence, we specified binomial error distribution, and for diversity, we constructed a GLMM with the Markov chain routine (Hadfield 2010). We used the ‘dredge’ function in the ‘MuMIn’ R package to fit and rank models with all possible combinations of the independent variables (Bartón 2011). Three random factors were used, the first was cell ID, to ensure that spatial autocorrelation is controlled for, and the second was Gi* to control for sampling bias. The third random factor was species ID, nested in taxonomic order, to reduce the non-independence arising from the phylogenetic relationships between the species.

After fitting all possible models, we ranked them based on Akaike’s information criterion (AICc, adjusted for small sample sizes) and evaluated the relative importance of the climatic and environmental variables (Burnham & Anderson 2002) We used the best models (delta AICc < 2 from the best model) to calculate model-averaged estimates for parameters and their standard errors. The R statistical environment was used for all these analyses (R Core Team 2015).

Results

Distribution and diversity of amphibians

Our database held 1097 observations of amphibians, the first of which were from 1920. The number of observations was low during most of the 20th century until the 1990s, when Haxhiu's (1994) work was published (Fig. 1.2.). Observations from literature sources made up 49% of all records in our database and 51% of all records are new observations published in our work for the first time (Table 1.2., Fig. 1.3a.). Of the new observations (N = 555), 482 (87%) were obtained during our field work in over 20 expeditions. The rest of the observations were from other herpetologists (N= 47 observations), museum catalogs (N= 18), and online databases (N = 8). The number of observations per species ranged between three (*Pelobates syriacus*) and 339 (*Pelophylax* spp.). Roughly half of the observations were new for most species, and the number of new (previously unpublished) observations was at least one for each species except for *P. syriacus* (Table 1.2.). This species was present in only one grid cell, meaning that it was the rarest of all species. *Pelophylax* spp. were the most widespread with at least one observation in 181 grid cells. Of all the grid cells that covered the country's territory (N = 349), 238 had at least one observation of any amphibian species. Ten species had country-wide ranges (*Triturus macedonicus*, *Lissotriton graecus*, *Bufo bufo*, *Bufotes viridis*/*variabilis*, *Rana dalmatina*, *R. graeca*, *Pelophylax* ssp., *Hyla*

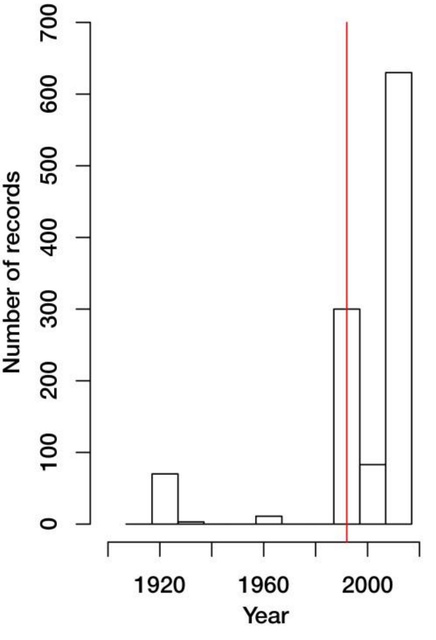


Figure 1.2. Number of amphibian records by year of publication (published sources) or year of data collection (unpublished sources). Vertical line indicates the year when the former isolationist political system ended in Albania (1991).

arborea), although two of them were more likely to occur in mountains (*Salamandra salamandra*, *Bombina variegata*). The others had limited ranges, for example, *Salamandra atra* was restricted to the Prokletije Mountains, *Ichthyosaura alpestris* and *Rana temporaria* to the mountains of East Albania and *Pelobates syriacus* to Prespa Lake. Detailed range maps of each amphibian species are presented in the supplementary material of Szabolcs et al. (2017).

Table 1.2. List of amphibians in Albania with their number of records and presences in 10×10-km cells.

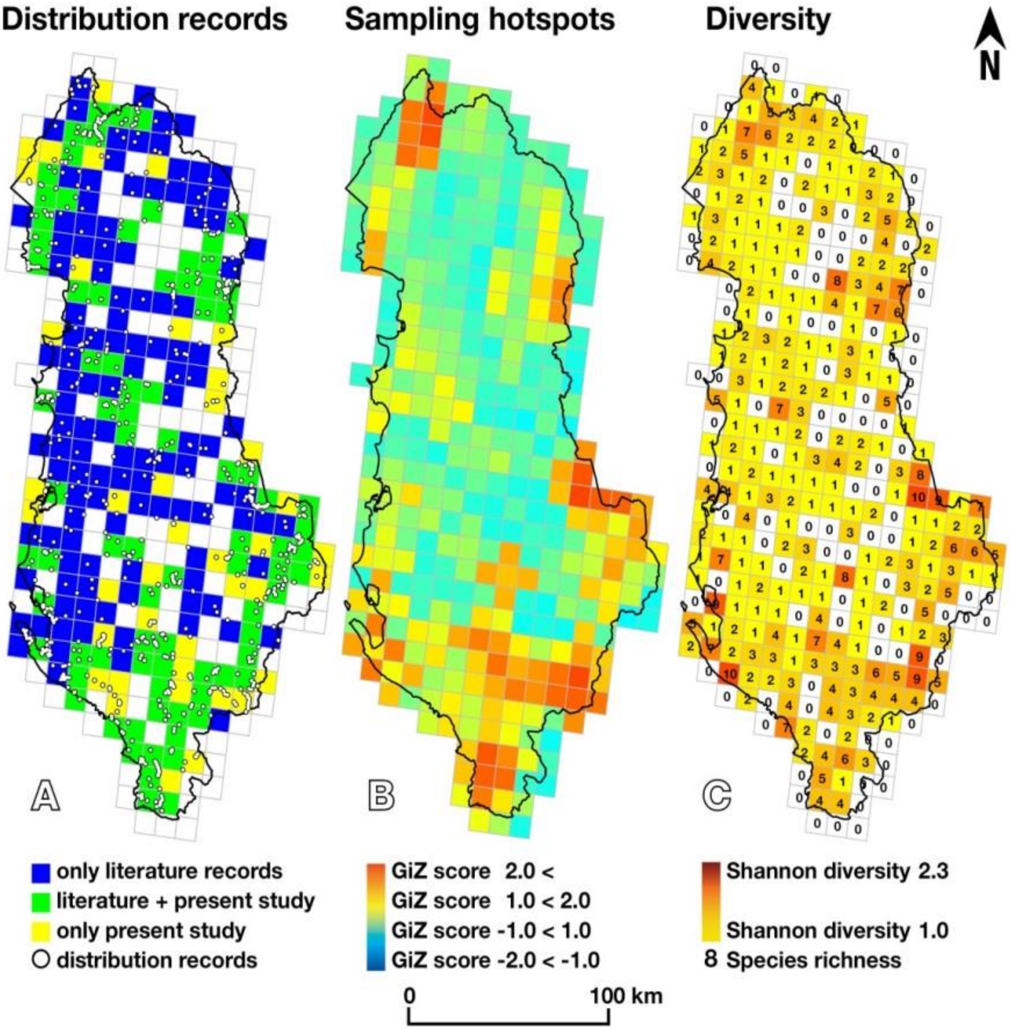
Species	**Number of all records**	**Published records**	**New records**	**N of presences**
Bombina variegata	136	46	90	67
Bufo bufo	70	27	43	50
Bufotes viridis/variabilis	96	36	60	53
Hyla arborea	48	26	22	33
Ichthyosaura alpestris	63	43	20	31
Lissotriton graecus	55	34	21	47
Pelobates syriacus	3	3	0	1
Pelophylax spp.	399	221	178	181
*Pelophylax epeiroticus**	8	5	3	5
*Pelophylax kurtmuelleri**	59	54	5	41
*Pelophylax shqipericus**	25	21	5	18
Rana dalmatina	54	28	27	35
Rana graeca	69	16	53	43
Rana temporaria	16	15	1	14
Salamandra atra	6	2	4	3
Salamandra salamandra	42	31	11	40
Triturus macedonicus	39	14	25	29
Total	**1097**	**539**	**558**	**238**

*We merged the three Pelophylax species in the analyses. Details about their records are only given here.

Moran's I index values showed a spatially clustered sampling effort (Z = 4.064, P < 0.0001). The Getis Ord Gi*index showed several sampling effort hotspots, mostly in areas with natural heritage sites (Prokletije, Pindos mountains; Butrint, Ohrid, Prespa lakes, Vlorë coast), whereas there were no coldspots for sampling effort (Fig. 1.1., Fig. 1.3B.).

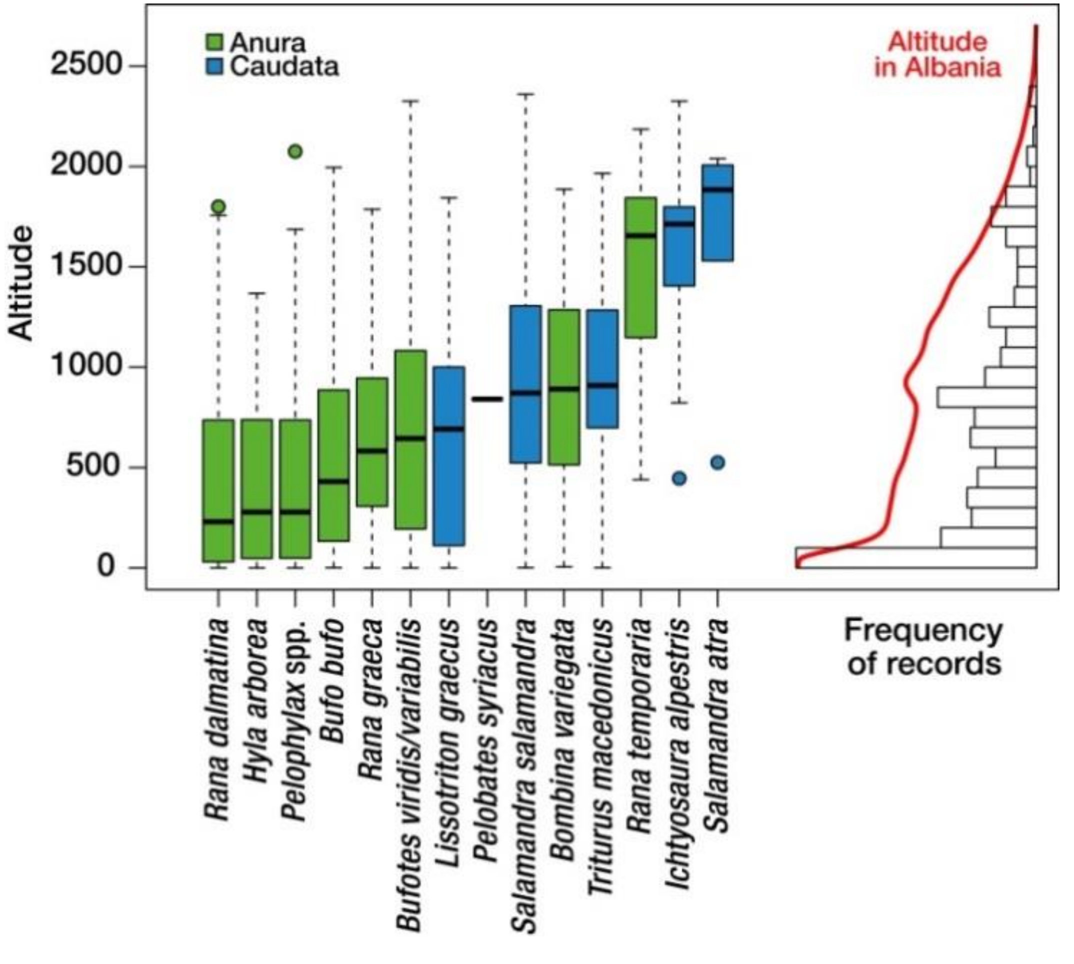
The maximum number of species in a cell was 10 (N= 2 cells), whereas the average was 1.8 ± (S.E.) 0.11 species per cell. High numbers of species were found in cells covering lakes Ohrid and Prespa, mountains Çikës, Grammos, Korab, Lura, Prokletije and the Vlorë coast (Fig. 1.1., Fig. 1.3C.).

Figure 1.3. A: Sources of occurrence records of amphibian species, B: sampling hotspots (GiZ score > 1.0) and coldspots (GiZ score < -1.0), C: amphibian species richness (numbers) and Shannon diversity index (shading) in Albania on a 10x10 km grid.



The altitudinal range of most amphibian species was below 1200 m and only a few alpine species were found higher than 1200 m (Fig. 1.4.).

Figure 1.4. Altitudinal distribution of amphibian species and frequency of occurrence records by altitude in Albania. Box-and-whiskers plots show the median (horizontal line), the 25th and 75th percentile (bottom and top of box, respectively), minimum and maximum values (lower and upper whiskers, respectively) and outliers (circles). The red line is the frequency distribution of altitudinal values in Albania.



GLMMs showed that for the presence of amphibian species in a cell, habitat (CORINE land cover) diversity, precipitation (PC2) and temperature variation (PC3) were important. For diversity, habitat diversity and precipitation were important for both presence and diversity (Table 1.3.). Each of the best models contained habitat diversity and either PC2, PC3 or both (Table 1.4.). The presence/absence of amphibian species in cells was negatively influenced by PC2 (precipitation) and PC3 (temperature variation) and positively by habitat diversity based on model-averaged parameter estimates (Table 1.5., Fig. 1.5.) The diversity of amphibian species was influenced only by habitat diversity, and its effect was positive (Table 1.5., Fig. 1.5.).

Table 1.3. The importance of predictors in the GLMM model for presence/absence and for Shannon diversity of amphibians in Albania. See table 1.1. for the description of the variables.

Presence		**Shannon diversity**	
Predictor	**Importance**	**Predictor**	**Importance**
CORINE DIV	1.000	CORINE DIV	1.000
BIO PC2	0.903	BIO PC2	0.427
BIO PC3	0.756	BIO PC4	0.291
BIO PC1	0.217	ALT SD	0.181
SEA DIST	0.204	ALT MEAN	0.087
ALT SD	0.122	BIO PC3	0.069
ALT MEAN	0.095	SEA DIST	0.000
BIO PC4	0.000	BIO PC1	0.000

Table 1.4. Parameter estimates and AIC values of the best GLMM models with substantial support (ΔAICc< 2) fitted on the presence and Shannon diversity of amphibians in Albania.

Response	**CORINE DIV**	**BIO PC2**	**BIO PC3**	**BIO PC1**	**SEA DIST**	**ALT SD**	**ALT MEAN**	**BIO PC4**	**ΔAICc**
Presence	1.375	-0.089	-0.104						0.000
	1.295	-0.088							1.222
	1.385	-0.096	-0.102			0.001			1.391
	1.379	-0.147		0.089	-0.000				1.568
	1.388	-0.089	-0.105	0.014					1.679
	1.358		-0.105						1.856
	1.383	-0.087	-0.103				0.000		1.896
	1.379	-0.084	-0.111		0.000				1.939
Diversity	0.317								0.000
	0.319	-0.029							0.154
	0.323							-0.043	0.455
	0.324	-0.031						-0.046	0.739
	0.294	-0.032				0.000			1.139
	0.324						0.000		1.193
	0.292					0.000			1.661
	0.333		-0.025						1.761

Table 1.5. Model averaged parameter estimates of GLMM fitted on amphibian presence and MCMCglmm fitted on Shannon diversity of amphibians. Significant parameter estimates are indicated in bold.

Response	**Main effect**	**Estimate**	**S.E.**	**z**	***P***
Presence	(Intercept)	-5.172	0.804	6.428	0.000
	CORINE DIV	**1.367**	**0.227**	**6.015**	**0.000**
	BIO PC2	**-0.096**	**0.046**	**2.093**	**0.036**
	BIO PC3	**-0.105**	**0.053**	**1.982**	**0.047**
	BIO PC1	0.053	0.049	1.072	0.284
	SEA DIST	0.000	0.000	0.766	0.444
	ALT SD	0.001	0.000	1.109	0.267
	ALT MEAN	0.000	0.000	0.326	0.745
	BIO PC4	0.000	0.000	-0.191	0.848

Response	**Main effect**	**Estimate**	**Lower 95% CI**	**Upper 95% CI**	***P***
Diversity	(Intercept)	0.224	-0.636	1.001	0.558
	CORINE DIV	**0.325**	**0.175**	**0.495**	**0.001**
	BIO PC2	-0.048	-0.101	0.004	0.078
	BIO PC4	-0.056	-0.145	0.028	0.228
	ALT SD	0.000	-0.001	0.001	0.772
	ALT MEAN	0.000	-0.001	0.000	0.606
	BIO PC3	-0.001	-0.069	0.063	0.962
	SEA DIST	0.000	0.000	0.000	0.474
	BIO PC1	0.037	-0.046	0.108	0.302

Distribution and diversity of reptiles

Our database held 3731 observations of reptiles, the first of which was from 1918. The number of records was low during most of the 20th century until the collapse of the isolationist political system in the 1990s (Fig. 1.6.). The majority (N = 2706 or 73%) of the observations were from the literature. Our field work added 885 new observations or 24% of the total. In addition, we gathered N = 97 observations (3%) from other herpetologists, N = 33 observations (1%) from online databases and *N* = 10 observations (0.3%) from museum catalogs (Table 1.6., Fig. 1.7.). The number of observations per species ranged between one (*Tarentola mauritanica)* and 379 (*Testudo hermanni*). The number of grid cells with observations per species ranged between one (*T. mauritanica*) and 191 (*Vipera ammodytes*). The most widespread species were *Anguis fragilis/graeca*, *Dolichophis caspius*, *Lacerta trilineata*, *Lacerta viridis* complex, *Natrix natrix*, *N. tessellata*, *Podarcis muralis*, *T. hermanni*, *Zamenis longissimus*, and *V. ammodytes*. Species that occurred in less than 10% of the grid cells were *Dalmatolacerta oxycephala*, *Dinarolacerta montenegrina*, *Eryx jaculus*, *Lacerta agilis*, *Podarcis melisellensis*, *P. siculus*, *Testudo graeca*, *T. marginata*, *T. mauritanica*, *Vipera berus*, *V. graeca*, *V. ursinii*, and *Zootoca vivipara*. The range of seven species was fragmented (*Ablepharus kitaibelii*, *Algyroides nigropunctatus*, *Coronella austriaca*, *E. jaculus*, *Mediodactylus kotschyi*, *Platyceps najadum* and *Xerotyphlops vermicularis*) and the ranges of 14 species were on the periphery of their entire distribution (*D. oxycephala*, *E.*

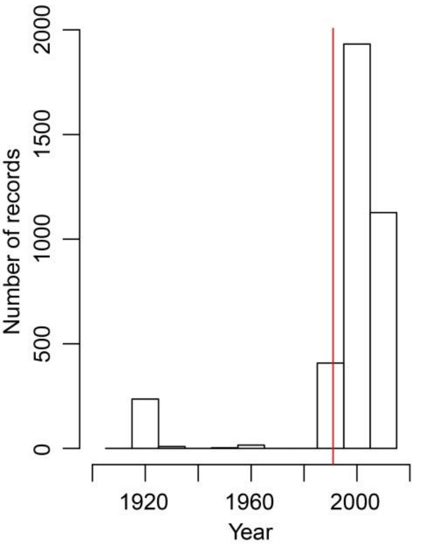


Figure 1.6. Number of reptile records by year of publication (published sources) or year of data collection (unpublished sources). Vertical line indicates the year when the former isolationist political system ended in Albania (1991).

jaculus, *L. agilis*, *P. melisellensis*, *P. siculus*, *T. graeca*, *T. marginata*, *T. mauritanica*, *V. berus*, *V. ursinii*, *X. vermicularis* and *Z. vivipara*). With the exception of *L. trilineata* and *V. graeca*, the number of published records exceeded that of the unpublished records.

Table 1.6. List of reptiles in Albania with their number of records and presences.

Species	Number of all records	Published records	Unpublished records	N of presences
Ablepharus kitaibelii	24	18	6	22
Algyroides nigropunctatus	105	80	25	81
Anguis fragilis/graeca	196	176	20	147
Coronella austriaca	39	30	9	33
Dalmatolacerta oxycephala	2	2	0	2
Dinarolacerta montenegrina	7	2	5	3
Dolichophis caspius	182	158	24	136
Elaphe quatuorlineata	98	87	11	75
Emys orbicularis	164	139	25	101
Eryx jaculus	8	4	4	5
Hemidactylus turcicus	47	39	8	32
Hierophis gemonensis	78	50	28	65
Lacerta agilis	10	7	3	7
Lacerta trilineata	106	45	61	135
Lacerta viridis complex	182	134	48	70
Malpolon insignitus	132	101	31	97
Mauremys rivulata	68	54	14	44
Mediodactylus kotschyi	19	15	4	17
Natrix natrix	241	192	49	173
Natrix tessellata	157	129	28	118
Platyceps najadum	49	35	14	44
Podarcis erhardii	46	11	35	24
Podarcis melisellensis	10	7	3	8
Podarcis muralis	298	218	80	186
Podarcis siculus	3	1	2	2
Podarcis tauricus/ionicus	150	105	45	95
Pseudopus apodus	101	81	20	75
Tarentola mauritanica	1	1	0	1
Telescopus fallax	78	73	5	63
Testudo graeca	2	2	0	2
Testudo hermanni	379	238	141	186
Testudo marginata	22	15	7	8
Xerotyphlops vermicularis	27	19	8	19
Vipera ammodytes	274	244	30	191
Vipera berus	19	13	6	13
Vipera graeca	208	1	205	11
Vipera ursinii	18	14	4	14
Zamenis longissimus	118	110	8	104
Zamenis situla	55	51	4	47
Zootoca vivipara	8	5	3	5
	3731	**2706**	**1025**	**303**

At least one species was found in 303 of the 349 grid cells (Fig. 1.7C.). The number of species per cell averaged 7.0 ± (S.D.) 5.79 and the maximum was 26. Twelve grid cells had 20 species or more, mainly in the West close to the Ionian and Adriatic seas (Fig. 1.7.). Zero or few (<5) species occurred in cells in the eastern and the central part of the country. Moran's I index suggested that the number of observations per cell was clustered in space (Z = 6.697, P < 0.0001) (Fig. 1.7B.). The Gi* index showed several sampling effort hotspots (Prokletije, Pindos mountains; Adriatic/Ionian sea coast) but no coldspots (Fig. 1.7B.). Detailed range maps of each reptile species are presented in the supplementary material of Mizsei et al. (2017b).

