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DISSERTATION

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Dadia National Park, Greece – an Integrated Study on Landscape, Biodiversity, Raptor Populations and Conservation Management

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Dedications

A graduate degree requires a commitment of time and energy that precludes the fulfillment of family obligations and normal social behavior. The degree that I earn with this dissertation is by no means an exception to this rule. For dealing with an often tired, cranky, forgetful and absent person, but also for the support that I received during the last six years (and before), I dedicate this dissertation especially:

♥ to my mother Luise Schindler, who encouraged me to dream and supported me for 35 years plus 9 months with all her strength, power, and love;

♥ to my deceased father Otto G. Schindler, who could not witness the final of this achievement, but influenced me heavily with his thirst for knowledge, encouraged me always for higher studies, and helped me also practically during parts of this thesis;

♥ to the habitants of Dadia National Park, who supported me with their friendliness and happiness, and transferred some long and exhausting field work sessions to the best years of my life.

♥ to my life partner Manuela Martins Cunha and to our baby Daniel, who will be born one of these days. I desire a birth without complications and more time in the future to enjoy both of you.

Dadia National Park, Greece – an Integrated Study on Landscape, Biodiversity, Raptor Populations and Conservation Management

Declaration

I hereby declare that this dissertation has not been submitted to obtain a degree at any other university. With the exception of the assistance noted in the acknowledgments, this thesis is entirely my own work. I authorize the University of Vienna to lend this document, or reproductions of this document by photocopying or any other means, in total or in part, at the request of other institutions or individuals for the purpose of research. It is a condition of use of this dissertation that anyone who consults, must recognize that the copyright rests with the author and that no quotation from the dissertation and no information derived from it may be published unless the source is properly acknowledged.

Summary

Global change is currently causing a strong decline of biodiversity and in the provision of essential ecosystem services. Assessments of status and trends of landscapes, biodiversity, and endangered species are needed and should be conducted in the framework of an integrated conservation management. This doctoral thesis is a cumulative work dealing with landscape, biodiversity, raptor populations, and conservation management of Dadia National Park (Dadia NP), a local hotspot of biodiversity in north-eastern Greece. In the twelve research articles that constitute this thesis, I investigated with my coauthors 1) habitat heterogeneity and biodiversity of Dadia NP, 2) landscape approaches and GIS applications for biodiversity management, 3) sets of landscape metrics for landscape structure analyses, 4) the multiscale performance of landscape metrics as biodiversity indicators for plants, insects and vertebrates, 5) the use of ecological heterogeneity to design reserve networks, 6) the performance of a telemetry system for Eurasian Black Vulture (*Aegypius monachus*), 7) GIS-based methodologies for territory analyses of raptors, 8) habitat, status and population trends of the diurnal raptor populations of Dadia NP, 9) decision support systems for the conservation of biodiversity in managed forest, and 10) the evidence base of conservation management in Dadia NP and the Bulgarian part if the Eastern Rhodpes Mountains. It can be concluded for Dadia NP and similar Mediterranean reserves that landscape surveillance should be integrated into the ecological monitoring of key and indicator species to aid the evaluation of management effects on habitats and wildlife. Further and consistent ecological monitoring and research is crucial for establishing integrative biodiversity conservation and management. Conservation research is providing important evidence for conservation managers and decision makers, but lack of political will for competent conservation authorities leads to weak rates of implementation and evaluation.

Zusammenfassung

Der globale Wandel verursacht derzeit einen starken Verlust an Biodiversität und für den Menschen notwendigen Ökosystemleistungen. Die Erfassung von Status und Trends von Landschaften, Artenvielfalt und Populationen bedrohter Arten ist notwendiger denn je und sollte im Rahmen von integrativem Naturschutzmanagement geschehen. Diese Dissertation fasst Studien über Landschaft, Biodiversität, Greifvogelpopulationen und Naturschutzmanagement für den Dadia National Park (Dadia NP) zusammen, ein Schutzgebiet in Nordost-Griechenland, das durch seine außergewöhnlich hohe Biodiversität gekennzeichnet ist. In den zwölf wissenschaftlichen Artikeln, die diese Dissertation umfasst, werden Fragestellungen zu folgenden Themen bearbeitet: 1) Habitatheterogenität und Biodiversität im Dadia NP, 2) Landschaftsökologische Ansätze und geographische Informationssysteme (GIS) für Biodiversitätsmanagement, 3) Sets an Landschaftsmaßzahlen für Landschaftsstrukturanalysen, 4) Maßstabsübergreifende Anwendung von Landschaftsmaßen als Biodiversitätsindikatoren für Pflanzen, Insekten und Wirbeltieren, 5) ökologische Heterogenität als Werkzeug zur systematischen Planung von Schutzgebietsnetzwerken, 6) Evaluierung eines Telemetriesystems zur Erfassung von Mönchsgeiern (*Aegypius monachus*), 7) GIS-basierende Methoden zur Revierkartierung von Greifvögeln, 8) Habitat, Status und Bestandstrends von Populationen tagaktiver Greifvögel, 9) ein Entscheidungsunterstützungssystem zur Berücksichtigung bedrohter Arten im Forstmanagement, und 10) Zusammenstellung aller in wissenschaftlicher Literatur für Dadia NP und den bulgarischen Teil der Ost-Rhodopen empfohlenen Naturschutzmaßnahmen und Ermittlung ihrer Umsetzungsrate und Evaluierungsrate. Zusammenfassend lässt sich sagen, dass in Dadia NP und ähnlichen mediterranen Schutzgebieten ein Landschaftsmonitoring neben dem Monitoring von Schlüssel- und Indikatorarten wesentlich zur Erfassung der Effektivität einzelner Naturschutzmaßnahmen für Habitate und deren assoziierte Arten beitragen kann. Fortführende

ökologische Forschung und Monitoring sind besonders wichtig für die Etablierung von integrativem Naturschutzmangement zum Erhalt von Biodiversität und Ökosystemleistungen. Die Naturschutzforschung liefert die notwendigen Grundlagen für Naturschutzmanager und Entscheidungsträger, aber mangelnder politischer Wille für kompetente Naturschutzbehörden beeinträchtigt die Umsetzung und Evaluierung der erarbeiteten Naturschutzmaßnahmen.

Synopsis

This doctoral thesis is an integrated study on landscape, biodiversity, raptor populations, and conservation management of Dadia National Park, north-eastern Greece. Most ideas and data, and some of the analyses, emerged from a three year employment by WWF Greece in the area (2003-2005), while most of the papers were finished during my employment as research assistant at the Department for Conservation Biology, Vegetation & Landscape Ecology at the University of Vienna. “Part A” of this thesis, which I called “introduction”, consists of two book chapters (i.e. habitats and biodiversity of Dadia National Park, and landscape approaches as tools for conservation management) that deal intensively with topics that otherwise would have to be explained in this synopsis. Therefore, I limit this synopsis to brief explanations regarding the content of the chapters and the crucial importance of biodiversity research in times of global change.

Brief information about the content of the chapters

This thesis has four main parts, called hereafter A, B, C, and D. Each part consists of 2-4 peer reviewed scientific papers, most of them already published or in press in scientific journals and books, the rest being at least submitted. I tried to stay as close as possible to the original publications, and mainly adapted the formatting, such as tables, figures, style and quotation of the references, in order to be homogeneous throughout this thesis. I updated in press references of the original papers where necessary, but did not search for additional recent literature while keeping the original references. As each paper is a “stand-alone” paper, some parts of this cumulative thesis (e.g. the study area sections of the papers), might be to some extent repetitive.

Part A is a general introduction consisting of two papers that describe landscape and biodiversity of Dadia National Park and landscape approaches for biodiversity management. Part B called “Landscape structure and its use as ecological indicators” contains four papers, which investigate the landscape structure of Dadia National Park and different approaches to use measures of landscape structure and of vertical vegetation structure as indicators of species richness and biodiversity. Part C is dealing with methodological refinements of systematic raptor surveys and the status and trends of the populations of the diverse raptor assemblage of Dadia NP. It consists of one paper that assesses the precision of a telemetry system used for Black Vulture surveys in the Dadia NP, a second one that is highlightening different aspects of the developed GIS-based methodology for the applied systematic raptor monitoring, and two papers, which are focusing on the results of the systematic raptor monitoring including long-term population trends. Finally the last part, part D, is dealing with conservation management. Its first paper demonstrates how conservation of biodiversity can be integrated into forest management by developing an adaptive decision support system. The second paper of this part is the last one of this thesis. It is called "From research to implementation: nature conservation in the Eastern Rhodopes mountains (European Green Belt)", provides an overview about published conservation recommendations for Dadia National Park and the neighbouring Bulgarian reserves, and follows the tracks of these recommendations by assessing which have been implemented and evaluated regarding their effectiveness.

Importance of biodiversity research in times of global change

According to the recently adopted European Biodiversity Research Strategy (European Platform for Biodiversity Research Strategies 2010), protecting biodiversity is a 'Grand Challenge' for mankind, on a par with climate change, food security, energy security, health, etc. Human societies have benefitted from using and exploiting biodiversity and from converting ecosystems, while human activities, imperfect knowledge, and unsustainable use of natural resources are the main causes for biodiversity loss and ecosystem degradation. This often leads to loss of ecosystem functions and services, and consequently to difficulties of societies in achieving and maintaining human well-being. The Grand Challenge of conserving biodiversity is also reflected in major policy documents such as the Convention for Biological Diversity (CBD). Meeting the Grand Challenge of biodiversity loss requires a major research effort as a basis for effective action and societal change (European Platform for Biodiversity Research Strategies 2010).

Over the last decades, research has delivered essential information and knowledge for tackling this challenge. However, new investments in focused research are needed to: Ensure the long-term survival of species, their genetic diversity, the ecological integrity and functionality of ecosystems, and the provision of ecosystem services. This can be achieved according to the European Platform for Biodiversity Research Strategies (2010) with research on status, trends, and functional relationships (as presented in the Parts B and C of this thesis), on development and evaluation of effective management, conservation and restoration (Chapters C.2., C.3, C.4., D.1., D.2.), and on the improvement of sustainable management and use of ecosystem services, landscapes, and their biodiversity (Chapter D.1.).

Part A - Introduction

This introduction to the doctoral thesis gives an overview about geomorphology, landscape and biodiversity of Dadia National Park as well as on landscape approaches for biodiversity management. Both paper included into this part were originally written for scientific books. The first one titled “Habitat heterogeneity and biodiversity” for the book “The Dadia-Lefkimi-Soufli National Park, Greece: Biodiversity, Management and Conservation”, which is currently published by WWF Greece. It gives a wide overview on the natural features of Dadia National Park, and deals additionally with the questionable relation of habitat heterogeneity and biodiversity. The second paper called “Landscape approaches and GIS for biodiversity management” was written for the book Landscape modeling: geographical space, transformation and future scenarios, which is the 8^{th} Volume of Springer’s Urban and Landscape Perspectives Series.

Chapter A.1. Landscape and biodiversity in Dadia – Lefkimi – Soufli Forest National Park

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Running title: Landscape and biodiversity in Dadia National Park

Keywords: Heterogeneity, landscape structure, land use change, diversity of habitats, local biodiversity

Dadia village surrounded by the two core areas of Dadia National Park – photo by Petros Babakas

Own contribution:

Study design 20%, implementation 25%, writing 30%

Abstract

The relationships between heterogeneous landscapes and biodiversity have been well investigated and in many cases human activities have played a significant role in the creation of landscape patterns. In the Dadia – Lefkimi – Soufli Forest National Park (DNP), natural and anthropogenic disturbances during the last century, such as forest fires, uncontrolled logging and extensive livestock grazing created a mosaic of different land-cover categories. However, nowadays natural succession and forest management have altered the mosaic of habitats towards a more homogeneous forest area. More than 70% of the land is now covered with oak and pine forests in either pure or in mixed stands affecting negatively fauna species, which depend on heterogeneity and semi-open habitats negatively. Despite this alteration, habitat diversity is one of the main gradients of landscape structure in Dadia. Although an optimal level of heterogeneity can hardly be determined as it depends on the taxa under consideration, diversity and spatial configuration of landscapes were found to be important drivers of local biodiversity in DNP and must be considered in the management and conservation of the reserve.

Biodiversity and heterogeneity – a questionable relationship

Nowadays, there is much discussion about the human impacts on landscapes and biological diversity worldwide. Most landscapes have been influenced by human land use, and the resulting landscape mosaic is a mixture of natural and human-managed patches that vary in size, shape, and arrangement (e.g. Forman & Godron 1986; Krummel et al. 1987). The intrinsic value of biodiversity is widely recognized as is its ecological, social, economic, cultural and aesthetic value (Pimm et al. 1995; Mittermeier et al. 1999), but human-induced loss of biodiversity has currently reached alarming rates at the levels of genes, species and ecosystems (Barbault & Sastrapradja 1995; Brooks et al. 2002). But surprisingly, in some cases human activity had positive effects by increasing biological diversity through the creation of heterogeneous landscapes (Blondel & Aronson 1999; Brotons et al. 2004; Kati et al. 2004b; Saïd & Servanty 2005).

The relationships between landscape and biodiversity have been investigated intensively during the last two decades (e.g. Wiens et al. 1993; With & Christ 1995;

Miller et al. 1997; Pino et al. 2000; Poudevigne & Baudry 2003; Betts et al. 2005; Quevedo et al. 2006). It is believed that anthropogenic disturbances enhanced landscape heterogeneity and that the "mosaic effect" of landscape patchiness therefore had a beneficial, rather than impoverishing impact on species diversity (Le Houerou 1981; Forman 1995; Bignal & McCracken 1996; Blondel and Aronson 1999; Ernoult et al. 2003). In fact, mosaics play an important role for many animal groups, such as insects (e.g. Chust et al. 2004; Saarinen & Jantunen 2005), birds (e.g. Sanchez-Zapata & Calvo 1999; Brotons et al. 2004) and mammals (e.g. Jepsen et al. 2005; Saïd & Servanty 2005).

When human disturbance exceeds a certain threshold, however, it can have a disastrous impact on biodiversity. In such cases we refer to landscape fragmentation, loss and degradation, which are widely considered to be the most important threats to biodiversity on a global scale (e.g. Soulé 1987; Fahrig & Meriam 1994; Tilman et al. 1994; Fahrig 2001). In Mediterranean ecosystems human-induced disturbances, such as fires, clear-cutting, grazing and logging, are believed to have had a direct or sustained impact for thousands of years (Naveh & Dan 1973). On the other hand, this long-lasting exploitation of natural resources in the Mediterranean resulted in the extinction of several plant and animal species and in a severe reduction in the area of primary forest vegetation (Quézel 1976; Myers et al. 2000; Guo 2003). Human activities also led to a wide array of adaptations of vegetation structure and of individual species (Blondel & Aronson 1999).

The landscape of Dadia – Lefkimi – Soufli Forest National Park (hereafter DNP) is covered mostly by woodland. However, during recent centuries this area was never a "virgin" forest without any human impact on the succession history of its ecosystems. Natural or anthropogenic forest fires, uncontrolled logging and extensive livestock grazing created a fine mosaic of open land-cover categories. Many of the factors that created clearings inside the forest have nowadays been diminished (e.g. livestock grazing, uncontrolled natural fires), resulting in a significant decrease of forest clearings and natural grasslands. This has had a significant effect on landscape composition and configuration.

The current paper aims to summarize the research carried out in DNP on landscape features and their effects on species diversity. Its objectives are: (1) to describe different aspects of the landscape of DNP, particularly regarding geomorphology, land-cover types and landscape structure; (2) to review land-use changes during the last century and thereby explain current patterns of landscape heterogeneity; and (3) to review the influence of current landscape heterogeneity on local biodiversity.

Heterogeneity in the DNP

Geomorphology

Dadia NP is characterized by an undulating landscape with low hills and hundreds of gullies. The distribution of the geomorphologic parameters is irregular. Although the altitude ranges from 20 to 640 m above sea level, 90% of the area lies below 320 m and has gentle or moderate slopes, while steeper slopes are found mainly in the central and southwestern part of the area and are associated with the highest altitudes of the park (Figure A.1.1 a&b). This difference is also reflected in the diversity of the park's geomorphology. Half of the area – mainly the lowlands and the northwest – is characterized by a gently rolling relief, with low elevational diversity. In contrast, the highlands as well as the southwest have a highly diverse geomorphology (Figure A.1.1c).

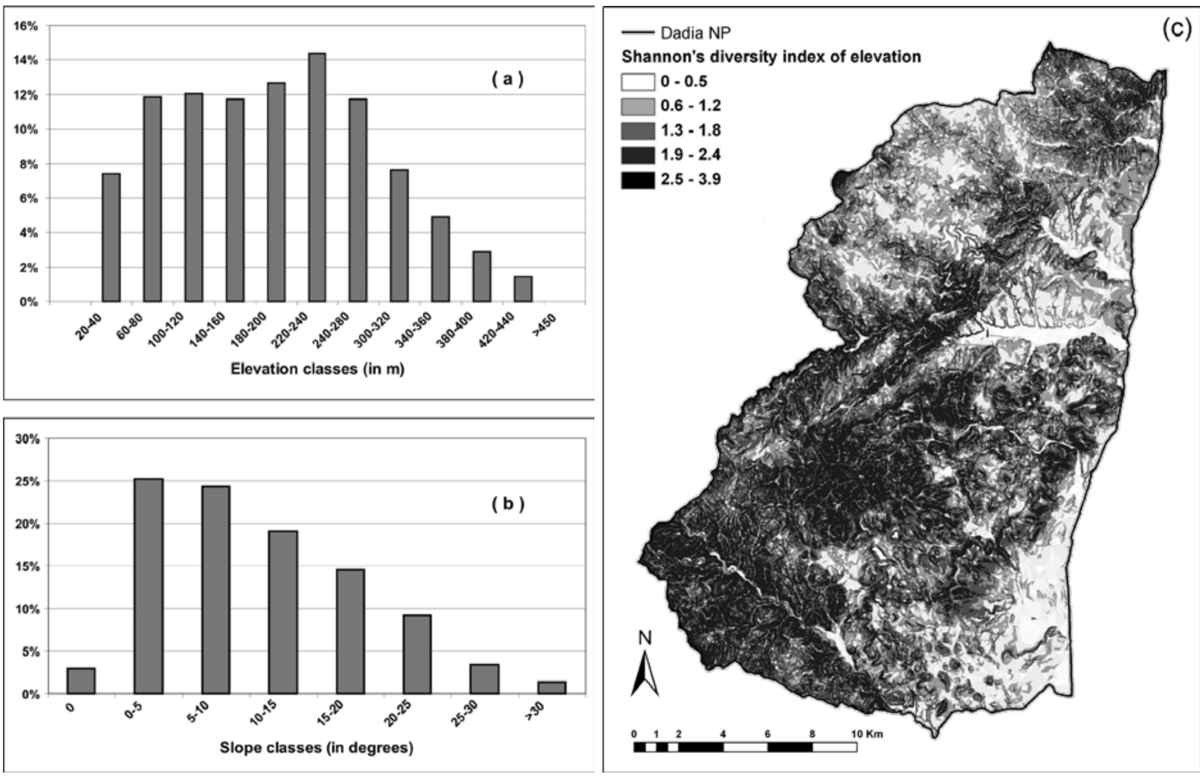

Figure A.1.1. *Proportional distribution of (a) elevation classes, (b) slope classes and (c) diversity of elevations in DNP (the latter as measured by Shannon's diversity index).3*

Land cover – types of vegetation

DNP is dominated by woodland. More than 70% of the area is covered by oak and pine forests in either pure or mixed stands. Most of the oak forest is present in the northern and the south-western parts of the area, while pine forests are concentrated in the central and eastern parts. Mixed forests cover the intermediate zones and the broad-leaved forest (mainly *Arbutus andrachne* and *Phillyrea media*) the south-west (Poirazidis 2003a). Fourteen different land-cover types have been recognized (Catsadorakis & Källander 2010). Intensive reafforestation has taken place in the area during the last 50 years (Triantakonstantis et al. 2006) which has resulted in a more homogeneous forest area with less forest edge but with a high diversity of habitat types still present. More than 55% of the forest belongs to mixed vegetation types in different proportions and variable patterns of composition and configuration (for an example, see Figure A.1.2).

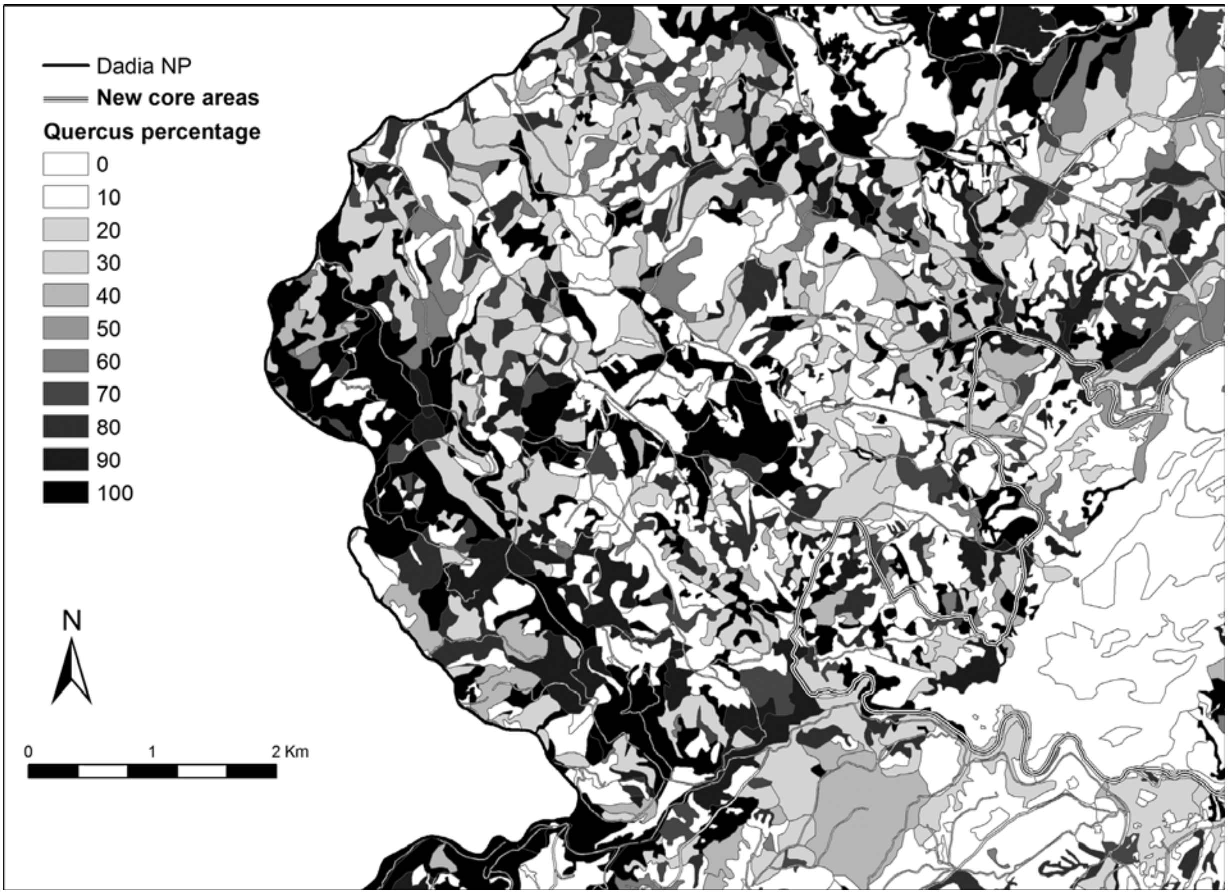

Figure A.1.2. *An example of the pattern of mixed forest in DNP (% of oak tree cover in the north-western section).*

Landscape structure

Landscape structure quantifies composition and configuration of a landscape and is characterized by measures such as patch size, edge density, patch shape, isolation, texture, connectivity, diversity, edge contrast, etc. (Turner et al. 2001). Gradients of landscape structure in DNP can be expressed optimally by variables such as landscape diversity, edge contrast (which is related to habitat fragmentation) and patch shape (Schindler et al. 2008 [= Chapter B.1 of this thesis]). The gradient of landscape diversity is especially pronounced and reaches from areas with very few and dominating habitats, towards ones with a high variety and interspersion of habitats. Diverse landscapes occur in several parts of the park but the highest values of landscape diversity are reached around the borders of the strictly protected areas where different forest types are mixed with clearings and fields. Low diversity is found in the eastern agricultural areas and in the oak forests at the northern and south-western borders of the park (Figure A.1.3).

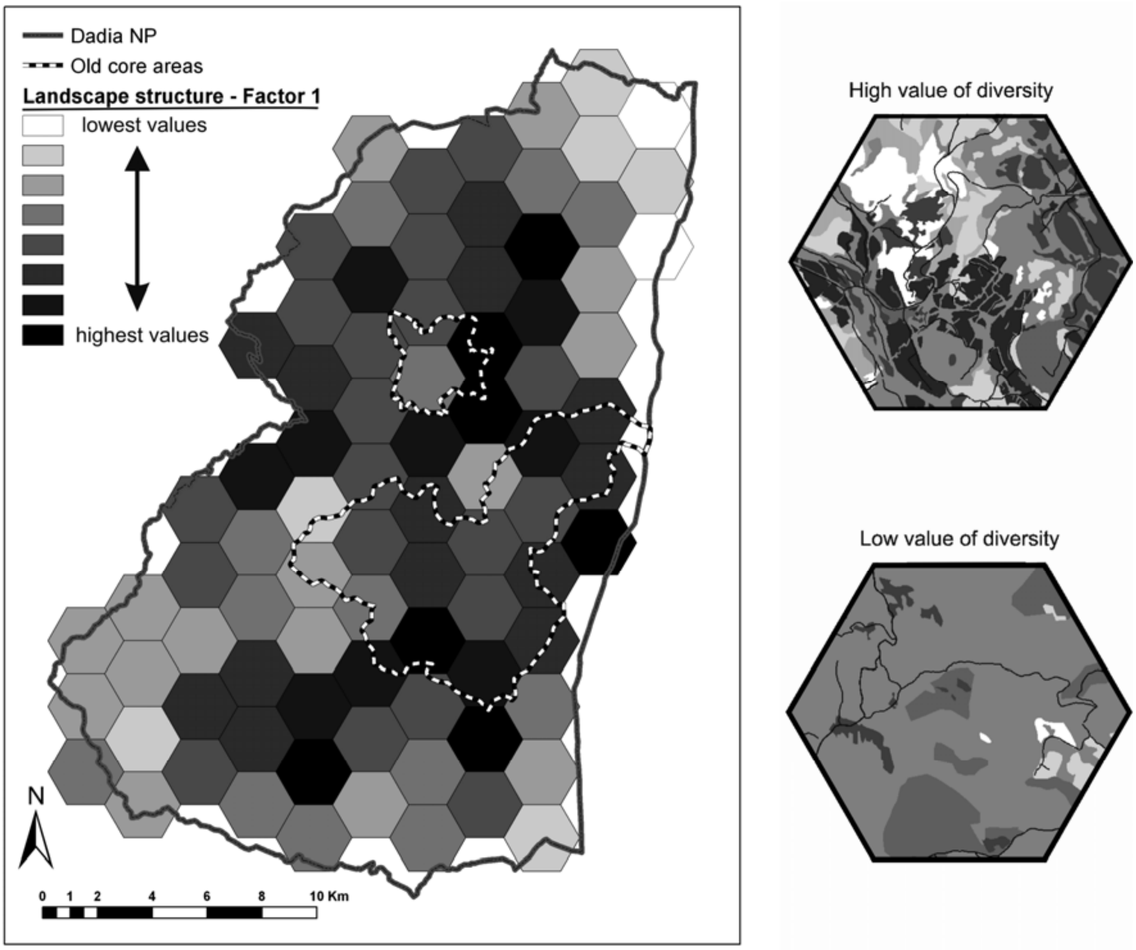

Figure A.1.3. *Pattern of landscape diversity in DNP and examples of areas with particularly high and low values.*

Edge contrast was the second most important gradient of landscape structure in DNP (Schindler et al. 2008 [= Chapter B.1 of this thesis]). It quantifies the contrast among different habitat patches, and high values are often related to anthropogenic fragmentation. The pattern of this gradient is clustered with the highest values occurring in the eastern part of the study area, which consists of agricultural land with many small patches of highly fragmented forest (Figure A.1.4). Two clusters of very low edge contrast coincide with the two strictly protected areas, which remain unfragmented due to the absence of forest roads and agricultural land. Another measure of landscape structure, "patch shape irregularity," was the dominant characteristic of the third main gradient that resulted from our research (Schindler et al. 2008 [= Chapter B.1 of this thesis]). Most irregularly shaped patches occurred in the two core areas of DNP.

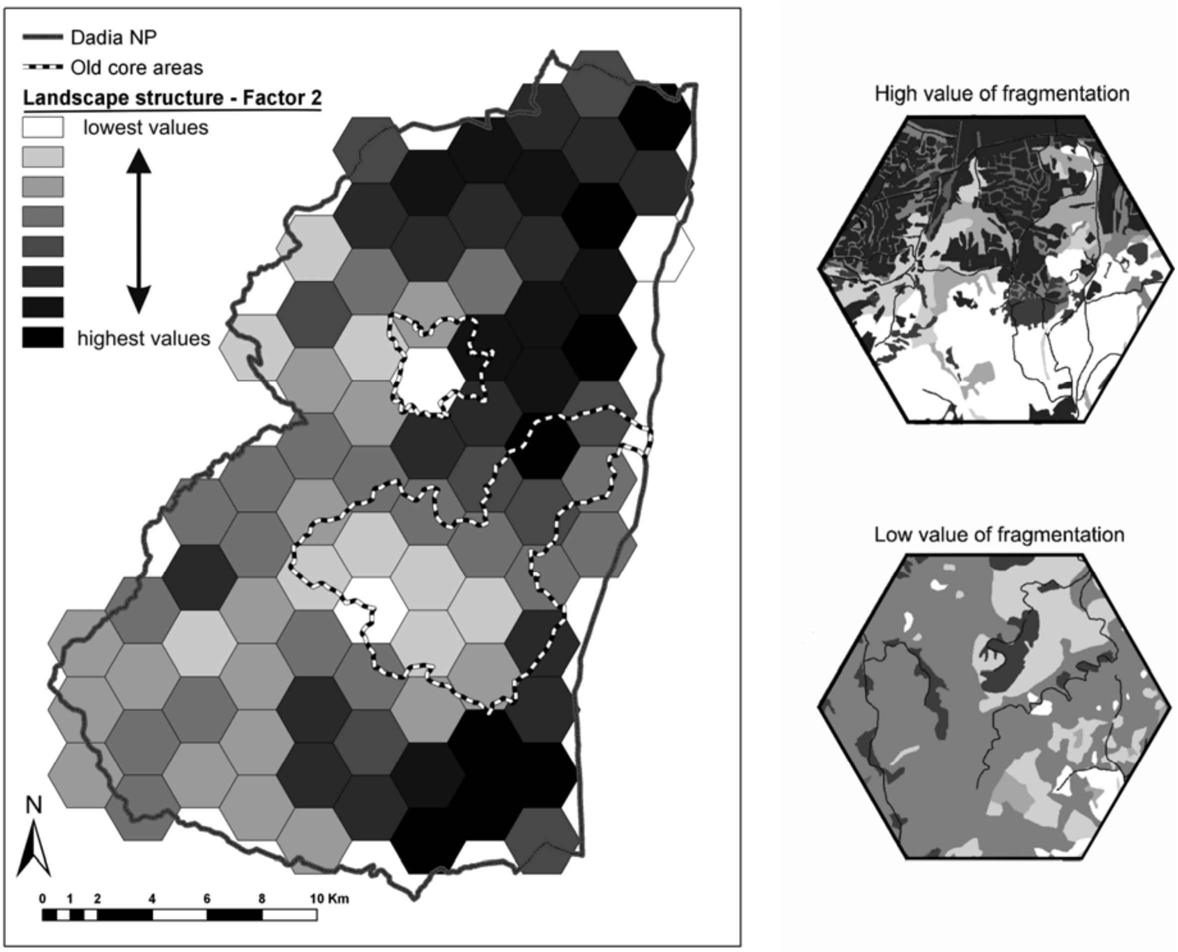

Figure A.1.4. *Pattern of fragmentation in DNP and examples of areas with particularly high and very low values.*

The three gradients of landscape structure provide a good overview of the effects of management on habitat heterogeneity and landscape characteristics (Table A.1.1). The strictly protected areas of the DNP are covered by unfragmented forests. The parts surrounding the core areas are characterized by a great diversity of habitat types and a low to medium level of fragmentation.

Table A.1.1. *Local differences of landscape structure in DNP.*

Region	Habitat diversity	Fragmentation	Patch shape irregularity
Core areas	medium	Low	high
Agricultural areas	low	High	low–medium
Mosaics	high	medium–high	medium
Managed forest	medium	Medium	varying (low–high)

Land-cover changes in DNP

As mentioned in the previous chapters, woodland covers most of the landscape of DNP. However, for the last 30 centuries DNP was not a "virgin" forest without any human impact on the succession history of its ecosystems. Especially during the past 60 years, many stochastic events played a leading role in creating what we now wish to conserve. In the past there was a higher percentage of open areas in the park, as can be seen from older aerial photos (Triantakonstantis et al. 2006). Natural or anthropogenic forest fires (e.g. during the Second World War and the civil war that followed), uncontrolled logging and extensive livestock grazing created a fine mosaic of open land-cover categories such as agricultural land, grassland, scrubland, rocky areas and degraded oak forest.

After the 1960s, many of the above-mentioned activities declined and a management plan for the forests was implemented. In 1980, a Nature Reserve was established with two areas under strict protection and with an adjoining buffer zone (Catsadorakis & Källander 2010). Together with other very important changes, this has resulted in many factors that in the past created open habitat nowadays having decreased in importance (e.g. livestock grazing, uncontrolled natural fires). This has led to a significant decrease in the number of forest clearings and the amount of semi-natural grassland. Environmental heterogeneity is one of the main factors generating biological diversity (Huston 1994) and it is obvious that many changes influencing habitat heterogeneity took place in the ecosystem of DNP. Although the detailed analysis of the changes in landscape structure is still on-going, it seems that a high level of habitat heterogeneity characterized the area in 1945, while in the following years natural forest expansion created more continuous and homogeneous forest habitats. According to recent research (Triantakonstantis et al. 2006), only 46% of the DNP was covered by forest in 1945, reaching 54% in 1973 and 72% in 2001. On the other hand, the proportion of clearings decreased from 35% in 1945 to 25% in 1973 and 9% in 2001. The extent of agricultural land was quite stable during this period, with 18, 20 and 16%, respectively (Figure A.1.5, Table A.1.2).

Table A.1.2. *Trends in land use changes in the buffer zone and the core areas of DNP from 1945 to 2001.*

Change	**1945–1973**	**1973–2001**	**1945–1973**	**1973–2001**
	Buffer zone (%)		Core areas (%)	
Forest → Forest	74	91	83	91
Forest → Clearings	23	4	15	7
Forest → Agricultural land	3	2	2	1
Clearings → Forest	50	69	55	65
Clearings → Clearings	40	19	43	31
Clearings → Agricultural land	10	10	3	3
Agricultural land → Forest	8	27	20	63
Agricultural land → Clearings	3	5	14	4
Agricultural land → Agricultural land	89	63	66	30

Forest expansion rates were high during the whole study period but were more evident after 1973 when the prescribed management of the forest was launched and the first protection status was implemented in the area. More than 60% of the forest expansion took place within a 200 m zone in the vicinity of the old existing forest patches resulting in more homogeneous forest ecosystems (Figure A.1.6). It is interesting, however, that forest expansion in what later became the strictly protected areas of the reserve was slower than in the managed forest. There are no scientific data that would explain the reasons for this difference, but it is possible that the forest policy in the managed area supported the re-establishment of forest in the clearings. Together with a decline in free-ranging livestock in many parts of the managed forests of the buffer zone, this may have acted towards a quicker natural regrowth. In contrast, these two factors never operated in the core areas.

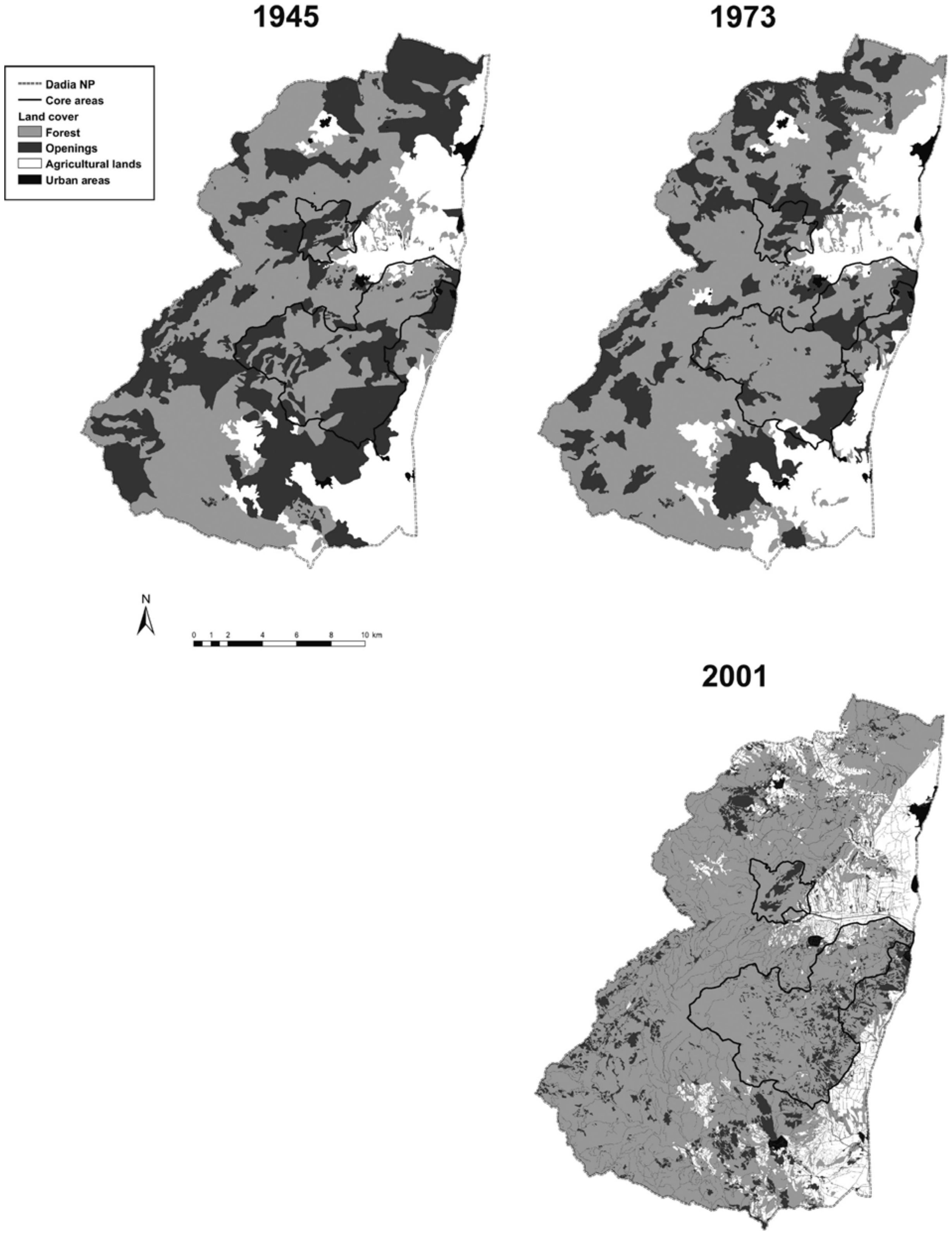

Figure A.1.5. *Trends in land-cover change, shown in maps for the years 1945, 1973 and 2001 (reprinted from Triantakonstantis et al. 2006).*

Both natural succession and anthropogenic management have acted in different ways during the last 50 years creating an increasingly homogenous and forested landscape in DNP. But how have these changes in landscape heterogeneity affected local biodiversity? How much forest or opening is optimal to support the highest biodiversity? To answer these questions, data from all past periods are necessary, but unfortunately this information is not available. Thus, present biodiversity in areas of different heterogeneity must be used to approach the correct answers.

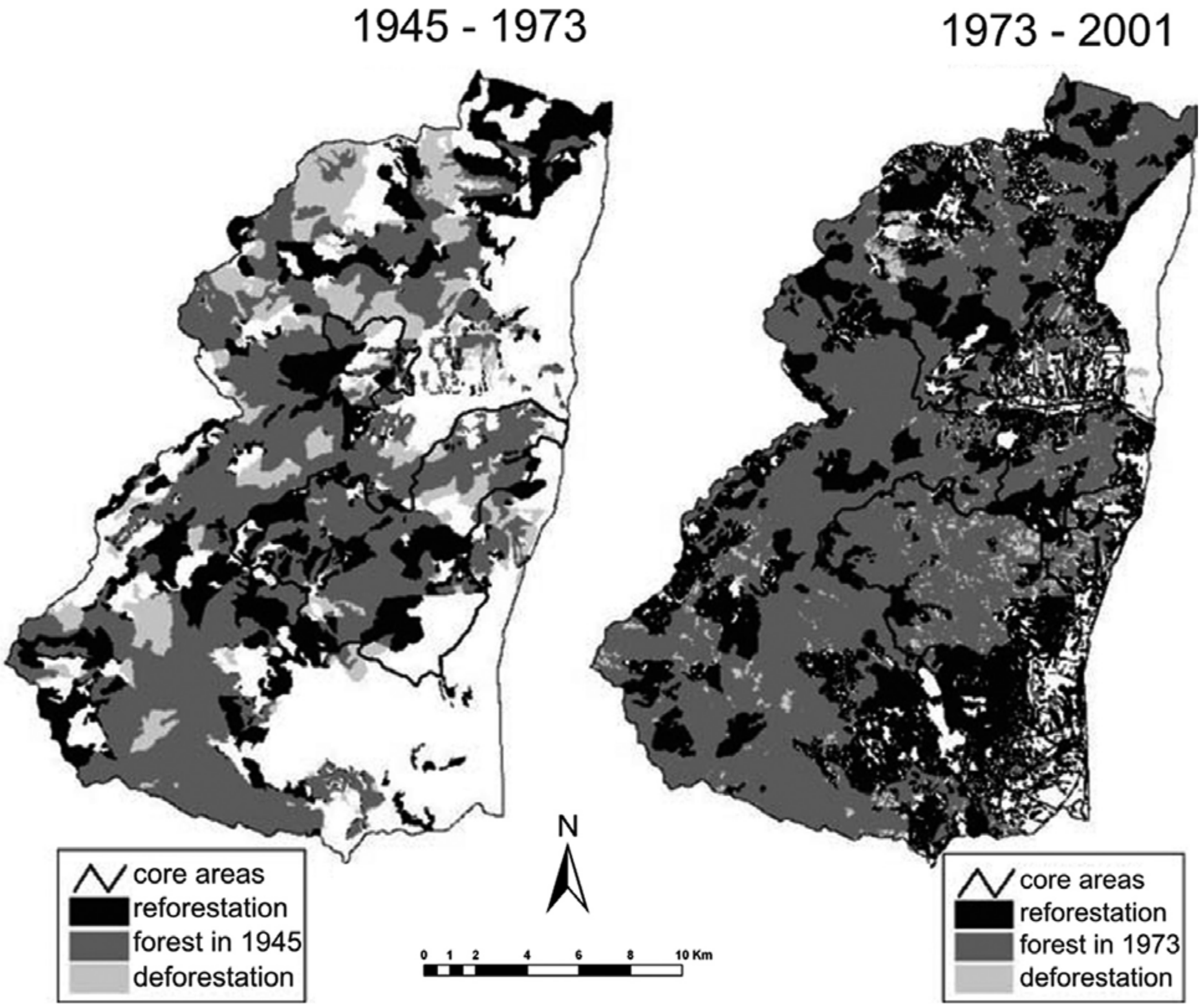

Figure A.1.6. *Maps of regeneration and deforestation during 1945–1973 and 1973–2001 (reprinted from Triantakonstantis et al. 2006).*

Landscape heterogeneity and local biodiversity in DNP

DNP is known for its high biodiversity, including unique and rare species of flora and fauna (e.g. Helmer & Scholte 1985; Adamakopoulos et al. 1995; Kati et al. 2000; Grill & Cleary 2003; Kati et al. 2004c; Korakis et al. 2006). The area is of great importance for raptorial birds because of the particularly high number of breeding species (17–18 diurnal breeding raptor species, of which 12 are tree-nesting), and also because of the sizable populations of some of these species (Hallmann 1979; Poirazidis et al. 1996; 2010c). A considerable breeding population of Black Stork *Ciconia nigra* also occurs in the area (Tsachalidis & Poirazidis 2006).

Heterogeneous landscapes provide a variety of breeding and foraging areas in close proximity and can maintain a high diversity and abundance of raptorial birds (Sanchez-Zapata and Calvo 1999; Anderson 2001). A definite reduction in the availability of open and semi-open habitats, as recorded for the mountain zone of DNP since the 1950s, affected the distribution of many raptor species, such as the Lesser Spotted Eagle *Aquila pomarina*, Long-legged Buzzard *Buteo rufinus* and Egyptian Vulture *Neophron percnopterus*. These species lost several of their traditional territories in the mountain zone (Poirazidis 2003a); they now mainly occupy the lowlands, reflecting their preference for nesting in mosaic habitats dominated by forest edges and small portions of mature forests (Alivizatos 1996; Väli et al. 2004; Poirazidis et al. 2007a). Non-intensive cultivated fields and pastures inside the forest are mainly used for foraging and are vital elements for raptor conservation in DNP (Bakaloudis et al. 1998a Xirouchakis 1999). On the other hand, raptor species adapted to the forest interior, such as Goshawk *Accipiter gentilis*, Booted Eagle *Hieraaetus pennatus* and Honey Buzzard *Pernis apivorus*, showed stable or increasing populations (Poirazidis 2003a). It is possible that the changes towards a more forest-friendly management, could have improved the nesting habitat of these species and consequently their population sizes (Poirazidis et al. 2007a).

Landscape heterogeneity has a positive influence on the community of smaller birds (passerines and woodpeckers) in DNP (Moskát & Fuisz 2002; Kati & Sekercioglu 2006). The highest diversity of these birds was detected at sites of a mosaic character that combined different kinds of vegetation patches within a limited area, such as grassy openings, hedges and forest plots. These sites were situated either in the agricultural zone of DNP, or were clearings in the pine forest. Several other studies have shown that horizontal heterogeneity (but also vertical heterogeneity) affects the distribution of small terrestrial birds positively (e.g. Blondel et al. 1973; Böhning-Gaese 1997; Farina 1997; Grand & Cushman 2003).

Spatial heterogeneity has a positive influence on the species richness of woody plants (Bascompte & Rodriguez 2001), and irregular shapes of patches have been shown to contain a higher diversity of vascular plants and bryophytes than regular ones (Moser et al. 2002). In accordance with these studies, we found that sites of a mosaic character in our study area were also the richest in species of woody plants also (Kati 2001).

Landscape diversity is also known to be one of the important factors for pond-breeding amphibians (Brodman et al. 2003). In our study area, the most important sites for the semi-aquatic herpetofauna (amphibians and freshwater terrapins) were the ones that combined a diversity of wet microhabitats, such as brooks, inundated land, puddles and ditches (Kati et al. 2007). Anthropogenic impact can be favourable for the semi-aquatic herpetofauna, making habitats more diversified by the creation of artificial aquatic microhabitats (puddles, ditches). Such new microhabitats can improve water availability during the arid season and thus favour the semi-aquatic herpetofauna, although they are far poorer in species richness than natural ones (Kati et al. 2007).

Semi-open or open habitats of a thermophilous character, such as oak woods and heaths, with a well developed shrub layer were found to be the most important sites for lizards and terrestrial tortoises (Kati et al. 2007). High densities of reptiles were also found in forests, mainly in mixed forest and oak forest, but they were dominated by just two to three species (Bakaloudis et al. 1998a). Although no strong evidence for links between habitat heterogeneity and reptile diversity was found in some studies of the herpetofauna in DNP (Helmer & Scholte 1985; Kati et al. 2007), when considering larger spatial scales, an increasing effect of landscape heterogeneity on reptile species richness was detected (Schindler et al. 2009, 2010 [= Chapter B.2, Chapter A.2 of this thesis]).

Considering six different taxonomic groups together to represent local biodiversity (woody plants, orchids, Orthoptera, semi-aquatic herpetofauna, terrestrial herpetofauna and birds), we found that landscape heterogeneity has significant positive effects on species richness (Kati & Poirazidis 2005; Schindler et al. 2009, 2010 [= Chapter B.2, Chapter A.2 of this thesis]).

According to existing knowledge, landscape heterogeneity could have significant positive effects on many taxa (Kati & Poirazidis 2005; Schindler et al. 2009, 2010 [= Chapter B.2, Chapter A.2 of this thesis]), but the extent of the studied area plays an important role for the detection of these relationships. For example, woody plants, Orthoptera and birds were related to landscape heterogeneity at smaller scales, while reptile diversity was predicted better at larger scales (Schindler et al. 2009 [= Chapter B.2 of this thesis]). An optimal level of heterogeneity can hardly be determined as it depends on the taxa of interest, but diversity and spatial configuration of landscapes are important drivers of biodiversity and must be considered in the conservation of managed forests (Radford & Bennett 2004; McDonald et al. 2005; Quevedo et al. 2006). However, special attention should be paid to the thresholds above which the effects of heterogeneity become negative.

Continuous research on the pattern of relations between landscape heterogeneity and species richness will be useful to understand the impact of heterogeneity on biodiversity, and to improve management decisions in DNP and other Mediterranean forest landscapes (Poirazidis et al. 2010b [= Chapter D.1 of this thesis]). A systematic monitoring of land use and land-cover changes and their effects on indicator species would improve management decisions in DNP.

Chapter A.2. Landscape approaches and GIS as a prerequisite for biodiversity management in a Mediterranean forest landscape.

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Running title: Landscape approaches and GIS for biodiversity monitoring

Keywords: Conservation, landscape metrics, landscape structure analysis, decision support system, Dadia National Park, Greece

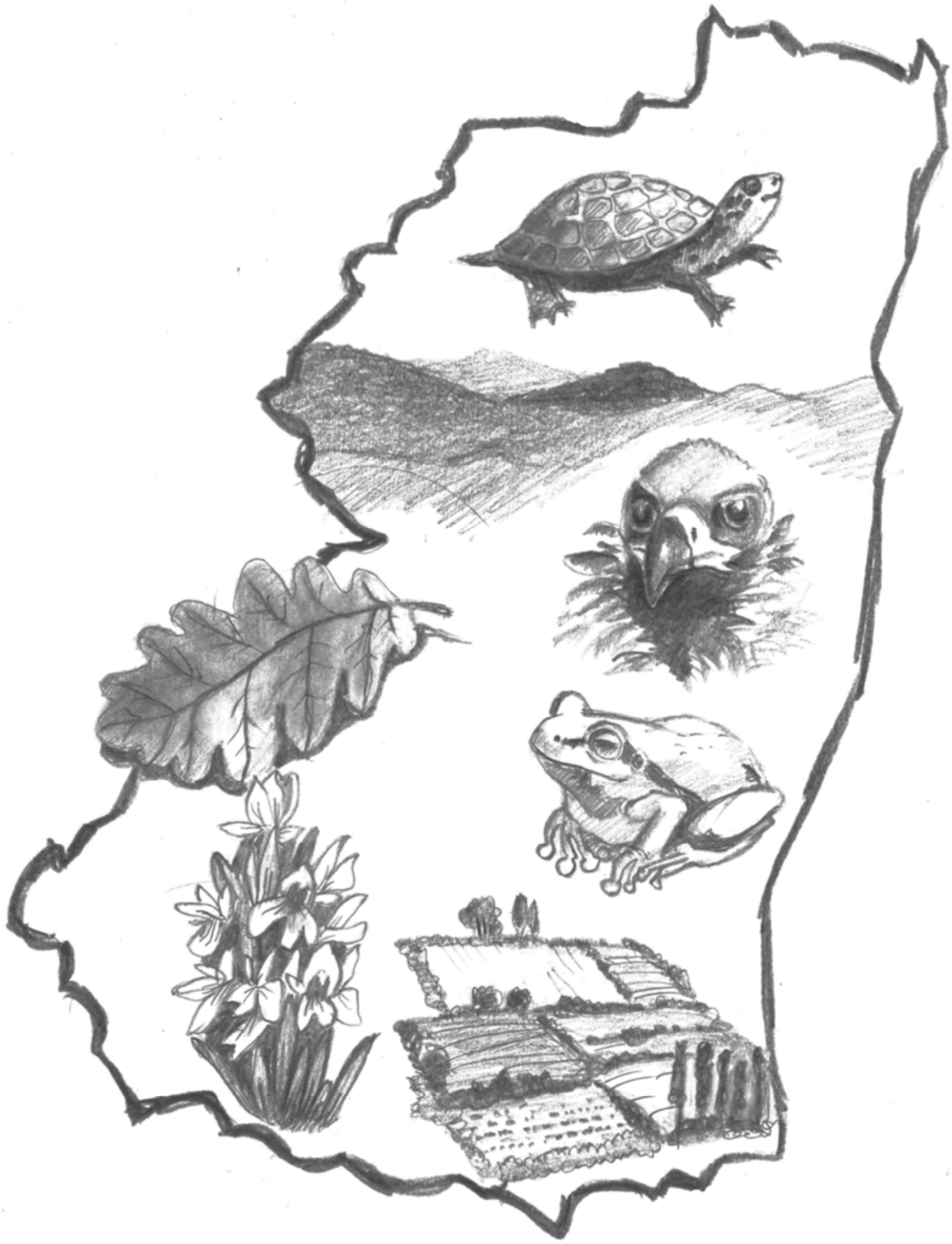

Illustration by Tuisku Sarrala

Own contribution:

Study design 70%, implementation 60%, writing 70%

Abstract

Remote sensing now routinely provides environmental information ranging from global to local scales, and geographical information systems provide, among other applications, necessary interfaces to store, analyse and visualise spatial data; increased computational capacities triggered even more such applications. In this chapter, we demonstrate how the combination of landscape approaches, remote sensing and GIS aids conservation and management of biodiversity. We therefore summarise six case studies from Dadia National Park (Dadia NP), in northeastern Greece. The studies aimed at (1) modelling of nesting habitat for a flagship species, (2) evaluation of land-use change, (3) detecting statistical dimensions and spatial patterns of landscape structure, (4) testing the performance of landscape metrics as indicators of biodiversity, (5) developing a GIS approach for a systematic raptor monitoring, and (6) developing a decision-support system to optimise conservation of biodiversity in managed forests.

Landscape approaches for biodiversity management

Landscape approaches and geographical information systems (GIS) have been playing an increasing role in biogeography and conservation biology over the last decade (Gaston 2000; Foody 2008; Gillespie et al. 2008). Within this period, the number of papers using GIS published in the journal Landscape Ecology has roughly doubled (Anderson 2008). Especially remote sensing applications became of growing importance within recent years; remote sensing now routinely provides environmental information ranging from global to local scales, and geographical information systems provide, among other applications, necessary interfaces to store, analyze and visualize spatial data; increased computational capacities triggered suchlike applications even more. In this paper, we demonstrate how the combination of landscape approaches, remote sensing and GIS aides conservation and management of biodiversity. We therefore summarize six case studies from Dadia National Park (Dadia NP), in north-eastern Greece. The studies aimed at 1) modelling of nesting habitat for a flagship species, 2) evaluating land use change, 3) detecting statistical dimensions and spatial patterns of landscape structure, 4) testing the performance of landscape metrics as indicators of biodiversity, 5) developing a GIS approach for a systematic raptor monitoring, and 6) developing a decision support system to optimize conservation of biodiversity in managed forests.

Study area and GIS data.

The study area, the Dadia NP, is situated in the Evros prefecture, north-eastern Greece (Figure A.2.1). Its extent of about 430 km^2 includes two strictly protected core areas covering 73.5 km^2. The mountainous area (altitudes ranging from 20 to 645m above sea level) is covered by extensive pine (*Pinus brutia, P. nigra*) and oak (*Quercus frainetto, Q. cerris, Q. pubescens*) forest, but it also includes a variety of other habitats such as pastures, agricultural fields, torrents and stony hills (Catsadorakis & Källander 2010). Dadia NP is an essential refuge for breeding populations of a unique assemblage of raptors (Poirazidis et al. 1996, 2010a [=Chapter A.1 of this thesis]). It contains the only remaining Black Vulture (*Aegypius monachus*) breeding colony in the Balkan Peninsula (Poirazidis et al. 2004; Skartsi et al. 2008), and a high diversity of passerines (Kati & Sekercioglu 2006), amphibians and reptiles (Kati et al. 2007), butterflies (Grill & Cleary 2003), grasshoppers (Kati et al. 2004c), and vascular plants (Kati et al. 2000; Korakis et al. 2006).

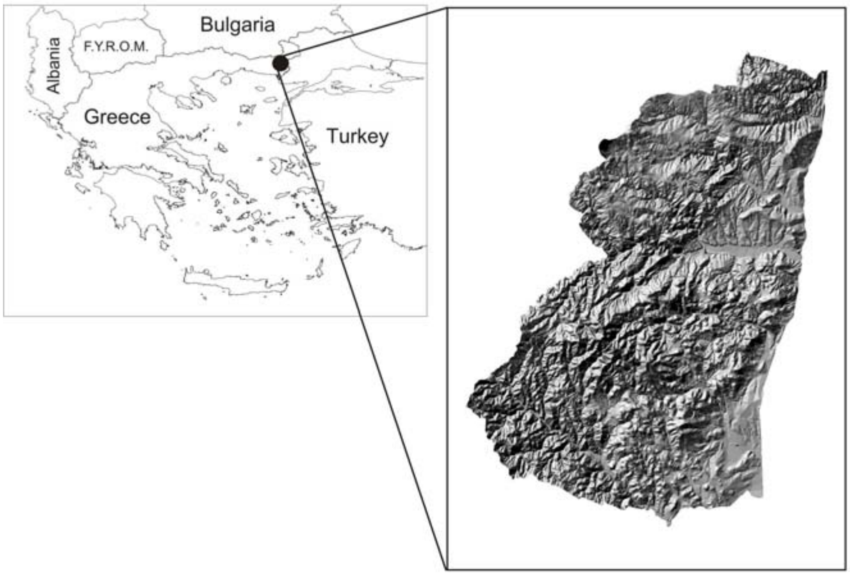

Figure A.2.1. *Map of the case study area, Dadia National Park, located in NE Greece.*

Satellite images (IKONOS, July 2001, pixel size 1 m) of the study area were digitized to produce a vector-map including 14 different habitat types related to the dominant forest tree species and six classes of the percentage of mixed forest. Out of this initial habitat base map, further maps differing in the number of land cover categories were produced for the case studies.

Case study 1 - Modelling nesting habitat as a conservation tool for the Eurasian Black Vulture (Poirazidis et al. 2004).

This study formulated habitat models in order to predict the potential nesting habitat of Black Vulture in Dadia NP, a priority breeding species for the area as well as over the rest of the Balkan Peninsula (Skartsi et al. 2008). The aims of this study were 1) to identify crucial determinants of suitable nesting habitat characteristics and 2) to build empirical models for the prediction of nesting habitat. Using logistic regression and 16 environmental variables, separate models regarding geomorphology, vegetation-types, and disturbance factors were obtained and combined using Bayesian statistics. At the final stage a Boolean map of the mature forest refined the present suitable nesting habitat (Figure A.2.2). The geomorphology contributed more than all other predictors to the final overall model of suitable Black Vulture nesting habitat. The nesting preference in areas with steep slopes seems to be adaptive, as such areas provide better foraging opportunities and protection from predators (Hiraldo and Donázar 1990; Fargallo et al. 1998; Donázar et al. 2002). The results of this study were used to improve the Black Vulture Monitoring, the zonation and the forest management of the National Park (cf. Poirazidis et al. 2010b [= Chapter D.1 of this thesis]).

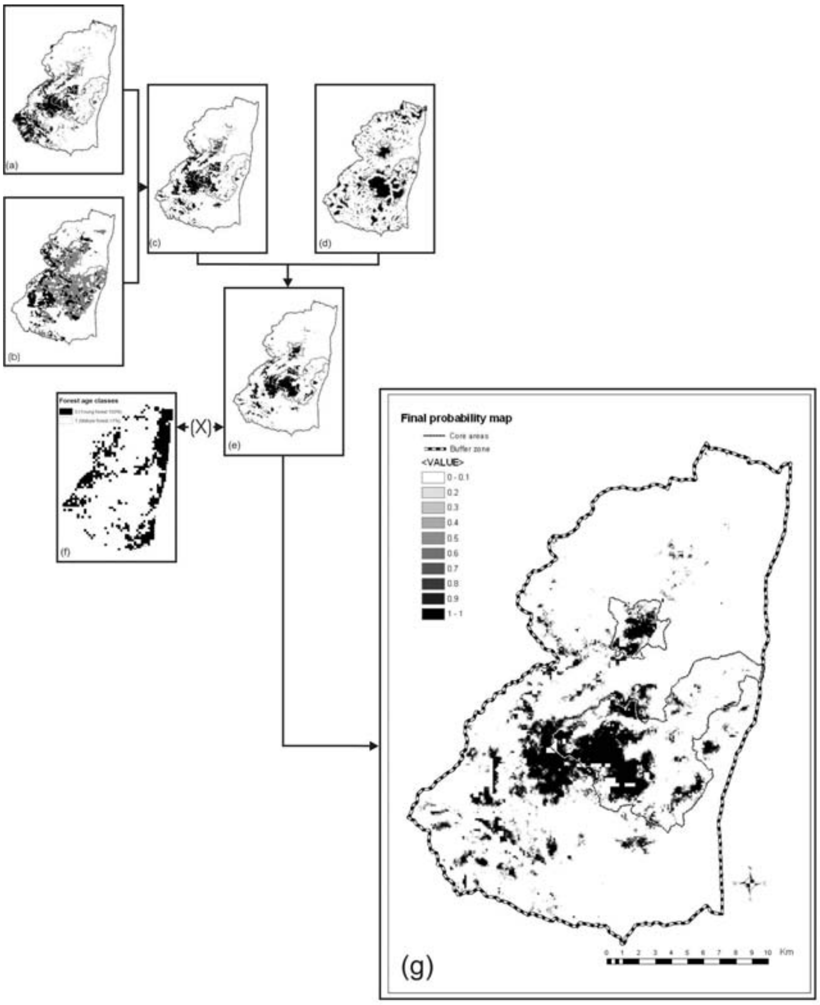

Figure A.2.2. *Maps of probability of occurrence for the nest sites of black vulture based on (a) a geomorphological model, (b) a vegetation-type model, (c) a model combining a and b, (d) a disturbance model, (e) a model combining c and d, (f) a Boolean map of mature forest, and (g) the final map combining e and f.*

Case study 2 - Forest re-growth since 1945 in the Dadia forest nature reserve (Triantakonstantis et al. 2006)

In this study, the focus was drawn on the interpretation of aerial photographs and satellite images in order to identify land use patterns in Dadia NP for 1945, 1973 and 2001, and thus to quantify the land use changes among these years. The landscape was classified to the three categories forest, openings, and agricultural land, and the most obvious change was a dramatic decline in forest openings (Table A.2.1), caused mainly by land abandonment and reforestation programs. During a period of 50 years, the landscape lost part of its characteristic heterogeneity and mosaic-structured character, landscape qualities that are very important for the maintenance of biodiversity of several groups of organisms (Atauri & De Lucio 2001; Torras et al. 2008).

Table A.2.1. *Land use change in Dadia National Park from 1945 – 2001.*

Land use	Zone	1945 km²	1945–1973 %	1973 km²	1973–2001 %	2001 km²
Forest	Core area	37.7	+ 33	50.1	+ 20	60.2
	Buffer zone	160.5	+ 15	183.9	+ 37	251.2
	Total area	198.2	+ 18	234.0	+ 33	312.6
Openings	Core area	33.3	− 40	20.1	− 50	10.1
	Buffer zone	119.4	− 27	87.0	− 67	28.6
	Total area	152.7	− 30	107.1	− 64	38.7
Agricultural land	Core area	1.9	+ 43	2.7	− 40	1.6
	Buffer zone	76.4	+ 12	85.4	− 21	67.2
	Total area	78.3	+ 13	88.1	− 22	69.0

Case study 3 - Towards a core set of landscape metrics for biodiversity assessments: A case study from Dadia National Park (Schindler et al. 2008 [= Chapter B.1 of this thesis])

Landscape metrics in GIS environment can be used to facilitate the investigation of the relation between landscape structure and biodiversity (Hill & Curran 2003; Honnay et al. 2003). Data reduction analyses have been applied to tackle the problem of highly correlated indices (Riitters et al. 1995; Cushman et al. 2008), but valid landscape predictors for fine scale Mediterranean forest-mosaics have been missing. In this study, we used a wide array of related variables of landscape structure, 1) to investigate correlations and statistical dimensions of landscape structure at landscape and class level, 2) to provide a core set of representative variables, 3) to evaluate the stability of the detected dimensions across scales, and 4) to describe characteristic landscape pattern of Dadia NP. Therefore, we produced a map of nine land cover categories that we converted to raster format with a grain of 5 m. We used FRAGSTATS (McGarigal & Marks 1995) for the computation of the 119 landscape metrics investigated in the study and applied correlation analysis and factor analysis, regarding both landscape and class level metrics in a parallel way. Landscape diversity, edge contrast (a measure related to fragmentation) and area-weighted mean patch shape were stable at landscape level across the three tested scales. The representative set of metrics consisted of Simpson's Diversity Index, Mean Edge Contrast Index, and the Area-Weighted Mean Shape Index. The pattern analysis revealed a dispersed pattern for landscape diversity, with high values in vicinity of the borders between core areas and buffer zone, and a clustered pattern for edge contrast, presenting a gradient from the unfragmented core areas to the agricultural land in the east of the reserve (Figure A.2.3).

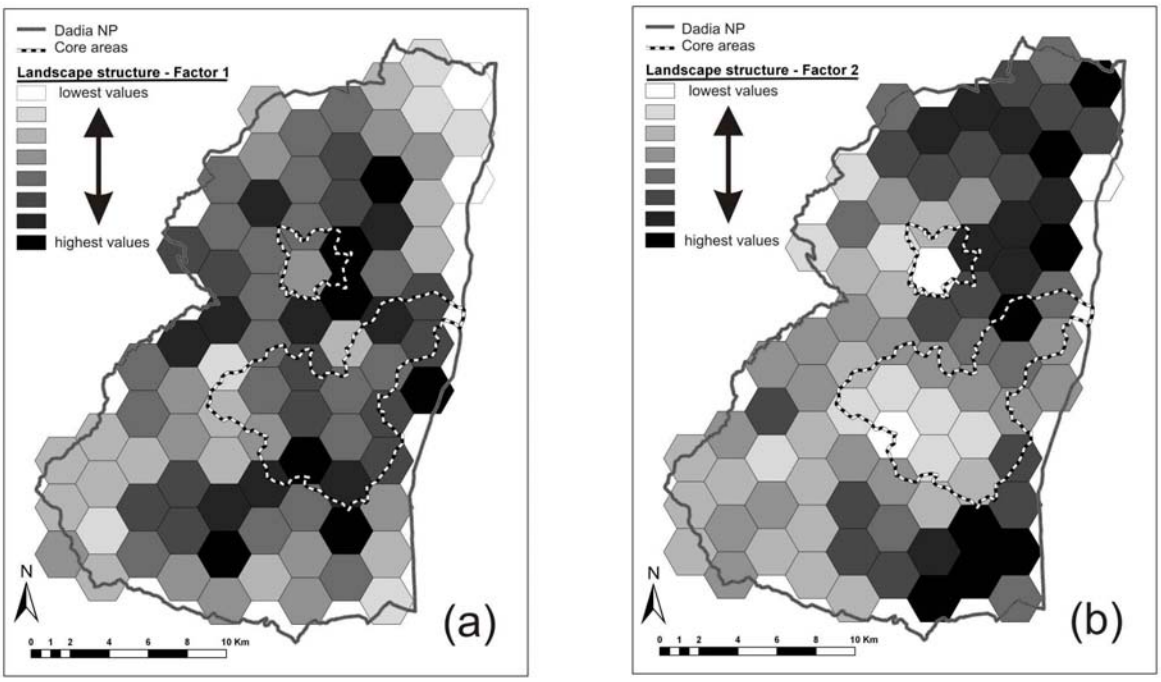

Figure A.2.3. *Pattern of the main dimensions of landscape structure in Dadia National Park. (a) landscape diversity (Factor 1) and (b) edge contrast (Factor 2).*

Case study 4 - Testing the performance of landscape metrics as indicators for biodiversity (Schindler et al. 2009 [= Chapter B.2 of this thesis (see also Chapter B.3 of this thesis)])

Since only some few empirical studies tested the relations between landscape structure and the species diversity of multiple taxa (Hernández-Stefanoni 2006; Yamaura et al. 2008), we tried to fill this gap in this case study. We analyzed the relations of 52 landscape structure variables with overall biodiversity and with species richness of the six taxa woody plants, orchids, orthopterans, amphibians, reptiles, and birds. Species data were collected by Kati et al. (2004b), based on standard methods; landscape structure variables were computed for circular areas of five different extents around the sampling plots. For each taxon the species richness was modeled with each individual landscape variable at each scale as predictor, based on a linear mixed model using the software R (R Development Core Team 2008). Additionally, we tested the performance of sets of three landscape structure variables as predictors of species richness, using AIC to compare sets composed by different methods such as expert knowledge, several methods of ordination (see previous case study or Schindler et al. 2008 [= Chapter B.1 of this thesis]), decision trees, random choice, and optimal sets after testing all possible combinations.

In this study, landscape metrics proved to be good indicators of species richness regarding the taxa woody plants, orthopterans, reptiles and for overall biodiversity. Metrics regarding patch shape, proximity, texture and diversity resulted frequently in significant univariate models, while metrics regarding similarity or edge contrast hardly contributed to significant models. Our results revealed that the scale affected the performance of landscape metrics. Woody plants, orthopterans and birds were better predicted at smaller scales, while reptiles were predicted best at larger scales. Regarding the different methods of composing sets, optimal sets performed always significantly better than all other methods. The statistical methods performed slightly better than random choice, while the expert knowledge performed slightly worse than random. The revealed pattern of relations and performances will be useful to understand landscape structure as driver and indicator of biodiversity, and to improve management decisions in Mediterranean forests and similar mosaic-landscapes.

Case study 5 - Development of a Geographic Information System for Territory Analysis of Raptor Species (Poirazidis et al. 2006, 2009a [= Chapter C.2 of this thesis])

Dadia National Park is well known for its high diversity of breeding birds of prey, a community exceeding totally 300 territories (Poirazidis et al. 2010a [= Chapter A.1 of this thesis]). An integrated monitoring plan was implemented by WWF - Greece in 1999, aiming the effective conservation of biodiversity and ecological values of the area. In this case study we describe the development of a GIS approach to estimate the territories of breeding raptors. All raptors within 34 permanent plots were counted and each plot was censuses five times during the breeding seasons 2001-2005. Raptor observations were labeled in GIS, showing flight trajectories, possible nest sites, the number of synchronously observed individuals, age, sex, and different territorial activities under different symbols to enable analyses that consider all the information obtained in the field. The progressive analysis per species was based on eight criteria related to territorial behaviour, general observations and biology of the species as well as to landscape features (Poirazidis et al. 2006, 2009a [= Chapter C.2 of this thesis]). Breeding territories were differently classified as confirmed or possible. The GIS approach for estimating raptor territories was particularly effective for strictly territorial species like most of the eagles, buzzards, hawks, and falcons (Table A.2.2). Less territorial species, such as the Egyptian Vulture (*Neophron percnopterus*) and the Short-toed Eagle (*Circaetus gallicus*) demanded a large amount of data to enable for precise territory estimation.

Table A.2.2. *Summary of the species-specific problems and advantages of the GIS-based methodology for the estimation of raptor population sizes at local scale.*

Species	**problems with low territoriality**	**problems with secretiveness or late arrival**	**frequent key observations, high accuracy**	**Total usefulness GIS method**
White-tailed Eagle	medium	very few	medium	medium
Golden Eagle	not any	very few	medium	very high
Imperial Eagle	very few	high	medium	medium
Lesser spotted Eagle	few	very few	high	very high
Short-toed Eagle	high	not any	medium	medium
Booted Eagle	very few	few	medium	high
Egyptian Vulture	very high	few	very high	high
Common Buzzard	very few	not any	high	very high
Long-legged Buzzard	very few	few	high	very high
Honey Buzzard	very few	high	medium	medium
Black Kite	very high	few	very few	low
Marsh Harrier	high	very few	very few	low
Goshawk	not any	medium	few	medium
Levant Sparrowhawk	very few	very high	few	low
Sparrowhawk	very few	medium	few	medium
Peregrine Falcon	few	very few	high	very high
Lanner Falcon	very few	very few	high	very high
Hobby	not any	high	few	medium
Eurasian Kestrel	not any	very few	medium	very high
Black Stork	very high	not any	medium	medium

Case study 6 - Conservation of biodiversity in managed forests: An integrated approach using multi-function forest services. (Poirazidis et al. 2008, 2010b [= Chapter D.1 of this thesis])

In this case study we developed a decision support system to optimize the conservation of biodiversity in managed forests. We investigated timber production and biodiversity, the main ecosystem services of the Mediterranean forest landscape of Dadia NP. We produced 1) a series of spatially explicit habitat suitability models for higher plants, amphibians, small forest birds and raptors and an overall model for total local biodiversity, 2) maps related to timber production and 3) three management scenarios and a decision support system based on a conflict assessment. Thus we were able to establish integrated management concepts, and to assess the effects of different management strategies on the two main ecosystem services.
Spatial modelling was based on data of several systematic field surveys. We used 23 eco-geographical variables to derive predictors for species habitat suitability, and modelled five taxa as surrogates for the total biodiversity in Dadia NP, namely grasses and shrubs (combined later to "higher plants"), amphibians, small forest birds (mainly Passerines) and raptors. For the three groups of fauna we created species distribution maps, while regarding plant species we used the accumulated number of plant species as proxy of biodiversity. For the raptor data set (Poirazidis et al. 2010c [= Chapter C.4 of this thesis]) we pooled data from five years and plotted the center of their yearly territories. All the data were converted to a raster grain of 50 x 50 m, and Environmental Niche Factor Analysis (ENFA) was performed within the BIOMAPPER software (version 3.2; Hirzel et al. 2002). The total timber standing volume per sub-section was estimated using the official forest service inventory for the current forest management plan 2006 – 2016. The relative thematic maps were classified into four bins, (1) unsuitable, (2) marginal, (3) suitable and (4) optimal regarding habitat suitability, and (1) minimum, (2) medium, (3) large and (4) maximum regarding timber stand volume. We considered four different forest management actions at the stand level: management (1) without limitations, (2) with temporal restrictions, (3) with temporal and spatial restrictions and (4) focused on the ecological values. Three general management scenarios were formulated: Conservation, timber production and

trade off. A major output was the map of the proposed forest management categories of the trade off scenario (Figure A.2.4).

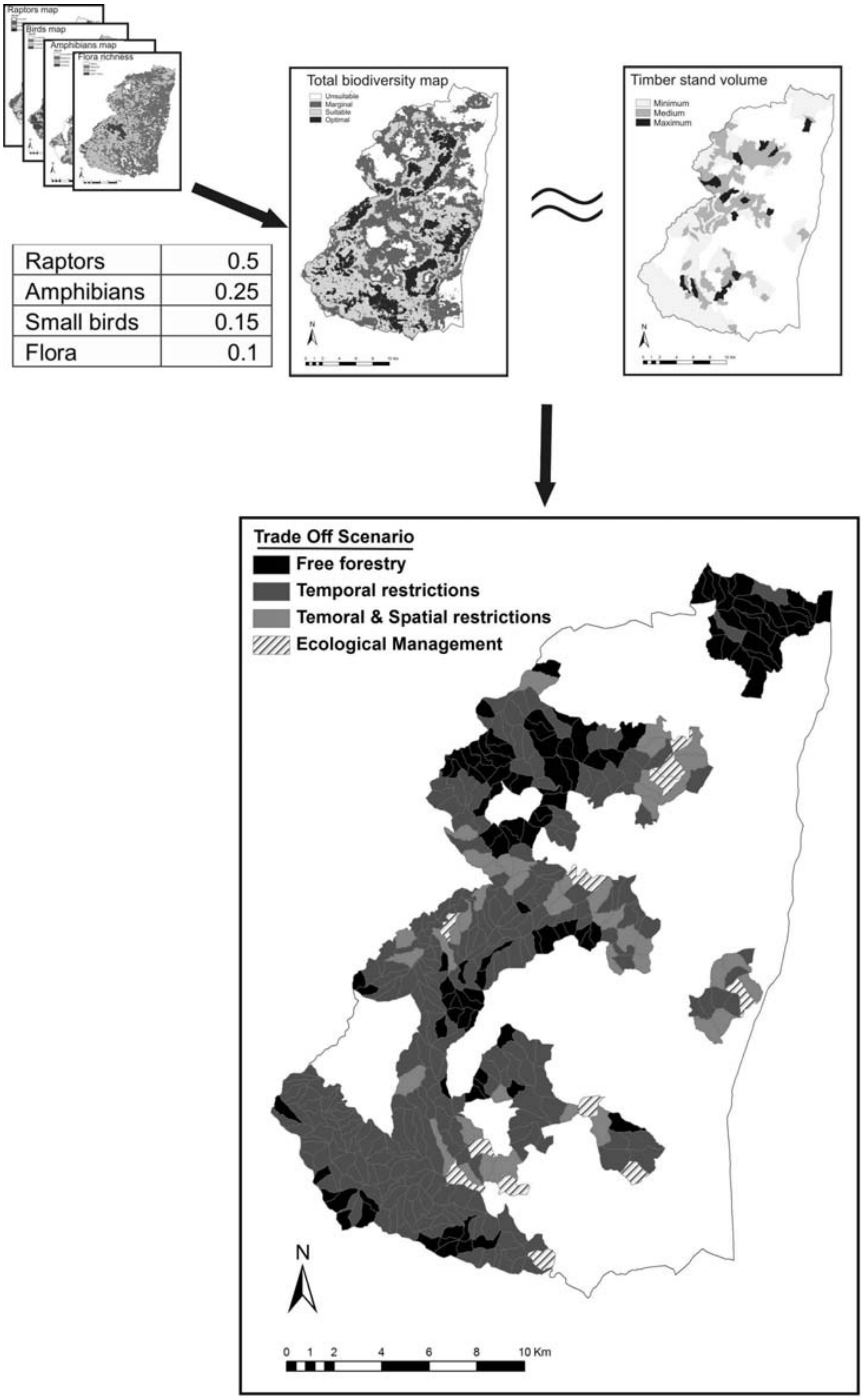

Figure A.2.4. *Map of Dadia NP after the trade off scenario considering conservation of biodiversity and timber production. The managed forests are categorized into the four* *management options free forestry, temporal restrictions, temporal & spatial restrictions, and ecological management.*

Conclusions and implications for biodiversity management

Landscape approaches involving GIS and integrated statistical approaches proved to be useful to understand the relations of pattern and changes of landscape structure with the present biodiversity and the habitat suitability for different groups of organisms. This knowledge was essential to establish conservation strategies for biodiversity, for instance regarding the maintenance of habitat heterogeneity in both core and buffer zone of the reserve (Grill & Cleary 2003; Kati et al. 2004c; Kati & Sekercioglou 2006), and for the optimization of other ecosystem services such as timber production. Habitat suitability modeling for selected groups of organisms to develop management scenarios for managed forests is highly recommendable. A landscape surveillance should be integrated into the ecological monitoring of key and indicator species to aid the evaluation of the management effects on both forest and wildlife. Further research regarding species, taxa and landscape indicators on a larger scale would be desirable to further extrapolate and validate the models, and enable an even more complete strategy for biodiversity conservation and management.

Acknowledgements

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Part B - Landscape structure and its use as ecological indicator

Dadia National Park is famous for its high diversity and abundance of rare raptor species, which are also the conservation flagships of the reserve. When dealing with raptors (and other wide ranging animals), it is needed to consider more than a single habitat patch, and landscape and landscape structure are gaining importance. In Dadia National Park, we started with landscape structure analysis in the year 2003. Overwhelmed by the multitude of available measures of landscape structure (e.g. McGarigal & Marks 1995), we decided to perform at first place a thorough study on the relations among the metrics, with the main aim to develop a core set of metrics for further studies in Dadia National Park, which should be also applicable for similar Mediterranean forest mosaics. This topic comprises the first paper of this chapter. Having obtained a core set of metrics for landscape level (e.g. considering all patched of a landscape), and for class level (considering only one certain landcover category), which were rather stable across three spatial scales, we wanted to test the performance of the sets of metrics as indicators of species richness and biodiversity. Thus, using a data set from Vassiliki Kati, we tested for several scales, 1) which metrics are well perfuming indicators for which groups of organisms, and 2) if our set of metrics composed by ordination methods performed better than other methods of composing sets such as expert knowledge, decision trees and random choice. This study is presented in the second paper of part B, which was written 2009 for the proceedings of the 1st European IALE conference. In the more elaborated version, which is presented here as the third paper of this part B and was recently submitted to Ecological Indicators, we had to omit the second aspect (i.e. the comparison of methods) to reduce the length of the paper.

Finally, in the forth chapter of this Part B, I present a study, organized by Vassiliki Kati, which I could have also included in the last part of this thesis regarding conservation management. This study develops and presents an approach of using ecological heterogeneity for reserve design, applying two measures of vertical heterogeneity and the core set of landscape metrics that resulted from Chapter B.1.

Chapter B.1. Towards a core set of landscape metrics for biodiversity assessments: a case study from Dadia National Park, Greece.

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Running title: Towards a core set of landscape metrics

Keywords: landscape structure, landscape pattern, factor analysis, heterogeneity, fragmentation, Fragstats

Illustration by Tuisku Sarrala

Own contribution:

Study design 65%, implementation 90%, writing 65%

Abstract

Spatial heterogeneity has an important influence on a wide range of ecological patterns and processes, and many landscape metrics in GIS environment are used to facilitate the investigation of the relation between landscape structure and biodiversity. Data reduction analyses have been applied to tackle the problem of highly correlated indices, but valid landscape predictors for fine scale Mediterranean forest mosaics are still missing. Therefore, we analyzed the landscape structure of Dadia National Park, Greece, a Mediterranean forest landscape of high biodiversity, characterized by pine, oak and mixed woodland. By distinguishing nine land cover classes, 119 variables were computed and factor analysis was applied to detect the statistical dimensions of landscape structure and to define a core set of representative metrics. At landscape level, diversity of habitats, fragmentation and patch shape and at class level dominance of mixed forest and the gradient from one pure forest type to another turned out to be the crucial factors across three different scales. Mapping the encountered dimensions and the representative metrics, we detected that the pattern of landscape structure in Dadia National Park was related to dominating habitat types, land use, and level of protection. The evaluated set of metrics will be useful in establishing a landscape monitoring program, to detect the local drivers of biodiversity, and to improve management decisions in Dadia NP and similar mosaic-landscapes.

Introduction

Fragmentation, loss and degradation of habitat are widely considered as the most important threats to biodiversity on a global scale (Wilcove et al. 1986; Soulé 1987; Fahrig & Meriam 1994; Tilman et al. 1994; Wiens 1995). On the other hand, in many European ecosystems, where human activities have shaped the landscape for many centuries, a positive relationship between spatio-temporal heterogeneity of ecosystems and local biodiversity has been detected (e.g. Brotons et al. 2004; Kati et al. 2004b; Saïd and Servanty 2005). Mosaics of seminatural habitats, which characterize forest landscapes of many parts of Europe (Forman 1995; Blondel & Aronson 1999; Ernoult et al. 2003), play an important role for many species of fauna (e.g. Chust et al. 2004; Carrete & Donázar 2005; Saïd & Servanty 2005). But the landscape structure, often regarded as important background for local biodiversity, underlies rapid changes due to current trends in socio-economic and agri- and silvicultural development (e.g. Rocchini et al. 2006). Thus, a negative impact on local and regional biodiversity has been encountered in several studies (e.g. Zechmeister et al. 2003; Scozzafava & De Sanctis 2006).