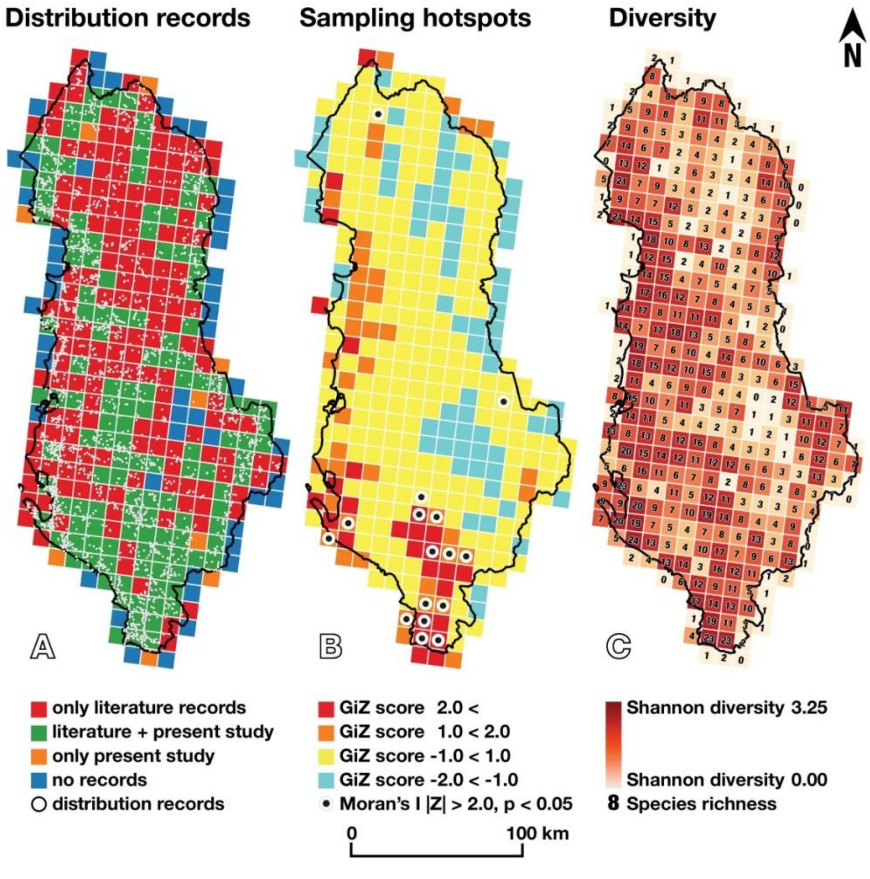


Figure 1.7. A: Sources of occurrence records of reptile species, B: sampling hotspots (GiZ score > 1.0) and coldspots (GiZ score < -1.0), C: reptile species richness (numbers) and Shannon diversity index (shading) in Albania on a 10x10 km grid.

The majority of species were recorded under 1000 m above sea level, and alpine species were typically recorded upwards from 1500 m (Fig. 1.8.).

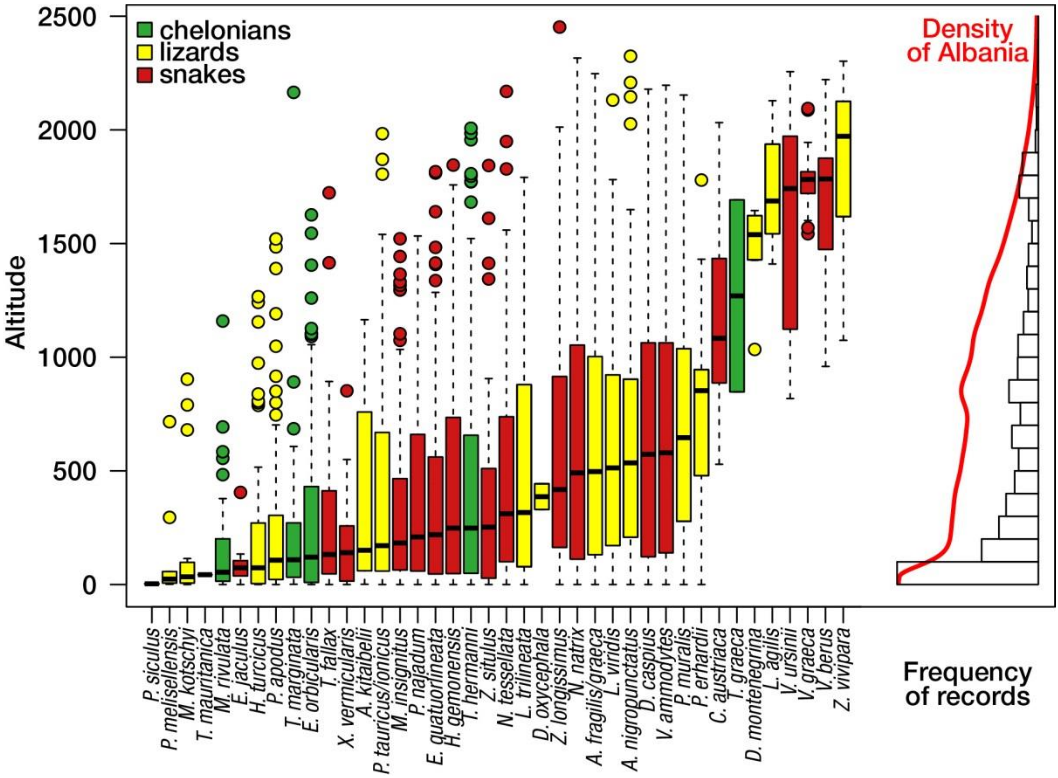


Figure 1.8. Altitudinal distribution of reptile species and frequency of occurrence records by altitude in Albania. Box-and-whiskers plots show the median (horizontal line), the 25th and 75th percentile (bottom and top of box, respectively), minimum and maximum values (lower and upper whiskers, respectively) and outliers (circles). The red line is the frequency distribution of altitudinal values in Albania.

GLMMs showed that habitat diversity (CORINE land cover), altitudinal diversity (ALT SD), precipitation variation (PC4) and temperature (PC1) had the highest importance as predictors of the presence and diversity of reptiles (Table 1.7.). Each of the best models contained these variables for both of the response variables (Table 1.8.). Estimates for parameters showed that habitat and altitudinal diversity as well as precipitation variation (PC4) influenced reptile presence positively, whereas the effect of temperature (PC1) was negative (Table 1.9.). For reptile diversity, only habitat and altitudinal diversity had significant, positive, effects.

Table 1.7. The importance of predictors in the GLMM model for presence/absence and for Shannon diversity of reptiles in Albania. See table 1.1. for the description of the variables.

Presence		**Shannon diversity**	
Predictor	**Importance**	**Predictor**	**Importance**
ALT SD	1.000	ALT SD	1.000
CORINE DIV	1.000	CORINE DIV	1.000
BIO PC1	1.000	BIO PC1	1.000
BIO PC4	1.000	BIO PC4	1.000
BIO PC3	0.838	BIO PC3	0.577
SEA DIST	0.198	BIO PC2	0.241
BIO PC2	0.153	SEA DIST	0.203
ALT MEAN	0.140	ALT MEAN	0.000

Table 1.8. Parameter estimates and AIC values of the best GLMM models with substantial support (ΔAICc< 2) fitted on the presence and Shannon diversity of reptiles in Albania.

Response	**ALT SD**	**CORINE DIV**	**BIO PC1**	**BIO PC4**	**BIO PC3**	**BIO PC2**	**SEA DIST**	**ALT MEAN**	**ΔAICc**
Presence	0.003	1.626	-0.163	0.235	-0.094				0.000
	0.003	1.612	-0.200	0.241	-0.129		0.000		1.131
	0.003	1.547	-0.164	0.245					1.531
	0.003	1.628	-0.164	0.237	-0.093	-0.022			1.648
	0.003	1.617	-0.140	0.208	-0.096			-0.000	1.826
Diversity	0.001	0.697	-0.083	0.117					0.000
	0.001	0.727	-0.082	0.112	-0.035				0.207
	0.001	0.718	-0.107	0.115	-0.059	0.000	0.000		0.734
	0.001	0.699	-0.083	0.118		-0.013			1.623
	0.001	0.728	-0.082	0.112	-0.034	-0.011			1.942

Table 1.9. Model averaged parameter estimates of GLMM fitted on reptile presence and MCMCglmm fitted on Shannon diversity of reptiles. Significant parameter estimates are indicated in bold.

Response	**Main effect**	**Estimate**	**S.E.**	**z**	***P***
Presence	(Intercept)	-6.228	0.674	9.232	0.000
	ALT SD	**0.007**	**0.000**	**3.542**	**0.000**
	CORINE DIV	**1.609**	**0.203**	**7.926**	**0.000**
	BIO PC1	**-0.167**	**0.043**	**3.921**	**0.000**
	BIO PC3	-0.102	0.054	1.885	0.059
	BIO PC4	**0.234**	**0.074**	**3.150**	**0.001**
	SEA DIST	0.000	0.000	0.994	0.320
	BIO PC2	-0.022	0.037	0.599	0.549
	ALT MEAN	-0.000	0.000	0.424	0.671
Diversity	(Intercept)	-1.320	0.807	-1.635	0.102
	ALT SD	**0.002**	**0.000**	**2.226**	**0.026**
	CORINE DIV	**0.689**	**0.213**	**3.227**	**0.001**
	BIO PC1	-0.074	0.085	-0.867	0.385
	BIO PC4	0.094	0.105	0.898	0.368
	BIO PC3	-0.049	0.080	-0.616	0.537
	BIO PC2	-0.011	0.057	-0.201	0.840
	SEA DIST	0.000	0.000	0.366	0.713
	ALT MEAN	-0.000	0.000	-0.350	0.726

Discussion

Our work presents the largest herpetological dataset from Albania so far. The database is comprehensive as it encompasses all species reported from the country and contains at least one observation of species in 87% of the grid cells covering the country's territory. Based on these data, the work also presents a first analysis of the diversity patterns and the most important climatic and environmental factors influencing these patterns in the case of amphibians and reptiles.

Our final database was more complete for reptiles than for amphibians. Although the proportion of cells with observations of at least one species did not differ much (68% for amphibians, 87% for reptiles), there were 37% more records per species for reptiles (average 93 records/species) than for amphibians (68 records/species). Moreover, there were more species per cell for reptiles (7.0) than for amphibians (1.8 or 26% of 7.0), and this difference was larger than what could be expected based on the difference in the number of species (40 reptiles and 16 amphibians or 40% of 40). This difference may be at least partly related to the differing amount of information available before the study. For instance, it is likely that reptiles had been better known before our study, in which only 24% of the observations were new compared to 51% for amphibians. Thus, the novel contribution our work was that it effectively doubled the amount of information on the distribution of amphibian species and that it organized previous knowledge and also considerably increased the amount of information on the distribution of reptile species.

Amphibians of Albania

In the case of amphibians, many cells had low species richness, suggesting that there are still considerable gaps in the knowledge of the amphibian fauna in a large proportion of the country's territory (Fig. 1.3B.). Areas with higher-than-average sampling effort corresponded with high amphibian diversity (i.e. in popular tourist destinations or areas under legal protection such as Prespa Lake, the Prokletije mountains, and Butrint World Heritage Site). Analysis of biases in a similar database for Romania by Cogălniceanu et al. (2013) found similar results: the country had uneven sampling, mainly due to differences in road density and high altitudinal diversity. Due to the generally low number of observations per cell, species with few observations had a great influence on where centers of amphibian diversity were located (Fig. 1.3.). Such species included three alpine species that had limited ranges altitudinally and *P*. *syriacus*, for which one cell in East Albania represents the western edge of the range (Szabolcs & Mizsei 2017; Table 1.2.; Fig. 1.4.). Our results suggest that for some rare species it will not easy to expand their known range by further sampling due to biogeographic constraints. Although no sampling coldspots were identified, it is clear that there is a need to collect more data of widespread species to address issues related to low species richness per cell.

Habitat diversity, i.e., the diversity of land cover types, had the highest importance as a predictor of the presence/absence and diversity of amphibians (Fig. 1.5.). This finding is in line with other studies conducted at larger (at least landscape) scales (Van Buskirk 2005; Denoël & Ficetola 2008; Hartel et al. 2009; Vági et al. 2013; Tsianou et al. 2016). The large role of habitat diversity can be explained because most amphibians require different habitats for their larval and their adult stages, often a wet and a terrestrial habitat type, that usually occur in landscapes with higher spatial heterogeneity. Some species also use wet habitat types even during their adult stage (Ficetola & De Bernardi 2004), and the distribution of such species mostly depends on the availability of freshwater habitats and less so on land use or climate (Fig 1.5.). The two commonest species in our database (*Pelophylax* spp., *Bombina variegata;* Table 1.2.) are typically found in freshwater habitats in each season (Arnold & Ovenden 2002). The high numbers of observations for these species, however, can be expected because they are easy to detect in their habitats during their long breeding season, when they are active also during the day and can be conveniently heard or seen in different types of wet habitat (Arnold & Ovenden 2002).

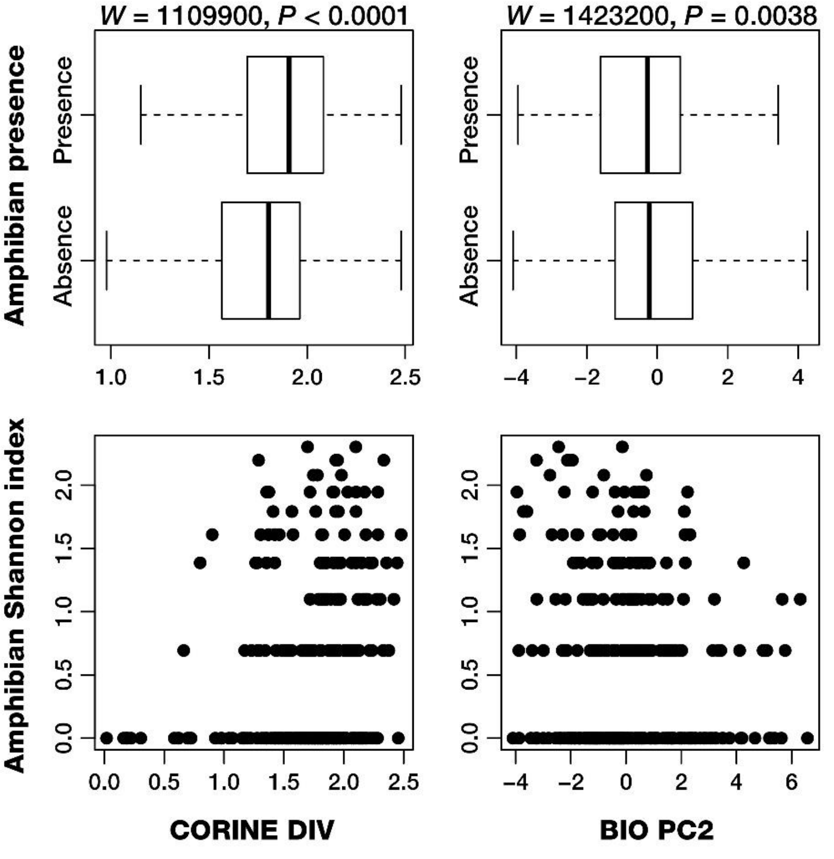


Figure 1.5. Species presence and Shannon diversity index as a function of the most important predictors identified by GLMM model selection (for abbreviations see Table 1.1.)

Other amphibian species are mostly found in terrestrial habitats during the whole year, they are typically active only under moist weather conditions, and during the night, which conditions make their detection more difficult. Several of such species are known only from a few cells, even though we can assume that they are distributed all over the country and over a wide altitudinal range. These species are the *Bufo bufo*, *Bufotes variabilis*, *Hyla arborea*, *Lissotriton graecus*, *Rana dalmatina*, *R. graeca*, *Salamandra salamandra*, *Triturus macedonicus*

(Table 1.2.). Future surveys of these species should focus on the spring breeding season, when they occupy water bodies where they are easily detectable. The probability of encountering individuals in different stages of the life cycle can be further increased by combining visual and acoustic surveys with dip-netting and newt traps (Ficetola & De Bernardi 2004; Mattfeldt 2007).

As morphological characters are not sufficient to identify *Pelophylax* frogs by species, we were unable to tell them apart. *Pelophylax epeiroticus* is genetically more closely related to *P. kurtmuelleri* and *P. ridibundus* and it lives along the Ionian coast of south Albania to southern Greece (Lymberakis et al. 2007). *Pelophylax shqipericus* is a closer relative of *P. lessonae* (Ragghianti et al. 2004) and it lives along the Adriatic coast from Lake Shkodra in Montenegro to the bay of Vlorë in western Albania. *Pelophylax kurtmuelleri* is widespread in the southern and western parts of the Balkans and is distributed over the entire Albania (Dufresnes et al. 2017). There is evidence that *P. kurtmuelleri* is capable of hybridising with the two other *Pelophylax* species (Schneider & Haxhiu 1994, Ragghianti et al. 2004) and can be sympatric with them. Because we know little about the coexistence and the ecology of these species and because two of them are threatened (*P. shqipericus* is Endangered, *P. epeiroticus* is Vulnerable), further research is urgently needed to aid their conservation (Uzzell & Crnobrnja-Isailović 2009; Uzzell et al. 2009).

The increasing economic development of Albania may have negative consequences on amphibians. An increase of habitat alterations and the abandonement of traditional land use practices are expected (Scribner et al. 2001; Hartel et al. 2009). Currently the number of hydropower projects on several rivers and road constructions are on the rise (Freyhof 2010; http://balkanrivers.net), which will likely lead to the destruction of freshwater habitats in many areas (Cushman 2006). Besides habitat alteration, climate change can also affect the future of amphibians in Albania. Climate change can alter important factors affecting amphibian occurences such as the distribution and amount of precipitation (Table 1.5., Rodríguez et al. 2005; Tsianou et al. 2016). Lastly, the chytrid fungus *Batrachochytrium dendrobatidis* that is spreading globally (Fisher et al. 2009) has already been found on the skin of eight amphibian species in Albania (Vojar et al. 2017), although chytridiomycosis outbreaks have not yet been reported from the Balkans.

Reptiles of Albania

Our final database was more complete for reptile species than for amphibians (see above). Furthermore, there was a notable difference in the location of the centres of species richness between amphibians and reptiles. For reptiles, the western parts of the country closer to the coast showed higher diversity than the central and eastern parts (Fig. 1.7C.). This can be explained by local differences in both sampling effort and diversity. First, many species only occupy the west of the country along the Adriatic and Ionian Sea, including *Elaphe quatuorlineata*, *Hemidactylus turcicus* or *Mauremys rivulata*, with fewer similar examples in the east, like *P. erhardii*. Eastern Albania is more characterised by common reptiles along with very rare species such as *T. graeca* and rare mountain specialists such as *Z. vivipara* and *V. ursinii*. Second, the number of species per cell showed much higher sampling effort in the west of the country than in the east (Fig. 1.7B.). This difference may be related to the fact that the western, coastal areas have higher human populations than the mostly mountaineous central and eastern areas of Albania (CIA, 1990). Areas with higher human populations usually also have a higher density of roads and urbanized areas, which increase the probability of encountering reptiles (e.g. as roadkills), which may lead to sampling bias (Kadmon et al. 2004; Beck et al. 2010). Further biases may emerge in picturesque and popular regions such as in Ohrid and Prespa Lakes, and Prokletije Mountains, which are more frequently visited by both tourists and herpetologists. Another hotspot of sampling effort was in the Pindos mountains where we have conducted extensive field studies on *V. graeca* (Mizsei & Üveges 2012; Mizsei et al. 2016b).

Biases could have also been introduced due to the different habitat requirements and behaviour of reptile species. It is not surprising that the species with the highest number of records in the database was *T. hermanni* (Table 1.2.), as this species is active during the day, easy to observe and has a wide range. Some other species e.g. *Dolichophis caspius*, *N. natrix*, and *V. ammodytes* can live in a wide variety of altitudes and habitats (Fig. 1.8.), while others, e.g. *Lacerta viridis* and *Podarcis muralis*, easily colonize areas heavily modified by humans such as urban areas (Arnold & Ovenden, 2004). Species which live in hardly accessible montane habitats under special climatic requirements such as *D. montenegrina*, *V. berus* and *ursinii*, and *Z. vivipara*, had fewer records (Fig. 1.8.). Secretive behaviour of some species also make them difficult to observe. Examples are *E. jaculus* and *Telescopus fallax* which are nocturnally active, whereas *X. vermicularis* lives in underground burrows (Arnold & Ovenden 2004). We presume that secretive species may be more common than they are represented in the database, as with *A. kitaibelii* or *C. austriaca*. Additionally, Albania lies on the periphery of the geographic range of some of the reptile species, e.g. *D. oxycephala*, *P. melisellensis*, *T. graeca*, or *T. marginata*) . Finally, two species are likely to have been introduced into Albania: *P. siculus*, which is found in a few sites in the north of Albania (Mizsei et al. 2016a), and *T. mauritanica*, found exclusively on Sazan Island. These species are effective invaders and sometimes they are accidentally picked up by humans or machines with rocks, wood or other goods. Such episodes occured both in ancient and recent times presumably with maritime shipping (Podnar et al. 2005; Mačát et al. 2014).

Most of the reptiles inhabit lower elevations in the Mediterranean landscape, and high alpine areas are home to only a few cold-adapted species (Fig. 1.8.). We found that temperature (BIO PC1) and the precipitation variability PC were the most important climatic variables both for reptile presence and diversity, although their model-averaged parameter estimates were not significant for diversity (Fig 1.9.). This is plausible because ectotherms usually cannot survive and breed in cold climates except for a few viviparous species (e.g., *Coronella austriaca*, *Vipera* spp., *Zootoca vivipara*). The effect of temperature PC, however, was negative, indicating that reptiles were less likely to occur in cells with higher average temperature. However, the effect of precipitation variability PC on reptile presence was positive (as in Rodríguez et al. 2005 and McCain 2010).

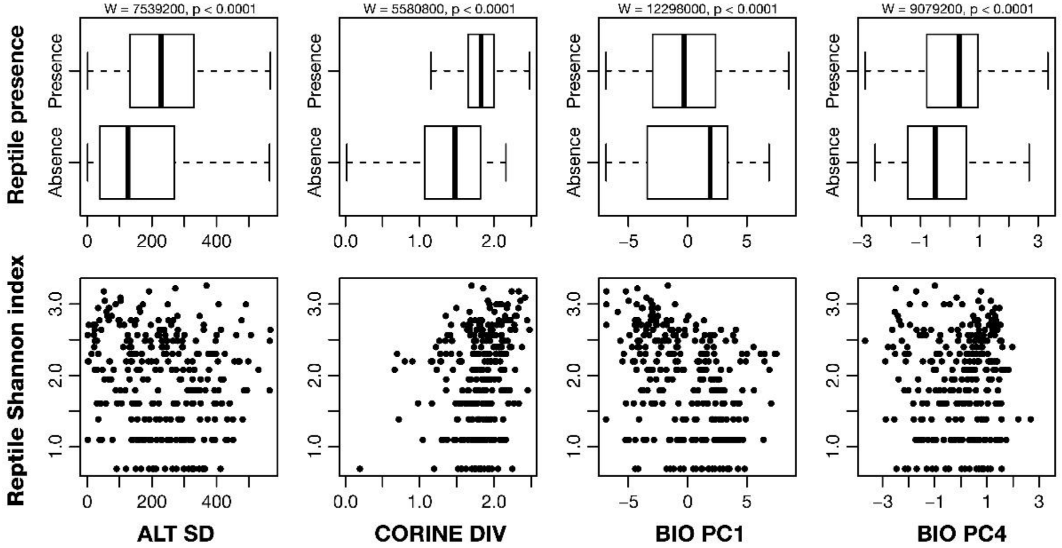


Figure 1.9. Species presence and Shannon diversity index as a function of the most important predictors identified by GLMM model selection (for abbreviations see Table 1.1.)