Landscape structure variables are easily obtainable over large areas (see Groom et al. 2006) and their calculation is less demanding in terms of time and money than collecting detailed data on species distribution and abundance. Thus, an increasing number of studies analyze relations of landscape structure and biodiversity, aiming at the use of related variables as predictors for modelling spatio-temporal distribution patterns of species and communities (Bisonette 1997; Dufour et al. 2006). Many landscape structure variables are currently available (McGarigal & Marks 1995; Riitters et al. 1995), and many of them can be computed for the overall landscape (landscape level) and for specific land cover classes (class level). It is often necessary to use several metrics to characterize a particular landscape, because different qualities of spatial pattern do exist (Tischendorf 2001; McAlpine & Eyre 2002; Neel et al. 2004), but the use of many highly correlated indices does not yield new information and leads to problems in the interpretation of the results (Jones et al. 2001; Li & Wu 2004). For these reasons, the analyst should select metrics that are relatively independent of one another, providing a unique and ecological meaningful contribution to our understanding of landscape structure (Hargis et al. 1998; Turner et al. 2001). In order to define an optimal set of metrics, theoretical considerations (Li & Reynolds 1994) and statistical data reduction analyses have been used to detect unique dimensions of landscape structure (McGarigal & McComb 1995; Riitters et al. 1995; Cain et al. 1997; Scånes & Bunce 1997; Tinker et al. 1998; Griffith et al. 2000; Lausch & Herzog 2002; Cifaldi et al. 2004). Despite these research efforts from mainly temperate and boreal regions, an optimal set of landscape metrics for Mediterranean landscapes – especially their biodiversity rich forest-mosaics – has not been defined yet.

We studied the landscape structure of the National Park of Dadia-Lefkimi-Soufli Forest (hereafter Dadia NP), Greece, a Mediterranean forest of high biodiversity (e.g. Kati 2001; Kati et al. 2004b, Poirazidis et al. 2004). Most of the area is under intensive forest management, thus a landscape monitoring should be established to determine effects of land use and management on landscape structure and to improve the conservation management (Poirazidis et al. 2002). The importance of the heterogeneity of the habitat for the local biodiversity has been recognized (e.g. Kati et al. 2004b), but the pattern of the landscape structure remains unidentified. For these reasons the objectives of this study were (a) to analyze the statistical dimensions of landscape structure at landscape and at class level, (b) to provide a core set of representative variables, (c) to evaluate the stability of the detected dimensions across different scales, and (d) to describe characteristic patterns of the landscape structure of Dadia NP.

Methods

Study area

Our study area, the Dadia NP (26º00′ - 26º19′ N, 40º59′ - 41º15′ E), is situated in the Evros prefecture, north-eastern Greece (Figure B.1.1). It has an extent of about 430 km^2, including two strictly protected core areas that cover 73.5 km^2. The mountainous area (altitudes ranging from 20-645 m above see level) is covered by extensive pine *(Pinus brutia, P. nigra*) and oak (*Quercus frainetto, Q. cerris, Q. pubescens*) forest, but it includes also a variety of other habitats such as pastures, fields (cultivations), torrents and stony hills. Dadia NP is an essential refuge for breeding populations of a unique assemblage of raptors (Poirazidis et al. 1996), contains the only remaining Black Vulture (*Aegypius monachus*) breeding colony in the Balkan Peninsula (Poirazidis et al. 2004), and a high diversity of passerines (Kati & Sekercioglu 2006), amphibians and reptiles (Helmer & Scholte 1985), butterflies (Grill & Cleary 2003), grasshoppers (Kati et al. 2004c), and orchids (Kati 2001).

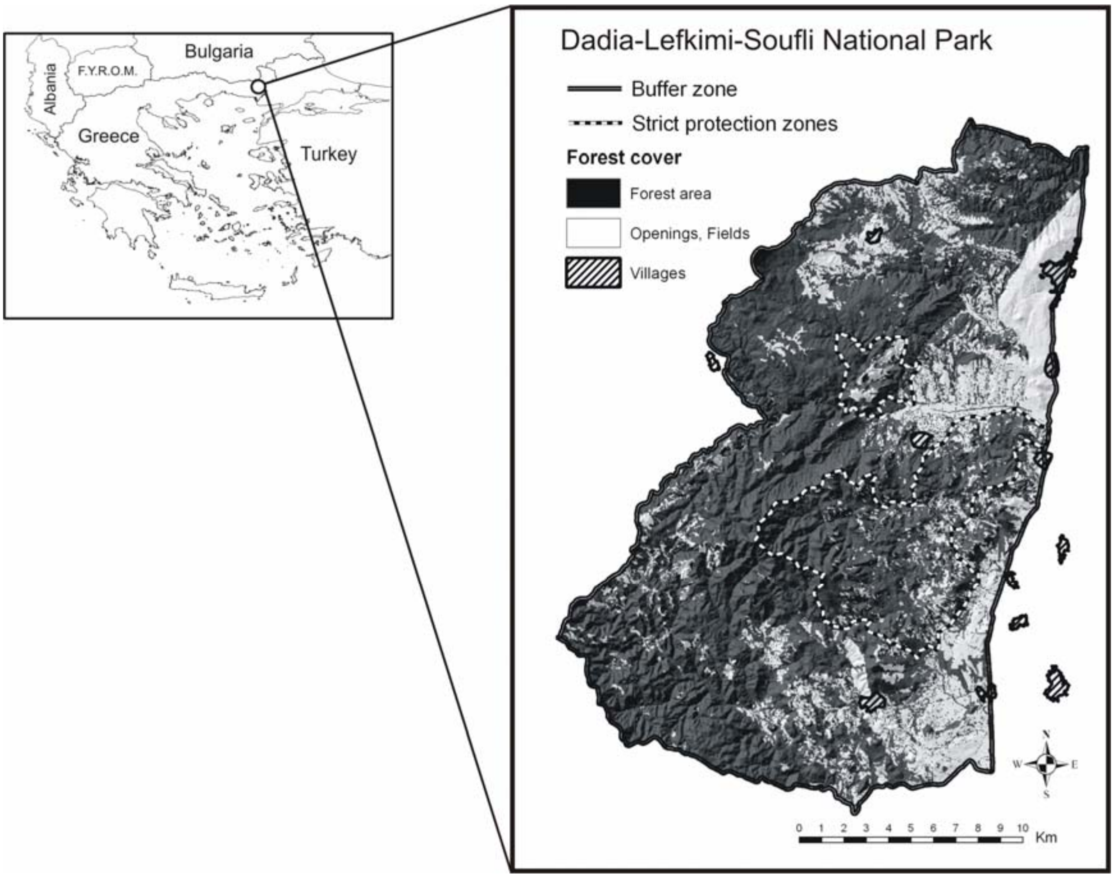

Figure B.1.1. *Habitat of Dadia NP, located in Evros, Greece.*

Land cover data set, hexagonal grid and landscape metrics

Satellite images (IKONOS, July 2001, pixel size 1 m) of the study area were digitized on screen to produce a vector-map including 25 different habitat types related to the dominant forest tree species and the percentage of mixed forest. The initial habitat map was merged into nine land cover categories, namely oak forest (OA), pine forest (PI), pine-oak forest (PO), oak-pine forest (OP), broadleaves (BL), openings (OO), fields (FI), roads (RO), and urban areas (UR). This map was then converted to raster format with a grain of 5 m, using the spatial analyst module of ArcGIS® (ESRI, Inc., Redlands, CA). In this study OA and PI are pure forests, while PO and OP are mixed forests, dominated by pine and oak, respectively. BL is dominated by broadleaves other than oaks, and OO includes several natural and semi-natural non-forested areas like patches of grassland, rocks and torrents.

In order to achieve homogenous spatial units for proper statistical analysis, we produced a hexagon grid and clipped samples from the land cover data set. Because changes in the extent of maps can produce unpredictable behavior of landscape metrics (e.g. Wu et al. 2002; Wu 2004), we used an adaptive approach, proposed by Turner et al. (1989) and tested for stability of the results across three different scales (grid units, i.e. extents of maps). Hence we chose the specific scale of 500 ha for the hexagon grid and assessed later the robustness of the results using grids of 1000 and 250 ha (see Figure B.1.2 for an overview of the methodology). The extent of 500 ha was chosen, because it guaranteed a representative sample of patches per hexagon (n = 230.2 ± 136.8, see O'Neill et al. 1996) and enough hexagonal maps for the total study area. After the exclusion of all hexagons with more than 20 % of their area uncovered by the land cover data, eighty-five 500 ha hexagonal maps of land cover categories (hereafter hexagons), covering 422.5 km^2, remained for further analysis.

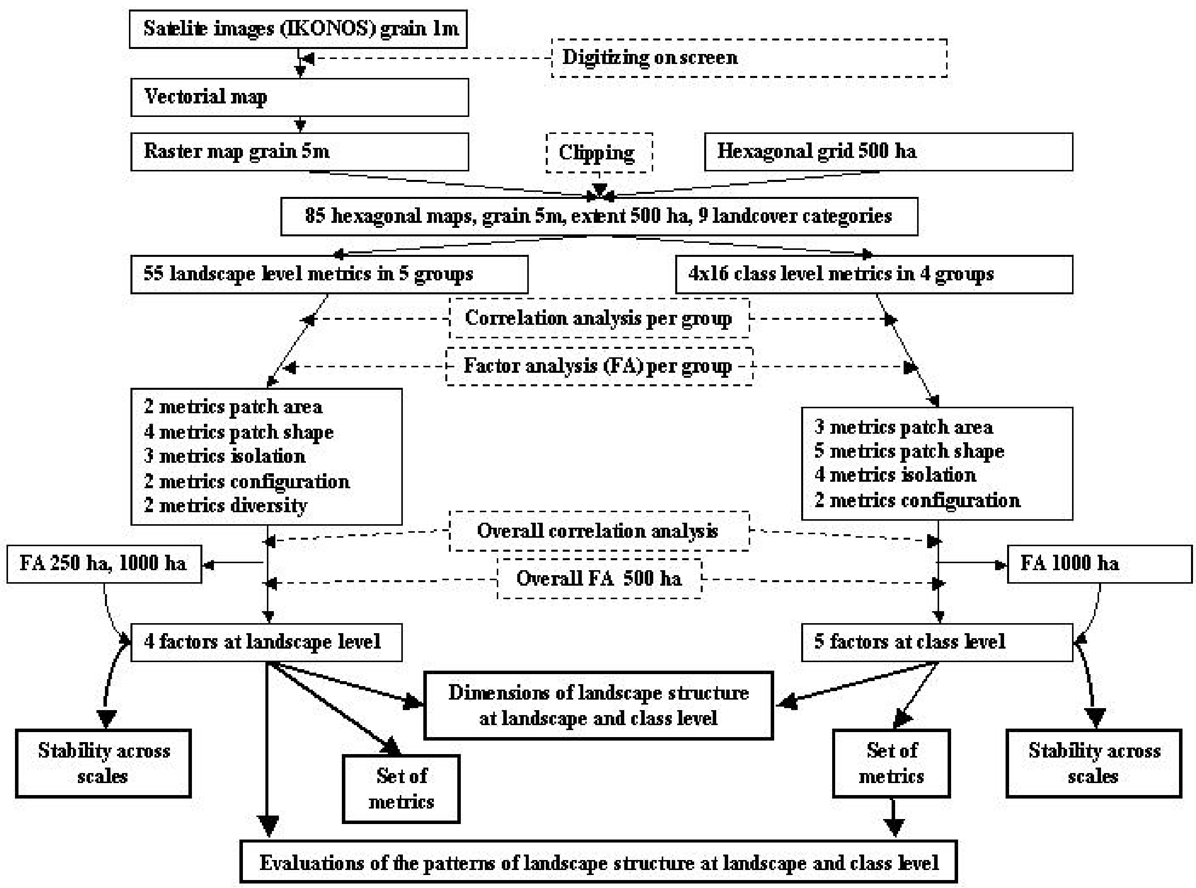

Figure B.1.2. *Overview of the methodology of the study.*

The landscape structure was analyzed at landscape level (considering all landcover categories) and at class level (considering one focus landcover category only), because the variables concerning the two levels contain different kind of information. Using FRAGSTATS 3.3 (McGarigal & Marks 1995), we computed for each hexagon 55 landscape level metrics, and 64 class level metrics (16 for each of the four forest categories PI, PO, OP, and OA). In order to even out the number of metrics, we modified the approach of Lausch & Herzog (2002) and applied correlation tests and factor analyses in a first step for five separated groups of metrics regarding the aspects (i) patch size and patch density, (ii) shape, edge and contrast, (iii) isolation, proximity and connectedness, (iv) texture, and (v) diversity of habitats (Table B.1.1). Thus, we evaluated smaller sets of metrics that explained most of the variance of these five aspects of landscape structure, and used the variables with the highest loadings per factor in a next step as input in an overall analysis to detect the statistical dimensions of landscape structure. We performed corresponding analyses for both the landscape and the class level (Figure B.1.2).

For the computation of the landscape metrics, the land cover patches were delineated applying the eight neighbor rule to guarantee that linear patches along a direction diagonal to the grid axes were identified as a single patch. Each hexagon was analyzed separately and hexagon boundaries were not counted as edges. The Proximity and Similarity Metrics as well as the Connectance Index were computed using search radii and threshold distances of 1000 m, respectively. In order to compute the Similarity Indices and the Contrast Metrics, a Similarity Matrix and an Edge Contrast Matrix were produced for the nine land cover classes, assigned weights were based on logical values according to the authors experience in the study area.

Data reduction analyses

Within each of the variable groups (five at landscape and four at class level, Table B.1.1) we examined pairwise Spearman correlation coefficients, and of the pairs of metrics with coefficients > 0.9, only one metric was retained (Riitters et al. 1995; Griffith et al. 2000). Density metrics were chosen over absolute metrics, because some of the hexagons were not fully covered by the land cover data set. In cases where the distribution statistics were highly correlated, the mean of the metrics was preferred to the area-weighted mean, which in turn was preferred to the coefficient of variation.

With respect to the diversity and evenness indices, Simpson-based metrics were preferred, because the use of Shannon-based metrics is recommended only if patch richness is greater than 100 (Yue et al. 1998). For all the other pairs of highly correlated metrics, we selected the metric, which is more commonly used in biodiversity literature.

Using this procedure, the original set of landscape level metrics was reduced from 55 to 35, and the class level set from 64 to 60. With the remaining metrics, within each of the groups, a factor analysis (FA, e.g. Johnston 1980) was performed. By using orthogonal (varimax) rotations of the axes, we accounted for additional variance and produced non-correlated factors. We retained factors by using two criteria: the shape of the scree plot and Kaisers rule of thumb that the eigenvalue of the factor should be greater than 1.0. In the cases of disagreement between these two criteria, both possibilities were evaluated and interpretability of results was the ultimate criterion for the final selection. For each retained factor of all groups, the metric with the highest absolute loading was defined as representative and included in the overall analysis. The selected metrics were checked first for high correlations and then an overall FA was performed (Figure B.1.2), applying the same methodology as described above. To detect the most important dimensions of landscape structure, we interpreted the overall factors using the variables that had high loadings and defined the optimal set of metrics to quantify landscape structure as the representative metrics of the overall analyses. At landscape level we used all the 500 ha hexagons for the factor analyses (n = 85), whereas at class level, we included only the hexagons that contained patches of all four forest types (n = 60).

Evaluation of the stability of the detected dimensions across maps of different extents

To evaluate the stability of the encountered factors across different extents of maps, we performed at landscape level FAs for the scales of 250 ha (n = 177 hexagons) and 1000 ha (n = 39). At class level, the FA was only performed for the 1000 ha scale (n = 36), because a high percentage of the hexagons lacked at least one of the four land cover categories at the scale of 250 ha. To permit the comparison of the resulting factors, we included the same metrics as for the overall FAs at 500 ha and retained the same number of factors. Finally, we calculated coefficients of congruence (Johnston

1980; Cain et al. 1997) to evaluate the similarity among the factors emerged at the different scales.

Mapping of the landscape structure and description of the resulting patterns

To detect the patterns of landscape structure at landscape level, we calculated the factor scores of each hexagon for each encountered dimension of landscape structure. Then we mapped the factor scores and compared the resulting patterns with dominating habitat type, land use and level of protection. Finally pattern analysis was performed at class level, using the values of the representative metrics instead of the factor scores, because they were available for more hexagons (76 to 85 instead of n=60), being of advantage when evaluating the landscape patterns.

Results

The number of land cover classes per 500 ha hexagon ranged from four to nine and the number of patches per hexagon ranged from 36 to 664. Oak forest (OA) accounted on average for 26.7%, PI for 12.9%, OP for 10.7%, and PO for 21.2% of the hexagons. The other land cover categories accounted on average for 1.9% (BL), 8.9% (OO), 14.7% (FI), and 3.2% (RO and UR together).

At landscape level, the number of factors retained per aspect of landscape structure ranged from two until four, and the cumulative variance explained by these factors from 66 % to 91%. Most factors were retained for the patch shape group (Table B.1.2). The metrics of the diversity group were highly correlated, and only the pair of metrics SIDI and PRD obtained a Spearman Correlation Coefficient less than 0.9. Thus, instead of performing a FA for this group, these two metrics were directly included in the overall analysis. At class level three until five factors were retained per aspect and the cumulative variances ranged from 70 % to 77%. Selecting the metrics with the highest absolute loading per factor, totally 13 metrics remained for the overall analysis at landscape and 16 at class level.

Table B.1.1. *Landscape metrics used in this study. Regarding the distribution statistics (DSt), Mean (MN), Area Weighted Mean (AM) and Coefficient of Variation (CV) were used at landscape level, but only the Mean at class level. Each class level metric was computed for each of the four forest types PI, PO, OP, and OA.*

Group	Acronym	Metric name	Landsc. L.	Class L.	Sum	Description
Group I. Patch size & patch density			***8***	***12***	***20***	
	AREA	Patch Area	3	4	7	DSt; size of the patches
	GYRATE	Radius of Gyration	3		3	DSt; radius of gyration, i.e. the mean distance for each cell of one patch to the patch centroid
	PD	Patch Density	1	4	5	Number of patches per area
	LPI	Largest Patch Index	1		1	Percentage of total area occupied by the largest patch
	PLAND	Percentage of Landscape		4	4	Percentage of area occupied by certain land cover class
Group II. Shape, edge & contrast			***23***	***24***	***47***	
	LSI	Landscape shape index	1		1	Ratio of the total edge to the minimum total edge
	NLSI	Normalized Landscape shape index		4	4	Ratio of the total edge to the minimum total edge per class, rescaled according the proportion of the classes
	ED	Edge Density	1	4	5	Total length of edge per unit area
	SHAPE	Shape Index	3	4	7	DSt; equals 1 when all patches are circular; increases with complexity of patch shapes; independent of patch size
	PARA	Perimeter-area ratio	3		3	DSt; patch shape complexity measure that measures perimeter per area
	CIRCLE	Related Circumscribing Circle	3	4	7	DSt; patch elongation measure; equals 1 minus patch area divided by the area of the smallest circumscribing circle
	FRAC	Fractal Dimension Index	3		3	DSt; patch shape complexity measure that approaches 1 for simple shapes and 2 for complex shapes
	CONTIG	Contiguity Index	3		3	DSt; equals 0 for a one-pixel patch and approaches 1 as patch contiguity, or connectedness increases
	PAFRAC	Perimeter-Area Fractal Dimension	1		1	Patch shape complexity measure, which approaches 1 for shapes with simple perimeters and 2 for complex shapes
	CWED	Contrast-Weighted Edge Density	1	4	5	Total amount of edge per area, weighted by the contrast between the different land cover types
	TECI	Total Edge Contrast Index	1	4	5	Ratio of the contrast weighted total length of edge to the not-contrast weighted total length of edge per grid
	ECON	Edge Contrast Index	3		3	DSt; ratio of the contrast weighted to the not-contrast weighted edge length per patch
Group III. Isolation, proximity & connectedness			***10***	***16***	***26***	
	PROX	Proximity Index	3	4	7	DSt; considers size and proximity of all patches with the same land cover type inside a specified search radius
	SIMI	Similarity Index	3	4	7	DSt; considers size and proximity of patches within a search radius, weighted by their similarity to the focal patch
	ENN	Euclidean Nearest Neighbour Distance	3	4	7	DSt; minimum edge to edge distance to the nearest neighbouring patch of the same type
	COHESION	Patch Cohesion Index		4	4	Measure of the physical connectedness of the focal land cover class
	CONNECT	Connectance Index (%)	1		1	Percentage of patches which are joined, i.e. inside a specified threshold distance
Group IV. Texture			***6***	***12***	***18***	
	CONTAG	Contagion Index	1		1	Measure of the aggregation of the land cover classes
	PLADJ	Percentage of Like Adjacencies	1	4	5	Percentage of neighbouring pixel, being the same land cover class, based on double-count method
	AI	Aggregation Index	1		1	Percentage of neighbouring pixel, being the same land cover class, based on single-count method
	IJI	Interspersion & Juxtaposition Ind. (%)	1	4	5	Measure of evenness of patch adjacencies, equals 100 for even and approaches 0 for uneven adjacencies
	DIVISION	Landscape Division Ind. (Proportion)	1	4	5	Equals the probability that 2 randomly chosen pixels in the landscape are not situated in the same patch
	SPLIT	Splitting Index	1		1	Equals the number of patches of a landscape divided into equal sizes keeping landscape division constant
Group V. Diversity			***8***	***0***	***8***	
	PRD	Patch Richness Density (no./100 ha)	1		1	Equals the number of patch types (i.e. land cover categories) per 100 ha
	RPR	Relative Patch Richness	1		1	Percentage of present patch types out of all categories
	SIDI	Simpson's Diversity Index	1		1	Diversity measure, which equals 1 minus the sum of the squared proportional abundance of each patch type
	SHDI	Shannon's Diversity Index	1		1	Equals minus the sum of the proportional abundance of each patch type multiplied by the ln of that proportion
	MSIDI	Modified Simpson's Diversity Index	1		1	Diversity measure, which equals minus the ln of the sum of the squared proportional abundance of each patch type
	SHEI	Shannon's Evenness Index	1		1	Diversity measure, which considers only evenness of patch sizes, not the number of patches
	SIEI	Simpson's Evenness Index	1		1	Diversity measure, which considers only evenness of patch sizes, not the number of patches
	MSIEI	Modified Simpson's Evenness Index	1		1	Diversity measure, which considers only evenness of patch sizes, not the number of patches
SUM			**55**	**64**	**119**	

Table B.1.2. *Overall factor analyses for the 500 ha hexagon grid of landscape and class level, including the 13 respectively 14 variables determined as representative for the retained factors of the factor analyses per group.**

Landscape Level

Metrics	**Group**	**Factor 1**	**2**	**3**	**4**
Eigenvalue		4.186	2.148	2.017	1.844
% of variance explained		32.199	16.521	15.517	14.187
% of cumulative variance explained		32.199	48.720	64.237	78.424
SIDI	**diversity**	**0.923**			
PROX_MN	isolation	**-0.889**			
CIRCLE_AM	shape	**0.823**			
IJI	texture	**0.765**			
PLADJ	texture	-0.694	-0.463	0.387	
PRD	diversity	0.519		-0.453	
ECON_MN	**shape**		**0.928**		
SIMI_CV	isolation		0.619		
SIMI_MN	isolation	-0.429	-0.539	0.392	0.492
FRAC_MN	**shape**			**0.894**	
GYRATE_MN	Area		-0.581	0.658	
SHAPE_AM	**shape**				**0.950**
AREA_CV	Area	-0.546			**0.765**

Class Level

Metrics	**Group**	**Factor 1**	**2**	**3**	**4**	**5**
Eigenvalue		3.170	2.357	2.233	1.626	1.325
% of variance explained		22.642	16.838	15.952	11.615	9.463
% of cumulative variance explained		22.642	39.480	55.432	67.048	76.511
PO_PLADJ	**texture**	**0.963**				
PO_PLAND	area	**0.897**		0.385		
PO_NLSI	shape	**-0.841**		0.335		
PO_PROX_MN	isolation	0.679		0.418	-0.362	
OA_PLAND	**area**		**-0.826**			
PI_ED	shape		**0.782**		0.320	
OP_COHESION	isolation		-0.668			
PI_AREA_MN	area		0.646	-0.548		
PO_SIMI_MN	**isolation**			**-0.770**		
PO_CWED	shape	0.340		**0.737**		
PO_IJI	**Texture**				**0.722**	
OA_CIRCLE_MN	Shape				0.623	
OP_CIRCLE_MN	**Shape**					**0.866**
OP_ENN_MN	Isolation			-0.387	-0.395	0.679

Bold *metrics are chosen as representative for the corresponding factors,* ***bold*** *numbers indicate factor loadings > |0.7|, loadings < |0.3| are not presented*

**Due to limitation in space, the tables concerning the analysis per group are not presented here, but they can be obtained from the corresponding author.*

Statistical dimensions of landscape structure

At landscape level, none of the metrics included in the overall analysis was redundant. We found four statistical dimensions of landscape structure, which explained 78 % of the variance of the 13 metrics included (Table B.1.2). They were labeled: diversity of habitats, fragmentation, mean patch fractal dimension and area-weighted mean patch shape, respectively. The first factor was characterized by a high negative loading of PROX_MN and high positive loadings of SIDI, CIRCLE_AM and IJI. It described a gradient from areas with few, dominating and clustered habitat classes towards areas with high diversity, high interspersion and a large amount of area covered by elongated patches. The second factor was characterized by a high positive loading of ECON_MN, obtaining high values for hexagons with high edge contrast, thus very fragmented areas. The third factor was characterized by a high loading of FRAC_MN, obtaining the highest values for hexagons with many irregular shaped patches, while the fourth factor was determined by high positive loadings of SHAPE_AM and AREA_CV, indicating a gradient from areas with regular patches towards those with large variation in patch size and a large amount of area covered by very irregularly shaped patches.

To provide a visual impression of the emerged factors and to demonstrate the differences between the gradients they represent, we inspected hexagons with very high and very low factor scores (Figure B.1.3). As expected, landscape mosaics with a high value for habitat diversity contained many land cover classes of even distribution and little variation in patch size, whereas highly fragmented forest areas were characterized by the additional occurrence of non-forest habitats like openings, fields and roads. When comparing hexagons with low values for the factors three and four (mean patch fractal dimension vs. area-weighted mean patch shape), it is obvious, that the decreasing importance of area is related with a high number of small regular shaped patches.

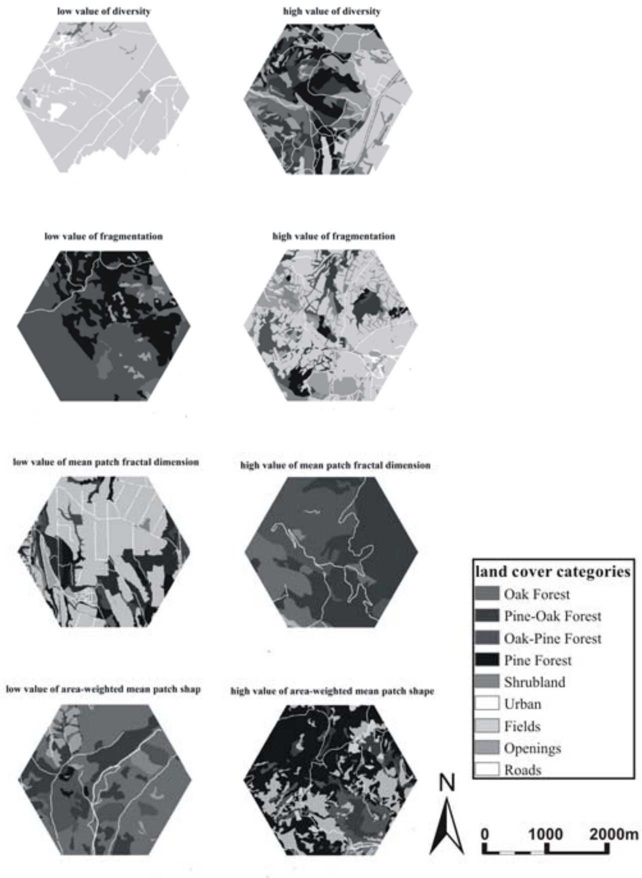

Figure B.1.3. *Resulting landscape level gradients of landscape structure, described by characteristic 500 ha hexagons.*

Regarding the overall analysis at class level OA_DIVISION and PI_PLAND were redundant with OA_PLAND and PI_AREA_MN, respectively, and excluded from the analysis. The five retained factors of the overall analysis explained 77% of the variance of the remaining 14 metrics (Table B.1.2). The factors were labeled PO dominance, OA - PI gradient, PO fragmentation, forest interspersion and OP patch elongation.

Evaluation of the stability of the detected dimensions across maps of different extents

The retrieved factors were remarkably stable across hexagons of different extents (Table B.1.3), when comparing them by applying coefficients of congruence (hereafter CoC – the measure approaches an absolute value of |1| when the loadings are proportional). At landscape level, the factors 1 & 2, concerning habitat diversity and fragmentation, obtained specifically high values (CoC: range |0.87| - |0.97|), and factor

4 of the 500 ha scale emerged clearly as the third factor in the FAs of the scales of 250 and 1000 ha (CoC: range |0.89| - |0.96|). Only factor 3 of the 500 ha scale was not stable. This factor was moderately correlated with different factors at the other scales. At class level, the result was analogous, with very stable factors 1 and 2 (CoC: |0.85| and |0.95|, respectively) and lesser congruence among the factors 3, 4 and 5 (Table B.1.3).

Table B.1.3. *Coefficients of Congruence for the combinations obtained from hexagons of 250, 500 and 1000 ha at landscape level (LAND), and of 500 and 1000 ha at class level (CLASS). Note that the measure approaches a value of one, when the loadings are proportional, and that the absolute value (not the sign) of the congruence statistic is important for the comparison.*

	factors	**1**	**2**	**3**	**4**	**5**
LAND						500 ha
1000 ha	1	**-0.97**	-0.23	0.35	0.48	
	2	-0.27	**-0.87**	**0.81**		
	3	-0.51			**0.96**	
	4	0.27	0.56	**-0.64**		
LAND						500 ha
250 ha	1	**-0.92**	-0.33	**0.62**	0.33	
	2		**0.95**	-0.51		
	3	-0.34	0.28	-0.28	**0.93**	
	4	-0.59	-0.41	0.52		
LAND						250 ha
1000 ha	1	**0.86**		0.39	**0.66**	
	2	0.49	**-0.80**	-0.40	0.51	
	3	0.43		**0.89**		
	4	-0.48	**0.61**	0.24		
CLASS						500 ha
1000 ha	1	**0.85**	-0.30		0.28	
	2		**0.91**		0.43	
	3			**-0.60**	-0.30	**0.66**
	4	0.59		**0.71**	0.27	
	5				**0.67**	0.48

Bold *numbers indicate values ≥|0.6|, values < |0.2| are not presented*

Sets of metrics for landscape monitoring

Regarding the overall landscape level analysis, the metrics SIDI, ECON_MN, FRAC_MN and SHAPE_AM contributed with the highest loadings on the four factors representing the dimensions of landscape structure (Table B.1.2). In similar way, at class level, the metrics PO_PLADJ, OA_PLAND, PO_SIMI_MN, PO_IJI, and OP_CIRCLE_MN contributed with the highest loadings on the five emerged class level factors (Table B.1.2). In this set, metrics concerning the three land cover types PO, OA, and OP were included, while metrics regarding pure pine forest (PI) became rejected during the data reduction analysis. These nine metrics were the optimal surrogate of the nine factors, including a maximum of the information provided by the other metrics, and forming a core set of structural features for landscape monitoring.

Description of the patterns of landscape structure

When mapping the factor scores at landscape level (Figure B.1.4), the first factor, diversity of habitats, resulted in a dispersed pattern with highest values around the borders of the strictly protected areas. The pattern of the second factor, concerning fragmentation, was clustered and the differences between neighboring hexagons were on average smaller than for the first factor. Highest values of the second factor occurred in the eastern part of the study area, indicating a higher level of fragmentation than in the western part and in the strictly protected areas. Regarding the third factor, mean patch fractal dimension, the pattern was homogeneous and gradients were slighter than for the other factors. The pattern of the forth factor, area-weighted mean patch shape, was again clustered with lowest values for the western part of the study area (Figure B.1.4). At class level different patterns were observed, (Figure B.1.5), as the first four metrics were clustered, while the pattern of the fifth metric, OP_CIRCLE_MN, was homogenous. Clusters of high values in the center of the park and in two small areas in the periphery characterized the pattern of the metric PO_PLADJ, representing the first factor. Highest values of the second metric, OA_PLAND, occurred in the periphery of the park, while PO_SIMI_MN, the third metric, obtained clusters of high values in the southwest and in the strictly protected areas. PO_IJI, the forth metric, obtained clusters of high values around and inside the strictly protected areas of Dadia NP (Figure B.1.5).

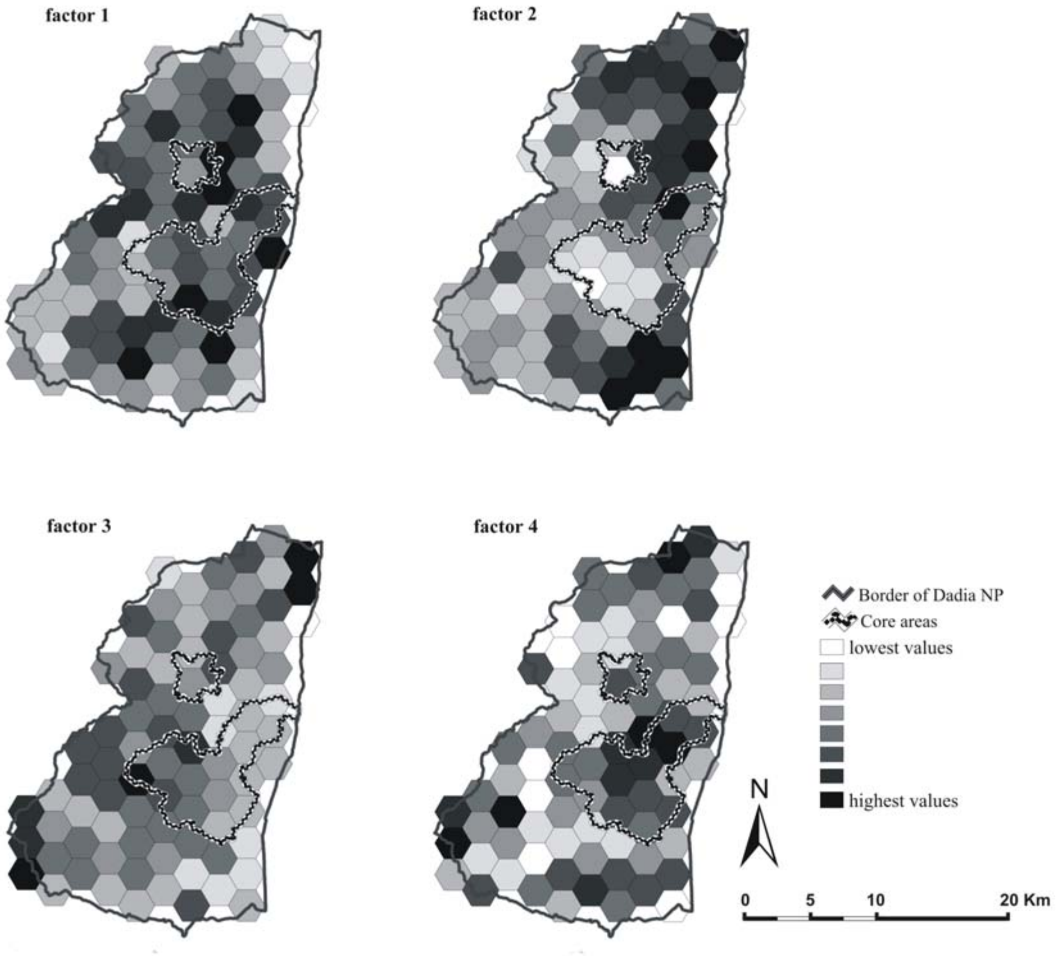

Figure B.1.4. *Landscape level patterns of landscape structure. The maps present the factor scores of each hexagon for the four factors diversity, fragmentation, mean patch fractal dimension, and area-weighted mean patch shape.*

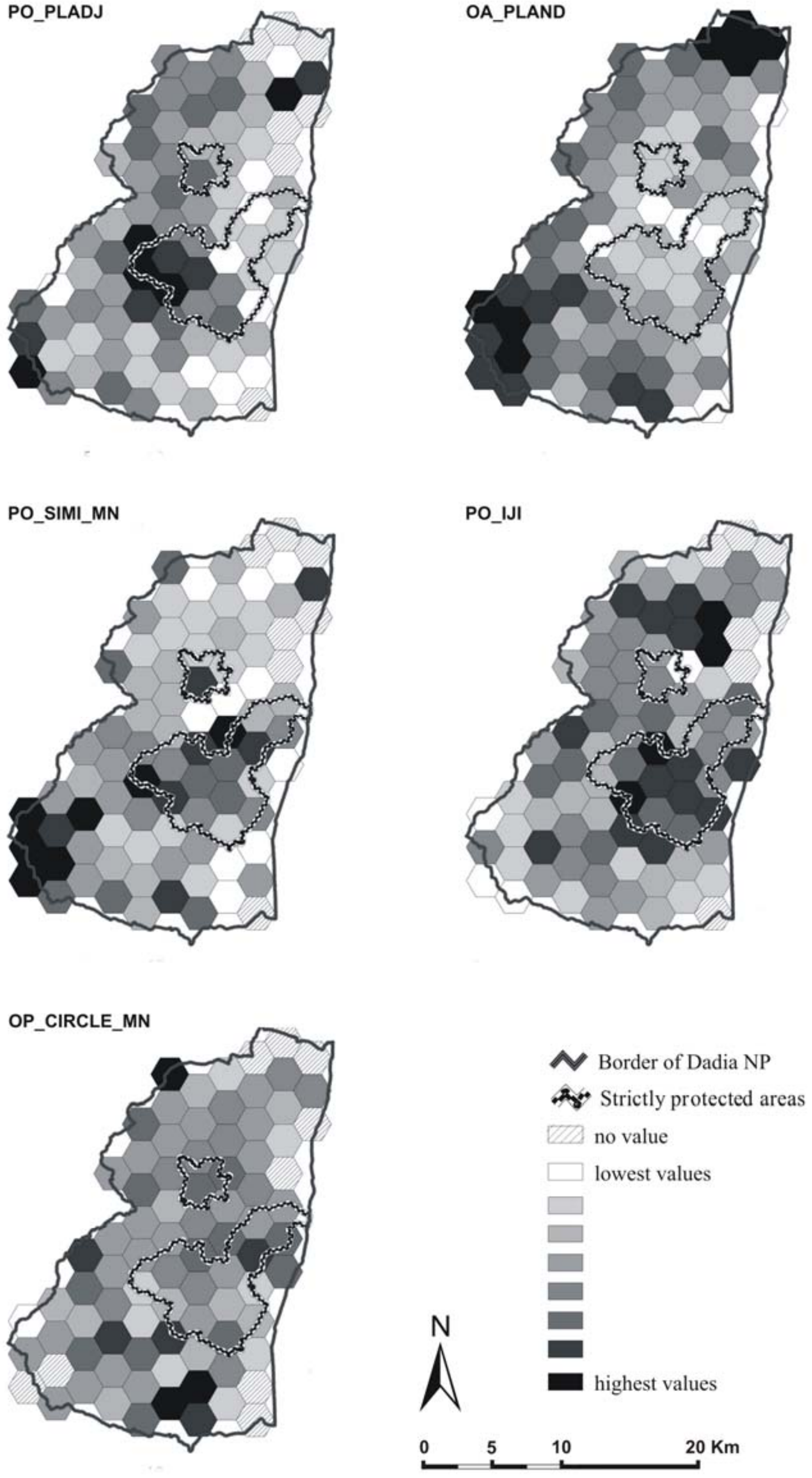

Figure B.1.5. *Class level pattern of landscape structure. The maps present the scores of the five metrics PO_PLADJ, OA_PLAND, PO_SIMI_MN, PO_IJI, and OP_CIRCLE_MN, which represent the five retained factors. Note that the 2nd and 3rd factors have high loadings of "–OA_PLAND" and"–PO_SIMI_MN" (Table B.1.2), thus the pattern of these factors is reverse to the pattern of the representing metrics.*

Discussion

Dimensions and patterns of landscape structure at landscape level

The total amount of variance explained by the overall analysis at landscape level was very similar to the variance explained by the first four factors of similar studies in other ecosystems (Riitters et al. 1995; Cain et al. 1997; Tinker et al. 1998; Griffith et al. 2000; Cifaldi et al. 2004). Other researchers retain in addition a fifth or sixth factor, but in most cases these factors do either explain little variance (Riitters et al. 1995; Cain et al. 1997), or are related to class level attributes (Griffith et al. 2000; Cifaldi et al. 2004).

According to previous research in different mosaics of temperate and boreal biomes, the most important dimensions of spatial structure at landscape level are usually related to diversity/aggregation of landcover categories and patch shape aspects (Riitters et al. 1995; Cain et al. 1997; Griffith et al. 2000; Cifaldi et al. 2004). In our study additionally fragmentation was an important and stable factor. Our results indicate the importance and the independence of the aspects diversity of habitats and fragmentation in a Mediterranean forest like Dadia NP. Although in some parts of the study area, both factors coincide, in other areas high diversity of habitats coincides with low fragmentation. Areas with a high level of habitat diversity were located mainly where different forest types were mixed with openings and fields, like around the borders of the core areas of the National Park (Figure B.1.4). The lowest values of habitat diversity were caused by dominance of agricultural areas in the northeast and southeast and of oak forests close to the northern and southwestern border. The first factor was determined very well by the pair of metrics SIDI and PROX_MN, being measures of diversity and dominance. The high positive loadings of CIRCLE_AM and IJI on this factor indicate that high diversity of habitat is related in our study area to elongated patches and a high interspersion and juxtaposition of landcover categories. The four metrics defining this factor were obtained from the four different groups, diversity, isolation, patch shape, and texture. Because in factor analysis, the composition and order of the emerged factors is a result of the number of indicators that are included in the analysis (Cain et al. 1997), this factor could probably be encountered in this composition only by reducing the large amount of metrics that measure very similar values during the data reduction process per group.

As outlined above, new insight could be gained by this study as two contrast-weighted structural attributes - edge contrast and Similarity Index – determined an important and stable factor. However, as the related metrics have been scarcely used by other researchers so far, we could not evaluate, if this is a specific characteristic for Mediterranean fine grained landscapes or should be regarded as factor of general importance. Edge contrast was included at class level in the analysis of Griffith et al. (2000), who recommend further studies, and it is supposed to be important for quantifying fragmentation and thus to distinguish between fragmented and undisturbed landscapes (McGarigal & McComb 1995). A similar approach, using edge contrast metrics at landscape level, has been presented by Tinker et al. (1998) for forest-dominated landscapes in Wyoming. Since a large and dominating set of core area metrics was included in this study, comparisons with our findings and general conclusions are difficult. Cifaldi et al. (2004), analyzing the dimensions of landscape structure of two watersheds of Michigan, detected one factor strongly related to fragmentation, and highest values occurred where agricultural and natural land was converted to urban. But using only four land cover categories, and excluding contrast metrics, it was not possible to differentiate between heterogeneity of habitats and fragmentation. Neither Hargis et al. (1998), testing the behavior of six metrics with artificially generated landscapes, could detect a good measure of fragmentation. Hargis et al. (1998) also recommend the use of metrics concerning interpatch distances, which we added to the commonly used sets of metrics (e.g. Riitters et al. 1995; Cain et al. 1997; Lausch & Herzog 2002; Cifaldi et al. 2004). Out of these variables, PROX_MN obtained a high loading on the first factor, SIMI_MN and SIMI_CV formed the contrast weighted character of the fragmentation factor, while the nearest neighbor metrics became rejected during the data reduction analysis per group. However, including the contrast metrics, our evaluated dimensions have come closer to the five attributes, Li & Reynolds (1994) identified based on theoretical considerations: (a) number of cover types, (b) proportion of each type, (c) spatial arrangement of patches, (d) patch shape, and (e) patch contrast.

The choice of appropriate scales is fundamental in landscape analysis (Gustafson 1998; Meisel & Turner 1998; Turner et al. 2001; Wu et al. 2002). Due to the strong influence of scale on the behavior of landscape metrics (e.g. Baldwin et al. 2004; Wu 2004), landscape pattern should be analyzed at a local scale when applied for local land management and conservation (Cifaldi et al. 2004). In this study the use of fine grain data permitted us to quantify the landscape structure of the diverse mosaic of habitats. The high stability of the factors habitat diversity, fragmentation and area-weighted mean patch shape across the three scales proved that our samples sizes have been large enough to reduce the effects of the map boundaries on the values of the metrics and indicated that our results could be applicable for a wider range of conditions. Also Cain et al. (1997) detected that the stability of the six factors that emerged in their study, decreased from the first to the last when analyzing maps of different resolution, numbers of attributes, and methods of delineating landscape unit boundaries. Their second and third factors were still relatively stable in composition, but the remaining three factors were very unsteady.

Statistical dimensions of landscape structure at class level

The five class level factors are not directly comparable with factors other researchers detected, because of differences in land cover categories, the area under concern and in the applied methodologies. Griffith et al. (2000), for instance, analyzed the landscape structure of Kansas (USA), using class level metrics for grassland and cropland and performing a mixed data reduction analyses including both, class and landscape level metrics. McGarigal & McComb (1995) and Tinker et al. (1998) performed class level analyses for several forest types separately, thus, factors presenting the gradient from one type to another could not emerge. In our study, the emerged factors describe gradients related to class attributes. They explain a high proportion of the variance of the class level metrics, provide additional information to the dimensions at landscape level, and resulted in different pattern when mapped. We included class level metrics only for the forest land cover categories PI, PO, OP, and OA, because these categories appeared in most of the hexagons and formed the matrix of the study area. Class level metrics regarding the interspersed forest types were often related, and as a result metrics of all four categories contributed with important loadings on the overall factors.

Sets of metrics for landscape monitoring

It is proposed to develop a suite of metrics that measure the fundamental dimensions of landscape structure and can be applied for a landscape monitoring (Riitters et al. 1995; Botequilha Leitão & Ahern 2002). Single metrics as surrogates of the factors have the advantage that they simplify the mental model and facilitate comparisons among different sets of maps. The simplest rule for the choice is the single metric with the highest absolute loading on each factor, being especially reasonable when the metric has a high loading for only that factor (Riitters et al. 1995). In this study the four highest loading metrics at landscape level fulfill the criteria and are proposed as a core set of variables for a landscape monitoring. At class level the representative metrics of the overall analysis also fulfill the criteria, but obtained on average lower loadings. Although it is more difficult to obtain general conclusions at class level, our results indicate that a core set of metrics for a monitoring of the landscape structure of Dadia NP or a similar forest should contain class level metrics concerning (1) the amount of mixed forest types, (2) the gradient from one pure forest type to another, (3) the quantification of the fragmentation of a mixed forest type, (4) the interspersion of the forest types, and (5) the patch shape of a mixed forest type.

It is remarkable that also Botequilha Leitão & Ahern (2002), reviewing previous works that studied dimensions of landscape structure and core sets of metrics (Li & Reynolds 1994; McGarigal & McComb 1995; Riitters et al. 1995; Hargis et al. 1998; Tinker et al. 1998), proposed a core set of nine landscape and class level metrics to address the principal needs of applied landscape structure analyses. They also included edge contrast in the core set and coincide with our study in totally five of the nine cases. However, we recommend to evaluate the importance of the detected dimensions of landscape structure across other landscape mosaics and to consider the evaluated sets of metrics for landscape monitoring and assessments of the effects of landscape structure on Mediterranean biodiversity.

Implications for management and conservation

In order to improve conservation management of Dadia NP, a monitoring plan has been established, mainly focusing on the assemblage of birds of prey (Poirazidis et al. 2002). Birds of prey seem to be good indicators of biodiversity (Sergio et al. 2005), and it is likely that the high abundance and diversity of birds of prey in Dadia NP is related to characteristics of landscape structure. However, the relation of landscape structure and biodiversity must be assessed yet for our study area, where the strictly protected areas, delineated to protect the Black Vulture breeding colonies, are dominated by pine and mixed forest, while the surrounding parts of the managed buffer zone are characterized by the highest diversity of habitats (see Figure B.1.4). As these parts of the buffer zone are of particular interest, because they host a great number of different taxa of flora and fauna (e.g. Grill & Cleary 2003; Kati et al. 2004b; Kati & Sekercioglu 2006), changes in composition and configuration must be monitored and effects of land use and management on landscape structure must be analyzed. This knowledge can then be used to achieve better conditions in the impoverished parts of the park, to assess progress in conservation efforts, and to improve management decisions not only in Dadia NP, but also in similar landscape mosaics and other Mediterranean forests.

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Chapter B.2. Landscape metrics as biodiversity indicators for plants, insects and vertebrates at multiple scales

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Runnung title: Landscape metrics as biodiversity indicators

Firesalamander (Salamandra salamandra) – photo by Alexandors Gassios

Own contribution:

Study design 80%, implementation 75%, writing 80%

Abstract

Landscape metrics are widely used to investigate spatial structure and pattern of landscapes. Numerous metrics are available, yet only little empirical research examined their indicator value for species richness pattern. In this study we explored the relations of 52 standard landscape metrics with the species richness patterns of six taxa (woody plants, orchids, Orthoptera, amphibians, reptiles, and birds) and overall biodiversity, taking as a case study a Mediterranean forest landscape - Dadia National Park in Greece. We computed landscape structure variables for circular areas of five different extents (hereafter "scales") around the sampling plots. Additionally, we tested the performance of sets of three landscape structure variables as predictors of species richness, comparing sets composed by different methods such as expert knowledge, ordination methods, decision trees and random choice. We also evaluated optimal sets, testing all possible combinations of three variables.

Landscape metrics were good indicators of overall biodiversity, and of the species richness of woody plants, Orthoptera and reptiles. Metrics quantifying patch shape, proximity, texture and diversity resulted in numerous significant univariate models, while metrics describing patch area, similarity and edge rarely contributed to significant models. Scale affected the performance of the metrics. Woody plants, Orthoptera and birds were usually better predicted at smaller scales, and reptiles frequently at larger scales. Regarding the different methods of composing sets, optimal sets always performed significantly better than all other methods. Among these, expert knowledge performed even slightly worse than random, while the statistical methods performed slightly better. The revealed pattern of relations and performances will be useful to understand landscape structure as driver and indicator of biodiversity, and to improve management decisions in Mediterranean forests and other mosaic-landscapes.

Introduction

Land use change and fragmentation are widely considered as important threats to biodiversity (Tilman et al. 1994). Landscape structure has an important influence on a wide range of ecological patterns and processes and landscape metrics are common tools to assess these relations (Turner et al. 2001). Their use increased over the last decade as remote sensing and GIS became standard data sources within biogeography and biodiversity research (Gaston 2000; Gillespie et al. 2008). Numerous metrics are available (McGarigal & Marks 1995), yet only little empirical research examined their indicator value for biodiversity. Biodiversity indicators are essential tools for ecological research, environmental NGOs, and national and regional agencies for nature conservation, forestry and agriculture, but a consensus regarding their use has not been reached (Duelli & Obrist 2003). A critical factor within landscape structuring is the examined scale, or grain size, thematic resolution and extent (Turner et al. 2001; Wu 2004). While the response of landscape metrics to grain and thematic resolution behaves rather consistently, their response to changing extent (i.e. the map size) does not (Wu 2004). Having uncovered recently the major components of landscape structure and the landscape pattern of the Mediterranean forest mosaic of Dadia National Park in Greece (Schindler et al. 2008 [= Chapter B.1 of this thesis]), we analyzed in this research a) the relations of 52 landscape metrics to overall biodiversity and to the species richness of the six taxa woody plants, orchids, Orthoptera, amphibians, reptiles and birds, b) the effect of the extent of the landscape plots on these relations, and c) the performance of different methods such as expert knowledge, ordination methods, decision trees and random choice to compose sets of predictors.

Methods

Study area, focal species and land cover data set

Our case study area, the Dadia National Park has an extent of about 430 km^2 and is located in north-eastern Greece. The mountainous area is dominated by extensive pine and oak forest, but it contains also a variety of other habitats such as pastures, cultivated land, torrents and stony hills. Dadia NP is a well known local biodiversity hotspot for many taxa (e.g. Kati et al. 2004a,b). For this study we used a species data set obtained from Kati et al. (2004b) at 30 sampling sites that were selected by random sampling. The six taxa woody plants, orchids, Orthoptera, amphibians, reptiles, and small terrestrial birds had been surveyed applying sampling techniques appropriate for each group under study (Kati et al. 2004b). Satellite images (IKONOS, July 2001, pixel size 1 m in the panchromatic channel and 4 m in the multispectrum) of the study area were digitized and used to produce a raster map with a grain of 5 m and a thematic resolution of nine land cover categories, namely: oak forest, pine forest, pine-oak forest, oak-pine forest, broadleaves, openings, fields, roads, and urban areas (Schindler et al. 2008 [= Chapter B.1 of this thesis]). We clipped the surrounding areas of 20, 50, 100, 200 and 500 ha of each sampling plot of organisms and computed for all these areas 52 landscape level variables of landscape structure using the software FRAGSTATS (McGarigal & Marks 1995). We used R to perform the statistical analyses described in the following paragraphs.

Univariate linear mixed models

We tested the indicator value of each individual landscape variable at each scale by using it as predictor to model the species richness of each taxa. Therefore we assigned the sampling plots to the five categorical habitat types forest, shrubs, heather, grassland and agricultural fields, excluded three plots representing mixed habitats, created linear mixed models with the categorical habitat type in the models as a random factor, and tested for significance. In a further attempt we modeled the overall biodiversity of the sampling plots. Therefore, to adequately represent species poor taxa, we used the sum of the relative species richness as proxy. The relative species richness was defined for each taxon as the number of species of a plot divided by the maximum number of species across all the sampling plots. Further we grouped the landscape structure variables into the six categories area, shape, isolation, contrast, texture and diversity, and evaluated for each taxon the number of categories containing significant variables across the scales.

Testing the performance of different methods to compose sets of metrics

In order to test the performance of different methods of composing sets of landscape metrics as indictors of species richness, we compared sets composed by A) random choice, B) expert knowledge, C) decision trees, D) ordination methods, E) PCA axes as predictors instead of the original variables, and F) the optimal set of predictors. The performance of the different methods of composing sets was compared by the Akaike Information Criterion (AIC) of the models. We used the AIC values as input in an ANOVA and applied Tukey post hoc tests and boxplots for further comparison. Regarding the optimal sets we calculated the Variance Inflation Factor (VIF) to detect and reject sets with extremely correlated predictors (VIF > 5.0) that produced erroneous AIC values.

Results

Landscape metrics contributed significant models of woody plants, Orthoptera, reptiles, birds and overall biodiversity, while virtually no significant relations were detected among the metrics and the species richness of orchids and amphibians. Landscape metrics quantifying patch shape, proximity, texture, diversity and patch size were often significant predictors within univariate models, while metrics regarding similarity or contrast of neighbouring patches hardly yielded any significant model. Regarding the distribution statistics, the area-weighted mean regularly outperformed the mean and the coefficient of variation of the variables.

Scale affected the number of landscape metrics, which were significantly related to species richness. Orthoptera and birds were better predicted by landscape metrics at the smaller scales of 20-50 ha and woody plants and overall biodiversity at 20-200 ha, while the performance was stable across scales for reptiles (Figure B.2.1).

Regarding the comparison of the six methods, the optimal sets performed always much better than the rest of the methods ($p < 0.01$). Expert sets performed worse than random (significantly, with $p < 0.01$ in the cases of Orthoptera and birds), and statistically obtained sets slightly but insignificantly better than random.

Implications for conservation management

This study revealed clearly that the heterogeneous landscape mosaics of fine texture are important regarding the maintenance of biodiversity in a seminatural Mediterranean forest ecosystem. Similar results were obtained for Italy and Spain (e.g. Torras et al. 2008), and it has to be supposed that they are valid for most parts of the Mediterranean basin. In Dadia National Park, land abandonment and homogenization of landscape already took place, and some decades ago the level of mosaic structure was clearly higher (Triantakonstantis et al. 2006). Kati et al. (2004c) suggested the maintenance of forest openings in the buffer zone, the maintenance of forest heterogeneity, and the enhancement of periodical livestock grazing. These proposed measures are clearly supported by our results. Also, social and political measures, e.g. against land abandonment, could help the maintenance of biodiversity, if they are targeted thoroughly (Wrbka et al. 2008).

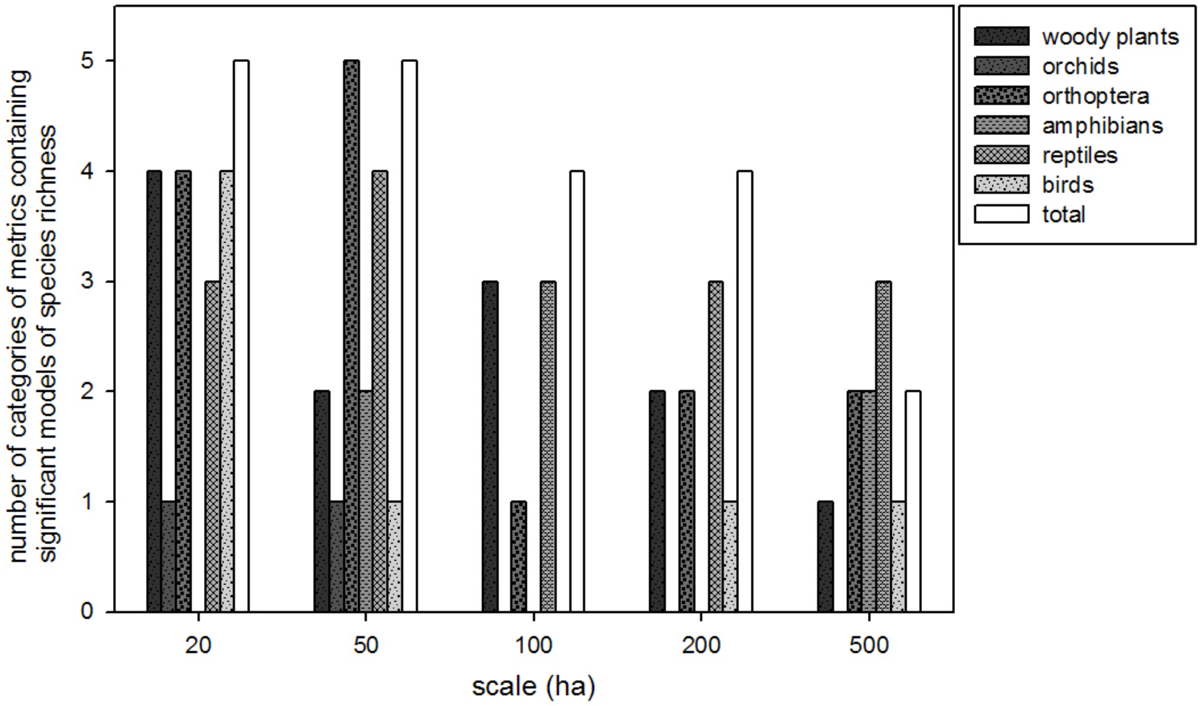

Figure B.2.1. *Relations between landscape structure, organism groups and scale expressed by the number of categories of univariate models (out of the six categories "area", "shape", "contrast", "isolation", "texture", and "diversity") containing at least one model that relates significantly a landscape metric with species richness. While reptiles were still predictable considering the surrounding 500 ha, for the other taxa the performance of landscape metrics as indicators of species richness declined clearly at the larger scales.*

Chapter B.3. Multiscale performance of landscape metrics as indicators of species richness of plants, insects and vertebrates

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Running title: Multiscale performance of landscape metrics as indicators of species richness

Keywords: biodiversity indicators, landscape structure, landscape heterogeneity, spatial extent, Dadia National Park, FRAGSTATS.

Kestrel (Falco tinnunculus) with reptilian prey – Photo by Alexandors Gassios

Own contribution:

Study design 80%, implementation 75%, writing 80%

Abstract

Landscape metrics are widely used to investigate the spatial structure of landscapes. Numerous metrics are currently available, yet only little empirical research has examined their indicator value for species richness patterns. Taking a Mediterranean forest landscape - Dadia National Park (Greece) - as a case study area, we explored the performance of 52 landscape level landscape metrics as indicators of species richness for six taxa (woody plants, orchids, orthopterans, amphibians, reptiles, and small terrestrial birds) and for overall species richness. We computed the landscape metrics for circular areas of five different extents around each of 30 sampling plots. We applied univariate linear mixed models to evaluate significant relations between metrics and species richness and to assess the effects of the extent of the considered landscape on the performance of the metrics. Additionally, we computed random sets of two to five metrics and tested for their parsimony.

Our results showed that landscape metrics were particularly good indicators of overall species richness, and of the species richness of woody plants, orthopterans and reptiles. Metrics quantifying patch shape, proximity, texture and diversity resulted in numerous significant univariate models, while metrics describing patch area, similarity and edge contrast rarely contributed to significant models. Spatial scale affected the performance of the metrics, since woody plants, orthopterans and small terrestrial birds were usually better predicted at smaller extents of surrounding landscape, and reptiles frequently at larger ones. Sets of metrics with a higher numbers of metrics performed better when using as criterion the best performing set, while they performed worse when the criterion was the average performing set. The revealed pattern of relations and performances will be useful to understand landscape structure as a driver and indicator of biodiversity, and to improve forest and landscape management decisions in Mediterranean and other forest mosaics.

Introduction

Landscape structure has an important influence on a wide range of ecological patterns and processes, and landscape metrics are common tools to assess these relations (Turner et al. 2001). Continously, new landscape metrics have been developed (e.g. McGarigal & Marks 1995; McGarigal et al. 2009; Moser et al. 2002), and their use has increased over the last decade as remote sensing and GIS became standard data sources within biogeography and biodiversity research (Foody 2008; Gillespie et al. 2008). Landscape datasets are obtainable over large areas (Groom et al. 2006). They enable an extrapolation based on a limited set of ground truth data and provide natural resources managers around the world with real-time data to support conservation efforts (Gaston 2000; Gillespie et al. 2008). Uuemaa et al. (2009) recently reviewed the applications of landscape metrics, they are applied in systematic reserve design (Kati et al. 2010 [= Chapter B.4 of this thesis]), evaluation of land use change (Rocchini et al. 2006), species habitat requirements (Quevedo et al. 2006), restoration ecology and landscape planning (Botequilha Leitao & Ahern 2002), sustainability indicators (Peterseil et al. 2004; Renetzeder et al. 2010), or species richness and biodiversity (Hernandez-Stefanoni 2006; Moser et al. 2002; Yamaura et al. 2008). Biodiversity indicators are essential for ecological research, environmental NGOs, and agencies for nature conservation, forestry and agriculture at local, national and international level. Yet a consensus regarding their use has not been reached (Duelli & Obrist 2003), and several crucial terms such as landscape heterogeneity and fragmentation are not well defined (Duelli & Obrist 2003; Fahrig 2003; Tews et al. 2004). Landscape metrics are potentially very useful indicator of biodiversity (Lindenmayer et al. 2002, Moser et al. 2002), but results of studies relating landscape structure to species diversity often differ widely. We are far from having a complete picture about the indicator value of the metrics for species richness, and depending on the applied landscape metrics and the landscape under consideration, patch size and patch shape can be related to species diversity positively, negatively or not at all (Fahrig 2003; Hernandez-Stefanoni 2006; Hill & Curran 2003; Honnay et al. 2003; Moser et al. 2002; Torras et al. 2008; Yakamura et al. 2008).

A critical factor within landscape structuring is the examined scale (Gustafson 1998; Turner 1989; Wu 2004; Wu et al. 2002), characterized by grain size, thematic resolution and extent (Lam & Quatrochi 1992; Turner et al. 2001). While the response of landscape metrics to grain and thematic resolution behaves rather consistently (Bailey et al. 2007; Wu 2004; Wu et al. 2002), their response to changing extent (i.e. the map size) does not (Saura & Martinez-Millan 2001; Wu 2004). Given a patchy landscape with underlying gradients, at small extents unpredictable behavior of metrics can be caused by too little a sample of patches, while at large extents environmentally different patches might be included in the sample. But also the spatial pattern of species richness changes with the scale of observation or analysis (Kallimanis et al. 2008). To discern the important elements of patch structure for a particular organism, an organism-centered view of the landscape must be adopted (Cushman et al. 2008; Li & Wu 2004; Lindenmayer et al. 2002; Turner 1989; Vos 2001). At larger extents landscape structure can influence metapopulation dynamics, and thus, local species richness (Gustafson 1998; Hunter 2002; Vos et al. 2001). According to the concept of ecological neighborhood (Addicott et al. 1987), the effects of extent on the performance of landscape metrics as indicators of species richness should depend on the body size, dispersal abilities and life history traits of the taxa under consideration. It can be expected that taxa with larger space demand and of higher mobility are affected by a wider extent of landscape than those that are small and sedentary.

Single-species conservation and conventional forestry are unlikely to be successful in maintaining the diversity of forest ecosystems, since landscape approaches and a suite of methods and tools are required for holistic management (Carey 2003; Mitchell et al. 2008). Regarding landscape metrics and their use as indicators of species richness, it is difficult to define an optimal set of metrics in advance, not least because only few empirical studies have so far explored their indicator value in a comprehensive way and for more than one taxon at once (Uuemaa et al. 2009). To aid both ecological management and conservation efforts two sets of analyses should be conducted, one describing the major components of landscape structure, and one relating pattern and processes (Cushman et al. 2008; Mitchell et al. 2006). Having recently examined the major components of landscape structure in the Mediterranean forest mosaic of Dadia National Park in Greece (Schindler et al. 2008

[= Chapter B.1 of this thesis]), in this study we analyzed the performance of landscape level landscape metrics as species richness indicators for the same study area. We screened 52 metrics, each for five different extents of landscape, in order to: a) provide an overview of their performance for six taxa, i.e. woody plants, orchids, orthopterans, amphibians, reptiles and small terrestrial birds, and for overall species richness, b) assess the effect of the extent of the landscape plots on these relations, e.g. if taxa with different space demand and mobility are affected by a different extent of landscape, and c) test the effect of the number of metrics on the parsimony of the models.

Methods

Study area, focal species and land cover data set

Our case study area, the Dadia-Lefkimi-Soufli National Park (hereafter Dadia NP) covers 430 km^2 and is located in north-eastern Greece (Figure B.3.1). The area is dominated by extensive pine (*Pinus brutia, P. nigra*) and oak (*Quercus frainetto, Q. cerris, Q. pubescens*) forest, but it also contains a variety of other habitats such as pastures, cultivated land, torrents and stony hills. Dadia NP is a well known local biodiversity hotspot (Catsadorakis & Källander 2010; Grill & Cleary, 2003; Kati et al. 2004a,b; Poirazidis et al. 2007a, 2010a [= Chapter A.1 of this thesis]) and contains the only remaining Black Vulture (*Aegypius monachus*) breeding colony of the Balkan Peninsula (Poirazidis et al. 2004; Skartsi et al. 2008).

We used a data set of six taxonomic groups (189 species), sampled within 30 randomly selected sampling plots of 20 ha or less, which represented the main vegetation types of the study area (Kati el al. 2004b). The species data set consisted of 48 woody plants, 19 orchids including one rare species (Kati et al. 2000), 38 orthopterans including one endemic species (Kati et al. 2004c), 18 amphibians and reptiles including 5 protected species that are listed under Annex II of Dir 92/43 EE (Kati et al. 2007), and 66 small terrestrial birds, includig 23 species of European conservation concern (SPEC 2 & 3) (Kati & Sekercioglu 2006). With the aid of a previously performed supervised classification of vegetation types, satellite images (IKONOS, July 2001, pixel size 1 m in the panchromatic channel and 4 m in the multispectrum) of the study area were digitized on screen, applying the classification criteria: vegetation type, percentage of cover, and pattern of forest mixture (clustered or random). The resulting vector map was verified with 120 random points and no error was detected. For landscape structure analyses, the vector map was transferred to a raster map with a grain of 5 m and a thematic resolution of nine land cover categories (Figure B.3.1): oak forest, pine forest, pine-oak forest, oak-pine forest, broadleaves, openings, fields, roads, and urban areas (Schindler et al. 2008 [= Chapter B.1 of this thesis]). For this study, we clipped circular areas of 20, 50, 100, 200 and 500 ha around the centroid of each sampling plot (Figure B.3.1) and computed landscape level variables of landscape structure for each of these extents using the software FRAGSTATS (McGarigal & Marks, 1995). Following Cushman et al. (2008) and Schindler et al. (2008), we selected 52 metrics in total, and kept Mean, Area-Weighted Mean and Coefficient of Variation of the Distribution Statistics (Table B.3.1).

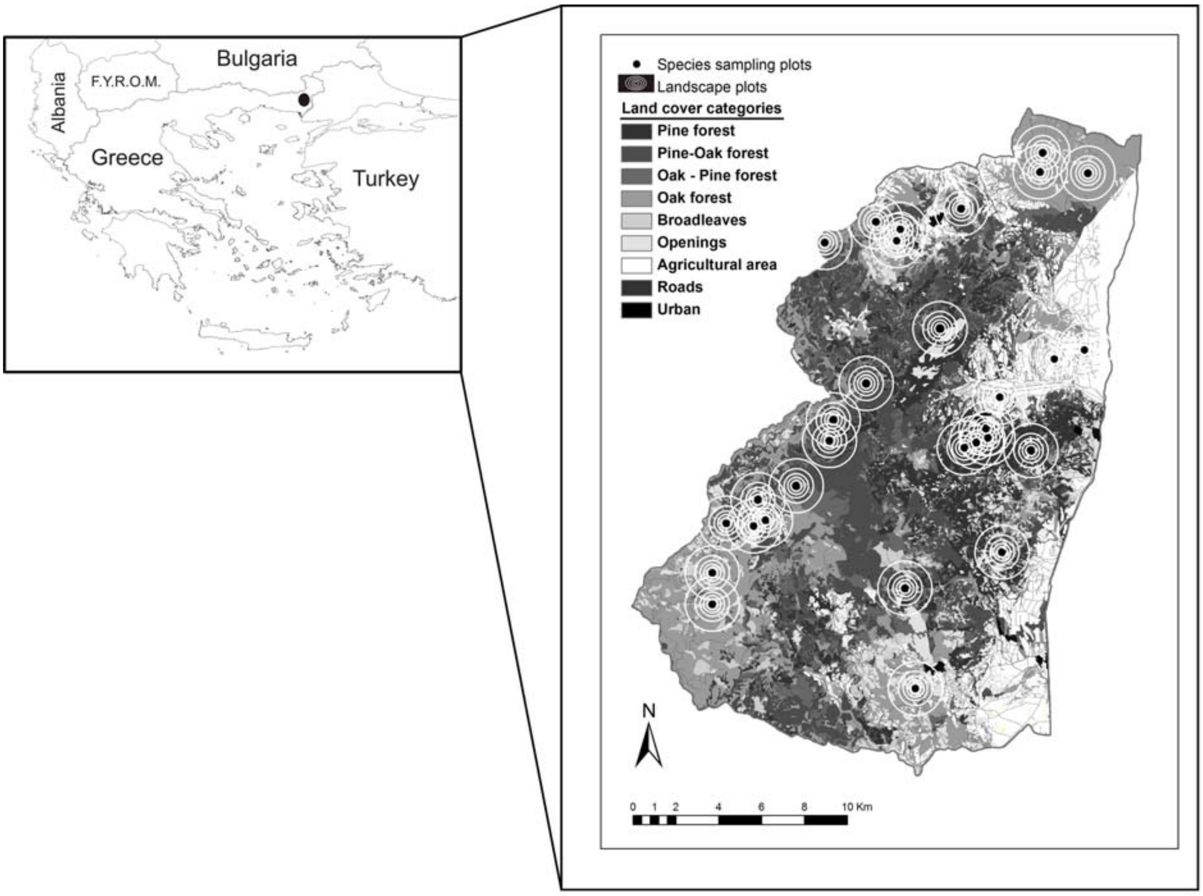

Figure B.3.1. *Dadia National Park, located in NE Greece and raster map of nine land cover categories derived from IKONOS satellite imagery. The circular areas of 20, 50, 100, 200 & 500 ha are shown for each of the 30 sampling plots.*

Table B.3.1. 52 landscape level landscape metrics tested in this study. Regarding the distribution statistics (DSt) (McGarigal and Marks, 1995), we used Mean (MN), Area Weighted Mean (AM) and Coefficient of Variation (CV).

Category	Acronym	Metric name	Number	Description
Category I. Patch size & patch density			***8***	
	AREA	Patch Area	3	DSt; size of the patches
	GYRATE	Radius of Gyration	3	DSt; radius of gyration, i.e. the mean distance for each cell of one patch to the patch centroid
	PD	Patch Density	1	Number of patches per area
	LPI	Largest Patch Index	1	Percentage of total area occupied by the largest patch
Category II. Edge & patch shape			***15***	
	LSI	Landscape shape index	1	Ratio of the total edge to the minimum total edge
	ED	Edge Density	1	Total length of edge per unit area
	SHAPE	Shape Index	3	DSt; equals 1 when all patches are circular; increases with complexity of patch shapes; independent of patch size
	PARA	Perimeter-area ratio	3	DSt; patch shape complexity measure that measures perimeter per area
	FRAC	Fractal Dimension Index	3	DSt; patch shape complexity measure that approaches 1 for simple shapes and 2 for complex shapes
	CONTIG	Contiguity Index	3	DSt; equals 0 for a one-pixel patch and approaches 1 as patch contiguity, or connectedness increases
	PAFRAC	Perimeter-Area Fractal Dimension	1	Patch shape complexity measure, which approaches 1 for shapes with simple perimeters and 2 for complex shapes
Category III. Edge contrast			***5***	
	CWED	Contrast-Weighted Edge Density	1	Total amount of edge per area, weighted by the contrast between the different land cover types
	TECI	Total Edge Contrast Index	1	Ratio of the contrast weighted total length of edge to the not-contrast weighted total length of edge per grid
	ECON	Edge Contrast Index	3	DSt; ratio of the contrast weighted to the not-contrast weighted edge length per patch
Category IV. Isolation, proximity & similarity			***9***	
	PROX	Proximity Index	3	DSt; considers size and proximity of all patches with the same land cover type inside a specified search radius
	SIMI	Similarity Index	3	DSt; considers size and proximity of patches within a search radius, weighted by their similarity to the focal patch
	ENN	Euclidean Nearest Neighbor Distance	3	DSt; minimum edge to edge distance to the nearest neighboring patch of the same type
Category V. Texture			***6***	
	CONTAG	Contagion Index	1	Measure of the aggregation of the land cover classes
	PLADJ	Percentage of Like Adjacencies	1	Percentage of neighboring pixel, being the same land cover class, based on double-count method
	AI	Aggregation Index	1	Percentage of neighboring pixel, being the same land cover class, based on single-count method
	IJI	Interspersion & Juxtaposition Ind. (%)	1	Measure of evenness of patch adjacencies, equals 100 for even and approaches 0 for uneven adjacencies
	DIVISION	Landscape Division Ind. (Proportion)	1	Equals the probability that 2 randomly chosen pixels in the landscape are not situated in the same patch
	SPLIT	Splitting Index	1	Equals the number of patches of a landscape divided into equal sizes keeping landscape division constant
Category VI. Diversity			***9***	
	PR	Patch Richness	1	Equals the number of patch types
	PRD	Patch Richness Density (no./100 ha)	1	Equals the number of patch types (i.e. land cover categories) per 100 ha
	RPR	Relative Patch Richness	1	Percentage of present patch types out of all categories
	SIDI	Simpson's Diversity Index	1	Diversity measure, which equals 1 minus the sum of the squared proportional abundance of each patch type
	SHDI	Shannon's Diversity Index	1	Equals minus the sum of the proportional abundance of each patch type multiplied by the ln of that proportion
	MSIDI	Modified Simpson's Diversity Index	1	Diversity measure, which equals minus the ln of the sum of the squared proportional abundance of each patch type
	SHEI	Shannon's Evenness Index	1	Diversity measure, which considers only evenness of patch sizes, not the number of patches
	SIEI	Simpson's Evenness Index	1	Diversity measure, which considers only evenness of patch sizes, not the number of patches
	MSIEI	Modified Simpson's Evenness Index	1	Diversity measure, which considers only evenness of patch sizes, not the number of patches
SUM			**52**	

Univariate linear mixed models

We tested the indicator value of each individual landscape metric (n=52) at each considered extent (n=5) by using it as a predictor to model the species richness of each taxon, resulting in a total of 260 models per taxon. For this purpose, we assigned the sampling plots to five categorical habitat types - forest (12 plots), shrubs (4), heather (2), grassland (3) and agricultural fields (6). We excluded three plots representing mixed habitats, and created linear mixed models with the categorical habitat type as the random factor of the models (Crawley 2007). Thus, we could control for the influence of the different habitat types of the sampling plots, which could have masked the effect of the landscape structure. We calculated a pseudo r^2 of the mixed model with a Pearson correlation to compensate for non-normal distribution within the data, and tested for significance. We were aware of the problem of Type I error inflation when testing various hypotheses on the same data set, but as the primary goal of our study was an exploratory screening of the indicator value of each landscape metric, we refrained from correcting the family-wise Type I error rate (Roback & Askins 2005).

Such as we did for each taxon, we also computed 260 univariate models for overall species richness. For this purpose, we computed an index of overall species richness (S_{OV}), using the sum of the taxa's relative species richness as a proxy to adequately represent species-poor taxa. Relative species richness was defined for each taxon as the number of species at a plot $S_{(i,T)}$ divided by the maximum number of species across all 27 sampling plots MAX (S_T).

$$S_{OV} = \sum_{i=1}^{27} S_{(i,T)} / MAX(S_T)$$

In an additional approach, we grouped the landscape structure variables into the six categories area, shape, isolation, contrast, texture and diversity (Schindler et al. 2008 [= Chapter B.1 of this thesis]), and evaluated for each taxon (and overall species richness) and each extent the number of categories containing at least one significant model (cf. Table B.3.3).

Testing the effect of the number of metrics on the parsimony of the models

In another approach we composed sets of metrics to test the effect of the number of metrics on the parsimony of the models. Therefore, we computed for each combination of extent and taxon 200 models with random sets of two, three, four and five metrics and calculated the average and minimum Akaike Information Criterion (AIC). Furthermore we included the AIC values of the univariate linear mixed models in this comparison.

We used R (R Development Core Team 2008) to perform the statistical analyses and Fragstats 3.2 and ArcGIS 9.1 to derive and process the landscape variables.

Results

Landscape metrics resulted in significant models for woody plants, orthopterans, reptiles, small terrestrial birds and overall species richness (Table B.3.2), while virtually no significant relations were detected between the metrics and species richness of orchids (only in two out of 260 models) or amphibians (only in four out of 260 models).

Landscape metrics quantifying patch shape, proximity, texture, diversity and patch size were often significant predictors within univariate models, while metrics regarding similarity or contrast of neighboring patches hardly yielded any significant model (Table B.3.2). Regarding the metrics based on distribution statistics (cf. Table B.3.1), the area-weighted mean (AM) regularly outperformed both the mean (MN) and the coefficient of variation (CV) of the variables. The Coefficient of Variation of the Patch Fractal Dimension (FRAC_CV) for a surrounding area of 500 ha was the best overall univariate predictor, significantly indicating species richness of orthopterans, amphibians, small terrestrial birds ($p < 0.05$), and woody plants ($p < 0.01$) as well as overall species richness ($p < 0.01$). Some metrics were significantly related to the species richness of one particular taxon at several extents, but at no extent to any other taxon. Examples of such taxon specific metrics were ED, LSI, CONTIG_AM, PLADJ and AI for woody plants and PAFRAC and GYRATE_CV for orthopterans (compare Table B.3.1 for the explanation of the abbreviations). Several metrics were significantly related to overall species richness, but did not show any significant relation to any single taxon (Table B.3.2). The metrics PARA_MN,

CONTIG_MN, CONTIG_CV, SIMI_AM, TECI, ECON_MN, and ECON_CV did not result in any significant univariate models for any combination of taxon and extent.

Table B.3.2. *Significance of univariate models among landscape metrics for different extents of landscape plots and species richness. W=woody plants, X=orchids, O=orthopterans, R=reptiles, A=amphibians, B=small terrestrial birds. Letters indicate p < 0.05, ** indicates p < 0.01. Regarding the models for overall species richness, bright grey background shows significance p < 0.05, dark grey background p < 0.01. "+", "–" indicate direction of relation, PARA_MN, CONTIG_MN, CONTIG_CV, SIMI_AM, TECI, ECON_MN and ECON_CV did not result in any significant model for any taxa or extent and were thus not presented in the overview.*

Metric	+/—	20ha	50ha	100ha	200ha	500ha	Metric	+/—	20ha	50ha	100ha	200ha	500ha
Category I. Patch size & patch density							Category IV. Isolation, proximity & similarity						
AREA_MN	—						PROX_MN	—	O,R	O**,R	R**	R	
AREA_AM	—						PROX_AM	+/—	R—	A+**			R—
AREA_CV	+	W,O					PROX_CV	+	O**				
GYRATE_MN	—	O,R					SIMI_MN	—					
GYRATE_AM	—			W			SIMI_CV	—	X				
GYRATE_CV	—				O	O	ENN_MN	+					
PD	+	W,B	O	W			ENN_AM	+	W,B	O			
LPI	—						ENN_CV	+/—	B+	X—**			A+
Category II. Edge & patch shape							Category V. Texture						
LSI	+	W	W**	W	W		CONTAG	—		O,R		R	R
ED	+	W		W	W		PLADJ	—	W	W**	W	W	
SHAPE_MN	—	O		R		W	AI	—	W	W**	W	W	
SHAPE_AM	+	W,B	W,B	W			IJI	+		O			
SHAPE_CV	—		A				DIVISION	+					
PARA_AM	+	W	W**,B	W	W		SPLIT	+		W	W**	W**	
PARA_CV	+		W,R,B				Category VI. Diversity						
FRAC_MN	—						PR	+	O				
FRAC_AM	+	W,B	W,B		B		PRD	+	O				
FRAC_CV	—	O		W,O	W,O**	W**,O,A,B	RPR	+	O				
CONTIG_AM	—	W	W**	W	W		SHDI	+	O,R	O		R	R
PAFRAC	—	O**			O	O**	SIDI	+	O	O,R	R	R	R
Category III. Edge contrast							MSIDI	+	O	O			R
CWED	+	B					SHEI	+		O		R	
ECON_AM	+	B					SIEI	+		O,R	R	R	R
							MSIEI	+					R

Spatial extent affected the number of landscape metrics that were significantly related to species richness. Although single metrics generally performed better at small and intermediate extents (Table B.3.2), some important exceptions were detected, such as FRAC_CV and the Coefficient of Variation of the Radius of Gyration (GYRATE_CV). Orthopterans and small terrestrial birds were better predicted by landscape metrics at smaller extents of 20-50 ha, woody plants and overall species richness at extents of 20-200 ha, while models for reptiles performed best at extents of 200 and 500 ha (Table B.3.2). A similar pattern was revealed regarding the number of categories of metrics (i.e. patch size category, patch shape category, diversity category, etc.) containing at least one significant model. While the number was stable throughout all extents for reptiles, it declined from smaller to larger extents for woody plants, orthopterans, small terrestrial birds, and overall species richness (Table B.3.3).

Table B.3.3. *Relations between landscape structure, organism groups and scale (i.e. extent of the landscape plot) expressed by the number of categories of univariate models (out of the six categories "area", "shape", "contrast", "isolation", "texture", and "diversity") containing at least one model that relates significantly a landscape metric with species richness.*

Taxon	**extent in ha**				
	20	**50**	**100**	**200**	**500**
Woody plants	4	2	3	2	1
Orchids	1	1	0	0	0
Orthopterans	4	5	1	2	2
Amphibians	0	2	0	0	2
Reptiles	3	4	3	3	3
Small terrestrial birds	4	1	0	1	1
Overall Species Richness	5	5	4	4	2

When comparing the sets of one to five metrics, the effect of extent on AIC was negligible, and we pooled across extents obtaining 1000 random sets per taxon and number of metrics. The minimum AIC of the random sets decreased from univariate models towards the models with five metrics, indicating a better goodness of fit of the models with more predictors (Figure B.3.2a). In contrast, the mean AIC increased from univariate models towards the models with five metrics (Figure B.3.2b). Thus, the difference between the best random set (Min AIC) and the average random set (Mean AIC) increased with the increasing number of metrics. Among the different taxa the patterns were very similar; however models with a high minimum AIC (e.g. small terrestrial birds, orthopterans) revealed a stronger AIC decline with an increasing number of metrics, while those with lowest minimum AIC were almost stable (Figure B.3.2a).