In addition, altitudinal diversity and land cover diversity also had strong positive effects on both the presence/absence and diversity of species (Fig. 1.9.) (Keil et al. 2012). These results indicated a higher chance of reptile presence in cells with higher diversity in elevation, land cover and precipitation. Such geographic areas are likely to have higher number of available habitats and thus more niches available to be occupied by species than cells with less climatic and environmental diversity (Schall & Pianka 1978). Model-averaged parameters of climatic variables did not have significant effects on species diversity, whereas both altitudinal and land cover diversity had strong positive effects on diversity, which reinforced the importance of the geographical and physical environment for reptiles.

Several recent studies on the biogeography of reptiles present in Albania found hidden genetic diversity and evolutionary distinct lineages which will likely facilitate further species descriptions. For example, five distinct species have been described recently in the *Anguis fragilis* species complex in Europe, of which two species (*A. fragilis* and *A. graeca*) have been reported from Albania (Gvoždík et al. 2010, Jablonski et al. 2016). Furthermore, *Dinarolacerta montenegrina*, a species endemic to the Prokletije mountains, has also been separated from the formerly monotypic Dinaric endemic *D. mosorensis* (Ljubisavljević et al. 2007; Podnar et al. 2014). Recent studies reported distinct evolutionary lineages within the *Lacerta viridis* species complex (Marzahn et al. 2016) and the *Natrix tessellata* species complex (Guicking et al. 2009), and the taxonomy of these lineages have not yet been clarified. The *Podarcis tauricus* species complex has also been recently split into *P. ionicus* and *P. tauricus*, and both species can be found in Albania. There are five distinct lineages within *P ionicus*, and further species are likely to be described in the near future (Psonis et al. 2017). All these examples show that the mountaineous landscapes of Albania, extending to two major mountain regions of the Balkan Peninsula, have functioned both as barriers to migration and as isolated centres of speciation for many reptile species (Joger et al. 2007; Jablonski et al. 2016).

Several factors can enhance the extinction risk of species and the size (area) of the geographic range is known to be one of the strongest predictors of extinction (Harnik et al. 2012). We hope that the methodological approaches and the spatially explicit database presented here provide important baseline information and will be a good starting point towards an evaluation of the distribution of species and their correspondence with protected areas. For example, one important step would be to develop analyses to identify gaps and overlaps with current and future (planned) protected areas, with species distribution modelling applied if and as necessary (Carvalho et al. 2010; de Pous et al. 2011; de Novaes e Silva et al. 2014; Ribeiro et al. 2016). The database presented here has been developed in a way to ensure that it can provide a foundation for further studies (e.g. Sillero et al. 2014a) and can help answer macroecological questions (e.g. Estrada et al. 2015).

Chapter 2

Confirmation of the presence of a new amphibian and a new reptile species in Albania

Introduction

As I described in Chapter 1, the wildlife of Albania is among the least studied in the European continent despite the country's high biodiversity (Mizsei et al. 2017b; Szabolcs et al. 2017). Fortunately, several local and foreign specialists have recently been engaged to challenge this and started to publish their findings from various groups of organisms. In poorly known invertebrate taxa, it is also possible to find species new to science. For example, Szederjesi and Csuzdi (2012) described two new species and added several new ones to the earthworm list of Albania while Lemonnier-Darcemont et al. (2015) summarized the new descriptions and recent findings in their overview of the orthopteran fauna of Southern Albania. Although vertebrates are generally considered a better-known group, it was still possible to describe two fish species new to science recently from the upper segment of River Devoll (Bogutskaya et al. 2010) and to add four more mammal species to the fauna of Albania (Bego et al. 2014; Stolarik & Jablonski 2017; Stolarik et al. 2017).

There have been several herpetofaunistic discoveries since the last major syntethising work was published (Bruno 1989). Uhrin and Šíbl (1996) reported the presence of *Podarcis siculus*, Petrov (2006) found *Dinarolacerta montenegrina* and Korsós et al. (2008) reported *Vipera graeca*. These records have been integrated into our recently published database on the amphibian and reptile fanua of Albania (Mizsei et al. 2017b; Szabolcs et al. 2017).

My aim in this chapter was to discover and confirm the presence of two species in Albania to complete the list of the herpetofauna of the country: the

Italian wall lizard *Podarcis siculus* and the eastern spadefoot toad *Pelobates syriacus*.

Podarcis siculus was first mentioned from Albania in a Czech language conference paper, a source rather obscure to the general public (Uhrin & Šíbl 1996). We aimed to confirm the presence of this species in the original locality and then highlight the presence of this species in Albania to a wider audience.

Pelobates syriacus has long been assumed to be present but has never been found in Albania. Bruno (1989) reported finding *P. syriacus* on the shores of Ohrid and Prespa Lakes in F.Y.R.O. Macedonia and Greece. The species is listed in the official register of animal species occurring in Albania (Dhora 2010), which encompasses species that are found in or near transboundary rivers and lakes shared by Albania and its neighbours. To confirm the presumed occurrence of this species in the country, we searched for this species within the borders of Albania.

Material and methods

We visited northern Albania in May, 2015 and searched for *Podarcis siculus* around Velipojë village where the original observations were made (Uhrin & Šíbl 1996). This species is frequent in towns and other human dominated areas throughout its range and is also a common species in dry Mediterranean open habitats. We looked for individuals in such places by day with regular visual surveys as this species is easy to observe during walking.

We visited Prespa Lake near Kallamas village in May, 2015 to look for *Pelobates syriacus* close to the border of F.Y.R.O. Macedonia (Bruno 1989). We conducted visual surveys after sunset with torches and headlamps on the lakeshore. We also listened to the choruses of frogs as this species can be identified by its vocalization.

Results

P. *siculus* was first recorded in Albania on 13 April 1995 near the village of Velipojë (N41.86°, E19.41°) with fewer than 10 specimens (Uhrin & Šíbl 1996) (Fig. 2.1.). The habitat was a mixture of halophilous marshes dominated by *Juncus acutus* and interspersed with *Pinus halepensis* and with sand dunes of various sizes. We detected one individual of this species 2 km away from this location and 20 years later on a street in the village of Velipojë on 26 May 2015 (N41.86°, 19.43°) (Mizsei et al. 2016a). We found this individual in a built-up environment on a concrete wall belonging to a garden containing several *Cupressus sempervirens* trees. We observed 12 individuals of this species again (third observation) in a cemetery near the village of Trush 15 km away on 26 May 2015 (N41.98°, E19.48°). The cemetery had a few deciduous oaks *Quercus* spp. and grazed grassland in the understorey, and the lizards were found during their movements on tombstones.

We searched for *P*. *syriacus* on 5 May 2015 on the shores of Lake Prespa, a transboundary lake where the species was already known from neighbouring F.Y.R.O. Macedonia and Greece (Fig. 2.2.). At 22:00 p.m. near the village of Kallamas (now Tuminec, N40.89°, E20.93°), we found one adult individual during visual survey with torches (Szabolcs & Mizsei 2017). The individual was found on the sheep-grazed lakeshore meadow covered by short grassland vegetation and partially inundated by water from the lake. Subsequently, two additional individuals were found near the site of the first observation in the course of a nighttime survey a on 20 July 2015.

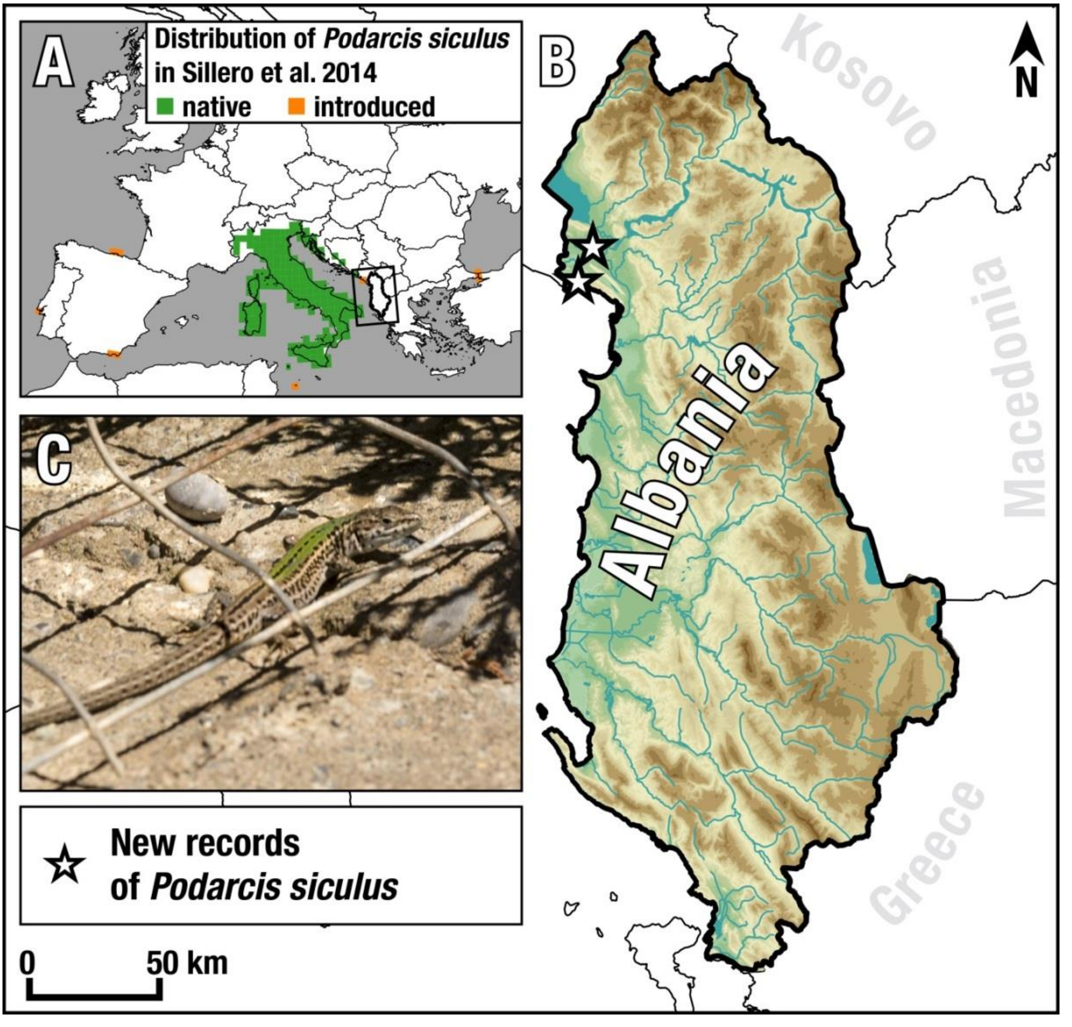


Fig 2.1. A: Overview of the distribution of *Podarcis siculus* in Europe with the delimitation of the inset (B); B: New records of *P. siculus* (white stars) with topography and main water bodies in background; C: *P. siculus* from Velipojë, Albania (photo by E. Mizsei).

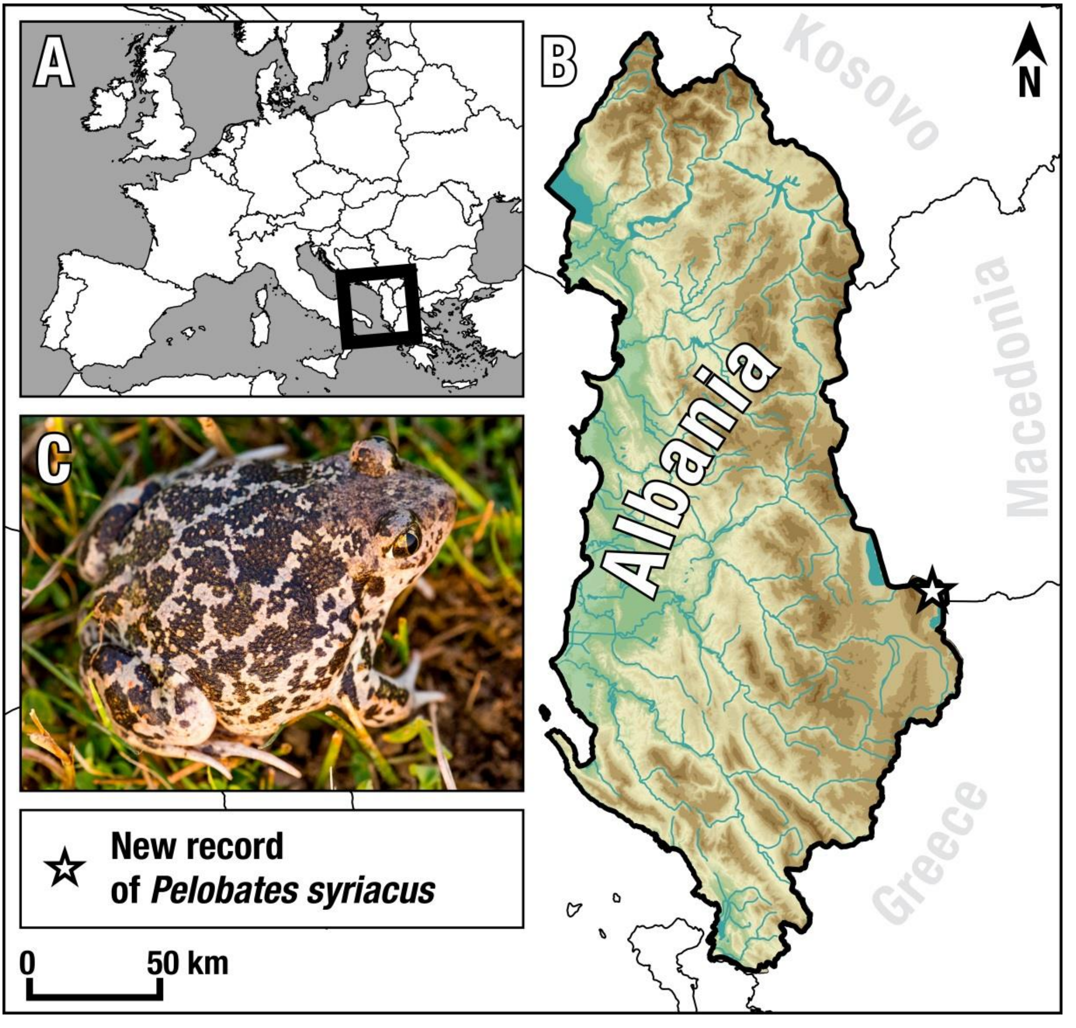


Fig. 2.2. A: The location of Albania in Europe; B: a topographical map of Albania with main water bodies, where the white star indicates the new locality of *P*. *syriacus*; C: a photograph of the first *P*. *syriacus* found in Albania (Photo by E. Mizsei).

Discussion

By targeting well-defined localities based on previous reports and assumptions we successfully confirmed the presence of *P*. *siculus* and found *P*. *syriacus* for the first time in Albania. The original range of *Podarcis siculus* includes the Apennine Peninsula, Sardinia, Sicily and other islands in the Tyrrhenian Sea, the

Northwestern Balkans, Southern France and the southern tip of Switzerland (Sillero et al. 2014a, Fig. 2.1A). The vicinity of Velipojë is currently the southernmost locality for the species in the Balkan Peninsula (Fig. 2.1B). Previously, the range boundary was assumed in Kotor, Montenegro (Podnar et al. 2005) with an uncertain occurence from Donji Štoj (Jovanović 2009). However, the situation is more complex as both native and introduced lineages exist in the Balkan Peninsula. Based on a phylogeographic analysis with mitochondrial genes, Podnar et al. (2005) suggested that the ancestors of populations living in the southeastern part of the Adriatic coast were likely introduced in ancient times by merchant ships. The haplotype of animals from Dubrovnik, Croatia was similar to those living in Southeast Italy, while the Kotor population originated from different introductions, one from Dubrovnik and one from another place in the Adriatic. It is likely that the Albanian populations were also introduced, or they could originate by natural dispersal from an introduced population. This species is highly capable of establishing new introduced populations as it did in multiple parts of the Mediterranean such as Turkey, Greece and Spain (Rivera et al. 2011; Ilgaz et al. 2013; Adamopoulou 2015) or in Great Britain and the U.S.A. (Kolbe et al. 2013; Silva-Rocha et al. 2014). Because this species may be a potential new invader in the Mediterranean, it would be informative to collect more data on potential occurrences in larger port cities in Albania, for example Durrës, Vlorë or Saranda and to expose specimens for genetic analysis.

The range of *Pelobates syriacus* comprises the southeastern and the central Balkan Peninsula, Turkey, Transcaucasia, the Middle East and northern Iran (Džukic et al. 2008). The Albanian location is the westernmost locality known for this species (Fig. 2.2.). It is likely that Prespa Lake serves as a breeding site for the species because there are almost no other sites suitable near the lake, which is surrounded by high mountains. We have no direct evidence of breeding as we did not hear the calls of the species while calls of other species (*Bufotes viridis*, *Hyla arborea*, *Pelophylax kurtmuelleri*) were omnipresent during the nights of our searching in May. *P. syriacus* has been reported from the Macedonian side of Ohrid Lake (Bruno 1989), thus, it is likely that the species is present in the Albanian part of that lake as well.

There are two other reptiles which have ambiguous records from Albania. The Moorish gecko (*Tarentola mauritanica*) was only known from Sazan Island, the largest island of Albania (Bruno 1989). The island is not open for the public as it serves as a military base for the Albanian army and NATO, thus, we could not visit it. This species is also capable of establishing introduced populations as it did in Korfu Island, Greece (Mačát et al. 2014) and is likely to have been introduced to Albania as well.

The spur-thighed tortoise (*Testudo graeca*) was also mentioned from Albania from around Lake Prespa (Haxhiu 1998). We also searched for this species both when we went to survey for *P. syriacus* and at other times on multiple occasions. Tortoises are easy to observe during daylight and we found several specimens of Hermann's tortoises (*T. hermanni*) but we never found *T. graeca*. It appears possible that the animals reported in Haxhiu (1998) were falsely identified as the two species are very similar. If *T. graeca* lives in Albania, it would be the western edge of its range.

Finally, according to a recent work on the reptile species of Serbia (Tomović et al. 2014), the meadow lizard (*Darevskia praticola*) is potentially present in some parts of Kosovo, an area neighbouring Albania. This species prefers open gaps in humid deciduous forests, usually oak, which are rather uncommon in the mostly Mediterranean Albania. However, there is a chance that this species also lives in North-East Albania because its range is close to the border of the country.

Chapter 3

Identífying catchments critical for the conservation of freshwater biodiversity in Europe

Introduction

Biological diversity is declining even faster in freshwater ecosystems than in terrestrial and marine ecosystems (Dudgeon et al. 2006). Freshwater ecosystems (rivers, lakes and wetlands) host disproportionately high species diversity relative to their surface area (Strayer & Dudgeon 2010), provide important ecosystem services e.g. food production, carbon sequestration, water purification and control of floods and soil erosion (Darwall et al. 2011), are vulnerable to pollution and hydromorphological alterations due to flood control and hydropower development (Zarfl et al. 2014), and mitigate the impacts of climate change (Kundzewicz et al. 2008). Yet freshwater ecosystems, habitats and species are poorly protected globally (Darwall et al. 2008; Revenga et al. 2005; Rodrigues et al. 2004). Although the correspondence between terrestrial and freshwater biodiversity is higher than expected by chance (Abell et al. 2011), many protected area networks were designated primarily to terrestrial or marine habitats and species (Abell et al. 2007; Linke et al. 2012b; Saunders et al. 2002). Major threats to freshwater biodiversity include habitat loss due to e.g. destruction of wetlands (Zedler & Kercher 2005), habitat fragmentation due to dams (Ligon et al. 1995) and bridges (Málnás et al. 2011), flow alteration (Vörösmarty & Sahagian 2000), introduction and spread of exotic or invasive species (Collares-Pereira & Cowx 2004) and pollution (Dudgeon 2006). There is thus an urgent need for conservation interventions to face these threats. Due to the global and omnipresent nature of threats, the limited resources, and the necessity for interdisciplinary and multi-sectoral approaches to design and implement conservation interventions for freshwater biodiversity, conservation measures need to be prioritised in a scientifically sound manner to find efficient solutions (Carrizo et al. 2017; Darwall et al. 2011; Linke et al. 2011).

The identification of Key Biodiversity Areas (or KBAs), i.e., globally important areas for the conservation of biodiversity is a well-regarded and increasingly used conservation tool. They can help improve and expand networks of current protected areas (Rodrigues 2004b; Langhammer et al. 2007) and they can be useful for meeting the criteria expressed in several Aichi Biodiversity Targets (IUCN and BirdLife International 2013) and the Biodiversity Strategy targets of the EU (EC 2011). KBAs can also be integrated into environmental policy both in the public and private sectors via online databases and tools e.g. the Integrated Biodiversity Assessment Tool (IUCN 2016).

Several previous studies embarked on identifying KBAs for freshwater species (Silvano et al. 2007; Holland et al. 2012; Darwall et al. 2014). However, we still know little on where areas important for the protection of freshwater biodiversity are located geographically. In addition, sites designated by the Alliance for Zero Extinction, i.e., areas that hold the only remaining population of species that are globally threatened (Ricketts et al. 2005) and which should be the cornerstone of any KBA network, have been largely neglected in previous efforts to identify KBAs. For instance, only one AZE site has been designated to date in Europe (for the brook newt *Calotriton arnoldi* in Catalonia, Spain, Carranza & Amat 2005).

Our first aim was to identify river and lake catchments that hold threatened species that could qualify the catchments as KBAs for freshwater biodiversity. We defined catchments with threatened species as “Critical Catchments”. The set of such catchments provides the basis for the selection of KBAs (Darwall & Vie 2005). To guide this selection process, our second aim was to apply spatial prioritization to narrow the first set of catchments to those catchments that provide as much protection to threatened species, to species with limited geographic ranges and to communities rich in endemic species as possible at the lowest cost. Ideally, catchments ending up with high priority should enjoy increased conservation attention and consideration as KBAs (Carrizo et al. 2017).

In the current Chapter 3, I describe the methodology and results of selecting critical catchments for improving the conservation of freshwater biodiversity. In Chapter 4, I describe the analyses and the results related to the spatial priorities in conservation, the role of current protected areas and the need for increased conservation effort in Europe.

Materials and methods

Data and study area

To identify and prioritize catchments, we used occurrence data of 1296 species of four taxonomic groups that are fundamental in freshwater biodiversity: 511 fishes, 617 molluscs, 73 odonates and 95 plants. Data were provided by IUCN and were the same as those used for the global Red List assessments of species (IUCN 2013). For nomenclature, taxonomy and threat level, we applied the IUCN Red List terminology. We used IUCN's definition of threatened species as either Critically Endangered, Endangered or Vulnerable. We also used data from other Red List categories (Near Threatened, Least Concern, Data Deficient) but did not consider Extinct or Extinct in the Wild species in the analyses. We also applied two filters based on data quality and considered occurrence data only if they had low uncertainty and if they were from the native (i.e. non-introduced) range of the species.