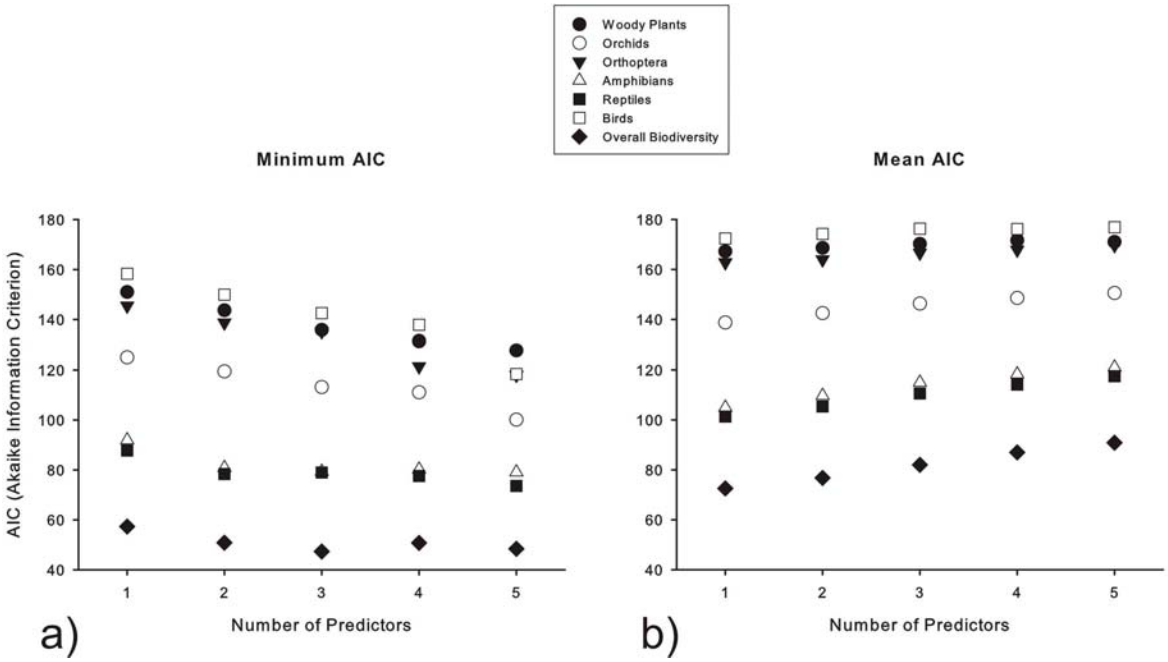

Figure B.3.2. *Effect of the number of metrics on a) Minimum and b) Mean AIC. n = 1000 random sets per taxon for each number of metrics (except the univariate models, where n = 260). Better goodness of fit of the models is indicated by a lower value of AIC*

Discussion

Landscape metrics as indicators of species richness

Our analyses revealed that landscape metrics can indicate species richness towards a significant level, although their indicator value strongly depends on the taxon examined. We detected a strong indicator value of landscape metrics for overall species richness, woody plants, orthopterans and reptiles, while the performance of the metrics was poorer for small terrestrial birds and the poorest for orchids and amphibians. The significant univariate relations confirm that a high species richness of woody plants is indicated by a surrounding landscape of fine texture and high edge density. This could have been expected according to the habitat heterogeneity hypothesis (e.g. McArthur & Wilson 1967) and the intermediate disturbance hypothesis (Connell 1978). Total phytodiversity may show an even stronger peak at disturbed habitats, but woody plants are also adapted to disturbance. Tree diversity in Spain (Torras et al. 2008) and plant diversity in Western Europe (Dufour et al. 2006; Honnay et al. 2003) are strongly related to landscape diversity, while the effects of patch shape are not consistent. These outcomes contrast results from tropical forests, although tropical plant diversity should also peak at intermediate disturbances (Connell 1978; Kessler 2001). Hill & Curran (2003) studied tree diversity in Ghana, which mostly depends on the total area of the forest fragments and to a lesser amount on other aspects of landscape structure. In Mexican forests, the diversity of trees, shrubs and vines is hardly affected by patch area, but strongly affected by patch shape and negatively affected by landscape diversity (Hernandez-Stefanoni 2006). Orchid diversity, on the other hand, was not affected by the surrounding landscape structure. Orchids are stenoecious organisms and their species richness seems to be mainly affected by their need for oligotrophe and sunny microhabitats of medium disturbance (Kati et al. 2000).

Orthopteran richness peaks in Dadia NP in open oak forest with thermophilous scrub undergrowth and wet grassy undergrowth (Kati et al. 2004c). At small landscape extents, our results show that orthopteran species richness was highest on plots with high landscape diversity, while at large extents, patch shape (expressed by FRAC_CV and PAFRAC, but not by the other metrics) had a complex effect on their species richness. Batary et al. (2007) detected little effect of landscape structure on orthopterans in Hungarian grasslands, but found that the most abundant species are even more abundant in homogenous landscapes. Amphibians are also stenoecious due to their dependence on water microhabitat diversity (Kati et al. 2007). This factor was not included in our set of metrics, mainly since small streams are covered by canopy and invisible on the satellite images. The few significant relations must therefore be considered cautiously. Atauri & De Lucio (2001) showed for central Spain that diversity of both amphibians and reptiles increases with increasing landscape heterogeneity, patch density and naturalness. Regarding reptiles, we found that landscape diversity at a micro- and meso-scale is an important factor positively affecting their species richness, as do other ecological factors such as the type of substrate or the degree of shade in the study area (Kati et al. 2007). On the other hand, landscape metrics regarding other aspects of landscape structure rarely were valid predictors of reptile species richness. Only the Mean Proximity Index and Contagion enabled valid models, but these metrics, being measures of aggregation and land cover dominance, are strongly correlated with the diversity indices (Schindler et al. 2008 [= Chapter B.1 of this thesis]; Turner 2005). Landscape heterogeneity and patch density do enhance avian diversity in case studies from Spain (Atauri & De Lucio 2001; Pino et al. 2000) and the south-eastern USA (Mitchell et al. 2006, 2008). Avian species richness in Dadia NP is related to landscape heterogeneity (Kati & Sekercioglu 2006; Kati et al. 2010 [= Chapter B.4 of this thesis]), but it was poorly indicated by our set of landscape level metrics. Only convoluted shapes in close vicinity of the sampling plots were of predictive value and had a clear positive effect. Reasons for the poor performance of landscape metrics as indicators for avian species richness may include the high mobility of birds. Therefore, homogeneity, isolation or poor connectivity might have less of an effect on the metapopulation dynamics and species richness of birds than on ground dwelling taxa. Furthermore, several guilds of birds are included in the data, which might even out different preferences of landscape structure (Mitchell et al. 2006, 2008; Pino et al. 2000; Yamura et al. 2008).

In this study, we detected that several landscape metrics indicated overall species richness much better than that of any single taxon. These cases are of special interest, as they imply that the overall significance was not caused by a very strong relation to a single taxon, but by a general pattern across most of the taxa. For our study area, woody plants and small terrestrial birds are considered the best surrogate taxa of overall species richness (Kati et al. 2004b, 2010). One reason for the good indicator value of these taxa might be, that they were the richest in species, thus for the current study we used an index of overall species richness, which was robust against big differences in species richness among the taxa. Also in Central Europe, birds and vascular plants have shown the highest correlations with overall species richness in a cross-taxon congruence assessment (Sauberer et al. 2004). However, in a recent review, Cabeza et al. (2008) assessed molluscs and fish best performing surrogates for other taxa, while vascular plants and birds only performed average.

Comparison among metrics

Metrics quantifying both aspects of landscape structure - composition (e.g. diversity) and configuration (e.g. texture and patch shape) - were valuable indicators of species richness (Andrén 1994; Mitchell et al. 2006; Vos et al. 2001). While particular species might need continuous and large patches, species richness for the studied taxa was always positively correlated with habitat diversity, patch and edge density. These results should be considered in conservation management of heterogeneous Mediterranean forest landscapes (Kati et al. 2004a,c; Pino et al. 2000; Rocchini et al. 2006) and are similar to results revealed throughout Europe. Wrbka et al. (1999,

2008) proved for plants, birds and bryophytes of Austria that species richness is positively related to landscape diversity, corresponding with results for plants in Belgium (Honnay et al. 2003) and for trees (Torres et al. 2008), butterflies, herpetofauna and birds (Atauri & De Lucio 2001) in Spain. Regarding the texture metrics, all but IJI were good indicators of overall species richness. IJI was significantly related to bird species richness in the Seine valley floodplain, France (Ernoult et al. 2006), and increased the predictability of plant diversity in Spain as the third independent measure next to landscape diversity and patch size (Ortega et al. 2004). However in this study, the performance of IJI in predicting species richness was much lower than that of the other texture metrics, where Contagion performed particularly well for reptiles, and PLADJ, AI and SPLIT for woody plants. Other metrics, e.g. the ones regarding edge contrast and similarity, generally performed worse. Due to anthropogenic disturbance, a wide range of contrast intensities appear in most of the landscape samples. Probably the effects of contrast metrics can neutralize each other over whole taxa, while they are important for specialized species such as the Redback Salamander (*Plethodon cinereus*) in North-American hardwoods (de Graaf & Yamasaki 2002). However for tropical forests, Hernandez-Stefanoni (2006) revealed that high edge contrast is related to lower species richness of the three groups of plants he was investigating, i.e. trees, shrubs and vines.

In this study, the metrics of the patch shape group were particularly good indicators of overall species richness and diversity of woody plants. Regarding the distribution statistics of this group, the area-weighted means performed better than the means, providing evidence that area-weighted metrics are ecologically more meaningful (Gustafson 1998). According to Saura (2002), however, large patches tend to have more irregular shapes, thus landscapes with larger patches could represent higher values for area-weighting patch shape indexes. This may cause them to be more related to patch size than to patch shape (Torras et al. 2008). Previous research revealed that irregular patch shape can indicate both high and low plant diversity (Hernandez-Stefanoni 2006; Hill & Curran 2003; Honnay et al. 2003; Moser et al. 2002; Torras et al. 2008), and Yamura et al. (2008) recently detected for Japanese boreal forests that irregular patch shapes have a positive effect only on edge species, while the effect is negative for interior species.

Effects of scale (i.e. landscape extent)

The effects of landscape structure on species richness depended strongly on the spatial scale, since no variable was constantly significant across all landscape extents for any taxon. An interesting pattern regarding scale was that an upper limit of relevant spatial extent was detected for all taxa but reptiles. The threshold between 100 and 500 ha does not necessarily imply that the animals cover such large home ranges, but rather that the landscape structure surrounding the sampling plots affects their metapopulation dynamics. Some metrics that performed particularly well at the larger extents are FRAC_CV and GYRATE_CV. One reason might be that for these complex distribution statistics (both quantify the statistical spread of patch shape, in the case of GYRATE combined with patch size) a larger extent is needed for their effects to become noticeable. The good performance of many metrics at the extent of 20 ha implies that this extent contains a representative sample of patches and thus enables fine scale modeling, at least with high resolution earth observation data in a heterogeneous landscape. While the indicator value of the metrics varied strongly with spatial scale, the most important components of landscape structure are rather stable across scales (Cain et al. 1997; Schindler et al. 2008 [= Chapter B.1 of this thesis]). We recommend widening scale research towards a comprehensive investigation of scale effects on the indicator values and other ecological applications of landscape metrics.

Effects of the number of metrics

The parsimony of the best random models, applying the minimum AIC as criterion, increased with the number of metrics. But for average sets, applying the mean AIC as criterion, new variables did not lead to an improvement of the models as parsimony increased with the number of metrics. Thus, the more metrics used for a model, the more important metrics choice becomes, and we recommend careful data mining and statistical optimization rather than by expert choice (Schindler et al. 2009 [= Chapter B.2 of this thesis]), before applying sets of landscape metrics to predict species richness.

Implications for landscape and forest management

In Dadia National Park, land abandonment and homogenization of landscape have already taken place, and have lead to an important decrease of landscape heterogeneity compared with some decades ago (Triantakonstantis et al. 2006). As conservation measures for safeguarding local biodiversity, maintenance of forest openings in the buffer zone, maintenance of forest heterogeneity, and enhancement of periodical livestock grazing have been suggested (Grill & Cleary 2003; Kati & Sekercioglu 2006; Kati et al. 2004c). Our results clearly support the above measures, by directly proving the predictive power of landscape heterogeneity (as expressed by various metrics) for species richness of several biological groups. The preservation of a mosaic character appears to be crucial for the conservation of biodiversity in landscapes of several parts of the Mediterranean biodiversity hotspot (Myers et al. 2000) such as Greece, Italy and Spain (Atauri & De Lucio 2001; Farina 1997; Pino et al. 2000; Rocchini et al. 2006; Torras et al. 2008). The Mediterranean spatial heterogeneity imitates that of a permanent disturbance regime and is threatened by land abandonment that leads to woodland recovery and a reduction of open space (Farina 1997).

Sustainable forest management should consider the maintenance of biodiversity and other traditionally undervalued ecosystem functions (Kohm & Franklin 1997; United Nations 1992). Management that leads to heterogeneous and convoluted forest patches should be promoted instead of intensive production forest, typically managed as mono-specific stands (Gil-Tena et al. 2007). An increased use of the forested area for the production of non-timber products may also be positive for maintaining species rich forests (Gil-Tena et al. 2007). For increasingly homogeneous forests, the creation and restoration of small forest openings by controlled logging and the promotion of traditional land uses such as extensive agriculture and low–intensity livestock grazing should show positive effects (Kati et al. 2009; Poirazidis et al. 2004, 2007a). We recommend integrating landscape monitoring into forest management plans. This enhances sustainability and promotes the evaluation of effects of forest management on landscape and wildlife. The metrics performed well for extents of 50 ha, which happens to be the average size of forest stands in Dadia NP (Poirazidis et al. 2010b [= Chapter D.1 of this thesis]). Landscape level metrics related to landscape diversity and patch shape could be applied as indicators of species richness for forest management plans that consider the conservation of biodiversity (Poirazidis et al. 2010b [= Chapter D.1 of this thesis]). For the optimal choice of metrics, case specific exploration of their indicator values is necessary. The present exploration provides a basis for the formal development of landscape structure indicators for forest landscape management and monitoring, and should promote further research regarding the indicator values of landscape metrics.

Conclusions

This study revealed clearly that landscape metrics represent a useful tool for the necessary integration of landscape approaches into conservation management. Regardless of the amount of open land versus forest, landscape diversity and landscape configuration proved to be related to species richness. Heterogeneous landscape mosaics of fine texture are crucial for the maintenance of biodiversity in seminatural Mediterranean forest ecosystems. Further key findings are that overall richness can be well indicated by several landscape level metrics, and that several of them are also good indicators for woody plants, orthopterans and reptiles. Species richness of orchids, on the other hand, is not predictable at all, while the poor performance of the metrics for amphibians and small terrestrial birds might be caused by particularities of the approach presented in this study. Scale has an influence on the indicator value of the metrics, which is generally better at smaller extents of surrounding landscape. Taxa with larger ranges and higher mobility seem to be affected by a wider extent of landscape than small and sedentary ones. With an increasing number of metrics, a careful choice becomes more important, and sets of metrics should preferably be composed after data mining and statistical optimization. To get a better picture of the underlying patterns and processes, we recommend further investigating and reviewing the performance of landscape metrics as indicators of species richness along environmental gradients, for multiple taxa, and multiple scales.

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Chapter B.4. Towards the use of ecological heterogeneity to design reserve networks: a case study from Dadia National Park, Greece

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Running title: Towards the use of ecological heterogeneity for reserve design

Keywords: biodiversity surrogates; complementarity; ecological heterogeneity; ecological networks; indicator; landscape metrics; Mediterranean; reserve design; vegetation complexity; vertical structure

Long Lipped Serapias (Serapius vomeracea) – photo by Grigoris Azoridis

Own contribution:

Study design 20%, implementation 5%, writing 20%

Abstract

In this paper, we present a novel approach for using ecological heterogeneity in reserve design. We measured five ecological heterogeneity indices (EHI) and we used a database of six biological groups (woody plants, orchids, orthopterans, aquatic and terrestrial herpetofauna and passerine birds) across thirty sites in a Mediterranean reserve (Greece). We found that all the five EHI were significantly related to the overall species richness and to the species richness of woody plants and birds. Two indices, measuring vertical vegetation complexity (1/D) and horizontal heterogeneity of landcover types (SIDI) in terms of Simpson's index, predicted well overall species richness and had significantly higher values inside the complementary reserve networks designed after five of the six biological groups. We compared five methods of forming reserve networks. The method of ecological heterogeneity (selecting those sites with the greatest 1/D and then SIDI) was less efficient (non significantly) than the species-based methods (scoring and complementary networks) but significantly more efficient than the random method (randomly selected network). We also found that the method of complementary ecological heterogeneity (selecting those sites where each EHI had its maximum value) was not that efficient, as it did not differ significantly from the random method. These results underline the potential of the ecological heterogeneity method as an alternative tool in reserve design.

Introduction

Reserve systems are the cornerstone for conserving biological diversity and supporting ecological processes in our rapidly changing world (Lee & Jetz 2008). Regarding species as a standard and measurable entity of biological diversity, planners aim primarily to maintain species richness. In this vein, they identify reserve systems that achieve explicit and quantitative conservation targets, such as the maximization of species richness or the conservation of a given threshold of species population size and range at least cost (Margules & Pressey 2000; Cabeza & Moilanen 2001; Naidoo et al. 2006). The most efficient methods are based on the principle of complementarity, which uses heuristic or more computation-intensive optimal algorithms to select those sites that add the greatest number of new species in an existing reserve system (Pressey et al. 1997; Cabeza & Moilanen 2001; Kati et al. 2004a; Arponen et al. 2005). Besides, more sophisticated complementary methods have been developed to maximize the long-term metapopulation persistence of selected species within optimal reserve systems of high connectivity (Nicholson et al. 2006; Arponen et al. 2007; Crossman et al. 2007; Moilanen et al. 2007). On the other hand, in real conservation world, simpler, faster but less efficient methods are often used, such as the scoring method that selects those areas with the greatest number of species (e.g. selection of areas with the greatest number of protected species under the European legislation, to form the Natura 2000 network, in some European countries). All the above species-based methods conserve significantly higher species richness than random reserve networks (Lombard 1995; Howard et al. 1998; Kati et al. 2004a).

However, species-based methods presuppose standard species taxonomy and are data¬intensive, whilst accurate distribution maps are lacking for the majority of species. Furthermore, biotic data are often of poor quality and biased towards charismatic vertebrate species or towards easily accessible sites, which undermines the effectiveness of conservation planning by generating suboptimal reserve solutions (Grand et al. 2007). Besides, the species-by-species mapping approach, though efficient, generates a critical time lag in network implementation, during which land conversion and degradation and subsequent biodiversity loss may continue (Balmford et al. 2002, Meir et al. 2004). To overcome the above problems, several surrogates of species richness such as flagship, umbrella or indicator species have been proposed to encourage faster conservation decisions for reserve selection, but their efficiency is debatable (Caro et al. 2004; Kati et al. 2004b; Hess et al. 2006; Williams et al. 2006; Cabeza et al. 2008). Non-biological proxies such as environmental diversity, land facets or vegetation types have been more rarely proposed in reserve design, because of their relatively low efficiency (Faith & Walker 1996; Wessels et al. 1999; Araújo et al. 2001; Kati et al. 2004a).

In our changing planet, there is a pronounced need to go further, shifting conservation planning from static targets such as the conservation of species diversity patterns, to dynamic targets such as the ecological and evolutionary processes that maintain and generate biodiversity (Cowling et al. 2003; Pressey et al. 2007). It is therefore essential to identify the mechanisms that regulate the patterns of species diversity, in order to integrate them to multi¬species conservation management and ecological networking. One of those well-known ecological factors that maintain and generate species richness at local scale is ecological heterogeneity, because complex habitats can provide more ecological niches, greater potential for resource exploitation, and thus support greater species richness (MacArthur & Wilson 1967; Huston 1994). Although well recognized, ecological heterogeneity was never considered in reserve design procedure. In the present paper we introduce a novel approach: we define ecological heterogeneity in a standard and explicit way at horizontal and vertical dimension and we test its efficiency as an alternative species-free tool in conservation planning, using two techniques (scoring and complementary method). Our results have a particular importance for the Mediterranean environment, where our study area was located (Dadia National Park, Greece). We attempted: (a) to estimate the correlations of five ecological heterogeneity indices (EHI) with the species richness patterns of six biological groups as well as with overall species richness, (b) to explore whether the EHI are significantly higher inside the complementary networks designed after each biological group and after overall species richness than outside them, (c) to compare the efficiency of the two reserve networks designed after ecological heterogeneity (scoring and complementary method) vis-a vis the respective species-based reserve networks (scoring and complementary method), and the random network.

Methods

Study area and sites

The study area of Dadia National Park (DNP) is situated in northeastern Greece (40o 59′ ¬41o 15′N, 26o 19′-26o 36′E). It is a hilly area extending over 43000 ha with altitudes ranging from 10 to 650m. The climate is sub-Mediterranean with temperature ranging from 19 to 40 oC and an arid summer season extending over three months, while the mean annual rainfall ranges from 556 to 916 mm. The forest complex is characterized by extensive pine (Pinus brutia, P. nigra) and oak (Quercus frainetto, Q. cerris, Q. pubescens) forests (Korakis et al. 2006) (Appendix B.4.1). Out of an existing dataset of 36 sites that were randomly selected to represent the vegetation types of the broader area of the National Park, we used a subset of those sites that occurred within the borders of the Park (Kati et al. 2004b). Our system included 30 sites on the whole: 4 sites of 5 ha, 8 sites of 10 ha, 3 sites of 15 ha, and

15 sites of 20 ha, described according to the European standard habitat typology (European Commission 2003) supplemented with Hellenic habitat types (Dafis et al. 2001) (Appendix B.4.2).

Ecological Heterogeneity Indices (EHI)

We created a map of the study area from an IKONOS satellite image (July 2001, 1m pixel resolution) by digitizing its vegetation using ArcGis software. The map consisted of 20 different landcover types, using the below criteria of vegetation composition and the percentage of vegetation cover: 14 forest types of pine, oak, broadleaved forests and their combinations, one type of agricultural land, and five types of openings with vegetation cover from 0 to 40% (see Appendix B.4.1). We then rasterized the resulting vectoral polygons (5m resolution) and calculated three horizontal heterogeneity indices within the area of the 30 sampled sites, using the spatial analysis program FRAGSTATS 3.3 (McGarigal & Marks 1995). We used three particular EHI that they have been determined as the most representative and stable indices describing the landscape structure in the DNP across different spatial scales (Schindler et al. 2008 [= Chapter B.1 of this thesis]). The Simpson's Diversity Index of the landcover types (hereafter SIDI) ranges from 0 to 1 and it is defined as SIDI = $1-\Sigma p_i^2$, where p_i corresponds to the proportional abundance of each patch type. It is higher in sites with greater number of landcover types and when their proportions are more evenly distributed. The mean edge contrast index ECON equals the sum of each patch perimeter lengths, multiplied by their corresponding contrast weights, and divided by the total of the patch perimeters. It ranges from 0 to 1 and increases when the structural differences between neighbouring patches increase. To assign contrast weights to neighbouring patches, we used a subjective scale with increasing weights according to the dissimilarity of the patch type: agriculture-opening patches (0.1-0.4), forest-forest patches (0.1-0.6), forest-opening patches (0.4-0.8), and forest-agriculture patches (1). The area-weighted mean patch shape index SHAPE is defined as p_{ij} / min (p_{ij}), where p_{ij} is the perimeter of the patch ij in terms of number of cell surfaces, and min p_{ij} is the minimum perimeter of patch ij in terms of number of cell surfaces. SHAPE has higher values when very irregularly shaped patches cover a high proportion of area, equals 1 when all patches are squares and increases without limits with the complexity of patch shapes (McGarigal & Marks 1995).

On the vertical dimension, we calculated the number of vegetation layers (NL) and the Simpson's diversity index of vertical structure (1/D). We defined the following five vegetation layers: dwarf shrub layer (<0.5m), lower shrub layer (0.6-2m), upper shrub layer (2.1-4m), lower tree layer (4.1-7m), and upper tree layer (>7m) (Mucina et al. 2000). Within five random quadrats in each site [50m x 50m], we recorded the number of vegetation layers and we determined the percentage cover (relative area occupied by the vertical projection of all aerial parts of plants as a percentage of the surface area of the sample plot) for the separate layers (van der Maarel 2005). We then calculated the average cover of the vegetation layers per site and we assigned them one of the following vegetation cover classes: 0=0%, 1=1-5%, 2=5-25%, 3=26-50%, 4=51-75% and 5>75% (Küchler 1988). Finally, we calculated the Simpson's diversity index (1/D), where $D = \Sigma p_i^2$, p_i corresponds to the above vegetation cover classes for each vegetation layer.

Species richness (S)

We used an existing dataset of six unrelated taxonomic group (189 species), representing different ecological, functional and spatial aspects of local biodiversity: 48 woody plant species, 19 orchid species, 38 Orthoptera species, 9 species of aquatic herpetofauna (terrapins and amphibians), 9 species of terrestrial herpetofauna (terrestrial tortoises and lizards), and 66 species of small terrestrial birds (Appendix B.4.3) (Kati et al. 2004b). We considered the species richness for each of the six groups studied (S) and overall species richness (S total) for all groups together.

Data analysis

We examined the correlations between EHI and the species richness of each group (S), using Pearson's pairwise correlation coefficient. We also tested the effect of site area (A) on the species richness S and on the EHI (univariate regression analysis, SPSS vers 15). We found that the area was weakly associated ($p>0.05$) with all EHI and with the species richness of each biological group, but had a marginal significance ($p=0.05$) for overall species richness, so as to consider it as a parameter besides EHI in our regression analysis. Seven predictive models for species richness (one model per group and one for overall species richness) were constructed, testing the predictive performance of the five EHI and area (A), using a stepwise backward multiple linear regression analysis with the option of presenting only significant predictors ($p<0.05$) (SPSS, vers. 15). We tested the goodness-of-fit of each model using the relative sum of squares (R^2) and the associated F-test.

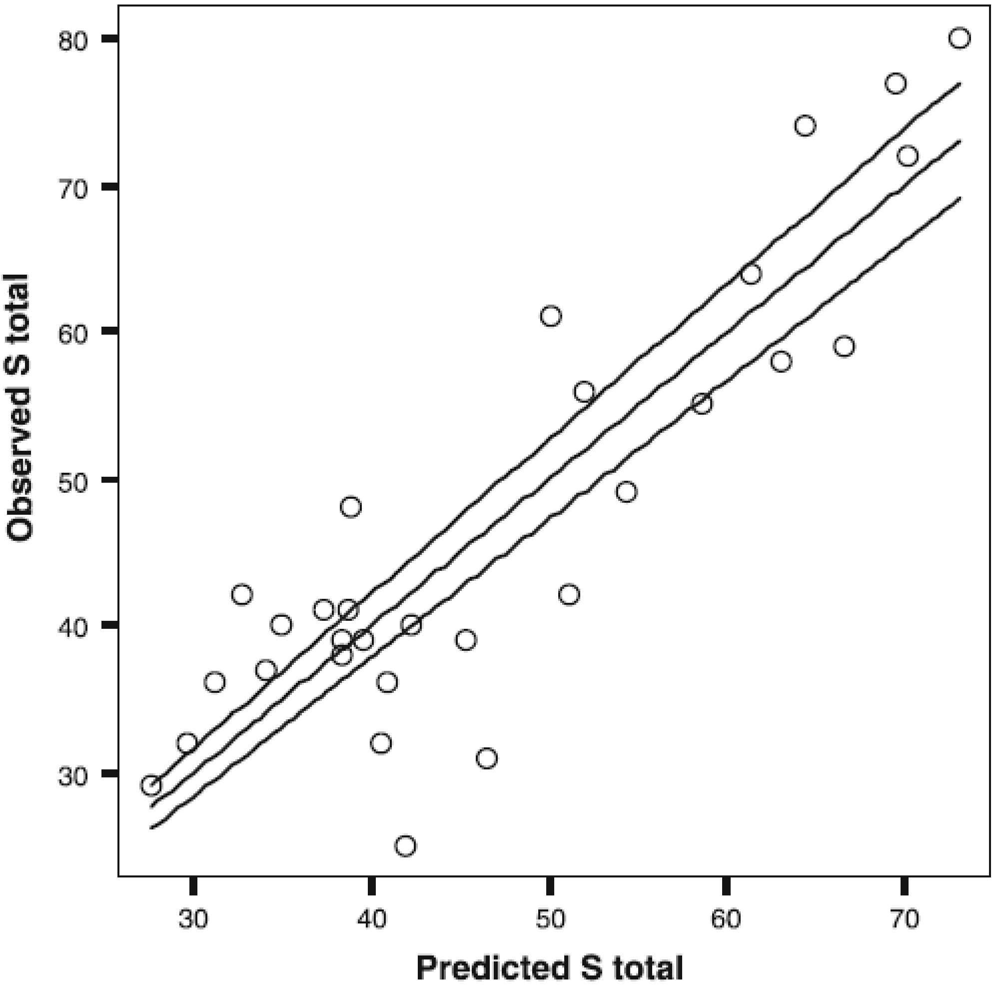
We investigated whether the EHI are higher inside a network designed in a complementary way than outside it. We ran an optimal selection algorithm for every group apart, picking up the complementary network of λ number of sites, where $\lambda=1, 2, \ldots\lambda_{max}$ sites. The network preserved always the maximum number of species each time (for every λ number of sites), until all species of the targeted biological group were preserved ($\lambda= \lambda_{max}$). The algorithm (100 000 permutations) (SAS 1985) could produce one or more solutions (N) for each network of λ sites. For each solution, the algorithm calculated the average value of each EHI inside the selected network (λ sites) and outside it (30-λ sites). We then tested whether the two average EHI values differed significantly (t-test). For example, we consider the group of birds, with $\lambda_{max} = 9$ and e.g. $\lambda = 3$. The algorithm picks up randomly three sites 100 000 times, calculates the number of species included in the above 100 000 solutions (i.e. combination of three sites), but presents only the best solution (i.e. combination of three sites with the maximum bird species richness). Then, the algorithm calculates the average value of each of our five EHI for the above solution (network of three sites) and for the remaining sites (30-λ=27 sites). In our example for birds, this procedure was repeated for λ = 1, 2, 3…9, to ultimately compare the average values of EHI inside and outside the networks.

Finally, we compared the efficiency of the networks designed after the ecological heterogeneity approaches with those designed after the species-based approaches (scoring and complementary versus respectively), to maintain the average species richness of each biological group for λ sites, where λ ranges from 1, 2… up to λ_{max} (as defined by the complementary algorithm for each group, see above). In the ecological heterogeneity method (EH), we selected a network of λ sites, using two sequential criteria (scoring): we selected the sites having the greatest diversity of vertical structure (1/D), and in case of equal values, we chose the ones with the greatest diversity of landover types (SIDI). These indices were selected because they were the two best predictors of overall species richness (see Figure B.4.1). To form the network after the complementary ecological heterogeneity method (CEH), we ranked the five EHI according to their correlation power with overall species richness (see Table B.4.1) and selected the respective sites where they reached their maximum values (sites M2, M1b, A2b, Q4 and A2a, see Appendix B.4.4). In the complementary approach (species¬based), we selected the best network of λ sites for each biological group, using the complementary selection algorithm while in the scoring approach (species-based) the most species-rich sites of the target group were chosen in descending order, to form a network of λ sites (Kati et al. 2004a). In the random approach, we ran a random choice algorithm (100 000 permutations) and we calculated the average number of species included (SAS 1985) in the network of λ sites. We then compared the average species richness included in the above networks (One way Anova, Tukey's post-hoc tests; SPSS vers. 15). More precisely, taking as an example the bird group, we calculated the species proportion out of the overall bird species that are included in the networks after the four approaches. The final network of birds consists of 9 sites (λ_{max} = 9) and conserves 100% of birds after the complementary approach. However, we did not present the results for the final bird network, but the average value of the bird species proportions across the 9 networks, allowing a sound comparison of the approaches without the bias of the number of sites selected each time.

Table B.4.1. *Pearson pairwise correlation coefficients between the ecological heterogeneity indices (EHI) and the species richness across different biological groups (N =30 sites).*

	Ecological heterogeneity indices (*EHI)*				
Species richness (S)	***SIDI***	***ECON***	***SHAPE***	***NL***	***1/D***
woody plants	0.60**	0.58**	0.62**	0.67**	0.68**
orchids	0.46*	–0.12	–0.09	0.56**	0.49**
orthoptera	0.33	0.29	0.33	0.03	0.38*
aquatic herpetofauna	0.45*	0.50**	0.61**	0.08	0.36
terrestrial herpetofauna	0.08	–0.19	–0.04	0.09	0.11
birds	0.72**	0.52**	0.65**	0.59**	0.75**
overall species richness	0.73**	0.52**	0.63**	0.59**	0.78**

**: P < 0.05, **: P < 0.01. SIDI: Simpson's Diversity Index of landcover types, ECON: mean edge contrast index, SHAPE: area-weighted mean patch shape index, NL: Number of vegetation layers, 1/D: Simpson's Diversity Index of vertical structure.*



$S_{total} = 19.90 + 19.76\ SIDI + 2.77\ 1/D + 0.71\ A$ $R^2 = 0.78$, $P<0.01$

$S_{birds} = 13.89 + 7.66\ SIDI + 2\ SHAPE + 0.79\ 1/D$ $R^2 = 0.74$, $P<0.01$

$S_{woody\ plants} = -5.45 + 3.02\ \mathrm{SHAPE} + 2.16\ NL$ $R^2 = 0.71$, $P<0.01$

$S_{orthoptera} = 7.09 - 1.82\ NL + 1.09\ 1/D + 0.32\ A$ $R^2 = 0.41$, $P<0.01$

$S_{orchids} = 1.64 - 1.11\ SHAPE + 0.56\ 1/D$ $R^2 = 0.37$, $P<0.01$

$S_{aq\ herpetofauna} = -1.05 + 1.30\ SHAPE$ $R^2 = 0.37$, $P<0.01$

Figure B.4.1. *Regression models of species richness across all the biological groups and scatter plots relating the observed overall species richness with the predicted one (lines indicate the linear regression model at 95% confidence interval)*

Results

EHI as predictors of species richness

Overall species richness (S total) was significantly correlated with all the five EHI; the diversity indices of vertical structure (1/D) and landcover types (SIDI) demonstrated the highest values (r = 0.78 and 0.73 respectively) (Table B.4.1). Similarly, the species richness of birds was significantly correlated with all the five EHI and best with 1/D and SIDI (r = 0.75 and 0.72 respectively). The species richness of woody plants was significantly correlated with all the five EHI but best with the vertical heterogeneity indices 1/D and NL (r = 0.68 and 0.67 respectively). On the contrary, the species richness of aquatic herpetofauna was significantly correlated only with the horizontal heterogeneity indices, and particularly well with the area-weighted mean patch shape index (SHAPE) (r = 0.61). The species richness of orchids was significantly correlated with the vertical complexity indices NL and 1/D and to a lesser amount with SIDI. The species richness of the Orthoptera group was not well correlated in general with the EHI, except for 1/D. No heterogeneity index presented significant correlation with the species richness of terrestrial herpetofauna.

We also found that all the models predicting species richness were highly significant ($p<0.01$), except for terrestrial herpetofauna, where we failed to produce any significant model. The models had strong predictive power for the overall species richness and the species richness of birds and woody plant richness (74% to 78% of the variance explained) and less strong for Orthoptera, orchids and aquatic herpetofauna (37% to 41% of the variance explained) (Figure B.4.1). We found that the indices 1/D and SHAPE were important predictors in four of the models and that the indices SIDI, NL together with area A were important predictors in two models.

EHI in complementary networks

All the five EHI were significantly higher inside the complementary networks designed for overall species richness, but also in those designed for woody plants, birds and aquatic herpetofauna (Table B.4.2). The horizontal diversity index (SIDI) and the vertical indices (NL, 1/D) were constantly higher inside the complementary networks for all groups except for Orthoptera (Table B.4.2).

Table B.4.2. *Average differences of the ecological heterogeneity indices (EHI) inside and outside the complementary networks of λ=1, 2… λ_{max} sites that conserve the overall number of S_{tot} species, on the basis of the N solutions produced by the optimal algorithm (S.A.S., 100 000 permutations).*

Biological group	S_{tot}	λ_{max}	N	Average difference (*EHI* in network – *EHI* outside network) *SIDI*		*ECON*		*SHAPE*		*NL*		*1/D*	
woody plants	48	8	14	0.25	**	19.34	**	1.18	**	1.50	**	3.49	**
orchids	19	5	6	0.34	**	-7.30	**	0.24	*ns*	1.43	**	4.97	**
orthoptera	38	5	8	0.18	*ns*	8.56	*ns*	-0.11	*ns*	0.09	*ns*	0.28	*ns*
aquatic herpetofauna	9	3	5	0.20	**	33.63	**	1.73	**	1.01	**	2.61	*
terrestrial herpetofauna	9	3	54	0.58	**	-0.46	*ns*	0.64	*ns*	0.56	**	1.02	*
birds	66	9	24	0.20	**	5.59	*	0.69	**	0.25	*	1.48	**
all groups	189	-	111	0.13	**	5.16	**	0.41	**	0.64	**	1.17	**

**: P < 0.05, **: P < 0.01, SIDI: Simpson's Diversity Index of landcover types, ECON: mean edge contrast index, SHAPE: area-weighted mean patch shape index, NL: Number of vegetation layers, 1/D: Simpson's Diversity Index of vertical structure.*

Reserve design methods

The networks designed after the ecological heterogeneity method (EH) succeeded to maintain on average 65% of overall species richness (123 out of the 189 species), whereas the performance of the complementary ecological heterogeneity method (CEH) was slightly worse, maintaining on average 61% of overall species richness. The networks designed after the complementary method, the scoring method and the random approach maintained on average a proportion of 84%, 79% and 40% of overall species richness respectively (Figure B.4.2, Appendix B.4.5). Hence, the EH network was less efficient but not significantly (p>0.05) than the complementary and scoring networks, by 19% and 14% of overall species richness respectively. It performed better than CEH network (4% more species of overall species richness maintained) and significantly better than random network (p<0.05), maintaining on average 25% more species of overall species richness (Figure B.4.2).

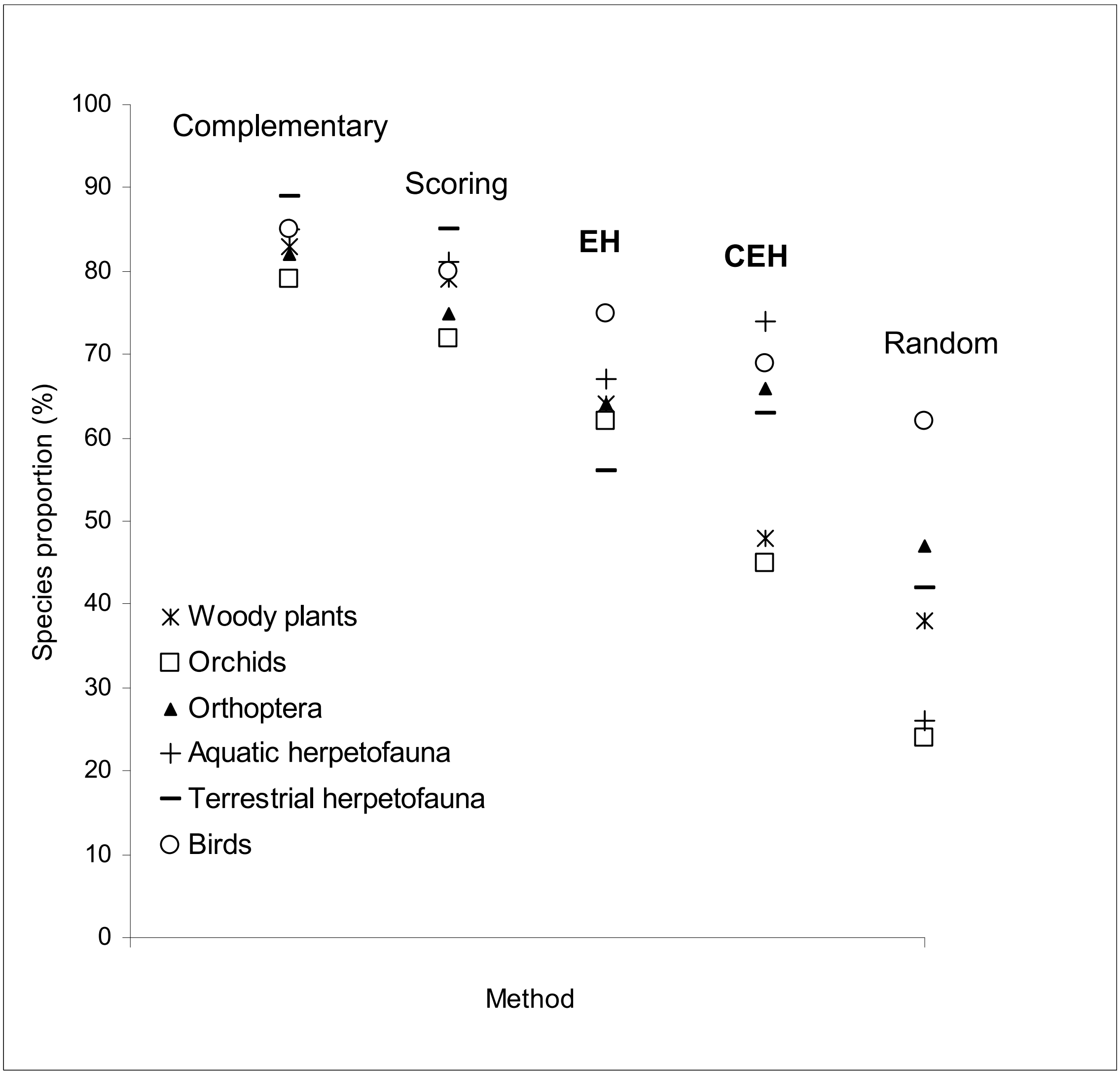


Figure B.4.2. *Comparison of reserve design methods, on the basis of the average proportion of species conserved inside their networks of λmax number of sites (see Appendix B.4.4).*

Discussion

Drivers of species richness

Ecological heterogeneity is a well known local process determining species richness at local scale (Huston 1994). Regardless of the methods used to measure ecological heterogeneity, it correlates often with species diversity (Honnay et al. 1999; Tews et al. 2004; Torras et al. 2008) and has been found to increase the slope of the species-area relationship (Kallimanis et al. 2008). In the present paper, we found that the response of species richness to the EHI was in general positive, but greatly varied depending on the biological group considered. The diversity indices of vertical structure (1/D) and landcover types (SIDI) were the best predictors of overall species richness. Our results showed that birds are positively affected by the complexity of vertical structure, the diversity of landcover types and the patch shape irregularity (1/D, SIDI, SHAPE respectively). Several small-scale and multiple-scale studies have shown that habitat heterogeneity at both horizontal and vertical dimensions affects small terrestrial bird distribution (e.g. Böhning-Gaese 1997; Farina 1997; Grand & Cushman 2003; Kati & Sekercioglu 2006; Kati et al. 2009). Shape irregularity is found to be positively associated with bird abundance across different habitats and spatial scales, but also negatively associated with the winter bird species abundance in deciduous forests, as well as with the species richness of boreal forest species (Brennan & Schnell 2005; Yamaura et al. 2008; Caprio et al. 2009). We also found that the vertical structural complexity, and patch shape irregularity affected the species richness of woody plants. The tree and bush species richness is known to be associated with forest stand structural complexity (Brokaw & Lent 1999; Lindenmayer & Franklin 2002), whereas irregularly shaped habitat patches contain usually more plant species, because of their higher number of environmental gradients (Honnay et al. 1999; Moser et al. 2002). Our results showed that the vertical vegetation structure is an important predictor for Orthoptera species richness. The gradient of vegetation density and particularly the cover and height of bushes affect substantially Orthoptera diversity patterns and abundance, given that Orthoptera communities avoid forest habitats with dense vegetation cover, but reach their optimum in semi-open habitats with great structural and microclimatic heterogeneity (Kati et al. 2004c; Fartmann et al. 2008). Ecological heterogeneity does not seem to be that important in regulating orchid and aquatic herpetofauna species patterns (models explaining 37% of variance). We found that vertical structural complexity is positively associated, but shape irregularity negatively associated with orchid species richness. Orchids depend most on other factors than landscape characteristics, such as altitude, soil characteristics or specific microhabitats types (Kati at al. 2000; Tsiftsis et al. 2008). We also found that patch shape irregularity was well associated with aquatic herpetofauna. The presence of linear vegetation features on the landscapes bordering streams and connecting habitats account for increased patch shape irregularity, thus explaining the above relationship, because of the dependence of the aquatic species on appropriate microhabitats and their connectivity (Cushman 2006; Kati et al. 2007). Finally, none of the ecological heterogeneity indices predicted the species richness of terrestrial herpetofauna. The community of terrestrial herpetofauna species is more dependent on the availability of open or semi open suitable habitats than landscape characteristics (Atauri & De Lucio 2001; Kati et al. 2007; Ioannidis et al. 2008).