We received the data in a form where the occurrence of species was tied to the catchments. For the spatial representation and analysis of catchments, we used level 8 of the global HydroBASINS database (Lehner & Grill 2013). Our study area encompassed the entire geographic Europe (Fig. 3.2; over 10 million km^2), which was covered by 18,816 river and lake catchments (mean area 538.3 ± S.D.

649.45 km^2) that we defined as planning units. Using catchments as planning units has several advantages over arbitrary systems of grid cells or hexagons (Abell et al. 2007; Linke et al. 2008) to facilitate the transferability of the results to real-world conservation policy and practice. Catchments provide naturally delineated, ecologically meaningful and easily identifiable units that represent the conservation and water management units for river networks (Moss 2008). Mapping and analyzing freshwater biodiversity at the catchment scale thus ensures compatibility between the scales of assessment, planning and management (Nel et al. 2007). Moreover, freshwater biodiversity differs from both marine and terrestrial biodiversity in that freshwater species are confined to stream/river systems and lakes which are connected in a dendritic, hierarchical river network (Abell et al. 2007).

Identification of Critical Catchments

First we identified the set of critical catchments by considering three criteria that are meaningful biologically and that were studied in detail by Holland et al. (2012). A catchment was considered as critical if one or more of these criteria were fulfilled by the species or the species assemblage within it.

Criterion A: A catchment holds at least one species that is threatened globally. Because all critically endangered or endangered species that have only one remaining population ('AZE' species) are defined as threatened by IUCN, this criterion also fulfilled the criterion that AZE sites are included in the set of critical catchments.

Criterion B: A catchment holds at least one species with a range that is geographically restricted. We considered a species as having a restricted range if the area of its range was less than 20,000 km^2 (fishes, molluscs, plants) or 50,000 km^2 (odonates – these species usually have high capability for dispersal).

Criterion C: A catchment holds a species community that is rich in endemic species. We considered a species endemic if all its occurrences were within one freshwater ecoregion (Abell et al. 2008). Because endemic species are, by definition, confined to a biogeographic region, they can be exposed to extinction even if they are not directly threatened. We defined a community as rich in endemic species if at least a quarter (25%) of the species in any of the four taxa was endemic. Criterion C ensured that biogeographic aspects and community aspects are also taken into consideration in the analyses.

Results

We identified 8423 catchments as critical, which covered 45% or more than 4,5 million km^2 of Europe (Fig. 3.1.). These catchments are mainly located in the Mediterranean peninsulas of southern Europe and they were classified as critical because they held one or more species qualifying under either Criterion A, B or C (these are called „trigger species” from now on). There were 766 separate trigger species (Table 3.1.). Trigger species number was highest (n = 69) in the catchment of Lake Ohrid .

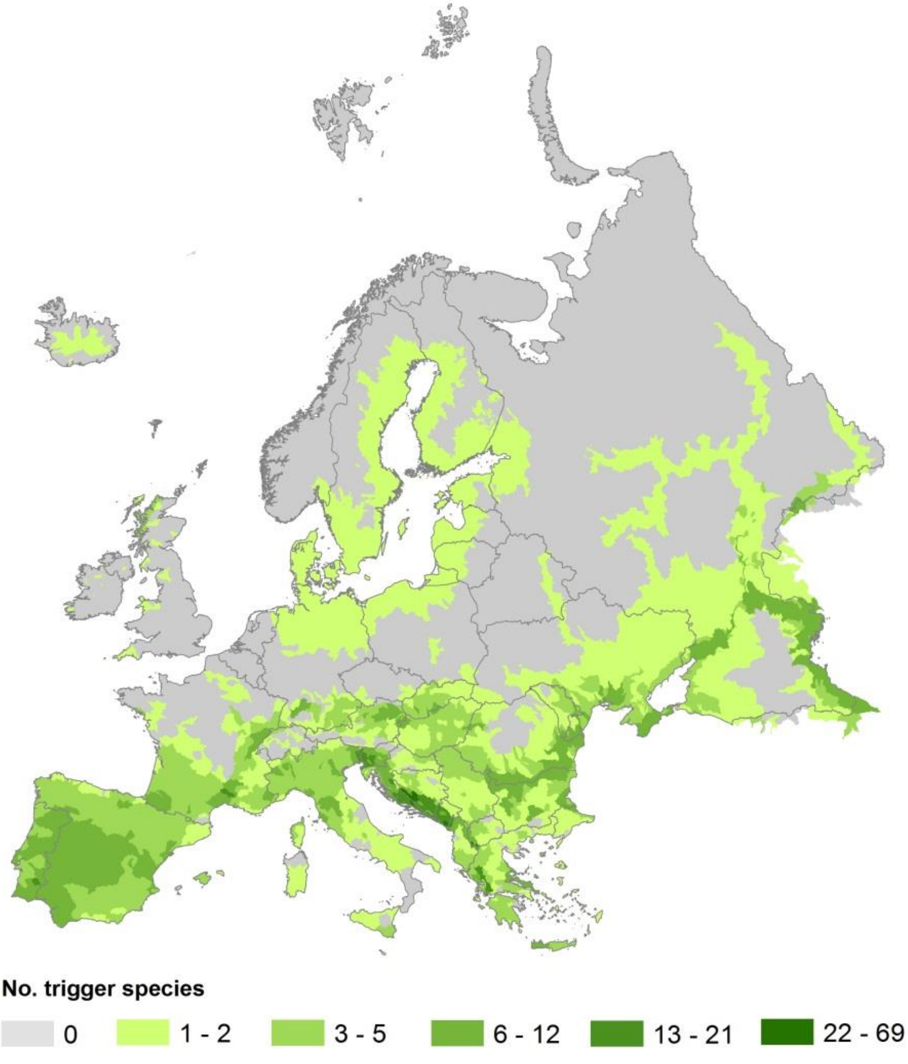


Figure 3.1. Critical catchments for fishes, molluscs, odonates and aquatic plants, with catchments shaded by the number of distinct trigger species.

We found 65 catchments that qualified under the criterion of the Alliance for Zero Extinction (AZE) (Fig. 3.2.). Threatened species with one last remaining population were fish, mollusc or plant species, of which 73 were Critically Endangered and 44 were Endangered. The catchment of Lake Ohrid held the highest number (n = 26) of AZE species and most catchments held only one AZE species.

Figure 3.2. Location of Alliance for Zero Extinction (AZE) catchments, defined as catchments that hold the last remaining population of one or more Critically Endangered or Endangered species.

Most (97%) of the critical catchments fulfilled Criterion A, and 26% fulfilled Criterion B. Each of the four taxa contained trigger species, but for Criterion C, only molluscs and fishes were triggers (Table 3.1). Each of the catchments qualifying under Criterion C was in one of only three ecoregions (SE Adriatic, N British Isles and Iceland).

Table 3.1. Number of trigger species and number of triggered catchments for threatened species (C-A), restricted range species (C-B), and ecoregion-restricted communities (C-C) and all criteria (C-A - C-C) for each taxon group. The total for catchments is the number of distinct catchments and is thus not the sum of the rows.

	Number of Trigger Species				**Number of Triggered**			
	C1-3	**C1**	**C2**	**C3**	**C1-3**	**C1**	**C2**	**C3**
Fishes	260	186	218	18	7547	7320	856	99
Molluscs	479	349	465	53	2724	2269	1621	1
Odonates	7	6	5	0	642	632	119	0
Plants	20	15	12	0	988	979	85	0
Total	766	556	700	71	8423	8144	2207	99

Discussion

Our most surprising result is that close to 60% of the freshwater species involved in the study (n = 1296) qualified as trigger species in one way or another (based on their threat status, restricted range or membership in a biogeographically unique species assemblage). We estimate that the proportion of threatened species is at least 44% in molluscs, 37% in fish, 15% in dragonflies and 7% in plants. Similar proportions are reported in Cuttelod et al. (2011). Future conservation efforts should employ much greater focus on the trigger species identified here (i.e. threatened, restricted-range and ecoregion-restricted species). It is essential to ensure a higher publicity and more efficient conservation interventions for these species.

Our maps showed that most of the critical catchments were in southern Europe and along large rivers in eastern Europe (Fig. 3.1.), whereas AZE catchments were scattered all over Europe (Fig. 3.2.). It requires further study to evaluate whether these critical catchments are included or not in the current system of protected areas. For instance, even though the Natura 2000 system of protected areas is often considered as complete, most (23 of 28) of the member states have been required by the European Commission to designate new Natura 2000 sites (Crofts 2014). Our results provide a basis for the scientifically sound designation of such protected areas as they draw attention to critical catchments that are potential freshwater KBAs (Juffe- Bignoli et al. 2016). Including such a focus in the designation of new protected areas, instead of a focus on terrestrial biodiversity, is absolutely necessary if we are to maintain freshwater ecosystems (Abell et al. 2007). However, it appears more realistic and pragmatic to manage both biodiversity and human activities at the scale of catchments (Moss 1999; Nel et al. 2009). This thought has been incorporated in the 'wider countryside measures' concept of the Habitats Directive and in the development and implementation of the Water Framework Directive of the European Union (Crofts 2014).

The framework focusing on critical catchments presented here is open for several future developments. It is not realistic, however, to expect that all critical catchments will be adequately protected because probably none of the European countries is willing to set aside 45% of its land for the conservation of freshwater species and habitats. Furthermore, changes in the natural status of catchments due to urbanisation, intensive land use by agriculture or altered hydromorphology (e.g. dams) have probably rendered many catchments sub-optimal for protection. Therefore, there is a dire need for a refined spatial prioritization of critical catchments that not only considers conservation status of species and habitats but also includes assessments of catchment vulnerability, opportunity costs, and feasibility of conservation measures. Additional aspects that should be added include the integration of information on currently common species which may become threatened in the future due e.g. to climate change and more knowledge on habitats that are likely to hold species that are poorly known currently and on ecosystems that are essential to maintain populations of threatened species (Khoury et al. 2010). Such an integrated approach requires the utilization of frameworks and methods of systematic conservation planning (SCP) in the future. It is important to nail down that spatial prioritisation is a step in SCP but not the whole process or the final answer to a problem. The first step in an integrated approach needs to be a spatial prioritization of catchments based on current patterns in biodiversity and threat status. Such a prioritization needs to be iterated and developed further when more input from stakeholders and better knowledge on economic and social costs and benefits of protection becomes available (Margules & Pressey 2000). A full SCP process should also incorporate data other than species distribution, for example, different socioeconomic variables to develop solutions that ensure the fulfillment of biodiversity conservation targets with as little socioeconomic cost to fishery, agriculture, mining and forestry as possible (Carwardine et al. 2008a). Assessments of the ecosystem services in the catchments and the adaptability of catchments to climate change are also essential to include in a full-blown evaluation (Groves et al. 2012; Markovic et al. 2014). We applied species-based analyses but alternative methods could focus on ecosystem status or condition or on species assemblages representative of different regions before they become threatened (e.g. Khoury et al. 2010), but it can lead to the identification of alternative sets of catchments (Heiner et al. 2011). Finally, the focus from species-based prioritization could shift to prioritizations that target ecosystem condition or status or species communities that are more natural (not yet threatened) or are more representative biogeographically than others (e.g. Khoury et al. 2010), although it has to be noted that such shifts in focus can result in different sets of areas qualifying as critical or high-priority catchments.

Chapter 4

Spatial priorities in freshwater biodiversity conservation in Europe

Introduction

Conservation planning is often used to evaluate the capacity of existing protected areas to conserve biodiversity (Lawrence et al. 2011). However, few studies investigated the spatial correspondence between hotspots of freshwater biodiversity and protected areas (Keith 2000; Abellán et al. 2007; Herbert et al. 2010). Similarly, there are only a few prioritisations that consider threat status or degradation (Hermoso et al. 2011; Moilanen et al. 2011; Linke et al. 2012). Management should be allocated to catchments that have high levels of biodiversity, are well protected but are vulnerable to future threats, whereas restoration is necessary in catchments that have high levels of biodiversity, are not adequately protected and/or are degraded (Thieme et al. 2007; Nel et al. 2011).

In this study, we aimed to establish spatial priorities for river and lake catchments in Europe based on their importance to the conservation of freshwater biodiversity. We used an extensive, continental-scale database on four representative taxa of freshwater ecosystems mapped to an ecologically meaningful layer of planning units. We used irreplaceability as a proxy for the conservation priority of catchments and data on protected areas to address four questions: (1) Where are the areas that have high priority in freshwater biodiversity conservation? (2) Does the inclusion of well-protected catchments improve the efficiency of prioritisation? (3) Is there correspondence between the conservation priority of catchments and their level of protection? (4) Where are the priority areas that urgently need conservation interventions?

Material and methods

Similarly to Chapter 3, we used catchments as planning units from the global HydroBASINS database (Lehner 2012) at level 8 (of 12), where geographical Europe (10,128,044 km^2) is delineated into 18,816 catchments. We used data on species distributions obtained from the IUCN, which provided the foundation of the Red List of Threatened Species (IUCN 2013). Data were from four taxon groups: fishes (n = 512 species), molluscs (n = 656), odonates (n = 124), and aquatic plants (n = 339). These groups represent a variety of trophic levels and dispersal types, and are important in ecosystem functioning and services. The final database contained 4,493,267 occurrence records of 1631 species in 18,816 catchments. The planning unit (catchments) layer was downloaded from http://hydrosheds.org/page/hydrobasins.

We prioritised catchments based on the conservation status, range-restriction and endemism of their species (corresponding to Criteria A-C in Chapter 3). For conservation status, we used the global Red List status, or, if it was not available (n = 334 species), the European Red List status. We considered a species threatened if it was listed as Vulnerable, Endangered or Critically Endangered (VU, EN or CR, respectively) (IUCN 2013). Because CR species are faced with imminent extinction (IUCN 2013), we targeted 100% of their occurrences to be included in the optimised protected area network. For EN and VU species, for which extinction risks are lower (IUCN 2013), targets were lower (75% and 50% for EN and VU species, respectively). We only used "extant" and "probably extant" occurrences of species in their native range. Three exceptions were the critically endangered molluscs *Belgrandia moitessieri* and *B. varica,* which had only "possibly extant" occurrences but were included because of their conservation status, and the critically endangered fish *Scardinius scardafa*, which only has a single introduced population. For non-threataned species whose range was not restricted, we specified two separate occurrences as targets. Occurrence targets were considered fulfilled when at least 98% of the targeted area of the occurrences was included in the MARXAN solution.

To estimate range restriction, we calculated range size as the total area of catchments in which the species occurred. We considered a range restricted under 20 000 km^2 for fish and molluscs and aquatic plants (taxa with low dispersal ability) and 50 000 km^2 for odonates (good dispersal ability) (Holland et al. 2012). We targeted 25% of the occurrences of range-restricted species to be included in the optimised network. When a species qualified both as threatened and as range-restricted, conservation status was considered first as it required higher percentages of occurrences to be included.

A species was considered endemic if it was restricted to one biogeographic unit. For biogeographic classification, we considered the freshwater ecoregion delineation of the World Wildlife Fund (Abell et al. 2008, http://www.feow.org/). We considered a catchment to hold unique species assemblages if the proportion of fish and mollusc species restricted to only one ecoregion was at least 5% of all fish and mollusc species (Holland et al. 2012). There were no ecoregion restricted species in the two other groups. We implemented this criterion by fixing the qualifying catchments (n=190) *a priori* into the prioritisation.

For spatial prioritization, we used MARXAN (Ball et al. 2009), a software developed for systematic conservation planning. MARXAN is most often used for decision support that applies spatial optimisation using a simulated annealing algorithm to iteratively estimate and identify the most cost-efficient solution to reserve design problems. MARXAN heuristically searches a user-defined number of spatial configuration of reserves (areas for protection) to identify the one that best meets the user-defined targets concerning the representation of species based on the principles of complementarity and irreplaceability while also minimising the total cost of protection.

MARXAN aims to minimise the total cost of a reserve network while maximising the total number of biodiversity features adequately covered by the network:

Objective function = Σ $\text{Cost}_{\text{Planning unit}}$ + Σ SPF $\times$ Feature penalty + BLM $\times$ Σ Boundary costs,

where Cost is the expense of selecting a planning unit for the network, SPF is the Species Penalty Factor that scales with the importance of species for which a Feature penalty is added when the target specified for the species is not met by the network. BLM is the Boundary Length Modifier that determines the importance of boundary costs that increase if the reserve network is too fragmented and decrease as the solution becomes more clumped. Although the real cost of protection depends on a variety of factors such as loss of land used for agriculture or other economic activity, the area of the planning units has been proposed as a general proxy to estimate protection cost (Moilanen, Leathwick and Elith 2008). As an estimate for the cost of protection, we used the area of each catchment (km^2) and we imposed a limit on the total cost of the network at 17% of Europe's area. We chose this value as it corresponds to Aichi Target 11 of the UN Biodiversity Strategy, that stipulates that the protection of terrestrial and water ecosystems should reach 17% of the Earth's surface by 2020.

Because the topology of catchments in the HydroBASINS database was not fully resolved, we controlled for only a basic level of connectivity by varying the Boundary Length Modifier (BLM) in Marxan. The BLM parameter exerts control over the shape of the resulting protected area network, and increasing BLM values cause more aggregated, less fragmented solutions. We varied the BLM between 0.001 and 25 at six levels along an exponential scale to identify the optimal value as recommended in the literature (Stewart & Possingham 2005). We found that BLM = 10 was an appropriate trade-off between coverage of threatened species, fragmentation and geographical representation, thus we used this value in all the prioritizations.

Because CR species are faced with imminent extinction (IUCN 2013), we targeted 100% of their occurrences to be included in the optimised conservation area network. In other words, catchments holding CR species always had to be part of the optimal solution. For EN and VU species, for which extinction risks are lower (IUCN 2013), lower targets were set (75% and 50%, respectively). For non-threatened species, we targeted a minimum of two spatially separated occurrences as in Holland et al. (2012). We forced the algorithm to ensure the fulfillment of targets specified for threatened species by setting the species penalty factor to 1,000,000, 1000, and 10 for CR, EN, and VU species, respectively. There were 99 catchments (1% of the total) in which we identified uniqe species assemblages (Criterion C) and these were built in the final solution *a priori* ("locked in”).

We started each Marxan run by randomly selecting 10% of the catchment planning units and used the default values for the simulated annealing algorithm in each run (Ardron et al. 2010). Each scenario was run 1000 times. To measure the irreplaceability, or the conservation value, of catchments, we used selection frequency, i.e., the number of occasions that a given catchment was selected by the algorithm as part of the optimal network in the 1000 runs.Catchments that were always selected (1000 times of 1000 runs) were considered as ‘irreplaceable’.

Spatial data on Natura 2000 protected areas were obtained from the European Environmental Agency and spatial data on areas protected by national laws were downloaded from the World Database on Protected Areas (https://www.protectedplanet.net). We then calculated the proportion of both types of protected areas and their combined proportion in each catchment. Data preparations and calculations were done in R (R Core Team 2015) and ArcGIS 10.0 for Windows was used for visualising the results on maps. All calculations were conducted in the area-constant Lambert Azimuthal Equal Area projection, which is recommended for statistical analysis and display in Europe (INSPIRE).

For Questions 1 and 2, we set up four scenarios: (i) catchment protection not considered (Scenario 1A), (ii) catchments rich in Natura 2000 areas locked in *a priori* in the solution (1B), (iii) catchments rich in nationally protected areas locked in (1C), and (iv) catchments rich in both types of protected areas locked in (1D). We defined catchments as rich in protected areas when the proportion of such areas was above 70%. This threshold was chosen to reflect that the macroinvertebrate and fish fauna of pristine and degraded rivers do not differ substantially as long as not more than 30% of the catchment area is transformed to agriculture (Allan 2004).

For Question 3, we used irreplaceability from Scenario 1A and the proportion of protected areas. We characterised correspondence in three ways, assuming that if protection is ideal, there should be a strong positive relationship between the proportion of protected areas and irreplaceability. First, we classified catchments into four groups by dividing them into high/low irreplaceability and high/low proportion of protected areas categories at the median values. Second, we calculated residuals from an ordinary least-squares linear regression of irreplaceability as a function of proportion of protected areas. Finally, we measured the deviation of each catchment from an ideal, hypothetical 1:1000 line that is expected if the correspondence between proportion of protected areas (range: 0 to 1) and irreplaceability (range: 0 to 1000) is perfect. For comparison, we also calculated the amount of protected areas in each catchment without the consideration of species.

For Question 4, we developed a second set of scenarios: (i) catchments rich in Natura 2000 areas (>70%) locked out *a priori* from the prioritisation (Scenario 2A), (ii) catchments rich in nationally protected areas locked out (2B), and (iii) catchments rich in both Natura 2000 and nationally protected areas locked out (2C). Catchments for which there was a conflict between the criterion of unique species assemblages and any of the above scenarios were locked in.

Results

In Scenario 1A, where protected areas were not considered, catchments with high irreplaceability were in S Europe (S Spain, S France, W, S and E Balkans), along major rivers (upper and lower Danube, lower Don, Dniester and Volga), and around large (e.g. Lake Ladoga) or smaller lakes in N Europe (Fig. 4.1A.). We identified 795 irreplaceable catchments (4% of total). Freshwater ecoregions with the highest average irreplaceability were in S and E Europe, including the catchments of the Volga delta and northern Caspian Sea, the Dalmatian coast, the Ionian coast, Crimea, the Caspian Sea, Crete, western Anatolia and the northern and southern Adriatic Sea, followed by southern Iberian and other western Mediterranean ecoregions (Fig. 4.2A.). In contrast, most of the single rivers with the highest irreplaceability were in the W Mediterranean (Guadiana, Duero, Ebro, Garonne, Guadalquivir, Tajo) and there were fewer rivers in this group from the eastern Mediterranean (Aoos/Vjosa, Po, Marica/Evros) (Fig. 4.2B.). Finally, countries with the highest average irreplaceability also were mostly in S Europe (Malta, Montenegro, Albania, Portugal, Bulgaria, Italy, Spain) and central Europe (Austria, Slovakia, Hungary) (Fig. 4.2C).

Locking in catchments rich (>70%) in Natura 2000 protected areas resulted in higher irreplaceability for catchments in N and central Europe (e.g. in Sweden, Finland, Poland) and parts of southern Europe (e.g. S Portugal) (Fig. 4.1B.). Locking in catchments rich (>70%) in nationally protected areas (Fig. 4.1C.) resulted in increased irreplaceability for catchments in N Russia, Iceland and Norway (countries without Natura 2000). Finally, locking in catchments rich (>70%) in both types of protected areas (Fig. 4.1D.) resulted in the most balanced distribution of irreplaceability across Europe and the clearest priorities (indicated by more catchments with zero or low selection frequency).

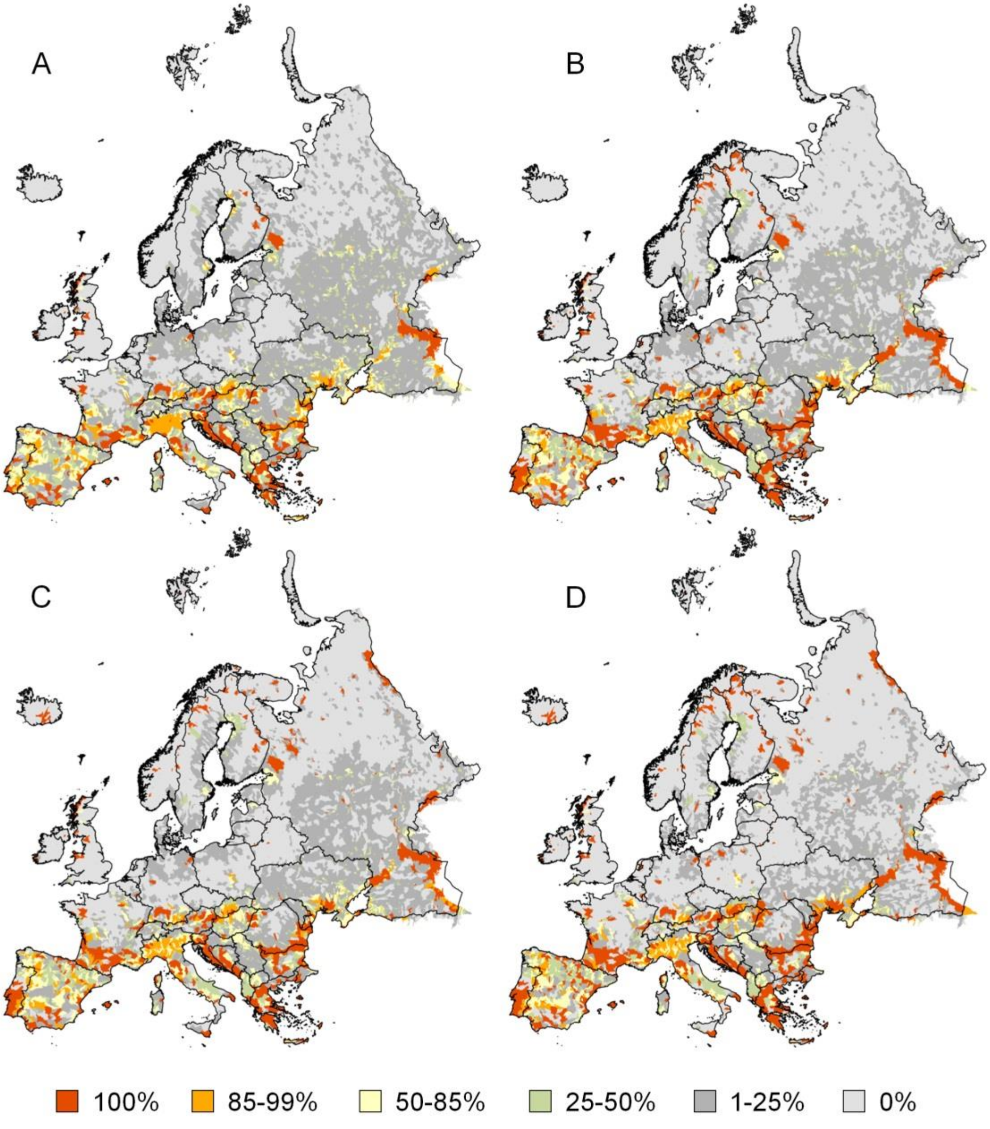


Figure 4.1. Conservation priority of European catchments (n=18,816), estimated as irreplaceability, i.e., selection frequency in 1000 runs of Marxan when protected areas are not considered (A), when catchments rich in Natura 2000 areas (>70% of total area) are selected first (locked in) (B), when catchments rich in nationally protected areas (>70% of total area) are locked in (C), and when catchments rich in both Natura 2000 and nationally protected areas (combined proportion >70% of total area) are locked in (D).

The inclusion of protected areas slightly increased the number of species with targets fulfilled from 1605 (98.4%) to between 1611 and 1613 (c. 98.9%), depending on the protected area network included (Table 4.1.). The number of threatened species with targets unfulfilled was highest when no protected areas were locked in (n=21) and was similarly lower in the three lock-in scenarios (14 to 16 species, Table 4.1.).

Table 4.1. Number of species for which representation targets were met (Yes) and not met (No) under four scenarios varying by whether catchments rich in protected areas (PA) were locked in at the start of prioritisation.

	Catchments added when coverage is adequate by							
	None		Natura 2000 PAs		National PAs		Both	
Conservation status	Yes	No	Yes	No	Yes	No	Yes	No
Critically Endangered (CR)	144	9	148	5	147	6	147	6
Endangered (EN)	142	5	144	3	143	4	144	3
Vulnerable (VU)	261	7	262	6	262	6	262	6
Near Threatened (NT)	127	0	127	0	127	0	127	0
Least Concern (LC)	831	3	832	2	832	2	832	2
Data Deficient (DD)	100	2	100	2	100	2	100	2

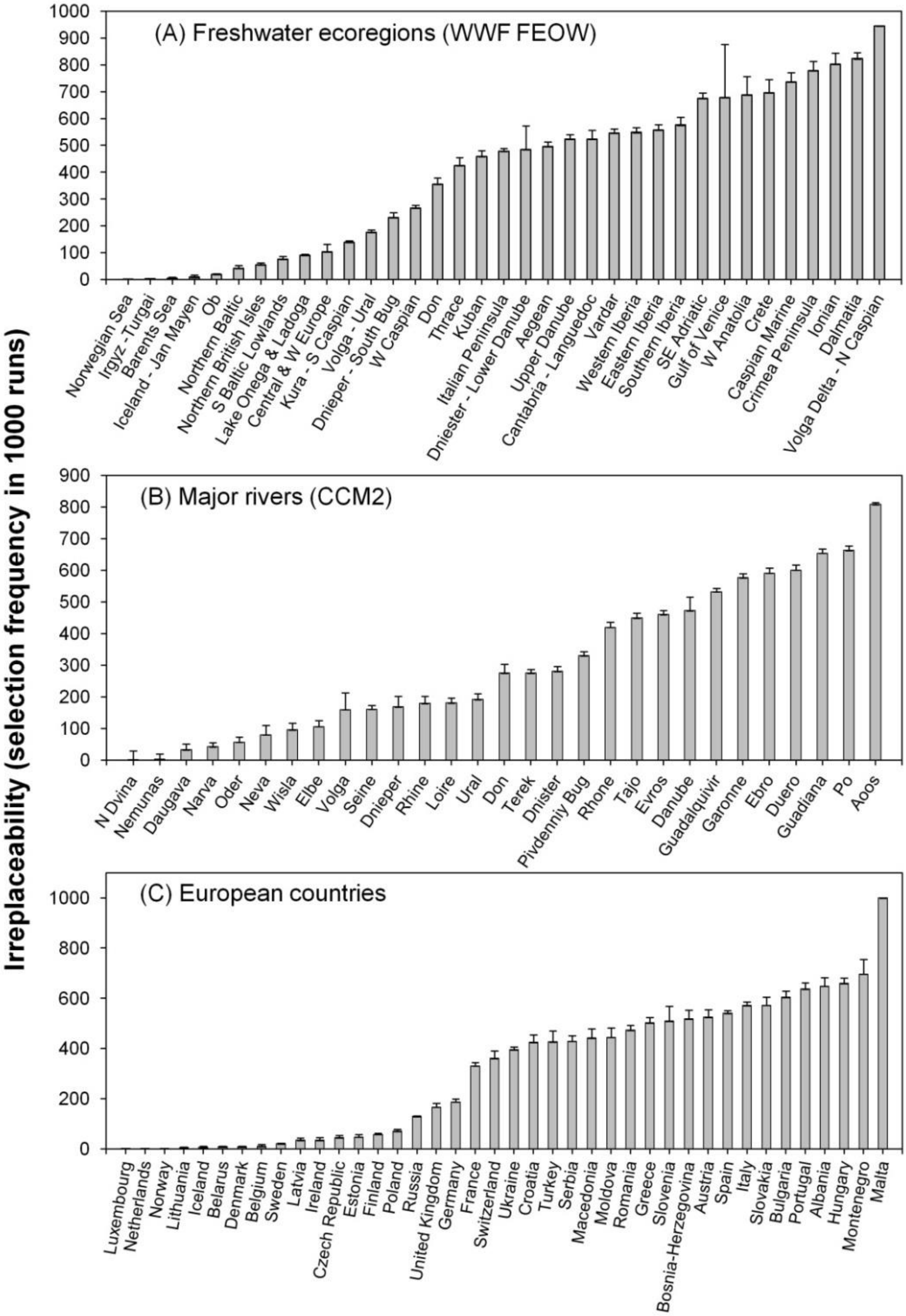


Figure 4.2 Mean ± SE irreplaceability (selection frequency) of freshwater ecoregions (A), major river systems (based on the European CCM2 system) (B), and countries (C) of Europe. Fig. 4.2D. shows the proportion of protected areas per catchment.

Catchments in S Europe generally had high irreplaceability and high protection, whereas catchments in NE Europe had low irreplaceability and low protection (Fig. 4.3A.). In contrast, catchments in the Balkans (Albania, Bosnia and Herzegovina, Croatia, FYR Macedonia, Montenegro, Serbia, Thrace in Turkey), in S Ukraine and central and S Russia had high irreplaceability and low protection. Most catchments in NW Europe had low irreplaceability and high protection (Fig. 4.3A.).

Irreplaceability and the proportion of protected areas in a catchment were positively related (slope $2.5 \times 10^{-4} \pm$ S.E. 1.69×10^{-6}; r=0.122; n=18,816; $p < 0.0001$). The distribution of residuals confirmed that catchments in the W Balkans, S Ukraine and Russia had higher irreplaceability than predicted based protection level (blue colours in Fig. 4.3B.). Catchments with lower irreplaceability than predicted were in N Europe (Iceland, Sweden, Finland, Ural region of Russia) and in central and S Europe (red colours in Fig. 4.3B.). Catchments on the Iberian peninsula showed a mixed pattern, whereas smaller residuals showed adequately protected catchments in central and southern Europe (Belgium, Bulgaria, Germany, Greece, Hungary, S Italy, Netherlands, Poland, , Romania, , Slovakia, Spain,) and the E Balkans but less so in N Italy and SW France (Fig. 4.3B.).

Lastly, deviations from the ideal 1:1000 line showed high irreplaceability and low protection in most catchments south of the 49° latitude (i.e., Upper Danube), and in the lower Volga and Lake Ladoga (blue colours in Fig. 4.3C.). The majority of catchments N from the 49° latitude did not show large deviations (light yellow colour in Fig. 4.3C.), indicating more or less adequate protection. In contrast, many catchments in northern Europe showed negative deviations, i.e., lower irreplaceability than expected (red colours in Fig. 4.3C.), with large deviations especially in N Europe (Iceland, Sweden, Finland, Russia) and smaller deviations in central Europe (Belgium, Netherlands, Germany, Poland, central Romania).

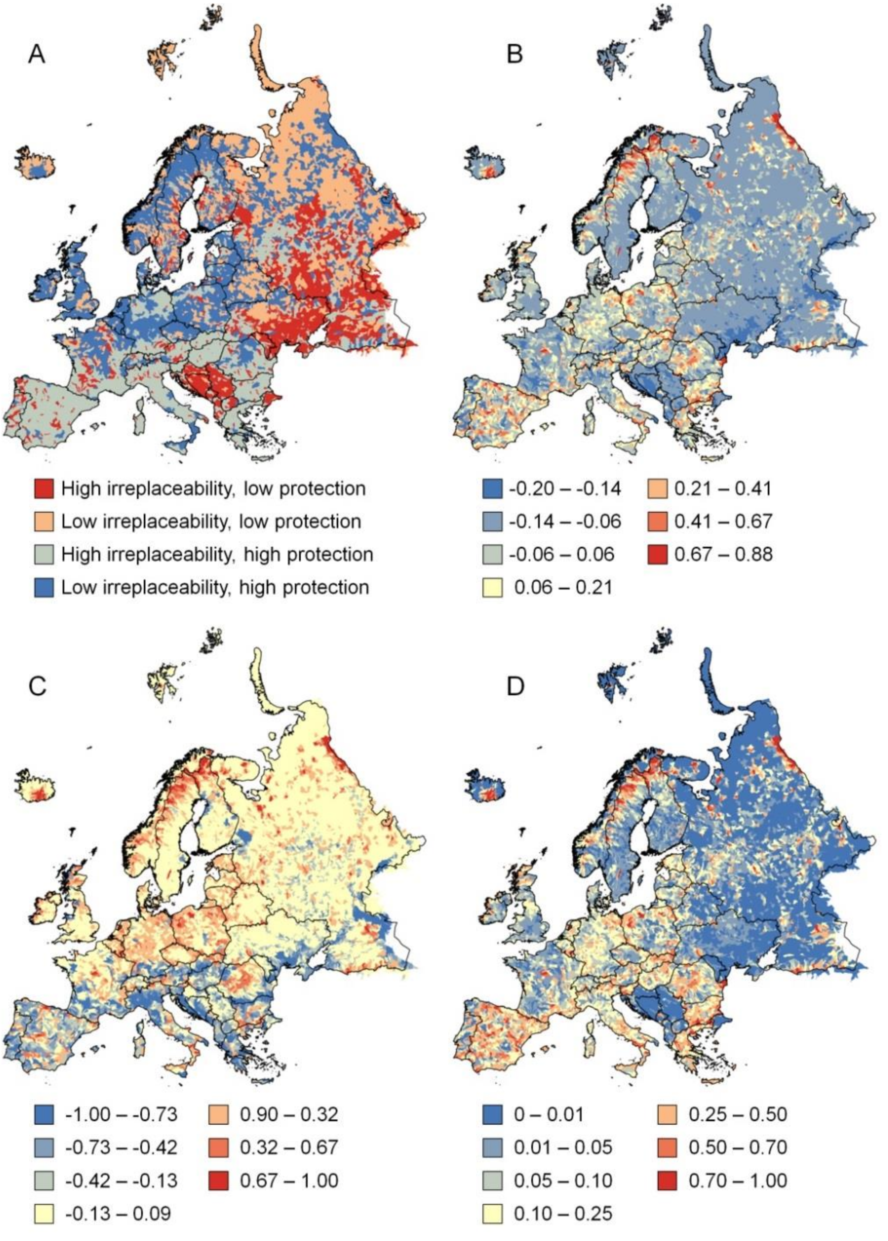


Figure 4.3 Evaluation of the correspondence between irreplaceability and proportion of protected areas. (A) Distribution of catchments in four combinations of irreplaceability and proportion of protected areas, divided at the medians as thresholds. (B) Map of residual values from an ordinary least squares linear regression of irreplaceability over the proportion of protected areas. (C) Deviations from a hypothetical line of 1:1000 of the ideal relationship between proportion of protected areas (ranging from 0 to 1, X axis) and irreplaceability (0 to 1000, Y axis). (D) The proportion of combined protected areas in European catchments (for reference only).

The exclusion of catchments rich (>70%) in protected areas identified a number of catchments that can be proposed for increased protection (Fig. 4.4.). Compared to Scenario 1A, the number of irreplaceable catchments decreased (to n=510), particularly in N and central Europe and along major river systems, and irreplaceability remained high in the W Balkans, S Spain, S France and N Alps.

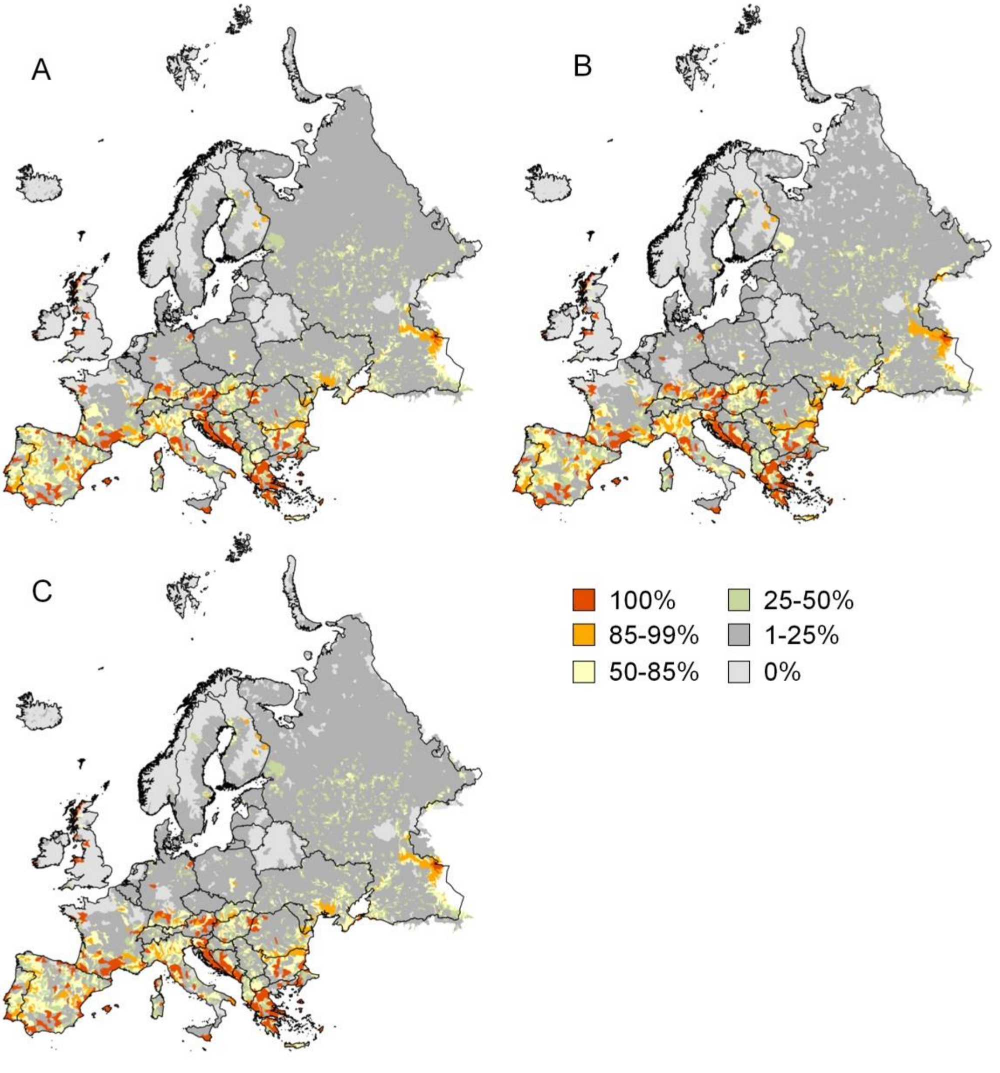


Figure 4.4 Prioritisation based on irreplaceability (selection frequency) of European catchments (A) when catchments rich in Natura 2000 areas (>70% of total area) are excluded (locked out), (B) when catchments rich in nationally protected areas (>70% of total area) are locked out, and (C) when catchments rich in both Natura 2000 and nationally protected areas (combined proportion >70% of total catchment area) are locked out.

The number of species with targets not fulfilled was higher in Scenarios 1A-D (well-protected areas included) than in Scenarios 2A-C (well-protected areas excluded) (Tables 4.1-2.). In addition, the exclusion of catchments rich in Natura 2000 areas resulted in a higher number of threatened species not adequately covered (n=63) than when catchments rich in nationally protected areas were excluded (n=28) (Table 4.2.).

Table 4.2 Number of species for which representation targets were met (Yes) and not met (No) under three scenarios varying by whether catchments rich in protected areas were excluded from the prioritisation.

	Catchments excluded when coverage is adequate by					
	Natura 2000 PAs		National PAs		Both	
Conservation status	Yes	No	Yes	No	Yes	No
Critically Endangered (CR)	125	28	139	14	126	27
Endangered (EN)	129	18	141	6	129	18
Vulnerable (VU)	251	17	260	8	252	16
Near Threatened (NT)	126	1	126	1	126	1
Least Concern (LC)	828	6	830	4	829	5
Data Deficient (DD)	100	2	100	2	100	2

Discussion

Our results showed high conservation priority of rivers and lakes in S Europe, large rivers in E Europe and lakes in N Europe, and of karst/limestone areas in the W Balkans and NW Greece, S France, W Bavaria and the E Alps. The inclusion of catchments rich in protected areas resulted in clearer priorities and better coverage of threatened species. S and E European catchments with high irreplaceability were adequately covered by protected areas in countries with Natura 2000 network (EU member states) but not in others (former Yugoslavia, Ukraine, S Russia). We found evidence of inadequate protection of catchments in S Europe and generally high protection of catchments in NW Europe relative to their conservation priority. The exclusion of catchments rich in protected areas also confirmed the importance of catchments in S and E Europe.

The overwhelming importance of catchments in S Europe corresponds well with biogeographical patterns and processes. First, the general latitudinal gradient in biodiversity, i.e., increasing species richness from north to south on the northern hemisphere, is also found in European freshwater fish (Griffiths 2006). The gradient is particularly expressed in Europe, where southern areas provided refuges during Pleistocene glaciation events, and the post-glacial recolonisation of northern areas took place from these refuges (Hewitt 1999), mainly from the middle and lower Danube basin in the case of freshwater fish (Griffiths 2006; Reyjol et al. 2012) and insects (Bálint et al. 2012). Refuges in glaciated areas (due e.g. to hydrothermal activity or proximity of oceans) could also be sites of reproductive isolation and subsequent speciation, resulting in unique species in some northern catchments (Stewart & Lister 2001). Similarly, southern areas separated from northern ones by mountain ranges that function as migration barriers (e.g. Pyrenées in Iberia, Alps in Italy, Dinarides in the western Balkans) evolved isolated faunas rich in endemics (Griffiths 2006). Finally, karst/limestone areas could serve both as glacial refuges for some species (e.g. fish) and interglacial refuges for cold-tolerant others (e.g. spring snails).

Our results provide relevant input for conservation policy and action. Our analysis identified several catchments and regions with lower or higher levels of protection relative to their conservation priority for freshwater biodiversity. Such information could provide a basis for the re-evaluation of the distribution/allocation of conservation resources to catchments and regions. As a general minimum rule, more resources should be allocated to irreplaceable catchments that are seriously underprotected (these are highlighted in Fig. 4.4C.). For example, the Western Balkans (mainly Croatia, Bosnia and Herzegovina, Montenegro) that have the highest concentration of irreplaceable catchments and catchments in urgent need of conservation in Europe are also a global hotspot for ongoing or future hydropower development (Zarfl et al. 2014). Our results provide a basis for a more efficient allocation of resources for the protection of freshwater biodiversity and will thus be of interest to conservation scientists, water management authorities, policy-makers and the general public.