Reserve design and conservation implications

Ecological heterogeneity can be an enriching rather than an impoverishing factor for biodiversity maintenance in Mediterranean landscape, up to a certain threshold (Blondel & Aronson 1999). Clearly, increasing heterogeneity increases the potential number of ecological niches and therefore the number of species that may exist in a given area. However, it is also clear that beyond a certain threshold increasing heterogeneity results in fragmentation, population decline and increase of the likelihood of stochastic extinction (Franklin et al. 2002; Kadmon & Allouche 2007). Ecological heterogeneity mirrors the combined effect of several interactive natural and anthropogenic processes (e.g. fire frequency, stochastic climatic events, human land use, grazing regimes) and furthermore can be measured and mapped, being an adequate factor for integration in conservation planning (Pressey et al 2007). However, usually ecological heterogeneity is only implicitly considered in conservation planning. Montigny & MacLean (2005) measured forest heterogeneity as a combination of biotic and geomorphologic factors (diversity of forest species composition, number of soil types and elevation classes), and found that the heterogeneity method is more efficient than the representation method in terms of selecting fewer reserves. In our study, we initiated a novel approach in conservation planning, by considering ecological heterogeneity and testing its efficiency using two techniques, scoring and complementarity. We measured ecological heterogeneity in two-dimensional space, in an explicit and replicated way, and we tested directly its efficiency in reserve design across diverse biological groups, which reflected different functional, spatial and ecological aspects of biological diversity. Previous work in our study area comparing species-based methods of reserve design proves that the method based on complementarity conserves more species than the scoring method, which in its turn is better than the random method (Kati et al. 2004a). The present work introduced a new approach in the above picture, EH method, which proposed to select the sites with the greatest vertical and then horizontal heterogeneity in terms of Simpson's diversity index (1/D and SIDI respectively). We showed that the method of ecological heterogeneity was less efficient but no significantly from the species-based methods and that it was significantly more efficient than the random method. According to our results, the principle of complementarity is not that important in reserve design when implemented by species-free methods. Selecting those areas with the maximum values of complementary indices of ecological heterogeneity (CEH method) did not differ significantly than random networks. Therefore we propose the Simpson's diversity index in both vertical (1/D) and then horizontal (SIDI) dimension as a good surrogate of species richness. Furthermore, these indices predicted best overall species richness and had significantly higher values inside the complementary reserve networks designed in favour of five out of the six biological groups examined

Although the EH method can be well implemented in local reserve selection procedure in other Mediterranean areas, further testing is required across different spatial scales and different areas, for different biological groups and particularly when considering threatened species or species with larger spatial requirements. Despite its limitations, the EH approach is a promising species-free alternative to taxonomy-based design of reserve systems. This novel approach focuses primarily on the speed and ease of monitoring of conservation decisions rather than on optimality. The EH method can be used as an alternative tool in the procedure of reserve design, ideal for situations where biological data are unavailable, at a preliminary stage of collection, or when resources do not permit a full biodiversity study.

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Part C – Methodological developments for monitoring raptor populations and trends of the diverse raptor assemblage of Dadia NP

Dadia National Park hosts about 350 territories of diurnal birds of prey during the breeding period. About 20 species are breeding in the reserve, many of which are priority species for conservation, and about ten further species use parts of Dadia National Park regularly. This high diversity and abundance is not only the reason for raptors being the flagship species of Dadia National Park, but also a big challenge for research, monitoring, and conservation management. When working on raptors in the area, very often established methods have to be adapted to be able to deal with the specific situation. So it was obvious after the first assessments (Poirazidis et al. 2002, 2009a [= Chapter C.2. of this thesis]) that a systematic monitoring of all breeding raptor species would not be realizable by searching for all the nests, at least not without disturbance of sensible species, nor with easily standardizable effort, a basic prerequisite for systematic assessments of population trends.

However, this part C starts with a chapter that assesses the precision of the applied system of Black Vulture telemetry. We evaluated angular and linear error, and compared the performance of eight different mathematical methods of estimating the source of a signal.

The other three papers are dealing with the systematic raptor monitoring: Developing its methodology (Chapter C.2), giving a brief overview on the species status and trends (Chapter C.3), and describing in an extended book chapter long term trends and habitat use each species (Chapter C.4.).

A further paper that would fit into this part was on the development of a methodology for the monitoring of Egyptian Vulture (Poirazidis et al. 2009b), but unfortunately this methodology was never implemented, and I decided not to include the paper into this thesis.

Chapter C.1. Error Assessment of a Telemetry System for Eurasian Black Vultures (*Aegypius monachus*).

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Running Title: Error assessment of a telemetry system for Black Vulture

Keywords: bearing analysis, Dadia National Park, linear error, location estimators, radio telemetry.

Illustration by Tuisku Sarrala

Own contribution:

Study design 60%, implementation 80%, writing 70%

Abstract

Telemetry is commonly used to study animals, but rarely its precision is considered. We evaluated the accuracy of a VHF telemetry system applied for Eurasian Black Vulture (Aegypius monachus) in the Dadia National Park, Greece. The system was without directional bias, the bearing standard deviation was 9.7°, and the most accurate location estimator was the Andrews estimator. The average linear error was 1 km for three-bearing estimates and 1.6 km for two-bearing estimates. Confidence ellipses were an accurate measure of confidence areas. We conclude that the applied system is precise enough to serve the purposes of the telemetry study.

Introduction

The globally endangered Eurasian Black Vulture is the largest bird of the Western Palaearctic and considered an umbrella species for the conservation of biodiversity (Carrete & Donázar 2005). Its breeding population in the National Park of Dadia-Lefkimi-Soufli Forest (hereafter DNP) is the last remaining in the Balkan Peninsula, and has been considered a central subject of conservation (Collar et al. 1994; Poirazidis et al. 2004; Skartsi & Poirazidis 2002). In 1979 the population was estimated at 26 individuals and 4-5 breeding pairs (Hallmann 1979), but it recovered due to several conservation measures and increased from 6 breeding pairs in 1987 to 21 pairs in 2002 (Skartsi & Poirazidis 2002). The present situation of the species in the area remains critical as many of the mortality factors continue to affect the population negatively (Elorriaga et al. 2004; Skartsi & Poirazidis 2002). Research has focused until now on the nesting areas, including monitoring of the breeding activities (Elorriaga et al. 2004; Skartsi et al. 2003) and modeling nest site availability (Poirazidis et al. 2004), while our knowledge of ranging habits remains limited.

Black Vultures are large scavenging birds that travel quickly and cover large and remote areas. Information about range use and movement patterns of the population, as well as the threats that the birds encounter in their foraging area, is essential for the management and conservation of the species (Skartsi & Poirazidis 2002). To obtain these data a radio telemetry project with the Eurasian Black Vulture has been developed in the DNP (Vasilakis et al. 2006).

An important concept essential to telemetry is that observed bearings and the resulting point locations are only estimates of the actual ones (White & Garrott 1990). However, few investigators have tested the accuracy of their telemetry systems, and point estimates derived from bearing intersections often were considered to be exact locations (for criticisms see Harris et al. 1990; Kenward 2001; Saltz 1994). No matter how much time and thought an investigator devotes to designing a radio tracking system, the quality of the produced location estimates is unknown until it has been tested in the field (White & Garrott 1990). The importance of testing the accuracy of telemetry systems was first suggested by Heezen and Tester (1967) and cannot be overemphasized (Harris et al. 1990; Saltz 1994; Saltz & White 1990; Samuel & Fuller 1994; White & Garrott 1990; Zimmermann & Powell 1995). Precise error estimates are needed for locations derived by triangulation in order to be used in an assessment of range use patterns or habitat selection, which are sensitive to location error (Marzluff et al. 1994). A radio telemetry system must be tested to determine the precision of the directional bearings (Saltz & White 1990; Springer 1979), the linear error between estimated and true locations (Marzluff et al. 1997; White & Garrott 1990; Zimmermann & Powell 1995), and thus whether the system can produce location estimates of adequate accuracy to meet the objectives of the study (Kenward 2001; White & Garrott 1990).

To obtain results that direct towards recommendations for the telemetry, the error must be assessed mimicking the study of radio tracking. Test transmitters should be placed in a variety of known locations through the study area and multiple bearing estimates on each transmitter location from the receiver stations should be obtained (Jenkins & Benn 1998; Marzluff et al. 1997; White & Garrott 1990; Zimmermann & Powell 1995). When using telemetry concerning raptors, it is of advantage to lift transmitters in the air to avoid additional error due to the low position of the transmitter on ground level (Marzluff et al. 1997).

The main aim of this study was to optimize the telemetry of Eurasian Black Vulture in DNP, concerning the best estimation of the point locations and the determination of their precision. Specifically, the objectives were: (1) to calculate method bias and sampling error of the directional bearings, (2) to find the optimal location estimators for the telemetry system applied in the study area, (3) to calculate the average linear error between estimated and true locations, and (4) to describe the confidence areas for the point locations and to evaluate their accuracy.

Materials and Methods

Data on the test transmitter were collected exactly like those for the telemetry study of the Eurasian Black Vulture (Vasilakis et al. 2006), using the same receiver stations (compass rosettes fixed on the ground), antennas (four-element Yagi, Televilt), receivers (ICOM R10 and Communication Specialists Inc. R-1000), methods to take bearings, and involved personnel. The study area was located in northeastern Greece, ranging from the Evros River forming the border with Turkey to the Bulgarian border in the Eastern Rhodope Mountains (Figure C.1.1). It covered the breeding colony of the population of Eurasian Black Vulture in DNP, as well as a large part of the potential foraging area of this population. The mountainous landscape ranged in elevation from 20 to 1200 m, for detailed descriptions see (Vasilakis et al. 2006) or (Vasilakis & Poirazidis 2004). The study area was divided in six watersheds, which were covered by twelve receiver stations, established at exposed hilltops. The study of error assessment was implemented in all the watersheds using three receiver stations per watershed.

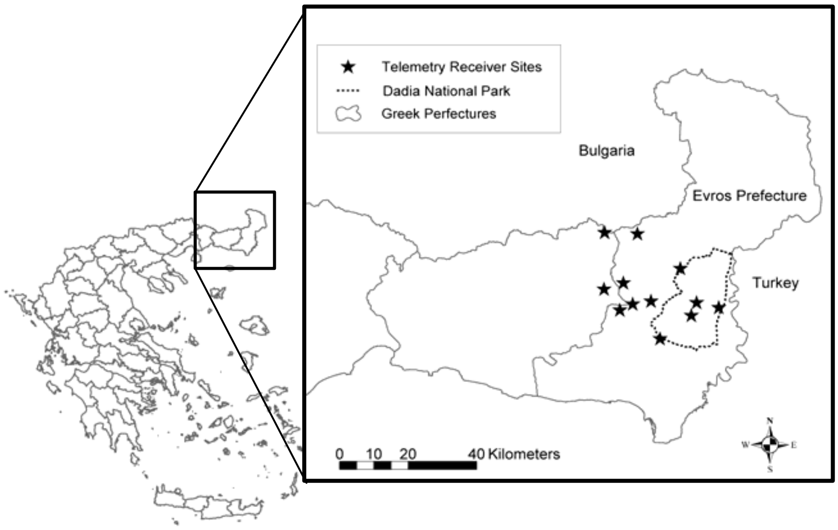

Figure C.1.1. *The study area located in Evros, northeastern Greece.*

Two different methods were used to obtain the bearings. One is based on the direction of the loudest signal (strongest bearing) and the other on the middle-direction between the directions where the signal disappears (null average) (Springer 1979). As for the vultures, strongest bearing was used, when the signal could be detected optically on the screen of the receiver, null average for the cases where the signal was received only acoustically. Observers communicated to permit simultaneous bearings and to detect signal bounce like erroneous bearings from the reverse side. Ten triangulation attempts were made per transmitter station. Transmitter stations were unknown for the observers, they consisted in a transmitter lifted in the air with a balloon. The balloon was filled up with Helium to a diameter of 1.2 m and fixed with a line of 50 m onto a car.

The software LOAS 3.0.1 (LOAS, Ecological Software Solution, Sacramento, USA) was used to calculate the true bearings from receiver to transmitter stations. The error angle, which equals the difference in degrees between true bearing and bearing taken by the observer, was calculated for each bearing. The magnetic declination (Boshoff et al. 1984; White & Garrott 1990) of the study area is 3.783° and it has been included in the bearing analyses. We tested for differences in accuracy between the bearings obtained by the methods "strongest bearing" and "null average". Finally, bias and precision of the bearings were determined, calculating mean and standard deviation of the error angle data set.

As the procedure of estimating the location of a signal depends on the amount of bearings that can be used to realize this estimation, triangulation attempts were classified according to the number of observers who obtained a bearing. Thus, three data sets resulted: (1) three bearing estimates (3BE) – three observers succeeded a bearing, (2) two bearing estimates (2BE) – two observers succeeded a bearing, and (3) single bearings – only one observer succeeded a bearing. Single bearings were not used in further analyses, the both remaining data sets 3BE and 2BE were analyzed separately. Seven different estimators (Andrews, Huber, Maximum Likelihood, Best Biangulation, Harmonic Mean, Geometric Mean, and Arithmetic Mean, for explanations see (ESS 1999) were used to estimate the point locations of the 3BE data. Thus, seven result data sets of location estimates were obtained and the linear error (LE) was measured for each of the location estimates in order to determine which estimator is optimal for the used telemetry system. For the 3BE data we also evaluated an optimal substitute for the cases the optimal estimator failed to produce a location estimate. For the 2BE data set the point estimates are located at the bearing intersections and for this reason only one estimator was used. For each point estimate, the linear error was calculated and compared with the linear error of each of the seven result data sets obtained from 3BE.

The confidence areas of the point estimates were calculated for the following data sets: (1) for the data set "3BE – best performing estimator", (2) for the data set "3BE – best substitute", and (3) for the data set "2BE". To compute the confidence areas, the evaluated bias and precision of the bearings were used. For the Maximum Likelihood (ML) based estimators (Andrews, Huber, and ML estimator), the Chi-Squared distribution was used to calculate 95% confidence ellipses, for the other estimators (Best Biangulation, Harmonic, Geometric and Arithmetic Mean) 95% error polygons were calculated. After the computation of the confidence interval areas, their accuracy was examined by evaluating the coverage, i.e. the proportion of true locations falling inside their corresponding confidence area.

Statistical treatment

Directional data like telemetry bearings are best described by a Von Mises distribution (Mardia 1972; Zar 1998), but the standard deviation of error angles, the common measure of bearing precision, assumes a normal distribution. The obtained distribution of error angles differed significantly from normality (Kolmogorov – Smirnov P = 0.009) and was leptokurtic. To obtain a normal distributed sample, extreme values and outliers were determined by box-plots and eliminated. For the detection of differences in accuracy between the bearings obtained by the methods "strongest bearing" and "null average", the independent sample t-test was applied. The two sample paired t-test was used to test if the evaluated bias of the bearings was significantly different from zero.

To detect the best location estimators, statistical differences between the linear errors provided by the different estimators were evaluated. First K-S was applied to test for normality. Being not normal distributed, the data were transformed using natural logarithm, square root, cubic root and 4th root. Lacking still normality, non-parametric approaches were used. Wilcoxon Signed Ranks test for paired related samples were applied to compare among the seven linear error data sets obtained by triangulation, and Mann-Whitney U tests for two independent samples were applied to compare each of these data sets with the 2BE linear error data set. The statistical procedures were completed using SPSS.

Results

Evaluation of bias and precision of bearings

In total 760 bearings were taken by six different observers. 29 transmitter stations were used, covering all six watersheds in the study area. Error angles obtained during this study did not show significant difference from normality (Kolmogorov – Smirnov Z = 0.953, P = 0.32) after eliminating 35 extreme values and outliers using box-plots. Using independent sample t-tests, no significant differences were detected between the two methods of taking bearings (P = 0.76), thus "strongest bearing" and "null average" data were pooled. For the resulting 725 bearings, method bias was 0.53° and the standard deviation 9.68°. The obtained bias was not significantly different from 0 (two sample paired t-test $t_{724} = 1.49$, $P = 0.14$), thus it was ignored for the following analyses.

Determination of the linear error and the optimal location estimator

The data set of 725 bearings was obtained during 288 attempts of triangulation. 164 three bearing estimates, 109 two bearing estimates, and 15 single bearings resulted. Regarding the 3BE data set, the locations of the transmitter station were estimated seven times, each time with another estimator based on the same 164 location estimates. The resulting distributions of linear error of the location estimates were not normally distributed and were examined with non-parametric approaches. The median of the linear error ranged from 1032 m when applying the Andrews estimator (Figure C.1.2) to 1303 m when the Arithmetic Mean estimator was used, the 95% percentiles as measure of variance ranged from 2422 m to 3772 m (Table C.1.1).

Table C.1.1. *Linear error (m) between estimated and true location of the transmitter station.*

Estimator	**N[a]**	**Med[b]**	**Percentiles**				**Max[c]**
			25%	**75%**	**90%**	**95%**	
3BE data set							
Andrews Estimator	156	1032	533	1529	2016	2422	4651
Huber Estimator	155	1033	522	1543	2024	2425	4648
ML Estimator	155	1033	522	1543	2024	2425	4648
Best Biangulation Estimator	163	1274	463	2047	2510	2886	5441
Harmonic Mean Estimator	163	1303	774	1784	2788	3761	27860
Geometric Mean Estimator	163	1303	774	1784	2788	3761	27882
Arithmetic Mean Estimator	163	1303	774	1785	2788	3772	28762
2BE data set	108	1570	1049	2198	3234	3984	19937

[a]number of estimates, [b]median, [c]maximum value

According to the evaluated precision, the estimators could be classified in two groups: the ML-based estimators (Andrews, Huber, and ML estimator) performed better than the second group, consisting in Best Biangulation, Arithmetic, Geometric, and Harmonic Mean estimator. The differences were highly significant when comparing any of the ML-based estimators with Arithmetic, Geometric and Harmonic Mean estimator (Wilcoxon Signed Ranks tests $P < 0.001$). Highly significant differences resulted when comparing the Huber and the ML estimator with the Best Biangulation estimator (Wilcoxon Signed Ranks tests $P < 0.01$), and significant differences for the comparison of the Andrews estimator with the Best Biangulation estimator (Wilcoxon Signed Ranks test $P = 0.011$).

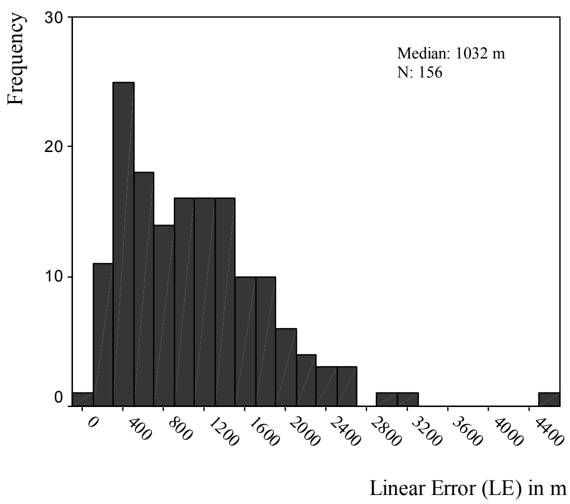

Figure C.1.2. *Linear error (LE) between true locations and locations estimates by three bearings. The Andrews estimator was used for location estimation.*

The Andrews estimator was the best performing estimator, providing smaller linear error than Huber and ML estimator (Wilcoxon Signed Ranks tests, P = 0.029). The later two provided exactly the same result data set (P = 1.00), and they failed in the same cases like the Andrews estimator. Thus, to find an optimal substitute for the Andrews estimator, the most adequate not-ML-based estimator was evaluated. Out of the four remaining location estimation techniques, no statistically significant differences were detected (Wilcoxon Signed Ranks tests, P values ranging from 0.06 to 0.33). Finally, the Best Biangulation estimator was chosen as the best performing out of this group, for having lower median, 25%, 90% and 95% percentile and maximum LE value than the other estimators (Table C.1.1). In the seven special cases that the Andrews estimator failed to estimate a location, the Best Biangulation estimator performed worse than generally. When considering only these seven cases, the linear error ranged from 1585 to 2528 m and its median increased from 1274 to 1897 m.

The analysis of the 2BE data set resulted in a median of 1570 m and a 95% percentile of 3984 m (Table C.1.1, Figure C.1.3). The precision of the 2BE locations was less than that of the 3BE locations. The differences were very highly significant when comparing the 2BE data set with the 3BE data sets of the ML-based estimators and the Best Biangulation estimator (Mann Whitney U tests P < 0.001), and highly significant when comparing the 2BE data set with the 3BE data sets of the estimators Arithmetic, Geometric and Harmonic Mean (Mann Whitney U tests P < 0.01).

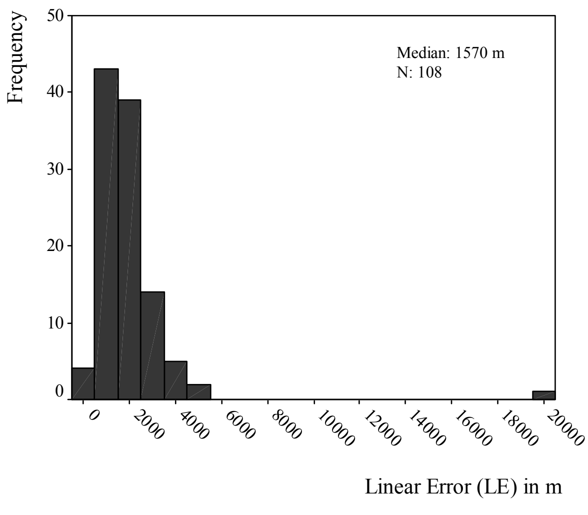

Figure C.1.3. *Linear error (LE) between true locations and locations estimated by two bearings.*

Description of the confidence areas of the location estimates

The confidence areas of the point location estimates were computed using the evaluated SD of the bearing errors of 9.68°. Being not normal distributed, a non-parametric approach was used for their description. For the data set "3BE – Andrews estimator" (N = 156), the median value of the area of the 95% Confidence Interval Ellipse was 1048 ha (Figure C.1.4). The true location was 141 times inside the 95% CI Ellipse, thus the coverage was 90.4% (Table C.1.2).

For the data set "3BE – Best Biangulation estimator" (N = 163), the median value of the area of the 95% Error Polygon was 301 ha and the coverage was only 42.3% (Table C.1.2). Regarding the 2BE data set (N = 108), the median value of the area of the 95% Error Polygon was 842 ha (Figure C.1.5) and the coverage was only 54.6% (Table C.1.2).

Table C.1.2. *Confidence Areas (ha) of the point location estimates.*

Data set	N[a]	Med[b]	Percentiles 25%	75%	90%	95%	Max[c]	Coverage (%)
3BE-Andrews	156	1048	516	2254	3941	4298	6588	90.4
3BE-Best Biang.	163	301	122	704	1069	1458	1799	42.3
2BE	108	842	580	2205	3571	27093	$1.6\ 10^5$	54.6

[a]*number of estimates,* [b]*median,* [c]*maximum value*

Note that confidence ellipses were computed for the Andrews estimator, but confidence polygons for the other data sets.

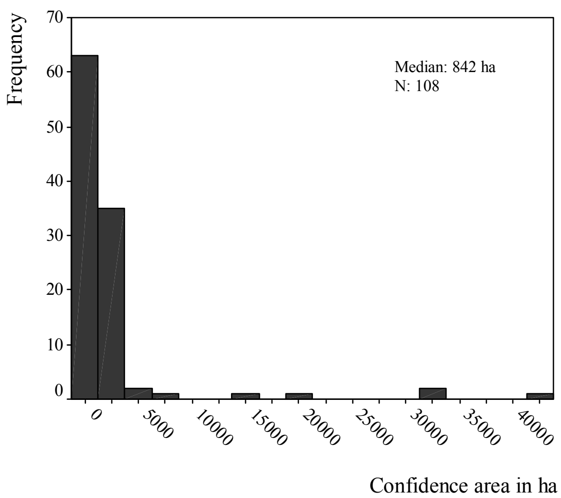

Figure C.1.4. *Area of the 95 % Confidence Interval Ellipses (in ha) of the estimates obtained with three bearings. Location estimation by Andrews estimator.*

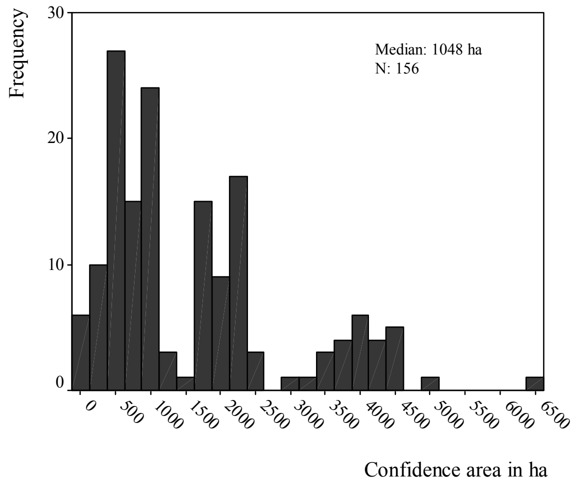

Figure C.1.5. *Area of the 95 % Confidence Interval Ellipses (in ha) of the estimates obtained with two bearings.*

Discussion

The analyses of the bearing errors clarified that the applied system of telemetry was without directional bias, thus the used equipment, the rosette, the observers and the net of established receiver stations provided accurate bearings. The evaluated bearing precision (SD = 9.68°) was similar to the values detected by other researchers with standard deviations ranging from 1.1° to 16.4° (Burger et al. 1991; D'Oleire-Oltmanns et al. 1989; Edge & Marcum 1985; Jenkins & Benn 1998; Marzluff et al. 1997; Mooty et al. 1987; Schmutz & White 1990; Springer 1979; Zimmermann & Powell 1995). Researchers use and recommend different methods of taking bearings (Kenward 2001; Marzluff et al. 1994; Springer 1979; Zimmermann & Powell 1995). We could demonstrate that in our case there was no significant difference between "strongest bearing" and "null average" method, although we had a large sample of bearings to detect any differences.

Small sample sizes of bearing error lead researchers to assume normality when their data may not be normal distributed (Zimmermann & Powell 1995). For the large sample of bearings obtained in this study (N = 760), the approach of the elimination of extreme values and outliers (Lee et al. 1985) was chosen to eliminate their pronounced effects on the estimation of bias and precision and to obtain normal distributed data. After excluding these outliers, all location estimates were retained (although some of them were acquired under unfavourable conditions like small intersection angles close to 0°), in order to examine the total range of the possible linear errors and confidence areas.

In this study of error assessment, the telemetry study of Eurasian Black Vulture was imitated as much as possible. After some preliminary attempts with test transmitters placed at the ground, a balloon was used to lift the transmitters in the air. Marzluff et al. (1997), comparing the accuracy of various methodologies of telemetry, found out that using a balloon the bearing precision was higher. Obtained values using a balloon should be more realistic when applied for flying or exposed animals than values obtained from ground level. However, Marzluff et al. (1994) also detected larger errors for moving test transmitters than for stationary ones. Also Schmutz & White (1990) and Zimmerman & Powell (1995) state that all methods of error analysis using stationary test transmitters may be biased towards underestimating true location error, when data are collected on moving animals. For the applied telemetry study of Eurasian Black Vulture in DNP this problem was minimized, because each triangulation of the bird location was coordinated to provide simultaneous bearings (Vasilakis et al. 2006).

Vultures are fast and wide ranging raptors, travelling over huge areas when searching for food. In a study concerning the Eurasian Black Vulture in Extremadura (Spain), a maximum distance from the nest of 80 km was determined for breeders, while the maximum annual home-range of non-breeding individuals was 500 000 ha (Corbacho et al. 2004). In the Sierra Morena, south-western Spain, Black Vultures cover home ranges of 135,430 ha (N = 14) during the breeding season and of 77,775 ha (N = 6) during the non-breeding season (Carrete & Donázar 2005). The preliminary analyses of telemetry data obtained in Dadia National Park showed that the vultures are covering similar areas like in Spain, for the breeding season 2004, the average home range of 6 birds was about 90,000 ha (Vasilakis et al. 2006).

Point locations estimated by three bearings were more accurate than point locations estimated by two bearings. The linear error of the point locations obtained by three bearings was on average about 1000 m, a distance easily travelled by Black Vultures. Only 5% of the point estimates were further than 2420 m from the true location (Figure C.1.2, Table C.1.1). Considering the amount of covered area and the distances between the receiver stations, the linear error was in the expected range, and was similar to findings of other researchers using large study areas with average linear errors ranging from of 261 m to 3000 m (Jenkins & Benn 1998; Marzluff et al. 1994, 1997; Zimmermann & Powell 1995). The average LE for locations obtained by two bearings was 1570 m, the 95% percentile of LE was 3900 m. Most of the locations obtained by two bearings should be precise enough for the telemetry study but care must be taken when using these data and they should be inspected. Marzluff et al. (1997) assessed an average linear error of 3000 m for data they used to compare the ranging behaviour of the Prairie Falcon (*Falco mexicanus*), a territorial raptor, much smaller than vultures, which range over smaller areas. The evaluated linear error in our study confirms that the applied telemetry system is precise enough to detect the patterns of range use and movement of the vultures.

All three estimators based on maximum likelihood theory provided precise location estimates. The Maximum Likelihood Estimator, developed by Lenth (1981), evaluates the most likely location for a given set of bearings using an iterative algorithm that tries to find the minimum angular error between the observed set of bearings and the estimated location of the signal. But the ML Estimator assumes data without outliers, an unrealistic assumption for data collected from wide-ranging animals in a mountainous study area (White & Garrott 1990). The M estimators (Lenth 1981) are also based on ML theory, but weight the bearings involved in a triangulation depending on their relative contribution to the estimated location (Lenth 1981; White & Garrott 1990). The most robust estimator, i.e. the estimator most insensitive against outliers, is the Andrews estimator and it is recommended for calculating location estimates when signal reflections are common, but it fails more often to produce successful estimates than the other ML-based estimators (White & Garrott 1990). In our study it was the most accurate estimator with significant difference. The Andrews estimator also provided in one more case a successful estimate than Huber and ML estimator.

The other, not-ML-based, estimators were less precise, but they always produced a location estimate. The Best Biangulation estimator only uses the two bearings with the intersection angle closest to 90° (ESS 1999; White & Garrott 1990). The Arithmetic, the Geometric and the Harmonic Mean of the bearing intersections use all available bearings, but they are estimators that are very sensitive for outliers. If one intersection point is very distant, the estimated location may be greatly displaced (White & Garrott 1990). In our study they produced some dislocated point locations, expressed as very high values of maximum linear error (Table C.1.1). They performed worse then the Best Biangulation estimator (Table C.1.1), thus the later was chosen to be the optimal substitute for the Andrews estimator. Concerning only the seven special cases, when the Andrews estimator failed to estimate the location, the Best Biangulation estimator performed worse than generally. The median linear error of these seven cases was 1897 m. Also Garrott et al. (1986) using a three-tower triangulation system evaluated the performance of other estimators for cases that the Andrews estimator failed to produce an estimate. They found out that MLE and Huber estimator provided correct estimates in only 12% and 10% of these cases. For these reasons, care must be taken when using locations obtained by substitutes of the failing Andrews estimator. For analyses that need high accuracy, these triangulations should be rejected or inspected with awareness.

Three factors determine the precision of a location estimated by telemetry (Saltz & Alkon 1985): variance around the bearings (error arc), distance from the receiving site (receiver station) to signal source (transmitter) and intersection angle of the bearings. The confidence ellipse (Lenth 1981) includes all these three independent factors, while knowledge of only one of them provides limited insight into the total variance of the estimated location (Saltz 1994). Other advantages of the confidence ellipse are that it can be computed easily and explicitly for each point estimate of the research data when the overall bearing error is assessed, and that it permits to set an objective threshold for data rejection (Enderson & Craig 1997; Marzluff et al. 1997; Morrison et al. 2003; Saltz & Alkon 1985; Tweed et al. 2003). The average size of the 95% confidence ellipses was about 1000 ha in our study, which is an area that can be described by a circle with a radius of 1800 m. The obtained coverage of 90.4% was close to the theoretical 95% and better than coverages evaluated by other researchers for ellipses ranging from 41% to 88% (Garrott et al. 1986; Saltz & White 1990; Zimmermann & Powell 1995). The coverage of the ellipse increased in a simulation study (Saltz & White 1990) when the SD of the bearing errors was increased form 1° to 5°. A precision of 5° appears to be more realistic for real data obtained by telemetry and closer to the value of 9.68° that was evaluated in our study and used for the computations of the ellipses. We recommend to compute confidence ellipses and to use them to describe confidence areas of telemetry data. Based on the distribution of ellipses we obtained in this study (Figure C.1.4), a threshold for data rejection between 2500 ha and 5000 ha can be recommended for DNP Eurasian Black Vulture data. Enderson & Craig (1997) applied a threshold of

5000 ha for location estimates of Peregrine Falcons (*Falco peregrinus*) after the evaluation of ellipses resulting from a study of error assessment. For point locations estimated by only two bearings, confidence ellipses are not available and were substituted by error polygons (Nams & Boutin 1991; Saltz 1994; Springer 1979), which consider also all three independent factors mentioned above. The computed error polygons offered a worse coverage than the ellipses (Table C.1.2). Only 55% of the true locations obtained by two bearings were actually inside the corresponding 95% error polygons. In the simulation study of Saltz & White (1990) the accuracy of the 95% Error Polygons increased from 75% to 89% when increasing the bearing precision from 5° to 1°, thus it seems that Error Polygons only describe confidence areas accurately for very low bearing errors. Regarding the study of Eurasian Black Vulture it is not recommended to use Error Polygons to determine confidence areas of the estimated point locations, but their shape can provide a useful tool to detect situations with unfavourable intersection angles.

Conclusions and Recommendations

1) The system of telemetry applied for the studies of Eurasian Black Vulture in DNP is without directional bias.

2) The accuracy of the applied telemetry system is determined with the evaluated standard deviation of 9.68°, and both methods of taking bearings provide the same accuracy.

3) The average linear error of three bearing point estimate is 1032 m.

4) The value of 2416 m can be applied as an overall 95% confidence distance around each point estimate obtained by DNP Eurasian Black Vulture telemetry.

5) The best performing estimator for the applied system in the topography of DNP is the Andrews estimator.

6) For the cases that the Andrews estimator fails to produce an estimate, the Best Biangulation estimator can be used, but the resulted point locations should be inspected carefully.

7) Two-bearing point estimates are less accurate than three-bearing point estimates. They provide on average a linear error of 1570 m. Point locations

based on only two bearings should be inspected and locations with suboptimal intersection angle should be rejected.

8) The 95% confidence ellipses should be computed for each point location. They provide an accurate measure of the confidence area and a useful tool for data rejection.

9) The 95% error polygons provide an inaccurate estimate of the confidence area. They can be used to detect situations with unfavorable intersection angles.

10) The average error distance of about 1000 m leads to the conclusion that the system is precise enough to estimate home ranges of the vultures and to determine main areas of foraging and the patterns of their movements.

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Chapter C.2. Monitoring breeding raptor populations – a proposed methodology using repeatable methods and GIS

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Running title: Monitoring breeding raptors population: a proposed methodology using GIS

Illustration by Tuisku Sarrala

Own contribution:

Study design 40%, implementation 40%, writing 50%

Abstract

Monitoring raptor populations is a difficult task, because birds of prey are wide-ranging, many are secretive and in some places very difficult to detect. In this paper, a systematic methodology for the monitoring of raptor populations is presented. This methodology was developed and implemented in Dadia National Park, north-eastern Greece, which hosts a diverse community of birds of prey in high abundance. It was applied by WWF – Greece in the framework of the monitoring plan established in the area, aiming at the evaluation of the raptor population trends in order to promote conservation measures. From 2001 until 2005, all species of diurnal raptors, except the large vultures *Aegypius monachus* and *Gyps fulvus*, were surveyed in 34 permanent plots, and a total of 4000-6000 annual observations of 22-24 species (17-18 breeding species) were collected during March to July. The observations were used to estimate raptor species' relative abundances and the numbers of territories. All the observations were entered in ArcGIS and the digitized observations were labelled, showing the number of individuals, age, sex, and bird behaviour under different symbols. For each species a spatially explicit territory analysis was performed, based on pre-defined criteria and the resulting breeding territories were classified in two categories: *confirmed* or *possible*. During the study period, the total number of territories was almost stable with an average value of 350 territories. Common Buzzard was the most abundant raptor having at average 120 territories and other nine species were found to have more than 10 territories.

Introduction

The decline of most bird of prey species has been relatively well investigated in Europe (Newton 1979, Cramp & Simmons 1980, Birdlife International 2004), but the estimation of their population status and trends pose special problems as raptors are usually dispersed, several are secretive, and in some places they are very difficult to detect due to the topography of the land (Fuller & Mosher 1987b). Additionally, their population may strongly fluctuate (Kirk & Hyslop 1998) and the monitoring of populations and the interpretation of their fluctuations requires specific and long-term studies (Catsadorakis 1994).

The assessment of population trends and the identification of the causes of population fluctuations could help in taking proper management measures (Vos et al. 2000), but comprehensive censuses and data collection on population dynamics have high requirements in personnel, time and cost (Noll West 1998). To overcome this problem, WWF Greece formulated a systematic monitoring plan for birds of prey in Dadia-Lefkimi-Soufli National Park (hereafter Dadia NP), northeastern Greece (Poirazidis et al. 2002) under the framework of Ecological Monitoring for Nature Management (Vos et al. 2000). Dadia NP holds one of the most diversified communities of raptor species across Europe, including endangered species such as the black vulture *Aegypius monachus*, the imperial eagle *Aquila heliaca*, and the white-tailed eagle *Haliaeetus albicilla,* and in fact 90% of European raptor species assemblage has been observed in this region (Hallmann 1979, Dennis 1989).

The main goal of the raptor monitoring was to estimate each year a relative abundance index of the breeding territorial raptor species through consistently repeatable methods, permitting data comparison throughout years. Relative abundance is used when it is difficult to overcome factors that impede the estimation of absolute densities. It is useful when comparing raptor populations against time, among sites or between species (Fuller & Mosher 1987b). The aim of this paper is to provide an overview of the methodology implemented in Dadia NP from 2001 to 2005 and to present the main findings of the five-year raptor monitoring.

Methods

Study Area

The Dadia NP is located in the centre of the Evros Prefecture, (E 260 20', N 410 15), and is part of the south-eastern tip of the Rhodope mountain range, with altitudes lying between 10 and 654 m, close to the border of Greece with Turkey (Figure C.2.1). Declared as a Protected Area in 1980, it includes now two strictly protected core areas, together covering 7290 ha, and a buffer zone covering 35170 ha. The landscape of the area is characterized by the sudden interchange of small and large valleys, by steep and shallow slopes, as well as an intricate hydrological network, composed of small and large streams. Seventy six percent (76%) of the area of Dadia NP is covered by forest vegetation, in which pine, mixed and oak forests are dominant while other vegetation types, such as broadleaf forests and maquis scrublands, participate with smaller proportions. The commonest pine forests are those dominated by calabrian pine *Pinus halepensis subsp. brutia*, while the corsican pine *Pinus nigra* develops smaller stands, usually close to streams. Four species of oak *Quercus spp.* are found in the oak and mixed forests of the area. In vegetation formations close to streams common alder *Alnus glutinosa* is dominant, and in some riparian places other species such as willow *Salix sp.*, black poplar *Populus nigra* and tamarisk *Tamarix spp*. The remaining area of Dadia NP is covered by grazing lands, fields and villages that interrupt the forested areas, creating characteristic mosaics of habitats and high landscape diversity (Schindler et al. 2008 [= Chapter B.1 of this thesis]).

Monitoring the populations of birds of prey

We conducted a systematic monitoring of raptor territories each year within the same area and for this reason the use of permanent plots was preferred to random plots (Millsap & Le Franc 1988). Several sampling methods exist to census breeding raptors. The three main ones are: a) line transects (surveys in a small area on either side along a line transect), b) point counts (surveys in specified areas around fixed points) and c) territory mapping (Fuller & Mosher 1987b). In this study we combined all three methods in the following way:

1. Surveillance of a fixed area from permanent view points with mapping of observations (*view points*).
2. Surveillance from a vehicle in predetermined transects with mapping of observations (*road transects*).