This study expands and complements our previous identification of critical catchments in Europe (Chapter 3, Carrizo et al. 2017) by providing novel insights regarding conservation priority and current protection. Our current study builds on a 26% larger database and prioritisation focuses on all catchments rather than only on critical catchments. Our current study also provides summary statistics on the conservation priority of ecoregions, major rivers and countries, and presents three novel methods for studying the match between conservation priority and current protection. Finally, we explicitly identify catchments that are in greatest need of increased conservation attention in terms of area protection, management, and restoration.

Three refinements can further improve our analysis. First, the inclusion of hydrological connectivity would be important because upstream effects largely determine biodiversity and conservation status downstream (Linke et al. 2011). Similarly to other studies of larger scale (Lawrence et al. 2011), our prioritisation did not consider connectivity because hydrological relationships were not fully resolved in our planning unit layer. As a partial remedy, connectivity was accounted for indirectly by the manipulation of boundary length. Second, information on ecological conditions or environmental threats would greatly improve the prioritisation of catchments (Linke et al. 2012a; 2012b). We could not identify such information readily available for use with our planning units at the scale of the study. Finally, our analysis could be further improved by more precise measures of protection. For example, an emphasis on protected areas specifically designated for freshwater species and habitats rather than for terrestrial ones would greatly enhance the applicability of our results in conservation practice.

In conclusion, our results draw attention to the high priority of catchments in southern Europe, particularly the Balkans, and in eastern Europe, particularly in southern Ukraine and Russia, and to karst/limestone areas in the conservation of freshwater biodiversity of Europe. These results are directly applicable in European, regional and local conservation efforts and provide a basis for potential future refinements. Similar continental-scale assessments based on ecologically meaningful planning units and data from terrestrial and marine biodiversity may greatly improve the efficiency of the allocation of international conservation effort.

General discussion

My dissertation provides the following key results: a database on the occurrence of amphibian and reptile species with knowledge on the factors influencing patterns in the diversity of these two groups in Albania (Chapter 1), the confirmation of one additional species and the finding of one species new to the fauna of Albania (Chapter 2), the foundations of a European conservation plan for freshwater biodiversity (Chapter 3), and a detailed analysis of spatial priorities for freshwater biodiversity in relation to protected areas in Europe (Chapter 4).

In Chapters 1 and 2, I showed how to establish a spatial database on occurences of selected animal groups in a data deficient area where previous records were dispersed in multiple sources. The steps taken here can help guide future efforts of data collection on species distributions in data-poor countries such as Albania. An atlas on the vascular plants of Albania has also been published recently (Barina 2017), however, there are no other comprehensive atlas-like publications for other taxonomic groups. Hopefully, my studies and others will catalyse future summaries of species distributions and analyses of diversity patterns in Albania or other countries of the Balkan Peninsula, a hotspot of many unique species and habitats.

My results from Chapter 1 show that land cover diversity is the most important explanatory variable for amphibians and reptiles in the scale of Albania. On the other hand, the other important factors differed for the two groups. Precipitation is more important for amphibians while temperature for reptiles. This indicates that local hotspots can be in different parts of the country for different animal groups which should be considered in future conservation actions.

In Chapters 3 and 4, I provide a case study on the usage of such databases in spatial conservation prioritisation. We revealed that many high-priority areas suffer from less-than-adequate protection in Europe. As expected, the Natura 2000 network of protected areas appeared more adequate to cover and protect biodiversity in Europe than only national networks of protected areas. This is expected because the Natura 2000 network was established based on an EU-wide policy framework discussed and agreed by the Member States, covers larger areas and was directly implemented to cover species and habitats of community importance (see Chapter 4, Gruber et al. 2012, Szabolcs et al. under review). For example, the rate of land use change has been slower in areas included in the Natura 2000 network compared to areas remaining outside the network in Hungary (Biró, Bölöni and Molnár 2018), although such an effect was not observed in hotspots for bat species on the Iberian peninsula (Lisón and Sánchez-Fernández 2017). Similarly, the current Natura 2000 network provides adequate coverage for amphibians in Europe, mostly because their distributions are centered in southwestern Europe, a part of the generally well-protected Western Europe (Thuiller et al. 2015). In contrast, the current network poorly covers other tetrapods (mainly squamate reptiles and mammals), whose distributions are centred in southeastern Europe, which is in the generally poorly protected Eastern Europe (Thuiller et al. 2015). My scientifically sound results in Chapter 4 provide further insights on the shortfalls in the Natura 2000 system, and offer direct knowledge to improve the protection of biodiversity, especially in guiding the implementation of the Natura 2000 network in countries aspiring to become EU member states in the Balkans.

Results presented in Chapter 3 and 4 showed an unparalleled importance of the Balkan region, especially the Dinaric coast and the Western Balkans, in the conservation of freshwater biodiversity of Europe. This latter area is the richest in terms of freshwater biodiversity in Europe. Unfortunately, it is also under serious pressure because multiple dams are currently under construction or in the permission-phase right now (Zarfl et al. 2014). From Slovenia to Albania, many river sections will be dammed and turned into reservoirs to produce electricity feeding the region’s growing energy demand. Dams will change flow regime, increase habitat fragmentation and reshape the landscape (Anderson et al. 2018). It is assumed that it will have negative effects on local biodiversity as most species living there are adapted to oxygen-rich, fast flowing rivers which will be turned into still water (Graf et al. 2018). Therefore, there is an urgent need to direct attention to these threatened habitats that hold many endemic species or unique species assemblages and to understand the potential effects of flow regime change and habitat fragmentation on species living in rivers planned for damming. For example, by investigating their functional traits, studies can elicit which species will be the winners and the losers of this large-scale landscape alteration and how species assemblages will look like in the future.

The Balkan Peninsula is also very rich in amphibians and reptiles. There are a total of 114 species (37 amphibians, 77 reptiles), which corresponds to about half of the European herpetofauna. This region is culturally and politically complex, with 11 sovereign countries with other subnational entities. Five of the countries, Slovenia, Croatia, Greece, Bulgaria and Romania are part of the European Union. The others, Bosnia and Herzegovina, Montenegro, Albania, Serbia, F.Y.R.O. Macedonia and Turkey are currently aspiring to join EU in the future. EU membership causes big differences among these countries in terms of conservation strategies and protected areas because EU member states have Natura 2000 areas beyond the areas protected at the national level. Among EU member states, the coverage by protected areas is highest in Slovenia with ~54% of the country territory being protected (which makes the country a world leader), while the others have ~30-40%. Among non-EU states, Albania has the highest coverage with ~17%, while the others have only around 10%. In our future work, we would like to expand the herpetological database of Albania to all countries of the Balkan Peninsula and to conduct a conservation assessment for the whole of the Balkans, which is expected to provide important support for the future development of protected areas in the region. As a biologist engaged in conservation, I hope that I can contribute to the prosperity of wildlife and people in the areas I studied in the thesis and that I can continue this in the future with meaningful, scientifically sound results.

Conclusions

The chapters of my dissertation are arranged along an arch of studies typical in conservation biogeography, ranging from data collection by literature search or field survey (Chapters 1 and 2), through mapping and spatial data analysis (Chapter 1) to application in conservation prioritisation (Chapters 3 and 4). Such series of studies are rarely done in Hungary and Europe. One reason is that the flora and fauna of most of the western and central European countries are well-known and the protected area system is largely fixed. This is not the case for the Balkans, where there are still considerable shortcomings in our knowledge on the occurrence of species and the protected area system is still being developed in several countries. I hope that my work will thus be relevant both in biogeography and conservation especially in that region.

Általános összefoglalás

Disszertációmban térbeli biodiverzitási adatbázisok létrehozásának ismertetésével és ezek felhasználhatóságával foglalkozom az ökológiában és a természetvédelemben. Az első két fejezet ezen adatbázisok elkészítésével foglalkozik Albánia kétéltű- és hüllőfaunájának példáján. A második két fejezet a térbeli természetvédelmi prioritizálást mutatja be Európa édesvízi biodiverzitásának segítségével.

Az első fejezet az Albániában előforduló kétéltűek és hüllők elterjedésével, diverzitásával és az ezeket befolyásoló tényezőkkel foglalkozik. Biodiverzitás szempontjából Albánia Európa egyik legkevésbé feltárt országa, annak ellenére, hogy területe a mediterrán biodiverzitási forróponton fekszik, ahol globális léptékben is kiemelkedően magas a fajgazdagság. Kétéltűekről és hüllőkről az utolsó átfogó faunalista és elterjedési munka 1989-ben jelent meg. Azóta tovább nőtt az előfordulási adatokat közlő cikkek száma, ezek azonban vagy nem országos léptékűek voltak, vagy nem tartalmazták a teljes fajkészletet. Az Albániából kimutatott fajok száma szintén tovább nőtt. Egyrészt a nehezen megközelíthető helyeken élő, korlátozott elterjedésű fajok eddig elkerülték a kutatók figyelmét, másrészt molekuláris genetikai módszereknek köszönhetően eddig egységesnek vélt fajok egyes evolúciós leágazásait ma már külön fajokba soroljuk. A fentiek fényében célul tűztük ki, hogy egységesítjük Albánia kétéltű és hüllő listáját, valamint elkészítjük az egyes fajok legfrissebb elterjedési térképeit. Munkánk során feldolgoztuk az irodalmi, múzeumi és közösségi tudományos oldalakon lévő adatokat. Összesen 21 expedíció alatt számos további előfordulási adatot gyűjtöttünk. Az adathiányos régiókra, valamint a kevés előfordulási adattal rendelkező fajokra külön figyelmet fordítottunk.

Összesen 16 kétéltű- és 42 hüllőfaj jelenlétét összegeztük Albániában, szinte minden fajnál sikerült eddig nem publikált adatokat is gyűjtenünk. Egyes fajok csak az ország határvidékén fordulnak elő, sokszor speciális élőhelyeken, mint a magashegyi alpesi szalamandra (*Salamandra atra*), vagy a homokkő dombokon élő homoki boa (*Eryx jaculus*). Más fajok országos elterjedésűek és sokféle helyen élnek, mint a sárgahasú unka (*Bombina variegata*) vagy a görög teknős (*Testudo hermanni*).

Általános lineáris kevert modellekkel választ kerestünk arra is, hogy milyen környezeti tényezők befolyásolják a fajok elterjedését. Mindkét csoportnál az élőhelyi változatosságnak döntő hatása volt a jelenlétre, értelemszerűen a sokféle élőhely sok fajnak ad otthont. Kétéltűeknél a csapadéknak volt még fontos szerepe, amely azt jelenti, hogy rendelkezésre állhat kellő számú vizes élettér szaporodásukhoz. Hüllők esetében a tengerszint feletti magasság változatossága volt fontos, amely közvetetten befolyásolja az élőhelyi változatosságot. Emellett a legtöbb faj inkább alacsonyabban, nagyjából 1000 m tengerszint feletti magasságig fordul elő, feljebb csak néhány hidegtűrő faj képes megélni. A hőmérséklet és csapadék változatosságának szintén fontos szerepe volt, a sokféle klimatikus viszony sokféle fajnak jelent lehetőséget (Mizsei et al. 2017b, Szabolcs et al. 2017).

A második fejezetben bemutom a korábban Albániából alig ismert *Podarcis siculus* ismételt megtalálását valamint a *Pelobates syriacus* kimutatását az országból. A *P. siculus*-nak korábban egy adata volt ismert az Észak-Albániában található Velipojë település mellől. Az előfordulást egy cseh nyelvű konferencia kiadványban ismertették (Uhrin and Šíbl 1996). Az adatbázis elkészítésekor külön figyelmet fordítottunk ritka fajok további előfordulásainak gyűjtésére. Ennek okán felkerestük e faj eredeti megtalálásanak helyszínét ahol egy további példányt sikerült megfigyelnünk, valamint innen 15 km-re újabb példányokat látnunk.

A *P. syriacus* nem volt ismert Albániából, de valószínűsíthető volt, mivel előfordul a szomszédos Macedónia Volt Jugoszláv Köztársaságban valamint Görögországban a Preszpa és Ohrid tavak közelében az albán határvidéken. Korábban szerepelt Bruno (1989) munkájában, mint Albániában potenciálisan előforduló faj, továbbá szerepelt egy az albán faunát ismertető listában (Dhora 2010) de konkrét előfordulási adata nem volt ismert. A faj felkutatásának érdekében utat szerveztünk a Preszpa-tó partjára, ahol sikeresen kimutattuk Albániából, két alkalommal összesen három egyedet találtunk.

A harmadik fejezetben bemutatom az előfordulási adatbázisok alkalmazását a természetvédelemben. Az édesvizekre (tavak, folyók, vizes élőhelyek) jellemző ökológiai rendszerek összes kiterjedése nem éri el a Föld felszínének 1%-át, ennek ellenére az eddig megismert fajok 10%-a megtalálható bennük. Az édesvizek továbbá alapvető ökoszisztéma-szolgáltatásokat nyújtanak az emberiségnek például táplálékforrásként és fontos szabályozó szerepük van például a víztisztítás, az árvízvédelem és a klímaváltozás elleni védekezés területén. Az édesvízi ökoszisztémák biológiai sokfélesége azonban jóval gyorsabban fogyatkozik, mint a szárazföldi vagy tengeri ökológiai rendszereké. Az édesvízi fajok közel harmada (29%-a) veszélyeztetett, mely messze a legmagasabb arány a három fő ökoszisztéma-típus között. A fogyatkozás legfőbb oka az édesvízi élőhelyek eltűnése és állapotának romlása a szennyezések és hidromorfológiai beavatkozások következtében. Az édesvízi ökoszisztémák veszélyeztetettsége magas, ennek ellenére ritkán állnak a természetvédelem fókuszában. A folyókat például ritkán veszik figyelembe védett területek céljaként, legtöbbször csak a védett területek határaként használják, és a védelem ritkán irányul magára a víztestre. Az édesvízi ökoszisztémák védelme ezért sürgető feladat, melyet a folyóvízi rendszerek hidrológiai kapcsoltsága miatt nemzetközi szinten, megfelelő tudományos alapozással kell megtervezni és kivitelezni.

Disszertációmban bemutatom az európai vízgyűjtőterületek természetvédelmi szempontú prioritizálását az édesvizek jobb védelmének elősegítése érdekében. A munka során közel 1296 édesvízi faj (halak, puhatestűek, szitakötők és vízinövények) elterjedési adatai és veszélyeztetettségi státusza alapján Európa 18816 vízgyűjtőterületének (kb. 10 millió km^2) határoztuk meg a természetvédelmi prioritását. Kritikus vízgyűjtőnek számított egy adott vízgyűjtő, ha onnan ismert volt legalább egy veszélyeztetett faj (IUCN kategória Kritikusan Veszélyeztetett, CR, Veszélyeztetett, EN vagy Sebezhető, VU), vagy legalább egy korlátozott elterjedésű faj (20,000 km^2 halak, puhatestűek és növények esetében, 50,000 km^2 szitakötőknél), vagy az előforduló fajok legalább 25%-a endemikus az adott édesvízi biogeográfiai régióban. A vízgyűjtőnél térképként a HydroBASINS (Lehner and Grill 2013) adatbázist használtuk.

Az eredmények szerint Európa mintegy 19000 vízgyűjtőterületének 45%-a ad otthont legalább egy veszélyeztetett vagy korlátozott elterjedésű fajnak vagy endemikus (bennszülött) fajokban gazdag közösségnek. A veszélyeztetett, korlátozott elterjedésű vagy endemikus fajok száma általában északról dél felé haladva nőtt Európában és a Balkán-félsziget nyugati és déli részén érte el a maximumát (pl. 69 faj az Ohrid-tóban Albánia és Macedónia határán). Kimutattuk továbbá az Európában található Alliance for Zero Extinction (AZE) helyszíneket is. Ezen a 65 helyszínen olyan CR és EN IUCN státuszú fajok élnek, amelyeknek már csak egyetlen előfordulási helye van.

A negyedik fejezetben szintén az európai édesvízi biodiverzitással foglalkoztam, de itt elsősorban a természetvédelmi értékesség és a védett terület hálózat közötti kapcsolaton volt a hangsúly. Itt továbbra is a HydroBASINS adatbázis vízgyűjtőit használtuk tervezési egységként, de az elemzéshez 1631 fajra kibővített fajlistával dolgoztunk. A vízgyűjtőket az ott élő fajok természetvédelmi státusza, elterjedése és endemizmusa alapján prioritizáltuk. CR fajok esetében az elterjedés 100%-a, EN fajoknál 75%-a, VU fajoknál 50%-a, míg korlátozott elterjedésű fajoknál 25%-a számított prioritásnak. Azokat a vízgyűjtőket, ahol az egyes édesvízi biogeográfiai régiókra endemikus fajok száma 5% vagy e fölötti volt szintén prioritásterületként kezeltük. Minden egyéb faj esetében két előfordulási adatot adtunk meg. Az elemzésekhez MARXAN nevű szisztematikus természetvédelmi tervezésben használt programot vettük igénybe, ami 1000 futtatás eredményét összegezve mutatatta meg az egyes vízgyűjtők értékességét.

A vízgyűjtők védett területek általi lefedettségét két szcenárió alapján vizsgáltuk. Első körben védett területek részét képezték az optimális prioritás hálózatnak. Külön és együtt is dolgoztunk az országos és a Natura 2000 hálózatokkal. Második körben ezen területeket kizártuk, hogy megvizsgálhassuk hol van lehetőség további védelemre.

A vízgyűjtők prioritási értéke és a védettség közötti kapcsolatot további három módszerrel is megvizsgáltuk. Először négy kategóriába soroltuk a vízgyűjtőket az alapján, hogy mekkora a prioritási értékük és a védettségük, a határérték pedig a medián volt. Másodszor a prioritási érték és a védettség alapján számolt lineáris regressziókor kapott reziduálisok értékét elemezük. Harmadszor egy 1:1000 hipotetikus ideális vonaltól való eltérést vizsgáltuk ahol az X tengely a védettség mértéke (0 és 1 között) az Y tengely pedig a választási gyakoriság (0 és 1000 között).

Eredményeinkből kiderült, hogy a legmagasabb prioritási értékkel elsősorban a Mediterrán félszigetek, Közép-Európa, valamint a nagy folyók egyes szakaszai (Duna, Volga) rendelkeznek. Kisebb mértékben Észak-Európában is megtalálhatóak fontos területek (pl. Ladoga-tó).

Az egyes vízgyűjtők és így az ott élő természetvédelmi szempontból fontos fajok lefedettsége akkor volt a legmagasabb, mikor figyelembe vettük a Natura 2000 hálózatot is, ami jelzi ezen országokon átívelő egyezmény fontosságát.

A magas prioritási értekkel rendelkező területek és a védett területek közötti kapcsolat vizsgálatából általánosságban az mondható el, hogy Közép- és Nyugat-Európában a védett területek általi lefedettség jónak mondható, Észak-Európában sokszor még aránytalanul magas is. Kelet-Európában és a Balkán-félszigeten azonban a magas értékesség dacára kevés a védett terület.

További védett területek kialakítására elsősorban a Mediterráneumban, azon belül is a Dinári-karsztvidéken lenne szükség. E terület nagy változásoknak néz elébe. Egyrészt, az itt lévő nyugat-balkáni országok csatlakozni kívánnak az Európai Unióhoz, ami maga után vonja a Natura 2000 hálózat kialakítását, így magassabb lesz a védett területek aránya. Másrészt a régióban tapasztalható gazdasági fellendülés miatt több száz vízierőmű áll tervezés vagy építés alatt, ami károsan hathat a biodiverzitásra (Zarfl et al. 2014).

Mint a természetvédelem iránt elkötelezett biológus remélem, hogy a disszertációmban szereplő eredmények hozzájárulnak majd az élővilág és az azzal kapcsolatban élő emberiség jólétéhez, valamint ezekre építve folytatni tudom az elkezdett munkát.

Acknowledgements

First of all I thank my supervisor Dr. Szabolcs Lengyel who helped me reach my goals since I was a university student and then gave me the possibility to participate in the PhD program and to work in the Hungarian Academy of Sciences, Centre for Ecological Research. He enabled for me a professionally, financially and atmospherically proper working environment where I have the chance to step forward in my career. Whichever part of the World I will land in the future I hope we will continue to investigate exciting research areas.

I thank Edvárd Mizsei with whom I worked closely during the years. He introduced me to his work with the Greek meadow viper in Albania which I joined and later we established our Albanian herpetological distribution project which is currently expanding towards the whole Balkan Peninsula.

I am grateful to all my coauthors I worked with on the manuscripts which are the basis of this thesis, namely Daniel Jablonski, Marcel Uhrin, Zsolt Végvári, Béla Mester, Balázs Vági, Felícia Kapusi, Savrina Carrizo, Hans Kasperidus, Mathias Scholz, Danijela Markovic, Jörg Freyhof, Núria Cid, Ana C. Cardoso and William Darwall.

I was funded by Balassi Institute, Campus Hungary Program (B2/1CS/11521, B2/1CS/19196), OTKA (K106133), GINOP (2.3.3-15-2016-00019), SEH Grant in Herpetology - M&D 2015 and the BioFresh Project (FP7-ENV-2008, no. 226874).

References

Abell R, Allan J, Lehner B 2007 Unlocking the potential of protected areas for freshwaters. Biological Conservation 134: 48–63.

Abell R, Thieme ML, Revenga C, Bryer M, Kottelat M, Bogutskaya N, Coad B, Mandrak N, Balderas SC, Bussing W, Stiassny MLJ, Skelton P, Allen GR, Unmack P, Naseka A, Ng R, Sindorf N, Robertson J, Armijo E, Higgins JV, Heibel TJ, Wikramanayake E, López HL, Reis RE, Lundberg JG, Pérez MHS, Petry P 2008 Freshwater Ecoregions of the World: A New Map of Biogeographic Units for Freshwater Biodiversity Conservation. BioScience 58: 403–414.

Abellán P, Sánchez-Fernández D, Velasco J, Millán A 2007 Effectiveness of protected area networks in representing freshwater biodiversity: the case of a Mediterranean river basin (south-eastern Spain). Aquatic Conservation: Marine and Freshwater Ecosystems 17: 361-374.

Adamopoulou C 2015 First record of *Podarcis siculus* (Rafinesque-Schmaltz, 1810) from Greece. Herpetozoa 27: 187-188.

Aliaj Sh, Baldassarre G, Shkupi D 2001 Quaternary subsidence zones in Albania: some case studies. Bulletin of Engineering Geology and the Environment 59: 313-318.

Allan JD, 2004 Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution and Systematics 35: 257–284.

Anderson EP, Jenkins CN, Hellpern S, Maldonado-Ocampo JA, Carvajal-Vallejos FM, Encalada AC, Rivadeneira JF, Hidalgo M, Cañas CM, Ortega H, Salcedo N, Maldonado M, Tedesco PA 2018 Fragmentation of Andes-to-Amazon connectivity by hydropower dams. Science Advances 4: eaao1642.

Ardron JA, Possingham HP Klein CJ 2010 Marxan Good Practices Handbook, Version 2. Pacific Marine Analysis and Research Association, Victoria, BC, Canada.

Arnold N, Ovenden DW 2002 Reptiles and amphibians of Europe. London, Collins. pp. 384.

Bálint M, Málnás K, Nowak C, Geismar J, Váncsa É, Polyák L, Lengyel S, Haase P 2012 Species history masks the effects of human-induced range loss - unexpected genetic diversity in the endangered giant mayfly *Palingenia longicauda*. PLoS ONE 7:e31872.

Ball IR, Possingham HP Watts M 2009 Marxan and Relatives: Software for Spatial Conservation Prioritisation. Spatial conservation prioritisation: Quantitative methods and computational tools (eds Moilanen A, Wilson KA, Possingham HP Chapter 14, pp. 185−195. Oxford University Press, Oxford, UK.

Barina Z (ed.) 2017 Distribution atlas of vascular plants in Albania. Budapest, Hungarian Natural History Museum. pp. 492.

Bartón K 2016 Mumin: Multi-Model Inference. R package version 1.15.6.

Beger M, Grantham HS, Pressey RL, Wilson KA, Peterson EL, Dorfman D, Mumby PJ, Lourival R, Brumbaugh DR Possingham HP 2010 Conservation planning for connectivity across marine, freshwater, and terrestrial realms. Biological Conservation 143: 565–575.

Bogutskaya NG, Zupančič P, Naseka AM 2010 Two new species of freshwater fishes of the genus *Alburnoides*, *A. fangfangae* and *A. devolli* (Actinopterygii: Cyprinidae) from the Adriatic sea basin in Albania. Proceedings of the Zoological Institute RAS 314 (4): 448-468.

Bolkay SJ 1921 Überpflanzen- und gesteinsliebende Lacerten. Blätter für Aquarien- und Terrarien-Kunde 32: 181-184.

Beck J, Böller M, Erhardt A, Schwanghart W 2014 Spatial bias in the GBIF database and its effect on modeling species' geographic distributions. Ecological Informatics 19: 10-15.

Bego F, Loçe E, Topoviti D 2014 First record of the lesser mole rat (*Spalax leucodon*, 1840) in Albania: updated list of small terrestrial mammals. Universi – International Journal of Education, Science, Technology, Innovation, Health and Environment 1: 10-15.

Bird TJ, Bates AE, Lefcheck JS, Hill NA, Thomson RJ, Edgar GJ, Stuart-Smith RD, Wotherspoon S, Krkosek M, Stuart-Smith JF, Pecl GT, Barrett N, Frusher S 2014 Statistical solutions for error and bias in global citizen science datasets. Biological Conservation 173: 144-154.

Biró M, Bölöni J, Molnár Z 2018 Use of long-term data to evaluate loss and endangerment status of Natura 2000 habitats. Conservation Biology 32 (3): 660-671.

Brodie JF 2016 Synergistic effects of climate change and agricultural land use on mammals. Frontiers in Ecology and the Environment 14 (1): 20-26.

Bruno S 1989: Introduction to a study of the herpetofauna of Albania. British Herpetological Society Bulletin 29.

Burnham KP, Anderson DR 2002 Model selection and multimodel inference. New York, Springer-Verlag. pp. 488.

Carranza, S. Amat, F. 2005 Taxonomy, biogeography and evolution of *Euproctus* Amphibia: Salamandridae, with the resurrection of the genus *Calotriton* and the description of a new endemic species from the Iberian Peninsula. Zoological Journal of the Linnean Society 145: 555–582.

Carrizo SF, Lengyel S, Kapusi F, Szabolcs M, Kasperidus HD, Scholz M, Markovic D, Freyhof J, Cid N, Cardoso AC, Darwall W 2017 Critical catchments for freshwater biodiversity conservation in Europe: identification, prioritisation and gap analysis. Journal of Applied Ecology 54: 1209-1218.

Carvalho SB, Brito JC, Pressey RL, Crespo E, Possingham HP 2010 Simulating the effects of using different types of species distribution data in reserve selection. Biological Conservation 143: 426-438.

Carwardine J, Wilson KA, Ceballos G, Ehrlich PR, Naidoo R, Iwamura T, Hajkowicz SA Possingham HP 2008a Cost-effective priorities for global

mammal conservation. Proceedings of the National Academy of Sciences of the United States of America 105: 11446–50.
Carwardine J, Wilson KA, Watts M, Etter A, Klein CJ Possingham HP 2008b Avoiding costly conservation mistakes: the importance of defining actions and costs in spatial priority setting. PLoS ONE 3: e2586.
Casale P, Margaritoulis D 2010 Sea turtles in the Mediterranean: Distribution, threats and conservation priorities. Gland, IUCN. pp. 294.
Central Intelligence Agency 1990 Atlas of Eastern Europe. Washington, USA, courtesy of the University of Texas Libraries. Austin, The University of Texas. http://www.lib.utexas.edu.
Cogălniceanu D, Rozylowicz L, Székely P, Samoilă C, Stănescu F, Tudor M, Székely D, Iosif R 2013a Diversity and distribution of reptiles in Romania. ZooKeys 341: 49-76.
Cogălniceanu D, Székely P, Samoilă C, Iosif R, Tudor M, Plăiaşu R, Stănescu F, Rozylowicz L 2013b Diversity and distribution of amphibians in Romania. ZooKeys 296: 35-57.
Collares-Pereira MJ, Cowx IG 2004 The role of catchment scale environmental management in freshwater fish conservation. Fisheries Management and Ecology 11: 303-312.
Crofts R 2014 The European Natura 2000 protected area approach: a practitioner's perspective. Parks 20 (1): 75–86.
Cushman SA 2006 Effects of habitat loss and fragmentation on amphibians: A review and prospectus. Biological Conservation 128: 231-240.
Cuttelod A, Seddon M, Neubert E 2011 European Red List of Non-marine Molluscs. Publications Office of the European Union, Luxembourg.
Darwall WRT, Vie J-C 2005 Identifying important sites for conservation of freshwater biodiversity: extending the species-based approach. Fisheries Management and Ecology 12: 287–293.
Darwall WRT, Holland RA, Smith KG, Allen D, Brooks EGE, Katarya V, Pollock CM, Shi Y, Clausnitzer V, Cumberlidge N, Cuttelod A, Dijkstra K-DB, Diop MD, García N, Seddon MB, Skelton PH, Snoeks J, Tweddle D, Vié J-C 2011 Implications of bias in conservation research and investment for freshwater species. Conservation Letters 4: 474–482.
Darwall W, Carrizo S, Numa C, Barrios V, Freyhof J, Smith K 2014 Freshwater Key Biodiversity Areas in the Mediterranean Basin Hotspot: Informing species conservation and development planning in freshwater ecosystems. Cambridge, UK and Malaga, Spain: IUCN. 86 pp.
Denoël M, Ficetola GF, 2008 Conservation of newt guilds in an agricultural landscape in Belgium: the importance of aquatic and terrestrial habitats. Aquatic Conservation: Marine and Freshwater Ecosystems 18: 714-728.
de Novaes e Silva V, Pressey RL, Machado RB, VanDerWal J, Wiederhecker HC, Werneck FP, Colli GR 2014 Formulating conservation targets for a gap

analysis of endemic lizards in a biodiversity hotspot. Biological Conservation 180: 1-10.
de Pous P, Beukema W, Weterings M, Dümmer I, Geniez P 2011 Area prioritization and performance evaluation of the conservation area network for the Moroccan herpetofauna: a preliminary assessment Biodiversity and Conservation 20: 89-118.
Dhora D 2010 Register of species of the fauna of Albania. Shkodër, Botimet Camaj-Pipa. pp. 207.
Dolezsai A, Sály P, Takács P, Hermoso V, Erős T 2015 Restricted by borders: trade-offs in transboundary conservation planning for large river systems. Biodiversity and Conservation 24 (6): 1403-1421.
Dudgeon D, Arthington AH, Gessner MO, Kawabata Z-I, Naiman RJ, Knowler DJ, Le C, Lévêque C, Prieur-Richard A-H, Soto D, Stiassny MLJ, Sullivan CA 2006 Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews of the Cambridge Philosophical Society 81: 163–182.
Dufresnes C, Denoël M, di Santo L, Dubey S 2017 Multiple uprising invasions of *Pelophylax* water frogs, potentially inducing a new hybridogenetic complex. Scientific Reports 7: 6506.
Džukić G, Beškov V, Sidorocska V, Cogălniceanu D, Kalezić ML 2008 Contemporary chorology of the spadefoot toads (*Pelobates* spp.) in the Balkan Peninsula. Zeitschrift für Feldherpetologie 15: 1-18.
EC 2011 Our life insurance, our natural capital: an EU biodiversity strategy to 2020. COM/2011/244, European Commission, Brussels.
EC 2012 Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the regions — a blueprint to safeguard Europe's water resources. European Commission, Brussels. Available at: http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52012DC0673&from=EN accessed 15 November 2015.
Eionet 2009 European Topic Centre on Biological Diversity 2009: Online report on Article 17 of the Habitats Directive (2001-2006). Available at: http://bd.eionet.europa.eu/activities/Reporting/Article_17/Reports_2007/index_html (accessed 15 November 2015).
Esselman PC, Allan JD 2011 Application of species distribution models and conservation planning software to the design of a reserve network for the riverine fishes of northeastern Mesoamerica. Freshwater Biology 56: 71-88.
Estrada A, Meireles C, Morales-Castilla I, Poschlod P, Araújo MB, Early R 2015 Species' intrinsic traits inform their range limitations and vulnerability under environmental change. Global Ecology and Biogeography 24 (7): 849-858.
Farkas B, Buzás B 1997 Herpetologische Beobachtungen in den Nordalbanischen Alpen. Herpetofauna 19 (109): 10-18.

Ferchaud A-L, Ursenbacher S, Cheylan M, Luiselli L, Jelič D, Halpern B, Major Á, Kotenko T, Keyan N, Behrooz R, Crnobrnja-Isailovič J, Tomovič L, Ghira I, Ioannidis Y, Arnal V, Montgelard C 2012 Phylogeography of the *Vipera ursinii* complex (Viperidae): mitochondrial markers reveal east-west disjunction in the Palearctic region. Journal of Biogeography 39 (10): 1836-1847.

Ficetola GF, De Bernardi F 2004 Amphibians in a human-dominated landscape: the community structure is related to habitat features and isolation. Biological Conservation 119: 219-230.

Fisher MC, Garner TWJ, Walker SF 2009 Global emergence of *Batrachochytrium dendrobatidis* and amphibian chytridiomycosis in space, time, and host. Annual Review of Microbiology 63: 291-310.

Fjeldså J, Lovett JC 1997 Biodiversity and environmental stability. Biodiversity and Conservation 6: 315-323.

Freyhof J 2010 Threatened mollusks and freshwater fish of the Balkans, Potential impacts of hydropower projects. Berlin, ECA Watch Austria & Euronatur, pp. 81.

Frishkoff LO, Karp DS, Flanders JR, Zook J, Hadly EA, Daily GC, M'Gonigle LK 2016 Climate change and habitat conversion favour the same species. Ecology Letters 19 (9): 1081-1090.

Frommhold E 1962 Herpetologische Studien in Albanien. Aquarium Terrarium 9 (12): 365-370.

Frost DR 2017 Amphibian Species of the World: an Online Reference v6.0. American Museum of Natural History, New York (http://research.amnh.org/herpetology/amphibia)

Gasc J-P, Cabela A, Crnobrnja-Isailovič J, Dolmen D, Grossenbacher K, Haffner P, Lescure J, Martens H, Martínez-Rica JP, Maurin H, Oliviera ME, Sofianidou TS, Veith M, Zuiderwijk A 1997 Atlas of amphibians and reptiles in Europe. Paris, Societas Europaea Herpetologica and Muséum national d'Histoire naturelle. pp. 494.

Gaston KJ 2000 Global patterns in biodiversity. Nature 405: 220-227.

Geiger M, Herder F, Monaghan M, Almada V, Barbieri R, Bariche M, Berrebi P, Bohlen J, Casal-Lopez M, Delmastro G, Denys G, Dettai A, Doadrio I, Kalogianni E, Kärst H, Kottelat M, Kovacic M, Laporte M, Lorenzoni M, Marcic Z, Özulug M, Perdices A, Perea S, Persat H, Porcellotti S, Puzzi C, Robalo J, Sanda R, Schneider M, Slechtova V, Stumboudi M, Walter S, Freyhof J 2014 Spatial heterogeneity in the Mediterranean Biodiversity Hotspot affects barcoding accuracy of its freshwater fishes. Molecular Ecology Resources 14: 1210−1221.

Graf W, Pauls SU, Vitecek S 2018 *Isoperla vjosae* sp.n., a new species of the *Isoperla tripartita* group from Albania (Plecoptera: Perlodidae). Zootaxa 4370 (2): 171-179.

Griffiths D 2006 Pattern and process in the ecological biogeography of European freshwater fish. Journal of Animal Ecology 75: 734-751.

Griffiths HI, Kryštufek B, Reed JM 2004 Balkan biodiversity: Pattern and Process in the European Hotspot. Dordrecht, Kluwer Academic Publisher.

Groves CR, Game ET, Anderson MG, Cross M, Enquist C, Ferdaña Z, Girvetz E, Gondor A, Hall KR, Higgins J, Marshall R, Popper K, Schill S, Shafer SL 2012 Incorporating climate change into systematic conservation planning. Biodiversity and Conservation 21: 1651−1671.

Gruber B, Evans D, Henle K, Bauch B, Schmeller DS, Dziock F, Henry P-Y, Lengyel S, Margules C, Dormann CF 2012 „Mind the gap!" – How well does Natura 2000 cover species of European interest? Nature Conservation 3: 45-63.

Guicking D, Joger U, Wink M 2009 Cryptic diversity in a Eurasian water snake (*Natrix tessellata*, Serpentes: Colubridae): Evidence from mitochondrial sequence data and nuclear ISSR-PCR fingerprinting. Organisms Diversity and Evolution 9: 201-214.

Gvoždík V, Jandzík D, Lymberakis P, Jablonski D 2010 Slow worm, *Anguis fragilis* (Reptilia: Anguidae) as a species complex: Genetic structure reveals deep divergences. Molecular Phylogenetics and Evolution 55: 460-472.

Hadfield JD 2010 MCMC methods for multi-response generalized linear mixed models: the MCMCglmm R package. Journal of Statistical Software 33 (2): 1-22.

Harnik PG, Simpson C, Payne JL 2012 Long-term differences in extinction risk among the seven forms of rarity. Proceedings of the Royal Society of London B: Biological Sciences. 279: 4969-4976.

Hartel T, Nemes S, Cogălniceanu D, Öllerer K, Moga CI, Lesbarrères D, Demeter L 2009 Pond and landscape determinants of *Rana dalmatina* population sizes in a Romanian rural landscape. Acta Oecologica 35: 53-59.

Haxhiu I 1994 The herpetofauna of Albania, Amphibia: species composition, distribution, habitats. Zoologische Jahrbücher 121: 321-334.

Haxhiu I 1998 The reptilia of Albania: species composition, distribution, habitats. Bonner Zoologische Beiträge 48 (1): 35-57.

Haxhiu I, Vrenozi B 2009 Species of amphibians and reptiles of lake Ohri with notes in their ecology. In: International Conference on Lakes and Nutrients, Pogradec (Albania), p. 382-387. Anonymus (ed), Pogradec.

Hedges SB, Marion AB, Lipp KM, Marin J, Vidal N 2014 A taxonomic framework for typhlopid snakes from the Caribbean and other regions (Reptilia, Squamata). Caribbean Herpetology 49: 1-61.

Heiner M, Higgins J, Li X, Baker B 2011 Identifying freshwater conservation priorities in the Upper Yangtze River Basin. Freshwater Biology 56: 89-105.

Herbert ME, McIntyre PB, Doran PJ, Allan JD, Abell R 2010 Terrestrial reserve networks do not adequately represent aquatic ecosystems. Conservation Biology 24: 1002-1011.

Hermoso V, Linke S, Prenda J, Possingham HP 2011 Addressing longitudinal connectivity in the systematic conservation planning of fresh waters. Freshwater Biology, 56: 57−70.

Hewitt GM 1999 Post-glacial re-colonization of European biota. Biological Journal of the Linnean Society 68: 87-112.

Higgins JV, Bryer MT, Khoury ML, Fitzhugh TW 2005 A freshwater classification approach for biodiversity conservation planning. Conservation Biology 19: 432−445.

Hijmans RJ, Cameron SE, Parra JL, Jones PG, Jarvis A 2005 Very high resolution interpolated climate surfaces for global land areas. International Journal of Climatology 25: 1965-1978.

Holland RA, Darwall WRT Smith KG 2012 Conservation priorities for freshwater biodiversity: the Key Biodiversity Area approach refined and tested for continental Africa. Biological Conservation 148: 167–179.

Ilgaz Ç, Kumlutaş Y, Sözen M 2013 New locality record for *Podarcis siculus hieroglyphicus* (Berthold, 1842) (Squamata: Lacertidae) in the western Black Sea region of Anatolia. Turkish Journal of Zoology 37: 123-127.

INSPIRE D2.8.1.1. INSPIRE Specification of Coordinate Reference Systems – Guidelines. (http://inspire.ec.europa.eu/documents/Data_Specifications/INSPIRE_Specification_CR S_v3.0.pdf (accessed 7 May 2014), 2009).

IUCN 2013 IUCN Red List of Threatened Species. Version 2013.2. Available at: www.iucnredlist.org. (Downloaded on 22 April 2014).

IUCN 2014 Joint Task force on Biodiversity and Protected Areas. Consultation Document on an IUCN Standard for the Identification of Key Biodiversity Areas. Draft 1 October 2014.

IUCN 2015 IUCN Red List of Threatened Species. Version 2013.2. Available at: www.iucnredlist.org.

IUCN 2016 A Global Standard for the Identification of Key Biodiversity Areas, Version 1.0. First edition. Gland, Switzerland: IUCN.

IUCN, BirdLife International 2013 Key Biodiversity Areas: Identifying areas of particular importance for biodiversity in support of the Aichi Targets. Seventeenth meeting of the Subsidiary Body on Scientific, Technical and Technological Advice, Montreal 14-18 October 2013. CBD.

Jablonski D, Jandzík D, Gvoždík V 2012 New records and zoogeographic classification of amphibians and reptiles from Bosnia and Herzegovina. North-Western Journal of Zoology 8 (2): 324-337.

Jablonski D, Jandzik D, Mikulíček P, Džukič G, Ljubisavljevič N, Tzankov N, Jelič D, Thanou E, Moravec J, Gvoždík V 2016 Contrasting evolutionary histories of the legless lizards slow worms (*Anguis*) shaped by the topography of the Balkan Peninsula. BMC Evolutionary Biology 16: 99.

Jarvis A, Reuter HI, Nelson A, Guevera E 2008 Hole-filled SRTM for the globe v4.1. http://www.srtm.csi.cgiar.org

Joger U, Fritz U, Guicking D, Kalyabina-Hauf S, Nagy ZT, Wink M 2007 Phylogeography of western Palaearctic reptiles – Spatial and temporal speciation patterns. Zoologischer Anzeiger 246 (4): 293-313.

Jovanović M 2009 Amphibia and reptilian of Štoj Plain (Ulcinj, Montenegro). Bulletin of the Natural History Museum 2: 137-152.

Juffe-Bignoli D, Harrison I, Butchart SHM, Flitcroft R, Hermoso V, Jonas H, Lukasiewicz A, Thieme M, Turak E, Bingham H, Dalton J, Darwall W, Deguignet M, Dudley N, Gardner R, Higgins J, Kumar R, Linke S, Milton GR, Pittock J, Smith KG, Soesbergen A 2016 Achieving Aichi Biodiversity Target 11 to improve the performance of protected areas and conserve freshwater biodiversity. Aquatic Conservation: Marine and Freshwater Ecosystems 26: 133−151.

Kadmon R, Farber O, Danin A, 2004 Effect of roadside bias on the accuracy of predictive maps produced by bioclimatic models. Ecological Applications 14 (2): 401-403.

Keil P, Schweiger O, Kühn I, Kunin WE, Kuussaari M, Settele J, Henle K, Brontos L, Pe'er G, Lengyel S, Moustakas A, Steinicke H, Storch D 2012 Patterns of beta diversity in Europe: the role of climate, land cover and distance across scales. Journal of Biogeography 39 (8): 1473-1486.

Keith P 2000 The part played by protected areas in the conservation of threatened French freshwater fish. Biological Conservation 92: 265-273.

Khoury M, Higgins J, Weitzell R 2010 A freshwater conservation assessment of the Upper Mississippi River basin using a coarse- and fine-filter approach. Freshwater Biology 56: 162−179.

Klein C, Steinback C, Scholz A, Possingham H 2008 Effectiveness of marine reserve networks in representing biodiversity and minimizing impact to fishermen: a comparison of two approaches used in California. Conservation Letters 1: 44–51.

Kolbe JJ, Lavin BR, Burke RL, Rugiero L, Capula M, Luiselli L 2013 The desire for variety: Italian wall lizard *Podarcis siculus* populations introduced to the United States via the pet trade are derived from multiple native-range sources. Biological Invasions 15: 775-783.

Kopstein F, Wettstein O 1920 Reptilien und Amphibien aus Albaniens – Verhandlungen der Zoologisch-Botanischen Gesellschaft in Österreich 70: 387-457.

Korsós Z, Barina Z, Pifkó D 2008 First record of *Vipera ursinii graeca* in Albania (Reptilia: Serpentes, Viperidae). Acta Herpetologica 3 (2): 167-173.

Kundzewicz ZW, Mata L-J, Arnell NW, Doell P, Jimenez B, Miller K, Oki T, Sen Z, Shiklomanov I 2008 The implication of projected climate for freshwater resources and their management. Hydrological Sciences Journal 53 (1): 3-10.

Langhammer PF, Bakarr MI, Bennun LA, Brooks TM, Clay RP, DarwallW, De Silva N, Edgar GJ, Eken G, Fishpool LDC, da Fonseca GAB, Foster MN, Knox DH, Matiku P, Radford EA, Rodrigues ASL, Salaman P, Sechrest W, Tordoff AW 2007 Identification and Gap Analysis of Key Biodiversity Areas: Targets for Comprehensive Protected Area Systems. Gland, Switzerland: IUCN.

Lawrence DJ, Larson ER, Reidy Liermann CA, Mims MC, Pool TK, Olden JD 2011 National parks as protected areas for U.S. freshwater fish diversity. Conservation Letters 4: 364-371.

Lehner B 2012 HydroBASINS Version1.b. Global watershed boundaries and sub-basin delineation derived from HydroSHEDS data at 15 second resolution.. Technical documentation.

Lehner B, Grill G 2013 Global river hydrography and network routing: baseline data and new approaches to study the world's large river systems. Hydrological Processes 27: 2171–2186.

Lemonnier-Darcemont M, Puskás G, Darcemont C 2015 First overview of the south Albanian Orthoptera fauna. Articulata 30: 63-80.

Ligon FK, Dietrich WE, Trush WJ 1995 Downstream ecological effects of dams. BioScience 45: 183-192.