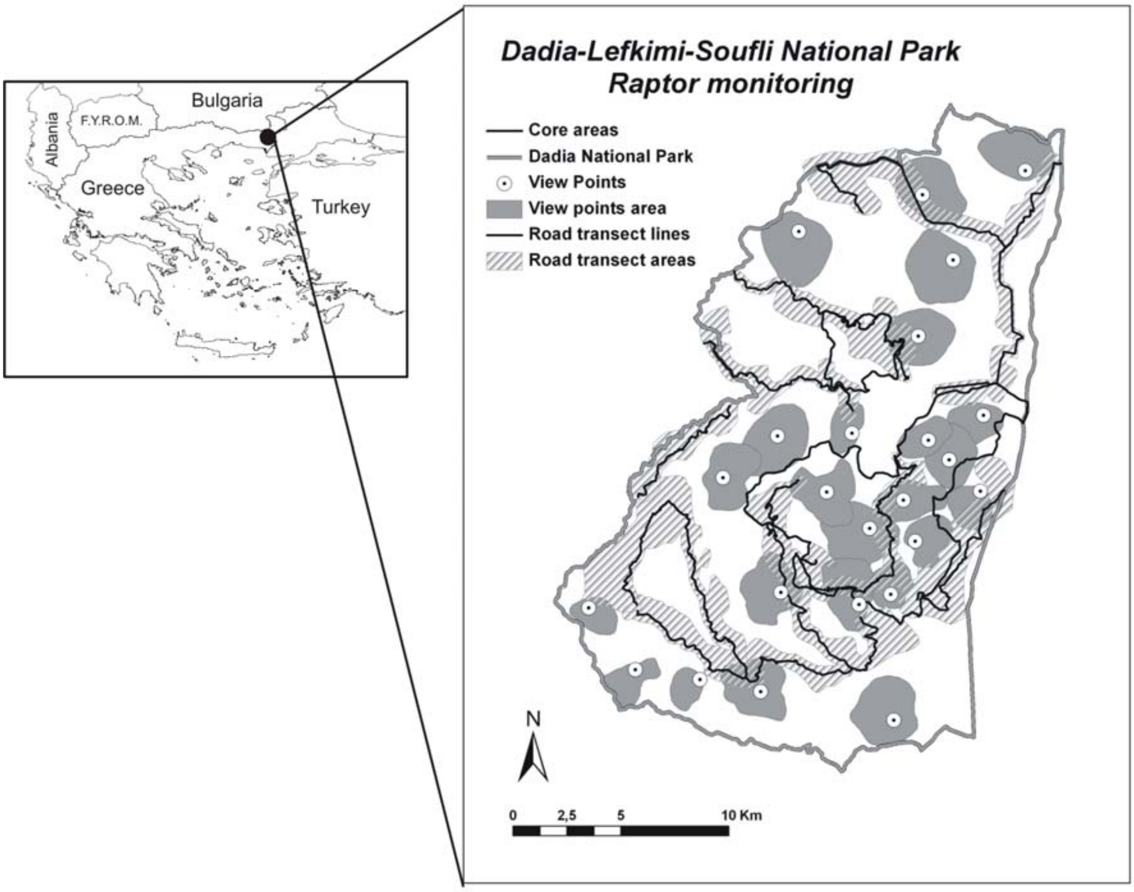

Figure C.2.1. *Sampling areas for the raptor monitoring in Dadia-Lefkimi-Soufli National Park.*

All territorial species of diurnal raptors were included in the systematic monitoring. These species were: white-tailed eagle, golden eagle *Aquila chrysaetos*, imperial eagle, lesser spotted eagle *Aquila pomarina*, short-toed eagle *Circaetus gallicus*, booted eagle *Hieraaetus pennatus*, egyptian vulture *Neophron percnopterus*, common buzzard *Buteo buteo*, long-legged buzzard *Buteo rufinus*, honey buzzard *Pernis apivorus*, black kite *Milvus migrans*, marsh harrier *Circus aeruginosus*, goshawk *Accipiter gentilis*, levant sparrowhawk *Accipiter brevipes*, sparrowhawk *Accipiter nisus*, peregrine falcon *Falco peregrinus*, lanner falcon *Falco biarmicus*, hobby *Falco subbuteo* and eurasian kestrel *Falco tinnunculus*. Furthermore, one non-raptorial species, the black stork *Ciconia nigra*, was included in this monitoring, as it shares the same ecosystem and has similar nesting and foraging requirements; additionally the local population is of national conservation importance (Handrinos & Akriotis 1997).

Since the reproductive periods of the species differ, it had to be ensured that the monitoring included the period in which each species emitted most cues of presence and reproductive behaviour (courting and pair formation displays, calls, clutches, etc). Furthermore, monitoring raptors presents difficulties due to their small population size and wide home-range. Thus, to increase both the sample size and the probability of key observations of all species, five surveys were carried out from March until July (one survey per month), covering each time all view points and road transects.

Twenty-four view points and 10 road transects were selected throughout the entire study area to monitor as much as possible of the raptor population (Figure C.2.1). Each survey was completed by two observers that alternated at sampling units, in order to reduce observer bias. Each observation was recorded in a field sheet and mapped on a field map with a scale of 1:10000 or 1:15000, and the following data were recorded: i) the species and the number of individuals, ii) the age and the sex of the individuals if feasible, iii) the time of the observation, iv) the type of activity of the individuals, v) the classification in migrating and local birds and vi) simultaneous observations with other individuals of the same species.

Selection of the permanent plots

Due to the topography of the area, the number of good vantage points was rather limited, and the definitive view points were selected using the following criteria:

- ✓ the point ensured the best and widest view of the neighbouring hillsides,
- ✓ the total area surveyed from all view points included all main habitat types in proportion to their availability,
- ✓ the points were distributed equally all over the expanse of the area without a bias towards habitats with already known high raptor presence,
- ✓ the access time to the view point from the nearest road should be short,
- ✓ the black vulture colonies were avoided to reduce disturbance.

The selection of road transects was based on the following criteria:

- ✓ their complementarities with view points and especially for covering raptor surveys within valleys where the positioning of good view points was impossible,
- ✓ the maximum coverage of the reserve jointly by the two methods.

The area covered by the established sampling plots was estimated as 66% of the total study area (12,668 ha covered by the 24 view points and 15,497 ha covered by the 10 road transects with an overall length of 149.6 km). Censuses from fixed view points offered great potential for detection of raptors in a radius of 1-1.5 km around the observation point; as the sampling plots were scattered all over the reserve, the uncovered area between them was small (Figure C.2.1), and most of the raptors (especially the bigger ones) that centered their territories in these intermediate zones could be detected from the neighboring sampling plots.

Territory estimation

The territory estimation processing followed a sequence of standard steps to permit comparison among the years:

1. The observation data were entered in seven different ArcGIS layers: general flights, territorial observations, landings, synchronous observations, nest areas, meeting points, meeting point flights. Each observation was represented as arrow and symbolized the movement of the observed birds. The labels showed the number of individuals, age, sex, and different activities under different symbols[1], as well as comments obtained during the field work. The GIS files were connected with the ACCESS database (where all the field data were initially entered and stored) to obtain all the available information in the GIS. Simultaneous observations were labelled as the maximum number of birds of the same species that had been seen at the moment of the observation. Characteristic symbols were used for Landmarks and Meeting Points and the important territorial observations were highlighted using thicker coloured lines.

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[1] *Activities recorded in the field and codes describing them. Those defined as territorial observations are marked as bold. Soaring S, Flying F, Gliding GL,* ***Display D, Landing L, Take off TOF****, Flying away AW,* ***Carrying food CF, Mobbing intraspecific Ma, Mobbing interspecific Mb, Calling CA,*** *Perching Pe,* ***Carrying nest material CNM, Family flight FF,*** *Meeting Point MP, Early Morning Flight EMF*

2. The territory estimation was done progressively per season based on the following criteria: a) possible nest sites, b) landings and take offs, c) territorial observations, d) simultaneous observations, e) non-intersection of bird flight lines, f) special circumstances per species, g) mean distance between nesting sites for species with marginal observations, and h) types of land cover and topography.

3. At the first stage, the estimation of each territory was done independently for each view point and each road transect, namely for 34 sampling plots. An example is given in Figure C.2.2a, where five territories of lesser spotted eagle were detected in the south-eastern part of Dadia NP in 2003. As territories extend beyond the boundaries of sampling plots and often the same territory continues onto the area of a neighbouring observation point, the results of the initial processing were used for further analysis combining and interpreting the territory polygons obtained by the estimations per view point and road transect. Based on this new interpretation, new polygons were created for the entire study area, representing the final result of the territory assessment per species. These polygons don’t necessarily cover the entire size of each territory, but include only the area confirmed by the raw data. In the previous example, the two territories identified by RT 4 and RT 6 were merged into one. In the area covered by VP 20 two territories were identified at the first stage (one confirmed and one possible) but at the overall analysis all the observations were consider to belong to the confirmed territories already identified (RT4-RT6 & VP20) and the possible territory was rejected (Figure C.2.2b).

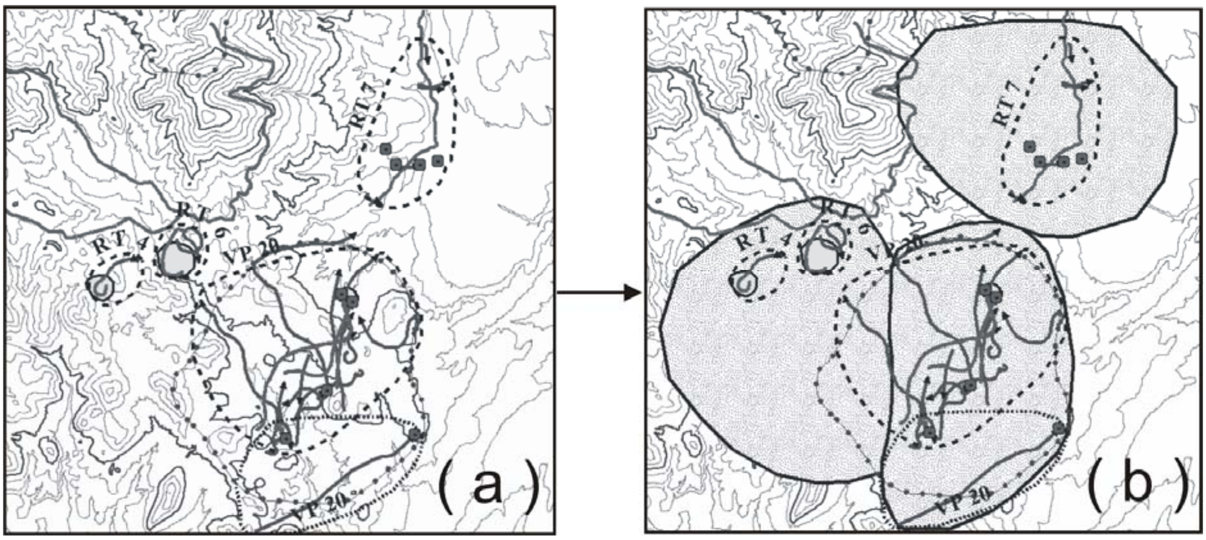

Figure C.2.2. *Example of territory estimation per sampling unit and overall estimation (lesser spotted eagle in the south-eastern part of Dadia NP in the year 2003, for details see text).*

We classified breeding territories as *confirmed* and *possible*, using *possible* when it could not be confirmed that the observations were obtained from separate individuals maintaining a separate territory. Considering the overall raptor population survey from 1999-2000 (Poirazidis 2003b), we made the assumption that the estimated number of confirmed territories was too conservative and that approximately 50% of the possible territories could be real territories. Therefore we estimated the total number of territories per species as the sum of confirmed territories plus 50% of the possible territories (Palma et al. 2004). An overview of the territory estimation is presented in Figure C.2.3.

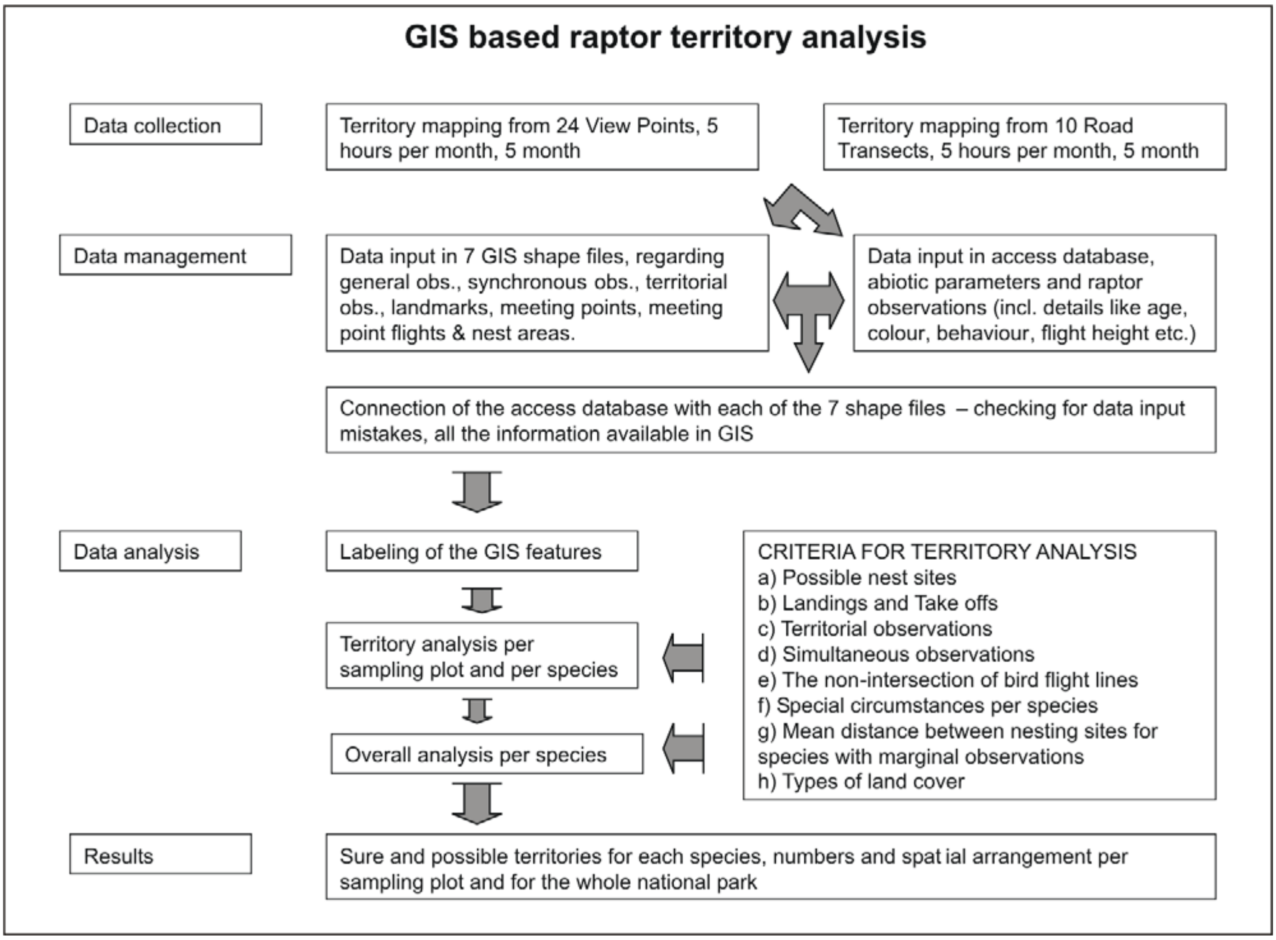

Figure C.2.3. *Methodological steps of the GIS based raptor territory analysis.*

The investigation of the fluctuation of the raptor population was done using a simple linear regression, with the total number of territories as the dependent variable and the five years of monitoring as the independent variable. Each data set was tested for normality with the Kolmogorov-Smirnov test. We report means ± S.D. for all measures of number of species and territories. Statistical tests were considered significant at $p < 0.05$.

Results

Total number of observations

By applying the sampling scheme of the systematic monitoring, several thousands of raptor records were collected each year. Most of the observations referred to the common buzzard; together with the observations regarding black storks and short-toed eagles, they comprised on average 74.9% of the total observations from view points and 75.1% from road transects. The egyptian vulture, lesser-spotted eagle and booted eagle formed a next group of species with an average, together, of 13.2 % of the total observations for both kinds of sampling plots. The rest of the species obtained less than 12 % of the observations. The proportions of the observations per species are presented analytically for each year in Table C.2.1 and Table C.2.2.

Table C.2.1. *Proportion of observations per species in view points during 2001 - 2005.*

Species	2001 (%)	2002 (%)	2003 (%)	2004 (%)	2005 (%)
Buteo buteo	39.56	26.57	26.94	25.45	29.27
Circaetus gallicus	21.12	24.54	24.39	23.68	23.88
Ciconia nigra	12.17	19.37	27.62	26.31	23.43
Neophron percnopterus	4.38	5.28	4.10	5.96	5.47
Aquila pomarina	4.43	5.99	3.99	5.12	4.23
Hieraaetus pennatus	3.92	3.37	3.69	3.29	2.80
Pernis apivorus	3.09	3.26	2.23	1.30	2.16
Accipiter nisus	2.44	3.34	2.43	1.20	1.49
Accipiter gentilis	1.89	2.57	1.26	0.88	1.63
Falco tinnunculus	2.07	1.78	0.91	2.22	0.96
others	4.93	3.94	2.43	4.58	4.69

Table C.2.2. *Proportion of observations per species in road transects during 2001 - 2005.*

Species	2001 (%)	2002 (%)	2003 (%)	2004 (%)	2005 (%)
Buteo buteo	42.26	31.84	25.34	31.15	29.70
Circaetus gallicus	18.45	27.14	31.72	25.02	20.33
Ciconia nigra	11.90	16.99	20.13	19.93	23.78
Aquila pomarina	4.76	5.02	4.76	7.30	6.33
Hieraaetus pennatus	4.17	3.95	4.04	2.95	3.50
Neophron percnopterus	3.57	3.31	3.14	5.66	3.81
Pernis apivorus	3.97	4.38	3.86	0.84	2.01
Milvus migrans	0.99	0.75	0.63	0.28	1.18
Accipiter nisus	3.97	1.82	2.34	1.22	1.60
Aquila chrysaetos	1.98	1.50	0.99	1.64	2.16
others	3.97	2.88	3.05	4.02	5.59

Number of species and territories

The total number of species observed in Dadia NP was 23-26 during the period 2001 – 2005 (March to July) and it reached 27-29 species, if black vulture and griffon vulture *Gyps fulvus* (species not included in the annual systematic raptor monitoring) and other raptor species observed by chance were included. The number of the observed species during the systematic monitoring was stable among the years having an average value of 24.8 ± 1.3 ($F_{1,3}$ = 0.045, p = 0.846). Among these species 19 to 20 bred in the area. The remaining species included raptors that used the area as a wintering place until March such as the spotted eagle *Aquila clanga*, or passage raptors like osprey *Pandion haliaetus*, bonelli's eagle *Hieraaetus fasciatus*, steppe eagle *Aquila nipalensis*, hen harrier *Circus cyneus*, montagu's harrier *Cyrcus pygargus*, pallid harrier *Circus macrourus* and the red-footed falcon *Falco vespertinus*. Finally the eleonora's falcon *Falco eleonorae* used the area late spring – early summer.

Number of territories per species

The number of territories of all the species ranged from 334 to 373 (Table C.2.3) and the average number was 349.4 ± 16.2, corresponding to a density of 82.4 terr/100km^2. Overall for all species, no statistical changes of the total number of territories was observed during the survey period ($F_{1,3}$ = 1.315, p = 0.335).

Table C.2.3. *Total number of estimated territories of the raptor species during 2001-2005.*

Territories	2001	2002	2003	2004	2005
Confirmed	305	331	325	311	352
Possible	58	53	43	50	42
Total*	334	357.5	346.5	336	373

The total numbers are the sum of the confirmed and the half of the possible territories

The number of territories was stable for most of the species and actually the eurasian kestrel was the only raptor species that showed a significant but marginal increase during the study period ($F_{1,3}$ = 10.208, p = 0.049). The average number of the territories was 17.4 ± 3.5 (4.1 terr/100km^2) and reached 22 territories in 2005 following an annual increase of 1.95 terr/year (Figure C.2.4). The commonest species in Dadia NP was the common buzzard with a density of 28.2 terr/100 km^2, representing 34% of the total number of breeding raptors in the area. The buzzard nested almost everywhere in Dadia NP with a nearest neighbour distance between nests of 1452 ± 358 m (Poirazidis 2003a). The density of the short-toed eagle was 8.7 terr/100km^2; it showed no significant population changes during the five years with an average number of 36.9 ± 3.8 territories ($F_{1,3}$ = 1.485, p = 0.31). The small fluctuation during the five years of monitoring reached high values of 40-41 pairs in 2002 and 2005 and a low value of 31 pairs in 2001 (Figure C.2.4). Other common species in descending order were the sparrowhawk, the black stork, the honey buzzard and the booted eagle (Table C.2.4).

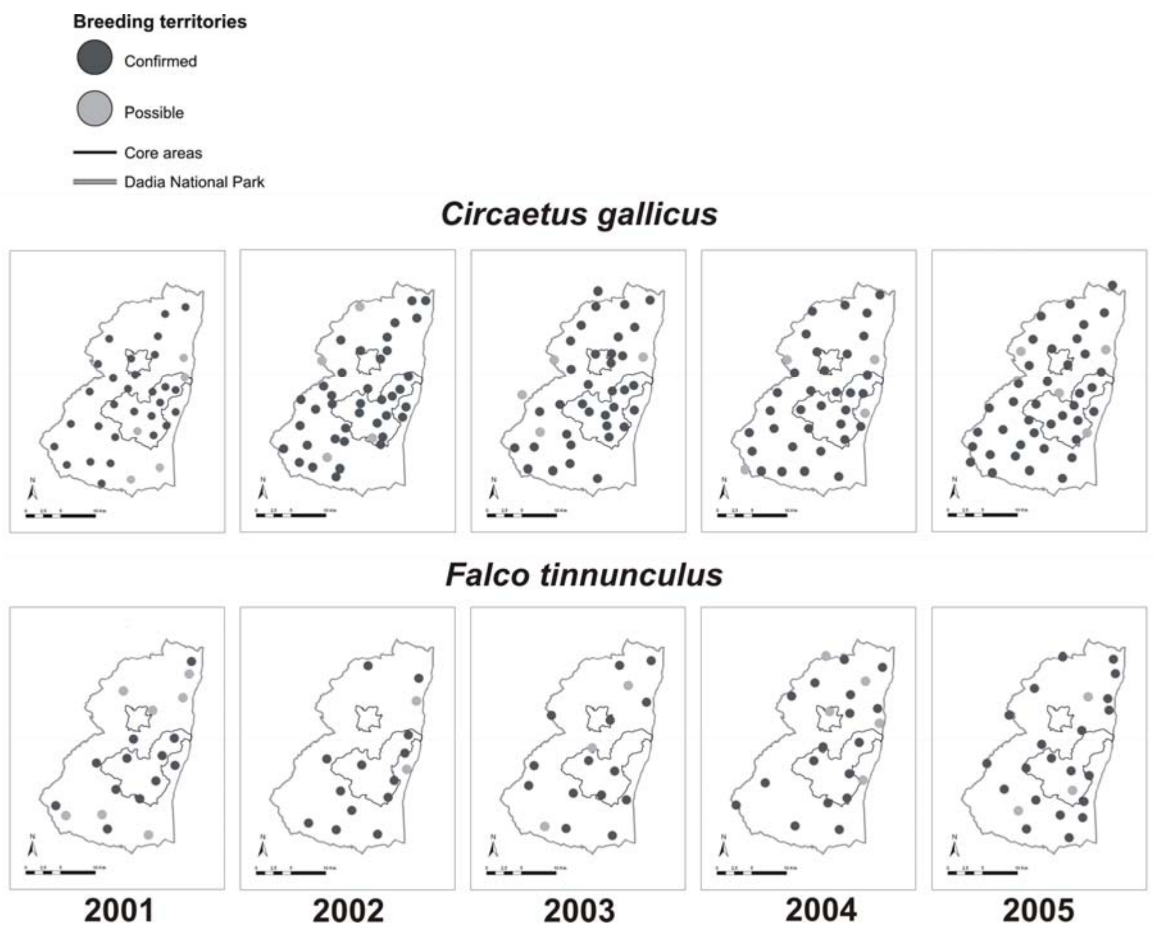

Figure C.2.4. *Centers of confirmed and possible territories of short-toed eagle and eurasian kestrel for the breeding seasons 2001-2005.*

Spatial distribution of territories

One advantage of the applied methodology based on the use of GIS in all stages is that the spatial distribution of the territories can be obtained as a direct result of the overall estimations per species. The spatial explicit output was stored in GIS (as example see Figure C.2.5 for the year 2005) and is therefore easily available for further analyses.

Table C.2.4. *Numbers of estimated territories per raptor species during 2001-2005.*

Species	2001	2002	2003	2004	2005
Vultures					
Egyptian Vulture	11	11.5	9.5	12	9.5
Eagles					
Golden Eagle	4	4.5	3.5	5	5.5
Imperial Eagle	1	0	1	1	0
Lesser-Spotted Eagle	17	20.5	18.5	20.5	22
Short-toed Eagle	31.5	40	37	35	41
Booted Eagle	20.5	20	18	21	20.5
Buzzards - Harriers - Kites					
Common Buzzard	110	128.5	125.5	112	122
Long-legged Buzzard	3.5	3	3	3	4
Honey Buzzard	28.5	27	23	18.5	24.5
Black Kite	0	0	0.5	0	0
Marsh Harrier	0	0.5	0.5	0	1
Hawks					
Goshawk	19	18.5	16.5	19.5	22.5
Sparrowhawk	36	29	31	27.5	32.5
Levant Sparrohawk	2.5	6	4	1	3
Falcons					
Hobby	6.5	9	3.5	8.5	7
Eurasian Kestrel	15.5	14	15.5	19.5	22.5
Peregrine Falcon	2	0.5	3	3	3
Lanner Falcon	1	0	0	0	0
Storks					
Black Stork	24.5	25	33	29	30.5
TOTAL	**334**	**357.5**	**346.5**	**336**	**373**

The presented numbers are the sum of the confirmed and the half of the possible territories

Discussion

Raptors are supposed to be good indicators of overall biodiversity (Sergio et al. 2006), but their monitoring is a time-intensive and difficult task. The monitoring of raptor populations has historically focussed on nests (Fuller & Mosher 1987b). But searching, observing and climbing of nests can include a high amount of disturbance and searching success can suffer from observer bias. On the other hand, the monitoring of territory occupancy has proved useful to trace the population trends of raptors in a feasible way (Katzner et al. 2007) and it was used to predict the implications of conservation measures (Carrete et al. 2002). Cost effectiveness is a key issue of assessments based on quantitative indicators (Atauri et al. 2005). In order to detect long term population changes of a diverse assemblage of birds of prey, a large amount of data is needed. and the integrated use of GIS based methods was found to be an effective tool for ecological monitoring (e.g. Joselyn 2003). Raptor populations can fluctuate considerably and if the monitoring is focusing only on rare species it is difficult to distinguish a directional trend due to external factors from "noise" o random elements (Palmer 1993). To distinguish chance fluctuation from actual trend, a long-term monitoring program is needed (Catsadorakis 1994). The methodology applied in this study permits cost effective overall surveys of raptor populations. Monitoring should not be viewed as a stand-alone activity, but instead as a component of a larger process of either conservation-oriented science or management (Nichols & Williams 2006) and the output of the implemented raptor monitoring (e.g. the spatial distribution of the territories) can be used effectively for management decisions and conservation. Due to the spatially explicit output, different years can be compared easily to evaluate stability and changes of the spatial distribution of the territories (Figure C.2.4). This latter aspect is very important especially in the case where one species could suffer from habitat degradation without showing any notable population decline. This has been observed for the lesser-spotted eagle in our study area as the breeding population of this species was stable during the last 25 years, but the spatial distribution of its territories has changed. The eagles abandoned their breeding sites in the interior of the forest, recorded by Hallmann (1979) in 1978, and nowadays all the pairs of this species have established their territories in the periphery of the National Park where the forest-meadow mosaic is still existing (Poirazidis et al. 2007a), thus making the population very sensitive to further reduction of suitable habitats (Väli et al. 2004b).

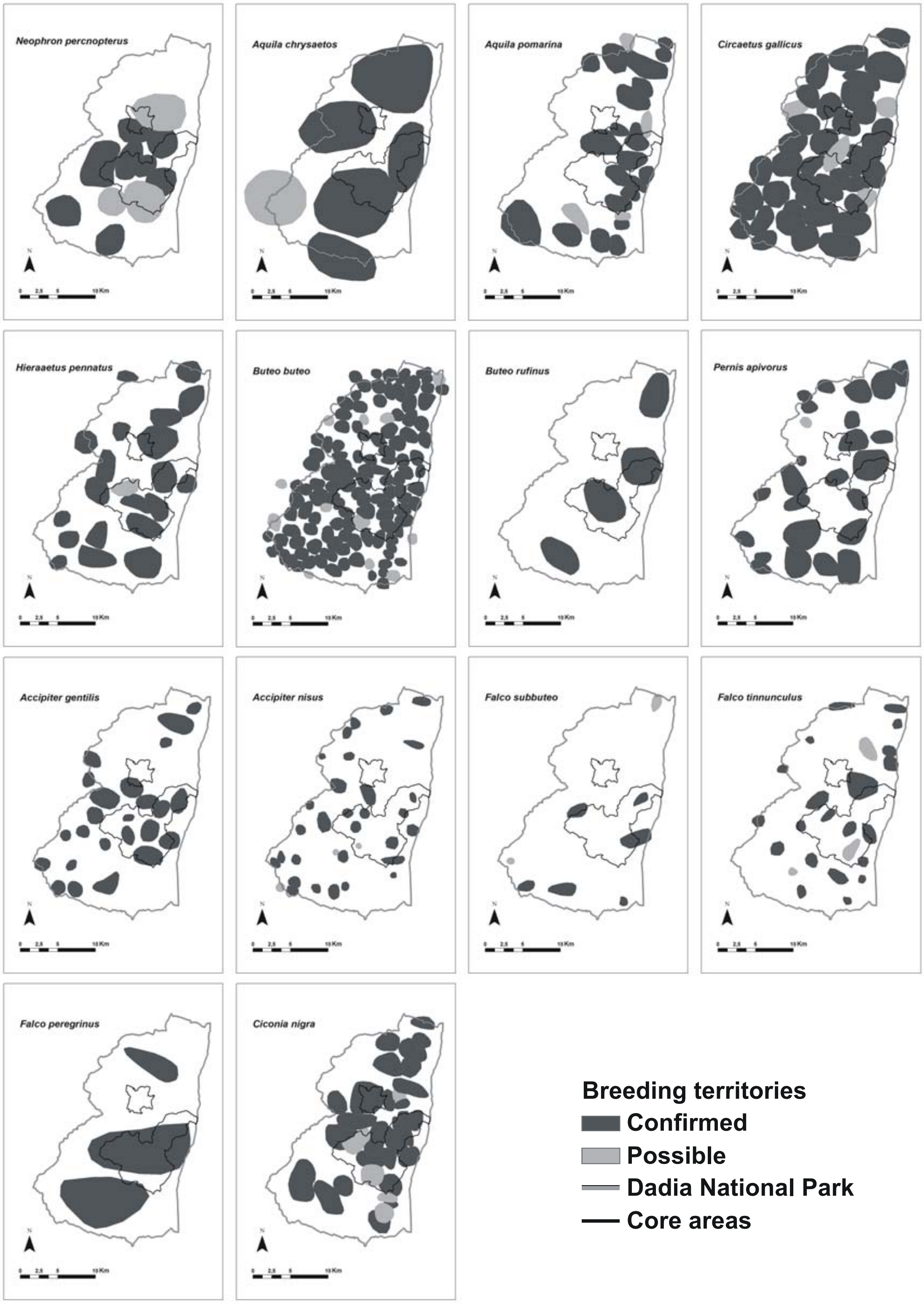

Figure C.2.5. *Confirmed and possible territories of 14 territorial raptor species, estimated for the breeding season 2005 in the framework of the systematic raptor monitoring. (These polygons don't necessarily cover the entire size of each territory, but include only the area confirmed by the raw data.)*

Differences regarding the usefulness of the method presented here could be due to the biological characteristics of the raptor species. The GIS approach of the analysis of raptor territories was very precise for typical territorial and relatively obvious species like most of the eagles, buzzards (*Buteo spp*.), and falcons (*Falco spp*.), but large amount of data is needed to increase the precision of the estimates for species that nest in high densities like the common buzzard. Some other species, like the hawks (*Accipiter spp*.), are very secretive and only few observations were obtained per sampling plot. For less territorial species, like the short-toed eagle, the black stork or the egyptian vulture, difficulties could arise. These species have a great overlap between neighbouring home-ranges, making the delineation of the different territories a difficult task. However, this problem was minimized by recoding about one thousand observations for both short-toed eagle and black stork every year. The territory estimation for the egyptian vulture were less problematic, because this species uses obvious nest sites in the rocks of Dadia NP, often easily detectable from view points or road transects and thus facilitating the overall territory estimation. The key issue for all the difficult estimations is to obtain more and good-quality data (like territorial observations, landings, etc.). An overview of the evaluation of the methodology per species is presented in the Appendix C.2.1.

The Dadia NP is one of the most important European forests for birds of prey. The integrated monitoring of their population trends combined with conservation-oriented management will contribute to safeguard their future (Poirazidis et al. 2010b [= Chapter D.1 of this thesis]). The proposed procedure can be applied to any ecosystem, region or country regardless of the raptor species being studied or their densities.

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Chapter C.3. Population trends in the diverse raptor assemblage of Dadia National Park, Greece

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Golden Eagle (Aquila chrysaetos) – Photo by Alexandors Gassios

Own contribution:

Study design 30%, implementation 40%, writing 65%

Abstract

Situated in northern eastern Greece, close to the border with Bulgaria and Turkey, the Dadia National Park is characterised by one of the most diverse range of breeding raptorial species in Europe. The first raptor survey was undertaken in the 1970s, but until 1999 most of the surveys were circumstantial and non-systematic. Considering some of these species are globally endangered and Annex 1 species of the Birds Directive, and raptors in general are considered key indicators of biodiversity and ecosystem health, a systematic raptor monitoring programme was established by WWF Greece in 2000. This paper presents the results of this programme including the population status, trends and breeding densities of raptors from 2001 to 2005. Between 18-19 species regularly bred in the area with a density ranging from 1 pair per 100 km^2 (e.g long-legged buzzard *Buteo rufinus* and peregrine falcon *Falco peregrinus*) to 30 pairs per 100 km^2 (common buzzard *Buteo buteo*). The total number of raptor territories was stable with an average of 321 ± 15.5 territories (77 territories/100km^2) with no overall trend and low fluctuations. Although the population size has increased for several species since the mid '90s, data from the first surveys in the 1970s suggest that some species are still recovering from the decline suffered in the '80s. Since the '70s the populations of six species have remained stable, whilst five species, such as the Egyptian Vulture *Neophron percnopterus* have shown a gradual decline. Black Vultures *Aegypius monachus* were the only species with a confirmed increase, with a further three species showing a probable increase. Due to insufficient data from the 70s the long term trend for four species, such as common buzzard, is unknown.

Resumen

Situado al noreste de Grecia, cerca de la frontera con Bulgaria y Turquía, el Parque Nacional de Dadiá está caracterizado por albergar una de las comunidades reproductoras de especies rapaces más diversas de Europa. El primer estudio sobre rapaces se llevó a cabo en los años 70', pero hasta 1999 la mayor parte de los estudios que se realizaron fueron ocasionales y no sistemáticos. Teniendo en cuenta que muchas de las especies presentes están globalmente amenazadas e incluidas en el Anexo 1 de la Directiva Aves y que las rapaces en general están consideradas como indicadores clave de biodiversidad y estado de conservación, en 2000 WWF-Grecia puso en marcha un programa sistemático de seguimiento de rapaces. En este artículo se presentan los resultados de este programa, incluyendo el estatus de la población, las tendencias y las densidades de territorios de nidificación desde 2001 hasta 2005. Habitualmente se reproducen en el área entre 18 y 19 especies de rapaces diurnas, con densidades que oscilan desde 1 pareja/km^2 (ej. busardo moro *Buteo rufinus* y halcón peregrino *Falco peregrinus*) hasta 30 parejas /km^2 (ej. busardo ratonero *Buteo buteo*). El número total de territorios de rapaces se mantuvo estable, con una media de 321 ± 15.5 territorios (77 territorios/km^2), sin una tendencia general y con fluctuaciones bajas. Aunque las poblaciones de varias especies se han incrementado desde mitad de los años 90, datos de los primeros estudios de los años 70, sugieren que algunas especies aún se están recuperando del declive sufrido en los 80. Desde los años 70 la población de seis especies se ha mantenido estable, mientras que cinco especies, como el alimoche *Neophron percnopterus*, muestran un declive gradual. Se ha producido un probable incremento para tres especies, un incremento seguro sólo para el buitre negro *Aegypius monachus*, mientras que la tendencia a largo plazo es desconocida para cuatro especies (ej. busardo ratonero) debido a la ausencia de datos de los años 70’.

Introduction

Raptors are supposed to be good indicators of overall biodiversity (Sergio et al. 2006). The decline of their European populations has been relatively well investigated (Newton 1979; Cramp & Simmons 1980; Birdlife International 2004), but meanwhile due to targeted conservation efforts, some vulnerable raptor species started to recover, e.g. in Central Europe (Literák et al. 2007; Kovacs et al. 2008; Probst 2009) and in Southern Europe (Olea et al. 1999; Suarez et al. 2000; Costillo et al. 2007; Skartsi et al. 2008). A systematic monitoring of the raptor populations is an essential tool to evaluate and improve conservation measures and to establish precise management actions (Witmer 2005). Monitoring raptor populations in large areas is a difficult task, because birds of prey are wide-ranging, many are secretive and in some places very difficult to detect. Comprehensive censuses for estimation of absolute densities have high requirements in personnel, time and cost, and relative abundance is used instead when comparing raptor populations against time, among sites or between species to assess population trends and proper management measures (Fuller & Mosher 1987a).

In this study we present an integrated monitoring plan for diurnal birds of prey based on GIS under the framework of the Dadia Systematic Scientific Monitoring (Poirazidis et al. 2002) in Dadia National Park, northeastern Greece. This protected area (hereafter Dadia NP) holds one of the most diversified communities of raptor species across Europe, including numerous breeding pairs of endangered species such as the black vulture *Aegypius monachus*, the lesser spotted eagle *Aquila pomarina*, the short-toed eagle *Circaetus gallicus*, and the booted eagle *Hieraaetus pennatus*, and in fact 90% of the European raptor species have been observed in this region (Table C.3.1; Hallmann 1979). The landscape in Dadia NP, although being still heterogeneous and diverse (Schindler et al. 2008 [= Chapter B.1 of this thesis]), has changed significantly during the last decades due to reforestation and land abandonment (Triantakonstantis et al. 2006) affecting the suitable habitat for many species (Poirazidis et al. 2007a; Bakaloudis 2009). The main goal of this monitoring was to estimate each year an index of relative abundance of the breeding territorial raptor species. We evaluated the number of raptor territories through consistently repeatable methods permitting unbiased data comparison throughout years.

After one year of evaluation, the Systematic Raptor Monitoring was implemented for five years (2001-2005). All the raptor species observed during the breeding season were included in the systematic monitoring, except black vulture and griffon vulture *Gyps fulvus*, because these species are colonial and the applied monitoring methods were not appropriate for colonial species. Additionally, an extensive survey for all breeding raptors of Dadia NP was carried out 1999-2000 to determine a base-line for the current status of the raptor populations (Poirazidis 2003b). Using also historical data for several species, the results of these surveys were used to evaluate both their population trends and the effectiveness of the conservation measures implemented in the area during the last 30 years.

Table C.3.1. *Status of the raptor species observed in Dadia NP. BM: Breeding-Migrating; M: Migrating; E: Extinct; FB: Former breeding; RM: Resident-Migrating; R: Resident; MW: Migrating-Wintering; RMW: Resident-Migrating-Wintering; BMW: Breeding-Migrating-Wintering; S: Summering.*

	English name	Latin name	Status
1	Honey Buzzard	*Pernis apivorus*	BM
2	Black Kite	*Milvus migrans*	M
3	Red Kite	*Milvus milvus*	E
4	White-tailed Eagle	*Haliaaetus albicilla*	FB
5	Bearded Vulture	*Gypaetus barbatu*	FB
6	Egyptian Vulture	*Neophron percnopterus*	BM
7	Griffon Vulture	*Gyps fulvus*	RM
8	Black Vulture	*Aegypius monachus*	R
9	Short-toed Eagle	*Circaetus gallicus*	BM
10	Marsh Harrier	*Circus aeruginosus*	MW
11	Hen Harrier	*Circus cyaneus*	MW
12	Pallid Harrier	*Circus macrourus*	M
13	Montagu's Harrier	*Circus pygargus*	M
14	Goshawk	*Accipiter gentilis*	RMW
15	Sparrowhawk	*Accipiter nisus*	RMW
16	Levant Sparrohawk	*Accipiter brevipes*	BM
17	Common Buzzard	*Buteo buteo buteo*	BMW
18	Steppe Buzzard	*Buteo buteo vulpinus*	BM
19	Long-legged Buzzard	*Buteo rufinus*	BMW
20	Steppe Eagle	*Aquila rapax orientalis*	E
21	Lesser-Spotted Eagle	*Aquila pomarina*	BM
22	Greater Spotted Eagle	*Aquila clanga*	W
23	Imperial Eagle	*Aquila heliaca*	RW
24	Golden Eagle	*Aquila chrysaetos*	R
25	Booted Eagle	*Hieraaetus pennatus*	BM
26	Bonelli's Eagle	*Hieraaetus fasciatus*	E
27	Osprey	*Pandion haliaetus*	M
28	Lesser Kestrel	*Falco naumanni*	BM
29	Kestrel	*Falco tinnunculus*	BMW
30	Red-footed Falcon	*Falco vespertinus*	M
31	Merlin	*Falco columbarius*	W
32	Hobby	*Falco subbuteo*	BM
33	Eleonora's Falcon	*Falco eleonorae*	S
34	Lanner	*Falco biarmicus*	R
35	Saker	*Falco cherrug*	E
36	Peregrine	*Falco peregrinus*	BMW

Study area and methods

Study area

Dadia NP is situated in the Evros Prefecture, north-eastern Greece (figure C.3.1) and has been declared a reserve since 1980 and National Park since 2003. It covers a forest complex extending over 427 km^2 including two zones of strict protection (core areas). Dadia NP is characterized by valleys and hills covered by extensive oak and pine forests including a variety of other habitats such as cultivations, fields, pastures, torrents and stony hills (Poirazidis et al. 2004; Schindler et al. 2008 [= Chapter B.1 of this thesis]; Catsadorakis & Källander 2010). The reserve is considered a local hotspot of biodiversity (Kati et al. 2004b; Poirazidis et al. 2010a [= Chapter A.1 of this thesis]).