Linke S, Hermoso V, Thieme M 2012a Preliminary results of a freshwater biodiversity Marxan analysis for the Democratic Republic of Congo.). Kinshasa, RDC: Program to Reinforce the Protected Area Network, Départment Technique & Scientifique ICCN.

Linke S, Kennard MJ, Hermoso V, Olden JD, Stein J, Pusey BJ 2012b Merging connectivity rules and large-scale condition assessment improves conservation adequacy in river systems. Journal of Applied Ecology 49: 1036–1045.

Linke S, Norris RH, Pressey RL 2008 Irreplaceability of river networks: towards catchment-based conservation planning. Journal of Applied Ecology 45: 1486-1495.

Linke S, Pressey RL, Bailey RC, Norris RH 2007 Management options for river conservation planning: condition and conservation re-visited. Freshwater Biology 52: 918–938.

Linke S, Turak E, Nel J 2011 Freshwater conservation planning: the case for systematic approaches. Freshwater Biology 56: 6–20.

Lisón F, Sánchez-Fernández D 2017 Low effectiveness of the Natura 2000 network in preventing land-use change in bat hotspots. Biodiversity and Conservation 26 (8): 1989-2006.

Ljubisavljevič K, Arribas O, Džukič G, Carranza S 2007 Genetic and morphological differentiation of Mosor rock lizards, *Dinarolacerta mosorensis* (Kolombatović, 1886), with the description of a new species from the Prokletije Mountain Massif (Montenegro) (Squamata: Lacertidae). Zootaxa 1613: 1-22.

Lukanov S, Tzankov N, Simeonovska-Nikolova D 2013 A comparative study of the mating call of *Pelophylax ridibundus* and *Pelophylax kurtmuelleri* (Anura:

Ranidae) from syntopic and allotopic populations. Journal of Natural History 49: 257-272.
Lymberakis P, Poulakakis N, Manthalou G, Tsigenopoulos CS, Magoulas A, Mylonas M 2007 Mitochondrial phylogeography of *Rana* (*Pelophylax*) populations in the Eastern Mediterranean region. Molecular Phylogenetics and Evolution 44: 115-125.
Mačát Z, Starcová M, Červenka J, Jablonski D, Šandera M 2014 A molecular assessment and first record of *Tarentola mauritanica* (Squamata: Phyllodactylidae) on Corfu, Greece. Salamandra 50 (3): 172-176.
Málnás K, Polyák L, Prill É, Hegedüs R, Kriska Gy, Dévai Gy, Horváth G, Lengyel S 2011 Bridges as optical barriers and population disruptors for the mayfly *Palingenia longicauda*: an overlooked threat to freshwater biodiversity? Journal of Insect Conservation 15: 823-832.
Margules CR, Pressey RL 2000 Systematic conservation planning. Nature 405: 243–53.
Markovic D, Carrizo S, Freyhof J, Cid N, Lengyel S, Scholz M, Kasperidus H, Darwall W 2014 Europe’s freshwater biodiversity under climate change: distribution shifts and conservation needs. Diversity and Distributions 20: 1097−1107.
Marzahn E, Mayer W, Joger U, Ilgaz Ç, Jablonski D, Kindler C, Kumlutaş Y, Nistri A, Schneeweiss N, Vamberger M, Žagar A, Fritz U 2016 Phylogeography of the *Lacerta viridis* complex: mitochondrial and nuclear markers provide taxonomic insights. Journal of Zoological Systematics and Evolutionary Research 54 (2): 85-105.
Mattfeldt SD, Grant EHC 2007 Are two methods better than one? Area constrained transects and leaf litterbags for sampling stream salamanders. Herpetological Review 38: 43-35.
McCain CM 2010 Global analysis of reptile elevational diversity. Global Ecology and Biogeography 19: 541-553.
Médail F, Diadema K 2009 Glacial refugia influence plant diversity patterns in the Mediterranean Basin. Journal of Biogeography 36:1333-1345.
Mester B, Szalai M, Mérő TO, Puky M, Lengyel S 2015 Spatiotemporally variable management by grazing and burning increases marsh diversity and benefits amphibians: a field experiment. Biological Conservation 192: 237-246.
Mizsei E, Jablonski D, Roussos SA, Dimaki M, Ioannidis Y, Nilson G, Nagy ZT 2017a Nuclear markers support the mitochondrial phylogeny of *Vipera ursinii–Vipera renardi* complex (Squamata: Viperidae) and species status for the Greek meadow viper. Zootaxa 4227 (1): 75-88.
Mizsei E, Jablonski D, Végvári Z, Lengyel S, Szabolcs M 2017b Distribution and diversity of reptiles in Albania: a novel database from a Mediterranean hotspot. Amphibia-Reptilia 38: 157-173.

Mizsei E, Uhrin M, Jablonski D, Szabolcs M 2016a First records of the Italian wall lizard, *Podarcis siculus* (Rafinesque-Schmaltz, 1810) (Squamata: Lacertidae) in Albania. Turkish Journal of Zoology 40 (5): 814-817.

Mizsei E, Üveges B 2012 Novel defensive behaviours of both sexes of *Vipera ursinii graeca* Serpentes: Viperidae. Herpetology Notes 5: 481-483.

Mizsei E, Üveges B, Vági B, Szabolcs M, Lengyel S, Pfiegler WP, Nagy ZT, Tóth JP 2016b Species distribution modelling leads to the discovery of new populations of one of the least known European snakes, *Vipera ursinii graeca,* in Albania. Amphibia-Reptilia 37 (1): 55-68.

Moilanen A, Leathwick J, Elith J 2008 A method for spatial freshwater conservation prioritization. Freshwater Biology 53: 577-592.

Moilanen A, Leathwick JR, Quinn JM 2011 Spatial prioritization of conservation management. Conservation Letters 4: 3833-93.

Moilanen A, Wilson KA, Possingham HP 2008 Spatial conservation prioritisation: Quantitative methods and computational tools. Oxford University Press, Oxford, UK.

Moss B 1999 The seventh age of freshwater conservation – a triumph of hope over experience? Aquatic Conservation-Marine and Freshwater Ecosystems, 9, 639–644.

Moss B 2008 The Water Framework Directive: Total environment or political compromise? Science of the Total Environment 400: 32–41.

Myers N, Mittermeier, RA, Mittermeier CG, da Fonseca GAB, Kent J 2000 Biodiversity hotspots for conservation priorities. Nature 403 (24): 853-858.

Nel JL, Reyers B, Roux DJ, Impson ND, Cowling RM 2011 Designing a conservation area network that supports the representation and persistence of freshwater biodiversity. Freshwater Biology 56: 106-124.

Nel JL, Roux DJ, Abell R, Ashton PJ, Cowling RM, Higgins JV, Thieme M, Viers JH 2009 Progress and challenges in freshwater conservation planning. Aquatic Conservation: Marine and Freshwater Ecosystems 19: 474–485.

Nel JL, Roux DJ, Maree G, Kleynhans CJ, Moolman J, Reyers B, Rouget M, Cowling RM 2007 Rivers in peril inside and outside protected areas: a systematic approach to conservation assessment of river ecosystems. Diversity and Distributions 13: 341-352.

Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos M, Stevens HH, Szoecs E, Wagner H 2016 vegan: Community Ecology Package. R package version 2.4-1. https://CRAN.R-project.org/package=vegan

Ord JK, Getis A 1995 Local spatial autocorrelation statistics: distribution issues and an application. Geographical Analysis 27 (4): 286-306.

Oruçi S 2010a Data on geographical distribution and habitats of the marginated tortoise (*Testudo marginata* Schoepff, 1792) in Albania. Natura Montenegrina 9 (3): 495-498.

Oruçi S 2010b Data on geographical distribution and habitats of the *Rana epeirotica* in Albania. Natura Montenegrina 7 (3): 419-423.

Özdemir N, Gül S, Poyarkov Jr NA, Kutrup B, Tosunoğlu M, Doglio S 2014 Molecular systematics and phylogeography of *Bufotes variabilis* (syn. *Pseudepidalea variabilis*) (Pallas, 1769) in Turkey. Turkish Journal of Zoology 38: 412-420.

Pabijan M, Zieliński P, Dudek K, Chloupek M, Sotiropoulos K, Liana M, Babik W 2015 The dissection of a Pleistocene refugium: phylogeography of the smooth newt, *Lissotriton vulgaris* in the Balkans. Journal of Biogeography 42: 671-683.

Pabijan M, Zieliński P, Dudek K, Stuglik M, Babik W 2017 Isolation and gene flow in a speciation continuum in newts. Molecular Phylogenetics and Evolution 116: 1-12.

Pecl GT, Araújó MB, Bell JD, Blanchard J, Bonebrake TC, Chen I-C, Clark TD, Colwell RK, Danielsen F, Evengård B, Falconi L, Ferrier S, Stewart F, Garcia RA, Griffs RB, Hobday AJ, Janion-Scheepers C, Jarzyna MA, Jennings S, Lenoir J, Linnetved HI, Martin VY, McCormack PC, Mcdonald J, Mitchell NJ, Mustonen T, Pandolfi JM, Pettorelli N, Popova E, Robinson SA, Scheffers BR, Shaw JD, Sorte CJB, Strugnell JM, Sunday JM, Tuanmu M-N, Vergés A, Villanueva C, Wernberg T, Wapstra E, Williams S 2017 Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. Science 355 (6332): 1-9.

Petrov PB 2006 *Lacerta mosorensis* Kolombatovic, 1886 new to the herpetofauna of Albania. Herpetozoa 19 (1/2): 92-93.

Pinheiro J, Bates D 2000 Mixed-effects models in S and S-PLUS. New York, Springer. pp. 528.

Podnar M, Mayer W, Tvrtkovič N 2005 Phylogeography of the Italian wall lizard, *Podarcis sicula*, as revealed by mitochondrial DNA sequences. Molecular Ecology 14: 575-588.

Podnar M, Mađarič BB, Mayer W 2014 Non-corcondant phylogeographical patterns of three widely codistributed endemic Western Balkan lacertid lizards (Reptilia, Lacertidae) shaped by specific habitat requirements and different responses to Pleistocene climatic oscillations. Journal of Zoological Systematics and Evolutionary Research 51 (4): 1-11.

Pressey RL, Humphries CJ, Margules CR, Vanewright RI, Williams PH 1993 Beyond opportunism: key principles for systematic reserve selection. Trends in Ecology & Evolution 8: 124-128.

Psonis N, Antoniou A, Kukushkin O, Jablonski D, Petrov B, Crnobrnja-Isailovič J, Sotiropulos K, Gherghel I, Lymberakis P, Poulakakis N 2017 Hidden diversity in the *Podarcis tauricus* Sauria, Lacertidae species subgroup in the light of multilocus phylogeny and species delimitation. Molecular Phylogenetics and Evolution 106: 6-17.

R Core Team 2016 R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://R-project.org/.
Ragghianti M, Bucci S, Casola C, Marracci S, Mancino G 2004 Molecular investigations on western Palearctic water frogs. Italian Journal of Zoology 2: 17-23.
Ragghianti M, Bucci S, Guerrini F, Mancino G 1999 Characterization of two repetitive DNA families (RrS1 and Rana/Pol III) in the genomes of Palaearctic green water frogs. Italian Journal of Zoology 66: 255-263.
Ribeiro J, Colli GR, Caldwell JP, Soares AMVM 2016 An integrated trait-based framework to predict extinction risk and guide conservation planning in biodiversity hotspots. Biological Conservation 195: 214-223.
Ricketts TH, Dinerstein E, Boucher T, Brooks TM, Butchart SHM, Hoffmann M, Lamoreux JF, Morrison J, Parr M, Pilgrim JD, Rodrigues ASL, Sechrest W, Wallace GE, Berlin K, Bielby J, Burgess ND, Church DR, Cox N, Knox D, Loucks C, Luck GW, Master LL, Moore R, Naidoo R, Ridgely R, Schatz GE, Shire G, Strand H, Wettengel W, Wikramanayake E 2005 Pinpointing and preventing imminent extinctions. Proceedings of the National Academy of Sciences of the United States of America 102: 18497–501.
Rivera X, Arribas O, Carranza S, Maluquer-Margalef J 2011 An introduction of *Podarcis sicula* in Catalonia (NE Iberian Peninsula) on imported olive trees. Buttletí de la Societat Catalana d'Herpetologia 19: 83-88.
Rocchini D, Hortal J, Lengyel S, Lobo JM, Jimenez-Valverde A, Ricotta C, Bacaro G, Chiarucci A 2011 Accounting for uncertainty when mapping species distributions: the need for maps of ignorance. Progress in Physical Geography 35: 211-226.
Rodrigues ASL, Akçakaya HR, Andelman SJ, Bakarr MI, Boitani L, Gaston KJ, Hoffmann M, Marquet PA, Pilgrim JD, Pressey RL, Schipper JAN, Sechrest WES, Stuart SN, Underhill LESG, Waller RW, Matthew EJ 2004a Global Gap Analysis : Priority Regions for Expanding the Global Protected-Area Network. BioScience 54: 1092–1100.
Rodrigues ASL, Andelman SJ, Bakarr MI, Boitani L, Brooks TM, Cowling RM, Fishpool LDC, da Fonseca GAB, Gaston KJ, Hoffmann M, Long JS, Marquet PA, Pilgrim JD, Pressey RL, Schipper J, Sechrest W, Stuart SN, Underhill LG, Waller RW, Watts MEJ, Yan X 2004b Effectiveness of the global protected area network in representing species diversity. Nature 428: 640-643.
Rodríguez MÁ, Belmontes JA, Hawkins BA 2005 Energy, water and large-scale patterns of reptile and amphibian species richness in Europe. Acta Oecologica 28: 65-70.
Reyjol Y, Hugueny B, Pont D, Bianco PG, Beier U, Caiola N, Casals F, Cowx IG, Economou A, Ferreira T, Haidvogl G, Noblé R, de Sostoa A, Vigneron T,

Virbickas T 2007 Patterns in species richness and endemism of European freshwater fish. Global Ecology and Biogeography 16: 65-75.

Saunders DL, Meeuwig JJ, Vincent ACJ 2002 Freshwater protected areas: Strategies for conservation. Conservation Biology 16: 30-41.

Schall JJ, Pianka ER 1978 Geographical trends in numbers of species. Science 201 (4357): 679–686.

Schneider H, Haxhiu I 1994 Mating-call analysis and taxonomy of the water frogs in Albania Anura: Ranidae. Zoologische Jahrbücher 121: 248-262.

Scribner KT, Arntzen JW, Cruddace N, Oldham RS, Burke T 2001 Environmental correlates of toad abundance and population genetic diversity. Biological Conservation 98: 201-210.

Sillero N, Campos J, Bonardi A, Corti C, Creemers R, Crochet P-A, Crnobrnja Isailovič J, Denoël M, Ficetola GF, Gonçalves J, Kuzmin S, Lymberakis P, de Pous P, Rodríguez A, Sindaco R, Speybroeck J, Toxopeus B, Vieites DR, Vences M 2014a Updated distribution and biogeography of amphibians and reptiles of Europe. Amphibia-Reptilia 35: 1-31.

Sillero N, Oliviera MA, Sousa P, Gonçalves-Seco L 2014b Distributed database system of the New Atlas of Amphibians and Reptiles in Europe: the NA2RE project. Amphibia-Reptilia 35: 33-39.

Silva-Rocha I, Salvi D, Harris DJ, Freitas S, Davis C, Foster J, Deichsel G, Adamopoulou C, Carretero MA 2014 Molecular assessment of *Podarcis sicula* populations in Britain, Greece, and Turkey reinforces a multiple-origin invasion pattern in this species. Acta Herpetologica 9: 253-258.

Silvano D, Angulo A, Carnaval ACOQ, Pethiyagoda R 2007 Designing a Network of Conservation Sites for Amphibians − Key Biodiversity Areas. Amphibian Conservation Action Plan (eds C. Gascon C, Collins JP, Moore RD, Church DR, McKay JE, Mendelson JRI), pp. 12−15. IUCN/SSC Amphibian Specialist Group, Gland, Switzerland and Cambridge, UK.

Speybroeck J, Beukema W, Bok B, Van Der Voort J, Velikov I 2016 Field guide to the amphibians and reptiles of Britain and Europe. London, Bloomsbury. pp. 432.

Sterijovski B, Tomovič L, Ajtič R 2014 Contribution to the knowledge of the reptile fauna and diversity in FYR of Macedonia. North-Western Journal of Zoology 10 (1): 83-92.

Stewart JR, Lister AM 2001 Cryptic northern refugia and origins of the modern biota. Trends in Ecology & Evolution 16: 608-613.

Stewart RR Possingham HP 2005 Efficiency, costs and trade-offs in marine reserve system design. Environmental Modeling and Assessment 10: 203–213.

Stojanov, A., Tzankov, N., Naumov, B. 2011 Die Amphibien und Reptilien Bulgariens. Frankfurt am Main, Chimaira. pp. 588.

Stolarik I, Grul'a D, Jablonski D 2017 Two new species of Cricetidae for the fauna of Albania. Biologia 72 (5): 581-585.

Stolarik I, Jablonski D 2017 First record and genetic affiliation of the Balkan snow vole *Dinaromys bogdanovi* (Rodentia: Cricetidae) in Albania. Journal of Threatened Taxa 9 (9): 10675-10678.

Strayer DL Dudgeon D 2010 Freshwater biodiversity conservation: recent progress and future challenges. Journal of The North American Benthological Society 29: 344–358.

Szabolcs M, Kapusi F, Carrizo S, Markovic D, Freyhof J, Cid N, Cardoso A-C, Scholz M, Kasperidus HD, Darwall WRT, Lengyel S -under review- Spatial priorities for the conservation of freshwater biodiversity in Europe.

Szabolcs M, Mizsei E 2017 First record of the eastern spadefoot toad (*Pelobates syriacus* Boettger, 1889) in Albania. North-Western Journal of Zoology 13: 175-176.

Szabolcs M, Mizsei E, Jablonski D, Vági B, Mester B, Végvári Z, Lengyel S 2017 Distribution and diversity of amphibians of Albania: new data and foundations of a comprehensive dataset. Amphibia-Reptilia 38: 435-448.

Szederjesi T, Csuzdi Cs 2012 New earthworm species and records from Albania (Oligochaeta, Lumbricidae). Acta Zoologica Academiae Scientiarum Hungaricae 58 (3): 259-274.

Thieme M, Lehner B, Abell R, Hamilton SK, Kellndorfer J, Powell G, Riveros JC 2007 Freshwater conservation planning in data-poor areas: An example from a remote Amazonian basin Madre de Dios River, Peru and Bolivia. Biological Conservation 135: 484-501.

Thieme ML, Sindorf N, Higgins J, Abell R, Takats JA, Naidoo R, Barnett A 2016 Freshwater conservation potential of protected areas in the Tennessee and Cumberland River Basins, USA. Aquatic Conservation: Marine and Freshwater Ecosystems 26: 60−77.

Thuiller W, Mairano L, Mazel F, Guilhaumon F, Ficetola GF, Lavergne S, Renaud J, Roquet C, Mouillot D 2015 Conserving the functional and phylogenetic trees of life of European tetrapods. Philosophical Transactions of the Royal Society B 370 (1662): 20140005.

Tomovič L, Ajtič R, Ljubisavljevič K, Uroševič A, Jovič D, Krizmanič I, Labus N, Đorđevič S, Kalezič ML, Vukov T, Džukič G 2014 Reptiles in Serbia – distribution and diversity patterns. Bulletin of the Natural History Museum 7: 129-158.

Tsianou MA, Koutsias N, Mazaris AD, Kallimanis AS 2016 Climate and landscape explain richness patterns depending on the type of species' distribution data. Acta Oecologica 74: 19-27.

Tulloch AIT, Mustin K, Possingham HP, Szabo JK, Wilson KA 2012 To boldly go where no volunteer has gone before: predicting volunteer activity to prioritize surveys at the landscape scale. Diversity and Distributions 19 (4): 465-480.

Uhrin M, Havaš P, Minařík M, Kodejš K, Bugoš I, Danko S, Husák T, Koleska D, Jablonski D 2016 Distribution updates to amphibian and reptile fauna for the Republic of Macedonia. Herpetology Notes 9: 201-220.

Uhrin M, Šíbl J 1996 Ďalšie faunistické poznámky z Albánska – pp. 5-6 in: Anonymus (eds): Sborník abstrakt III. konference České herpetologické společnosti. Česká herpetologická společnost, Mariánské Lázně, 11 pp.
Ursenbacher S, Schweiger S, Tomovič L, Crnobrnja-Isailovič J, Fumagalli L, Mayer W 2008 Molecular phylogeography of the nose-horned viper (*Vipera ammodytes*, Linnaeus (1758): Evidence for high genetic diversity and multiple refugia in the Balkan peninsula. Molecular Phylogenetics and Evolution 46: 1116-1128.
Uzzell T, Crnobrnja-Isailović J 2009 *Pelophylax shqipericus*. The IUCN Red List of Threatened Species.e.T58715A11829016.
Uzzell T, Lymberakis P, Haxhiu I 2009 *Pelophylax epeiroticus*. The IUCN Red List of Threatened Species.e.T58592A11793456.
Vági B, Kovács T, Bancilă R,Hartel T, Anthony BP 2013 A landscape-level study on the breeding site characteristics of ten amphibian species in Central Europe. Amphibia-Reptilia 34: 63-73.
Valakos ED, Pafilis P, Sotiropoulos K, Lymberakis P, Maragou P, Foufopoulos J 2008 The amphibians and reptiles of Greece. Frankfurt am Main, Chimaira. pp. 463.
Van Buskirk J, 2005 Local and landscape influence on amphibian occurrence and abundance. Ecology 86: 1936-1947.
Vojar J, Havlíková B, Solský M, Jablonski D, Iković V, Baláž V 2017 Distribution, prevalence and amphibian hosts of *Batrachochytrium dendrobatidis* in the Balkans. Salamandra 53: 44-49.
Vörösmarty C-J, Sahagian D. 2000 Anthropogenic disturbance of the terrestrial water cycle. Bioscience 50 (9): 753-765.
Vukov T, Kalezić ML, Tomović L, Krizmanić I, Jović D, Labus N, Džukić G 2013 Amphibians in Serbia – distribution and diversity patterms. Bulletin of the Natural History Museum 6: 90-112.
Werner F 1920 Zur Kenntnis der Reptilien- und Amphibienfauna Albaniens. Zoologischer Anzeiger 17 (1/2): 21-24.
Willig MR, Kaufman DM, Stevens RD 2008 Latitudinal gradients of biodiversity: patterns, process, scale and synthesis. Annual Review of Ecology, Evolution ans Systematics 34: 273-309.
Zachos FE, Habel JC 2011 Biodiversity hotspots – Distribution and Protection of Conservation Priority Areas. Berlin-Heidelberg, Springer-Verlag. pp. 546.
Zedler JB, Kercher S 2005 Wetland resources: Status, trends, ecosystem services, and restorability. Annual Review of Environment and Resources 30: 39-74.
Zarfl C, Lumsdon A, Berlekamp J, Tydecks L, Tockner KA 2014 Global boom in hydropower dam construction. Aquatic Sciences 77: 161-170.