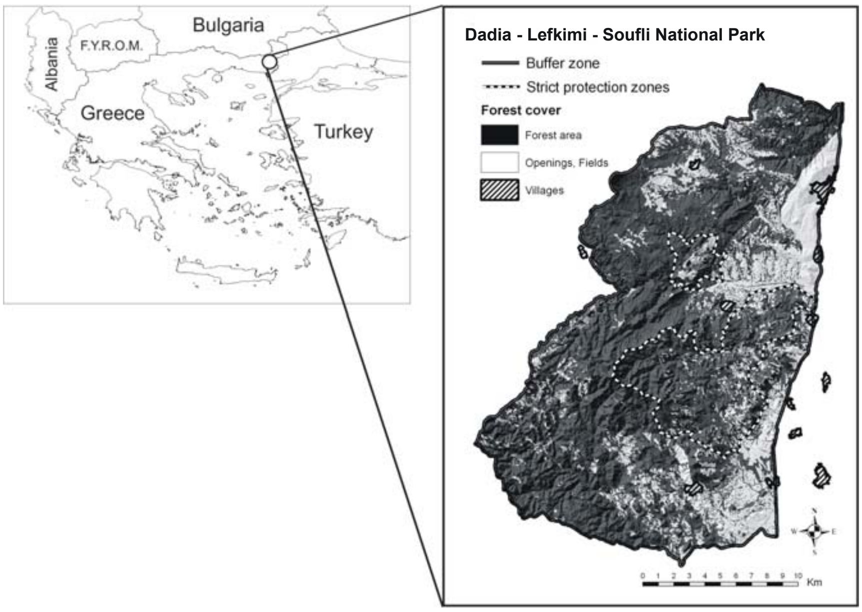

Figure C.3.1. *Dadia National Park located in north-eastern Greece.*

Population census and territories analyses

To survey raptors, we used well established sampling methods from permanent plots following Fuller & Mosher (1987a) and Millsap & Le Franc (1988). Given the habitat heterogeneity and relief diversity of Dadia NP (hilly relief with difficult access and lack of panoramic view points), we combined two sampling methods: 10 strip transects (surveys in a small area on either side along a line transect) and 24 point counts (surveys in specified areas around fixed points), and applied a particularly developed GIS based territory mapping approach for the analysis of these data (Poirazidis et al. 2009a [= Chapter C.2 of this thesis]). Five surveys were carried out each year from March to July (one survey per month) to be ensured that for each species monitoring included the highest cue-emission period for presence and reproductive behaviour (courting and pair formation displays, calls, clutches, etc.) and to increase the probability of important observations. Each observation was recorded and mapped, special attention was paid on the classification in migrating and local birds, territorial observations, type of activity and simultaneously observed individuals of the same species (Poirazidis et al. 2009a [= Chapter C.2 of this thesis]).

Data analyses for the territory estimation followed certain standard steps to permit comparison among years. The progressive analysis per season was based mainly on the following criteria: a) territorial observations, b) landings, c) simultaneous observations, d) space use and the non-intersection of bird flight lines, e) special circumstances per species, and f) topography and land cover types (Poirazidis et al. 2009a [= Chapter C.2 of this thesis]).

All the analyses were done with GIS and at the first stage the territory estimation was done independently for each view point and each road transect. As territories extend beyond the boundaries of sampling plots and often the same territory continues onto the area of a neighbouring observation point, the results per view point and road transect were interpreted and combined. Thus, new polygons were created for the entire study area, representing the final result of territory assessment per species. We classified breeding territories as 'confirmed' and 'possible', using 'possible' when it could not be confirmed that the observations were obtained from separate individuals maintaining a separate territory. Having in mind the overall survey of the raptor population during 1999-2000 (Poirazidis 2003b) many of the possible territories could be classified as real ones and so, the total number of territories per species for each year was estimated as the sum of confirmed territories plus 50% of the possible territories (Palma et al. 2004). The breeding density of each species was presented as territories per 100 km^2.

The investigation of the fluctuation of the raptor population was done using a simple linear regression, with the total number of territories as the dependent variable and the five years of monitoring as the independent variable. Each data set was tested for normality with the Kolmogorov-Smirnov test. We report means ± S.D. for all measures of number of species and territories. Statistical tests were considered significant at $p < 0.05$.

Compilation of the historical raptor data during 1979-2000

In this study we additionally reviewed previously published information of the raptors in Dadia NP to make some estimation of their long-term changes (Hallmann 1979; Vlachos 1989; Adamakopoulos et al. 1995; Alivizatos 1996; Poirazidis 2003b). Unfortunately, many of these studies were restricted to counts of vultures and larger eagles, while for the remaining species the data collected were clearly insufficient. Moreover, these studies used non-systematic methods to estimate the numbers of pairs, making the evaluation of the long-term population trend for several raptor species difficult or impossible. A general evaluation of the trend was done comparing the values of the estimates per species of the first published study (Hallmann 1979) with the latest survey (Poirazidis 2003b). The detected difference was presented as a percentage of the value of the Hallmann estimate. Actually, for 13 species enough data were available to permit the study of their population trends over a period of twenty two years (1979-2000). Based on this long-term general trend as well as on the results of the five years of systematic scientific monitoring, a first evaluation of the management actions in Dadia NP was completed and proposals for proper management measures were described.

Results

Populations status and trends of birds of prey during 2001-2005

The total number of observed species was 22-25 per year during the period 2001 – 2005 and it reached 26-28 species per year, if we add the species observed outside the systematic monitoring as well as the Black Vulture and the Griffon Vulture (species not included in the annual systematic raptor monitoring). The number of the species observed each year during the systematic monitoring was stable having an average value of 23.8 ± 1.3 ($F_{1,3} = 0.045$, $p = 0.846$). From these species 18 to 19 bred in the area. The remaining species include raptors that use the area for wintering until March such as the spotted eagle *Aquila clanga*, or passage raptors like osprey *Pandion haliaetus*, bonelli's eagle *Hieraaetus fasciatus*, montagu's harrier *Circus pygargus*, pallid harrier *Circus macrourus*, and the red-footed falcon *Falco vespertinus*. Finally the eleonora's falcon *Falco eleonarae* uses the area late spring – early summer.

The number of territories at breeding time (*confirmed plus 50% of the possible ones*) was very stable for most of the species (see figure C.3.2). The breeding density of the species is ranging from about 0.8 pairs per 100 km² (e.g. long-legged buzzard, levant sparrowhawk, peregrine) to about 30 pairs of common buzzard per 100 km² (Table C.3.2). The total number of territories of all the species ranged from 307 to 342 and the average number was 320.9 ± 15.5 having a density of 76.6 terr/100km². Overall for all species, no change of the total number of territories has been observed ($F_{1,3}$ = 1.315, p = 0.335).

Table C.3.2. *Diurnal raptors populations in Dadia National Park. Historical numbers of territories, percent of change between 1979 and 2000, and breeding densities (average value of territories per 100 km² from 2001 to 2005).*

Study reference	**Hallmann (1979)**	**Vlachos (1989)**	**Adamakopoulos et al. (1995)**	**Poirazidis (2003)**	**Percent change***	**Territories per 100 km²**
Year of survey	**1979**	**1987**	**1993-94**	**1999-2000**		**2001-2005**
Vultures						
Bearded Vulture	*no data*	1ind	1ind	0	-	0
Black Vulture	5	12-15	20	20	+ 300	4.7
Griffon Vulture	0	8-10	8-12	0	-	0
Egyptian Vulture	17	20-25	10-14	13-14	- 21	2.5
Eagles						
White-tailed Eagle	1	1	0	0	- 100	~0
Golden Eagle	5	4-5	3-4	4	- 20	1.1
Imperial Eagle	3	1	0	1	- 67	~0
Lesser-Spotted Eagle	19	16-20	14-17	20	+ 5	4.6
Short-toed Eagle	21	13-16	20-23	37-40	+ 83	8.7
Booted Eagle	9	8-10	20	21-25	+ 156	4.7
Medium-sized raptors						
Common Buzzard	*no data*	15-20	16-20	120-130	-	28.2
Long-legged Buzzard	7	5-10	7-9	4	- 43	0.9
Honey Buzzard	*no data*	2-4	10-12	25-30	-	5.7
Hawks						
Goshawk	18	10-15	10-12	21	+ 17	4.5
Sparrowhawk	*no data*	5-10	8-10	35	-	7.3
Levant Sparrohawk	*no data*	*no data*	8-12	7	-	0.8
Falcons						
Hobby	*no data*	?	3-5	12	-	1.6
Kestrel	*no data*	*no data*	5-10	20	-	4.1
Peregrine	1	*no data*	1	2-3	+ 150	0.5
Lanner	2	1	1	1-2	- 25	~0

**The evaluation of the trend between 1979 and 2000 was done comparing the mean values of the two estimates. Their difference is presented as the percentage of the value of the Hallmann (1979) estimate.*

The white-tailed eagle was extinct as a breeder in 1990. In 2004 immature and adult individuals started again to be present in the area during spring and summer, and in 2005 a possible territory was recognized, since a subadult pair used the old breeding area during the spring season. The number of breeding golden eagles increased from 4 pairs in 2001 to 5 and one possible in 2005, the average density was 1.1 terr/100km^2 (Table C.3.2) with an annual increase of 0.35 terr/year, but this tendency was not statistically significant ($F_{1,3}$ = 2.882, p = 0.188). The imperial eagle returned as a breeding species during the monitoring period, after being absent as breeder from Dadia NP since 1992. An adult pair used the area during 2000. In the following year (August 2001), we observed the common flight of one adult and one first calendar year bird in the core area of Dadia NP and proved the breeding of this species for 2001. In 2002 there were no observations, while they appeared again for the following two years but disappeared again in 2005. The lesser spotted eagle had an average number of 19.7 ± 1.9 territories (4.6 terr/100km^2) during the five year monitoring period, reaching 22 pairs in 2005 ($F_{1,3}$ = 5.66, p = 0.098).

The density of the short-toed eagle was 8.7 terr/100km^2 (Table C.3.2); it showed no significant population changes during the five years with an average number of 36.9 ± 3.8 territories ($F_{1,3}$ = 1.485, p = 0.31). The small fluctuation during the five years of monitoring reached high values of 40-41 pairs in 2002 and 2005 and a low value of 31 pairs in 2001 (figure C.3.2). The booted eagle appeared to be stable during the five years of censuses with a density of 4.7 terr/100km^2 and an average number of 20 ± 1.2 territories ($F_{1,3}$ = 0.056, p = 0.829). The egyptian vulture had an average number of 10.7 ± 1.2 territories ($F_{1,3}$ = 0.401, p = 0.572). The estimated density was 2.5 terr/100km^2 and the confirmed territories (n = 9) were stable during the five years of monitoring and constituted the main breeding population in Dadia NP, while the number of possible territories, probably belonging to non-breeding pairs, was unstable (figure C.3.2).

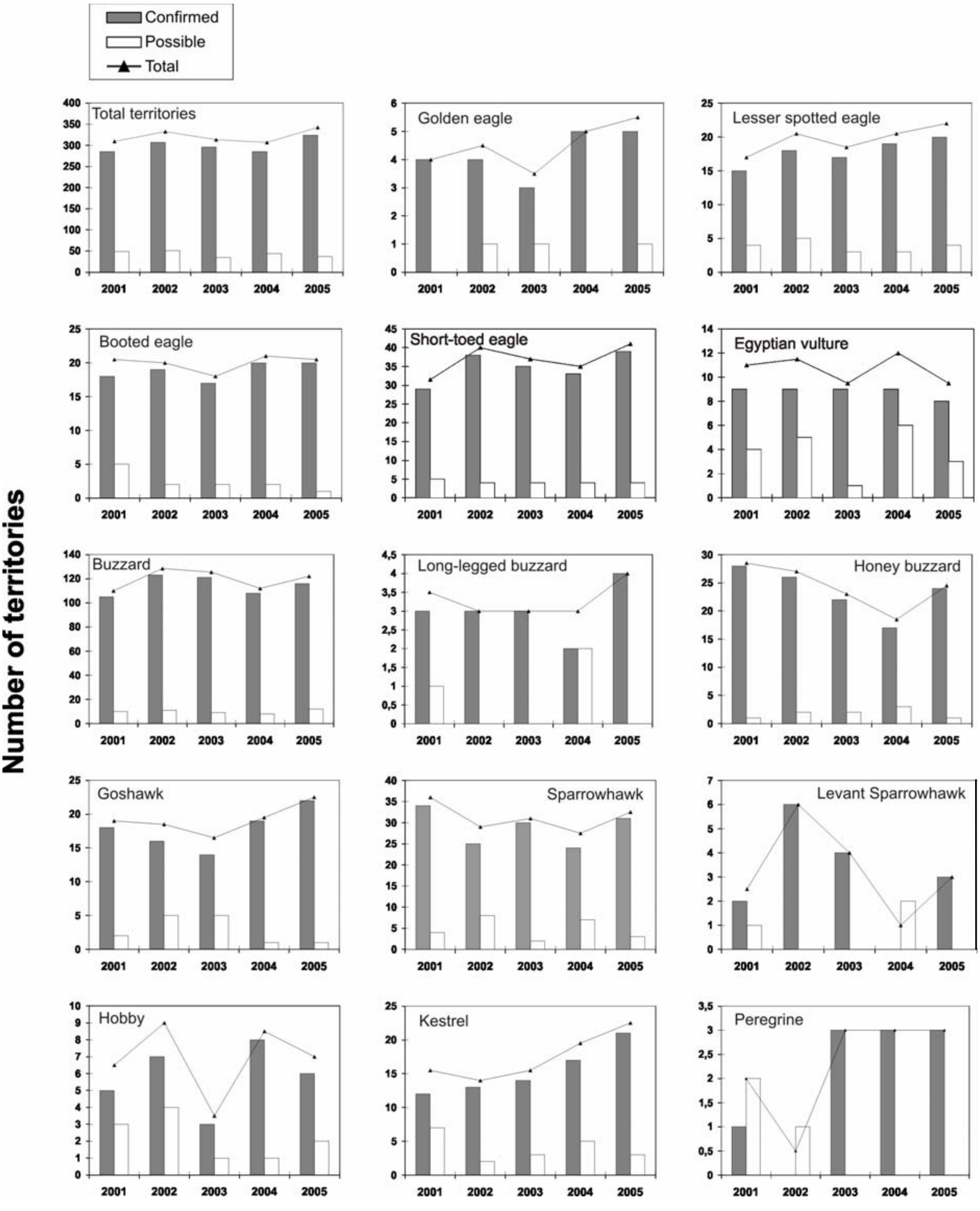

Figure C.3.2. *Changes in the number of territories of raptor species in Dadia National Park during 2001-2005 (Total territories = confirmed plus the half of the possible ones).*

The population of the common buzzard was very stable, having an average number of territories of 119.6 ± 8.2 ($F_{1,3}$ = 0.064, p = 0.817). This species was the most abundant raptor in Dadia NP with a density of 28.2 terr/100 km^2 representing 34% of the total number of breeding raptors in the area. The long-legged buzzard is meanwhile a rare species in Dadia NP. During the monitoring period the small population was very stable holding a core of 3-4 pairs (figure C.3.2) and a density of 0.9 terr/100km^2. The population of the honey buzzard changed considerably during the monitoring years. From a peak of 28 pairs in 2001, it declined to 18 pairs in 2004, but increased again in 2005 (figure C.3.2). The average number of pairs was 24.3 ± 3.9 ($F_{1,3}$ = 2.469, p = 0.214) and the density 5.7 terr/100km^2. The black kite holds a good breeding population along the riparian forest of the Evros river, and some of these birds visit often the eastern part of the National Park. In 2003, the pine forest close to Dadia village was possibly used for nesting. The marsh harrier breeds very close to the southeastern border of the National Park in an extensive reed bed. During 2002-2003, one or two females possibly reproduced inside the southeast border of Dadia NP.

The goshawk population was stable during the monitoring period with an average number of territories of 19.2 ± 2.2 ($F_{1,3}$ = 1.548, p = 0.302) and a density of 4.5 terr/100km^2. The last twenty years the overall population of this species didn't change significantly (Table C.3.2). The sparrowhawk population was very stable during the five years of censuses having an average number of territories of 31.2 ± 3.3 ($F_{1,3}$ = 0.601, p = 0.495) and a density of 7.3 terr/100km^2. The species is secretive, difficult to spot and due to the applied methodology it was expected that some fluctuations would result. The levant sparrowhawk's main breeding area is along the Evros river at the border with Turkey. Only a few pairs breed inside the National Park, where a maximum population of six pairs was observed in 2000 and 2002 (figure C.3.2). In the other years we observed even fewer pairs (only one in 2004 and three in 2005) but as for the sparrowhawk, the detectability of this species is low.

For the last 20 years, only one breeding pair of peregrine was believed to occupy Dadia NP, but a new territory was verified in 2001 and from 2003 on the estimated number of pairs were three. Unfortunately this increase of peregrines followed the extinction of the single pair of lanner in 2002. The hobby holds a very stable populations in Dadia NP where the maximum number of territories (n = 9) was estimated during 2002, with an average number of pairs of 6.9 ± 2.2 for the period

2001-2005 ($F_{1,3}$ = 0.004, p = 0.953) and a mean density of 1.6 terr/100km^2. The kestrel is actually the only raptor species that showed a significant but marginal increase during this period ($F_{1,3}$ = 10.208, p = 0.049). The average number of the territories was 17.4 ± 3.5 (4.1 terr/100km^2) during 2001-2005, and reached 22 territories in 2005 following an annual increase of 1.95 terr/year (figure C.3.2).

Long-term changes of the populations of birds of prey

Up to 1970, twenty-four raptor species used to breed in the area (Hallmann, 1979) and Dadia NP constituted one of the few European regions where four vulture species occurring in Europe were observed together: black vulture, griffon vulture, egyptian vulture and bearded vulture *Gypaetus barbatus*. The bearded vulture nested here until 1969 but then only one individual was observed until it disappeared in 1994. Over the last three decades, four more species ceased to nest in Dadia NP: the white-tailed eagle, the imperial eagle, the bonelli's eagle and the lesser kestrel *Falco naumanni* (Adamakopoulos et al. 1995). In 1999 seventeen (17) diurnal raptor species nested within the borders of Dadia NP while in 2000 the number of breeding species increased to 18, when an active territory of imperial eagle was confirmed after an absence of 8 years (Poirazidis 2003b). In 2005, a new territory of the white-tailed eagle, which successfully bred until 1990, was possibly re-established in the area. However, in contrast to these positive changes, no breeding attempts were observed for the lanner in Dadia NP after 2002. Seventeen (17) species are wintering in the area (Table C.3.1), three of which are present only during winter, among which, a considerable population of the spotted eagle. In addition, several individuals of the white-tailed eagle, the imperial eagle and the long-legged buzzard winter in the area.

Comparing the 1979 and 1999-2000 survey data (Hallmann 1979; Poirazidis 2003b) it can be seen that the populations of most raptors appear to be stable, while the black vulture showed a strong increase (Table C.3.2). On the contrary, the breeding populations of imperial eagle, long-legged buzzard and egyptian vulture have declined and the white-tailed eagle as well as the breeding population of the griffon vulture has become extinct (Table C.3.2). With 17 territories in 1979, the egyptian vulture reached 25 territories in 1987, but the next years the population declined dramatically to 10 - 14 pairs. The population of the long-legged buzzard consisted of seven pairs in 1979 (Hallmann 1979), but Alivizatos (1996) found only five in 1990, and the following years one more pair disappeared from the area. The species showed a considerable 43% decrease from 1979 to 2000 (Table C.3.2). The population size of the lesser spotted eagle seemed to be stable during the last twenty years (Table C.3.2). Nineteen (19) pairs of this species were recorded in 1979 (Hallmann 1979), while a population of 16-20 pairs was estimated for the year 1987 (Vlachos 1989), a number similar to the current population (Table C.3.2). Also other species such as the golden eagle, the goshawk, the short-toed eagle, and the booted eagle still preserve their traditional territories within Dadia NP as these were recorded in the '70s, although positive upward trends seem to have occurred for the short-toed eagle and the booted eagle (Table C.3.2). Bakaloudis et al. (2005) found 22 active territories of short-toed eagle in Dadia NP in 1997 a number closer to the first estimated population (Hallmann 1979), and Adamakopoulos et al. (1995) found in 1993-94 twenty (20) pairs of booted eagle a number similar to the current population of 21-25 pairs (Table C.3.2). For the most common species like common buzzard and sparrowhawk, no data were available from the first period (Table C.3.2).

Discussion

Two kinds of conclusions can be drawn from the results; the first one regards the methodology of the monitoring. Raptors are usually dispersed, nest at low densities and their population may strongly fluctuate (Fuller & Mosher 1987a; Kirk & Hyslop 1998), and monitoring of their populations and the interpretation of their fluctuations requires specific and long-term studies (Catsadorakis 1994). The monitoring plan implemented in Dadia NP contained an integrated GIS based method for the collection and analysis of the observations in order to manipulate the big amount of information (Poirazidis et al. 2009a [= Chapter C.2 of this thesis]). Due to particularities of raptor ecology and behaviour the used methodology presented biases for particular species so much during data collection in the field, as in the analysis. Low detection rates of small woodland raptors such as the hobby and sparrowhawk could have led to high fluctuations and affected the trend among the years. The territory estimation for the short-toed eagle presents some ambiguities and under- or over-estimation is possible. The weak territoriality of this species has been the main reason that the applied methods were incapable of recognizing correctly all territories (Bakaloudis et al. 2005). A similar problem occurred with the egyptian vulture estimation. But, precision of estimations increased with quality and number of observations and for most of the rather strictly territorial species, such as golden eagle, lesser spotted eagle, booted eagle, long-legged buzzard, common buzzard, peregrine, lanner and kestrel, the applied methodology was effective and precise (Poirazidis et al. 2009a [= Chapter C.2 of this thesis]).

The second conclusion is directly related to the estimation of the population trends during the last twenty-five years. According to the kind of population trends observed from 1979 to 2005 we can classify the species into five groups. In the first group are species with increasing populations. The black vulture actually was the only species that exhibited an increase, due to the protection of nesting sites, the supplementary feeding, and reduction of threats such as poaching and habitat degradation. But the increase lasted only until 1994, since then the population is stable (Skartsi et al. 2008; Skartsi et al. 2010a).

Species with a probable population increase can be classified into a second group. The short-toed eagle and the booted eagle showed a considerable increase (83% and 153% respectively). Although possibly a serious under-estimation in 1979 affected these results, the improvement of the forest conditions in Dadia NP during the last decade towards a more conservation-friendly management could have improved the nesting habitats of these forest species and consequently their population size (Bakaloudis et al. 2001; Poirazidis et al. 2007a). A major increase of booted eagle populations was recorded the last decades in western Europe and this probably was attributed to species adaptability in changing environments (Carlon 1996). In Donana National Park (south-west Spain) the booted eagle increased from six pairs in the early 1980s to 150 in 2000 (Suarez et al. 2000). According to the 2001-2005 surveys, both species had dense and very stable populations in Dadia NP indicating that they have reached the carrying capacity in the area. In central Italy a lower density (2.05 pairs/100km^2) was estimated for short-toed eagle (Petretti 1988) than the one reported by Bakaloudis et al. (2005) for Dadia NP (5.9 pairs/100km^2). Another species that showed an increase during the last twenty years was the honey buzzard, although this trend is likely an artifact caused by the under-estimation of the first years.

The large and sensitive raptor species belong to a third group which exhibited a population decrease the last 25 years. The species that were at the extremely lower limit of one pair such as the white-tailed eagle and the imperial eagle became extinct when conditions turned unfavorable. The habitats of the area have changed (Triantakonstantis et al. 2006) and the conditions that favoured the populations of these large-sized raptors i.e. a mosaic of open habitats in the forested area and a suitable number of key prey species, stopped to exist (Adamakopoulos et al. 1995; Poirazidis et al. 2007a; Bakaloudis 2009). Furthermore the human persecution seriously affected the last breeding pairs of the large eagles (Hallmann 1985; Jerrentrup 1988). At the end of the 1970s, the Evros region held 5-7 pairs of imperial eagle (Hallmann 1979) but in 1986 only two pairs remained in the area and the last confirmed nesting in Greece was recorded in Dadia forest in 1990 (Hallmann 1996). A similar chronology took place in the neighboring countries such as Bulgaria, where the population of imperial eagle dropped in 1993 to 15-20 pairs (Petrov et al. 1996) but recovered recently to a stable number of 20-25 pairs (Stoychev et al. 2004). The return of this species to Dadia NP as a breeding species is a very hopeful message for the effectiveness of the conservation measures of the last 15 years. The long-legged buzzard and the egyptian vulture lost many of their traditional territories mainly in the forested area and they occupy now the lowlands where a mosaic of habitats prevailed. Colonies of european susliks *Spermophilus citellus* occurred in 10 of the 16 territories of long-legged buzzard found in Evros region in 1993 (Alivizatos & Goutner 1997). The observed decline of the raptor species in Dadia NP was probably affected by the progressive disappearing of the colonies of susliks of which the last colony survived up to 1995 (Adamakopoulos et al. 1995, Alivizatos & Goutner 1997). The breeding pairs of egyptian vulture declined rapidly after 1987, to a current population of nine pairs. Many of the old nesting sites remain unoccupied and the operation of the vulture restaurant didn't improve the status of this species (Vlachos et al. 1995). The factors affecting the breeding population are still unknown, probably associated with the wintering grounds in Africa, and must be further investigated. The griffon vulture disappeared during this period as a breeding species, but the total number of individuals (mainly juveniles and immature) increased due to the long-term operation of the supplementary feeding. In 2007, three breeding pairs of griffon vulture nested at the traditional breeding rocks in Dadia NP (Skartsi et al. 2010b).

A fourth group of species showed generally stable populations during the last 25 years. The golden eagle lost some of the territories occupied in 1979, but in the recent years new pairs have been observed in the area. Many observations referred to immature birds flying around the unoccupied areas for many years, an indication of the establishment of new pairs. During the breeding season, the main prey of the golden eagle in Dadia NP are tortoises, which abound in the forest area (Adamakopoulos et al. 1995). The improvement of the environmental awareness of the local people about the importance of raptors might have minimized persecution of big raptors during the last years and helped the recovery of these birds. The lesser spotted eagle is a priority species for conservation, for which large-scale action was drafted in a recent European Action Plan (Meyburg et al. 2001). During the last decades major changes occurred in the landscape of Dadia NP (Triantakonstantis et al. 2006) and in the Evros region in general. These changes, which include intensification of the agriculture, intensive forest exploitation and reduction of livestock, had significant impacts on the birds of prey (Bakaloudis et al. 1998b). It was expected that land use change in the foraging areas of lesser spotted eagle (Vlachos & Papageorgiou 1996) such as decrease of wetlands and decrease of the mosaic character of habitats would have affected its populations but the breeding population of this species was stable the last 25 years. However, the spatial distribution of its territories has changed with the abandonment of the breeding sites in the interior of the forest and the establishment of new ones in its periphery (Poirazidis 2003b), thus making the population sensitive to further reduction of the suitable habitats (Vali et al. 2004). Five small ponds were created in the core areas of Dadia NP under the framework of a LIFE-Nature project to enhance the abundance of prey (amphibians) and to support the isolated pairs of this species (WWF Greece 2006). This action is also expected to affect positively the breeding population of the black stork *Ciconia nigra* in the area. Goshawks showed a very stable population and spatial distribution in their traditional breeding areas (Hallmann 1979; Poirazidis 2003b) and the raptor-friendly management of the forest in the National Park will further improve the suitable nesting habitats (Alexandrou et al. 2008). The density of Goshawk in Dadia NP is similar to populations in Italy (Penteriani & Faivre 1997a) and in Finland (Solonen 1993). The observed changes of the status of the large falcons (peregrine and lanner) are very difficult to explain as both species use similar nesting and foraging habitats. It is possible that the inter-specific competition among them combined with the expansion of peregrine caused the extinction of lanner.

Finally there is a group of species with unknown population trends as we do not have historical records or the relevant information is limited. The common buzzard is a flexible generalist species and nests almost everywhere having a 1452 ± 358 m nearest neighbour distance between nests (Poirazidis 2003a). The estimated number of common buzzards was not the complete population of the National Park, because probably 15-20 % of the territories remained uncovered. It seems that this species has covered all the available habitats and its population may be limited now mainly by intra-specific competition (Poirazidis 2003a). In Dadia NP, common buzzards have a density of 28-30 terr/100km^2, very similar to that found by Sergio et al. (2002) in the Italian Pre-Alps (28 to 31 pairs/100km^2). These populations are denser than in central Italy where they were estimated at 19.7 pairs/100km^2 (Cerasoli & Penteriani 1996) and 8.3 pairs/100km^2 with a mean distance between nesting territories of 2.5 km (Penteriani & Faivre 1997b). In the UK, Dare & Barry (1990) found densities ranging from 5.9 to 14.1 pairs/100km^2, although the mean nearest-neighbour distance between nests ranged from 1.5 km to 1.9 km and was not lower than the one estimated for Dadia NP.

The small raptors like the sparrowhawk, the levant sparrowhawk and the hobby exhibited also stable numbers in Dadia NP while the Kestrel has slightly increased. Also over a 17-year period in Scotland the number of sparrowhawks varied little, with no overall trend and nest numbers fluctuated by no more than 15% of the mean level of 34 pairs (Newton 1991a). The density of hobby was much higher for a population nesting along Po river plain poplar plantations in Northern Italy (Bogliani et al. 1994) than for Dadia NP (29 nests/100km^2 vs. 1.6 terr/100km^2). In our study area the hobby mainly nests in poplar plantations along the Evros river where its local densities could be comparable to those referred by Bogliani et al. (1994).

Conclusions

Conclusively, the assemblage of birds of prey of Dadia NP remains diverse and the population status has been improved for many species when comparing the recent years to the mid '90s. The slight increase of golden and short-toed eagles, and the possible return of the imperial eagle and the white-tailed eagle are some of the positive results of the protection measures implemented in the area the last 10-15 years. According to the results of the monitoring, most species have stable populations, with slight fluctuations and for this reason a detailed, long-term breeding population monitoring (> 20 years in duration) must be implemented. To improve knowledge on their needs and threads data and results from the systematic monitoring have been used for habitat suitability models in order to be considered in forest management scenarios (Poirazidis et al. 2010b [= Chapter D.1 of this thesis]). Dadia NP is still one of the most important European forests for birds of prey and the integrated monitoring of their population trends combined with conservation-oriented management will contribute to safeguarding their future.

Acknowledgments

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Chapter C.4. Diurnal birds of prey in the Dadia – Lefkimi – Soufli Forest National Park: Long-term population trends and habitat preferences

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Own contribution:

Study design 30%, implementation 40%, writing 40%

Abstract

Raptors are indicators of ecosystem health and may act as conservation flagship species for conservation. Twenty-four raptor species have been found to breed in the Dadia – Lefkimi – Soufli Forest National Park (DNP), which holds one of the most diversified raptor assemblages in Europe. While only 18–20 species still breed, during five years of systematic monitoring (2001–2005) most species exhibited stable populations. The overall number of territories of diurnal raptor was estimated at between 307 and 342, which corresponds to a density of 71.4 to 79.6 territories 100 km^2. The Common Buzzard *Buteo buteo* represented 35–38% of the total territories in the area, while other common species were Short-toed Eagle *Circaetus gallicus*, Booted Eagle *Hieraaetus pennatus*, Lesser Spotted Eagle *Aquila pomarina*, Honey Buzzard *Pernis apivorus*, Goshawk *Accipiter gentilis* and Sparrowhawk *A. nisus*. Some other important species, such as Golden Eagle *Aquila chrysaetos*, which have shown population declines during recent decades show signs of recovery, possible due to habitat protection and reduced persecution. Distance to foraging areas and territorial behaviour mainly determine the segregation of raptors in the DNP. Within their breeding territories raptors were selective with respect to nesting microhabitat, selecting specific forest structures and nest-tree characteristics.

Introduction

Raptors, being at the top of the food chain are considered biologically important and environmentally sensitive as well as being indicators of ecosystem health (Newton 1979; Sergio et al. 2005). Their unfavourable conservation status has attracted public interest (BirdLife International 2004) and they can act as a conservation flagship. The decline of most species of birds of prey has been relatively well documented in Europe (Newton 1979; Cramp and Simmons 1980; BirdLife International 2004). Greece lost large parts of its raptor populations during the last 30–50 years, but some areas still hold good numbers of these birds (Hallmann 1979; Catsadorakis 1994). The Evros region and particularly the Dadia – Lefkimi – Soufli Forest National Park (hereafter called DNP) holds one of Europe's most diverse raptor faunas including endangered species such as Black Vulture *Aegypius monachus*, Imperial Eagle *Aquila heliaca* and White-tailed Eagle *Haliaeetus albicilla.*

No less than 36 species out of the 39 occurring in Europe have been observed in this area (Hallmann 1979; Dennis 1989) (Appendix C.4.1). DNP is also one of the few places in Greece where research on the raptor populations and their habitats has been carried out for many years. The first pioneer study on the status and distribution of birds of prey in the DNP was made in 1979 by WWF International and IUCN giving accurate information for 10–13 species (Hallmann 1979). During the following years, more research was done on the status of the raptor community in DNP (Adamantopoulou & Androukaki 1989; Papageorgiou et al. 1994; Adamakopoulos et al. 1995) and on the ecology of individual species (Vlachos 1989; Alivizatos 1996; Bakaloudis 2000; Poirazidis 2003a). Unfortunately, many of these studies were restricted to counts of the vultures and large eagles, while for the remaining species the data collected were rather poor (see also Appendix C.4.2). Moreover, these studies did not use standardized methods to estimate numbers of pairs and, as a result, the assessment of the population trends after 20 years of protection was almost impossible.

The estimation of population status and trends of raptors poses special problems because raptors are usually dispersed, nest at low densities and their populations may fluctuate strongly (Fuller & Mosher 1987a; Kirk & Hyslop 1998). Monitoring of raptor populations and the interpretation of their fluctuations require specific and long-term studies (Catsadorakis 1994). To overcome this problem, in 2000 WWF Greece formulated a systematic monitoring plan for the birds of prey (Poirazidis et al. 2002). This monitoring should form the basis for annual relative abundance indices of the breeding territorial raptor species by using repeatable methods that would permit data comparison between years (Poirazidis et al. 2006, 2009). Relative abundance is used when it is difficult to overcome problems in estimating absolute densities. It is useful when comparing raptor populations over time, among sites or between species (Fuller & Mosher 1987a) and enables the assessment of population trends. Additionally, an extensive survey of all the breeding raptors in DNP was carried out during 1999–2000 to estimate the current status of the breeding raptor species (Poirazidis 2003b) and to provide base-line information for the monitoring plan.

The main objectives of this chapter are: (1) to describe the historical changes in the populations of birds of prey in DNP, (2) to review the historical information on the breeding raptor populations during 1978–2005 with an emphasis on their population trends during the five years (2001–2005) of systematic monitoring and (3) to describe aspects of their nesting habitats.

Historical changes of raptor populations in DNP

Until 1970, twenty-four raptor species bred in the DNP (Hallmann 1979). This area constituted one of the few European regions where four vulture species could be observed together: the Black Vulture, Griffon Vulture *Gyps fulvus*, Egyptian Vulture *Neophron percnopterus* and Bearded Vulture *Gypaetus barbatus*. The Bearded Vulture nested in this region until 1969, after which only one individual was observed until it disappeared in 1994. Over the last three decades, four more species ceased to breed in DNP: White-tailed Eagle, Imperial Eagle, Bonelli's Eagle *Hieraaetus fasciatus* and Lesser Kestrel *Falco naumanni* (Adamakopoulos et al. 1995). In 1999 seventeen (17) diurnal raptor species nested within the borders of DNP, while in 2000 the number of breeding species increased to 18, when an active territory of Imperial Eagle was confirmed after the species had been absent for eight years (Poirazidis 2003b). In 2005, a new territory of the White-tailed Eagle, which had bred successfully until 1990, was possibly re-established in the area. However, in contrast to these positive changes, no breeding attempts have been recorded for Lanner Falcon *Falco biarmicus* in DNP after 2002.

From 2001 to 2005 (during the monitoring period; March to July), 19–20 species were found breeding in the area. Seventeen species were found wintering in the area, among these a considerable number of Greater Spotted Eagles *Aquila clanga* and several individuals of White-tailed Eagle, Imperial Eagle and Long-legged Buzzard *Buteo rufinus*. Other species used the area on passage, such as Osprey *Pandion haliaetus*, Bonelli's Eagle, Montagu's Harrier *Circus pygargus*, Pallid Harrier *Circus macrourus* and Red-footed Falcon *Falco vespertinus*. Finally Eleonora's Falcon *Falco eleonorae* can be met with in the area during late spring – early summer (see Appendix C.4.1 for an analytical review of the observed species).

The raptor populations during 1978–2005, with some notes on their ecology

DNP has a diverse avifauna of raptorial birds. The first estimation of the total population of all breeding species was successfully made during the integrated survey in 1999–2000. The number of territories was estimated at between 307 and 342, which corresponds to a density of 71.4 to 79.6 territories/100 km^2 (Poirazidis 2003b). Black and Griffon Vultures were excluded, because they are colonial and the survey methods were not appropriate for them.

However, during these years, 22 pairs of Black Vulture bred (and 89 individuals were seen) while 112 individuals of Griffon Vulture were observed, however without attempting to breed (see Skartsi et al. 2010b).

The systematic monitoring was launched in 2001 to estimate the number of breeding territories with repeatable methods (Poirazidis et al. 2002, 2006, 2009). Twenty-four points that provided a good view of the surroundings and 10 road transects were selected; from these at least 66% of the total area could be covered (Figure C.4.1). To a large extent most of the raptor territories in the remaining uncovered zones were also possibly recorded through detailed mapping of flight paths even at the margins of censused areas and observations of the behaviour of birds.

During the first five years of monitoring (2001–2005) the total number of territories exhibited a reasonable stability and the same was true for most of the individual species (Figure C.4.2). Common and Steppe buzzards *Buteo b. buteo* and *B. b. vulpinus* represented 35–38% of the total number of raptor territories in the area, while other common species were Short-toed Eagle *Circaetus gallicus*, Sparrowhawk *Accipiter nisus*, Honey Buzzard *Pernis apivorus*, Booted Eagle *Hieraaetus pennatus*, Lesser Spotted Eagle *Aquila pomarina* and Goshawk *Accipiter gentilis* (Figure C.4.3).

Despite the methodological problems of earlier surveys (see Appendix C.4.2), below we shall attempt an assessment of the population trends of the different species during the last 28 years (1978–2005), with some notes on their nesting ecology.

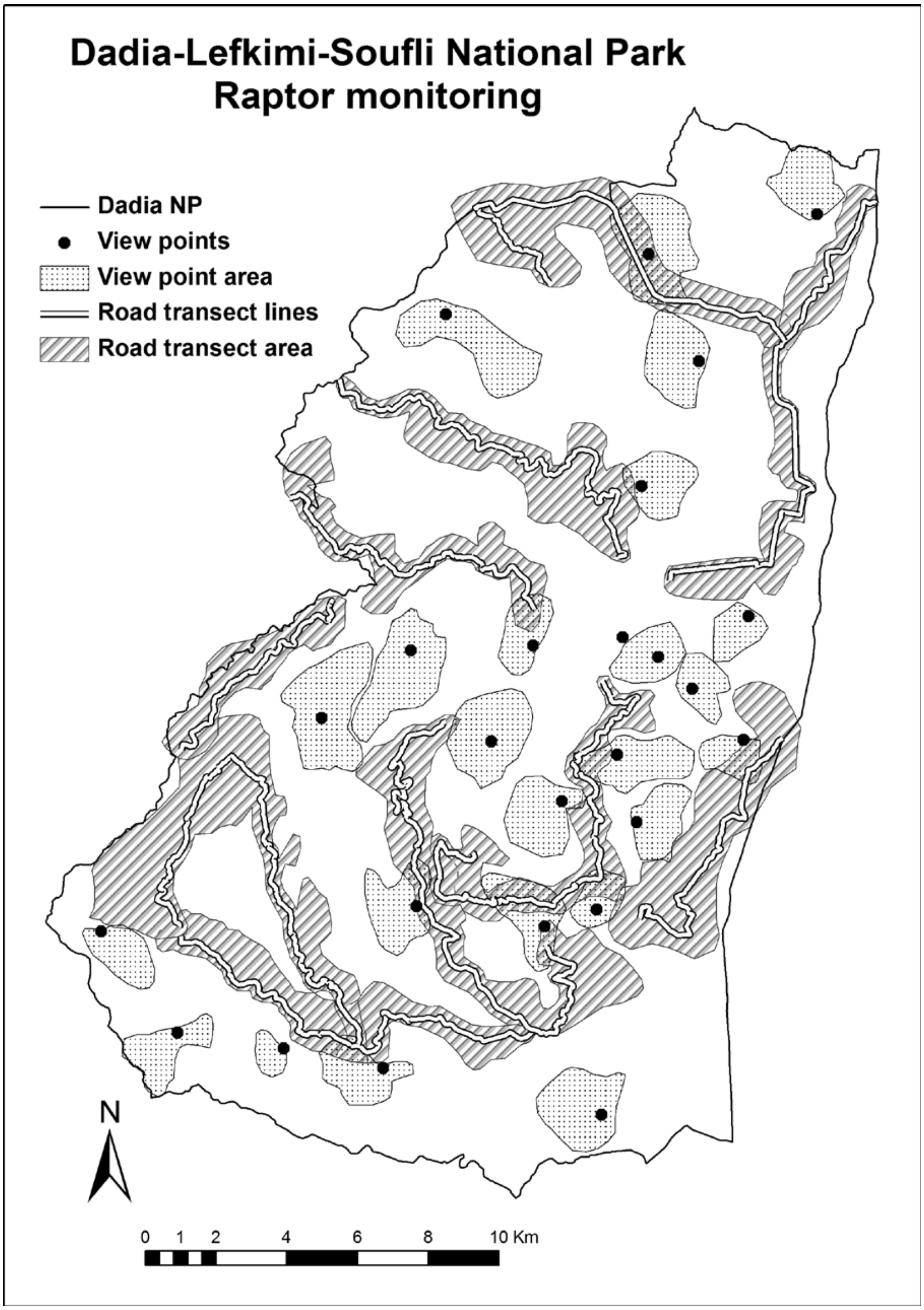

Figure C.4.1. *Sampling areas for the raptor monitoring in DNP.*

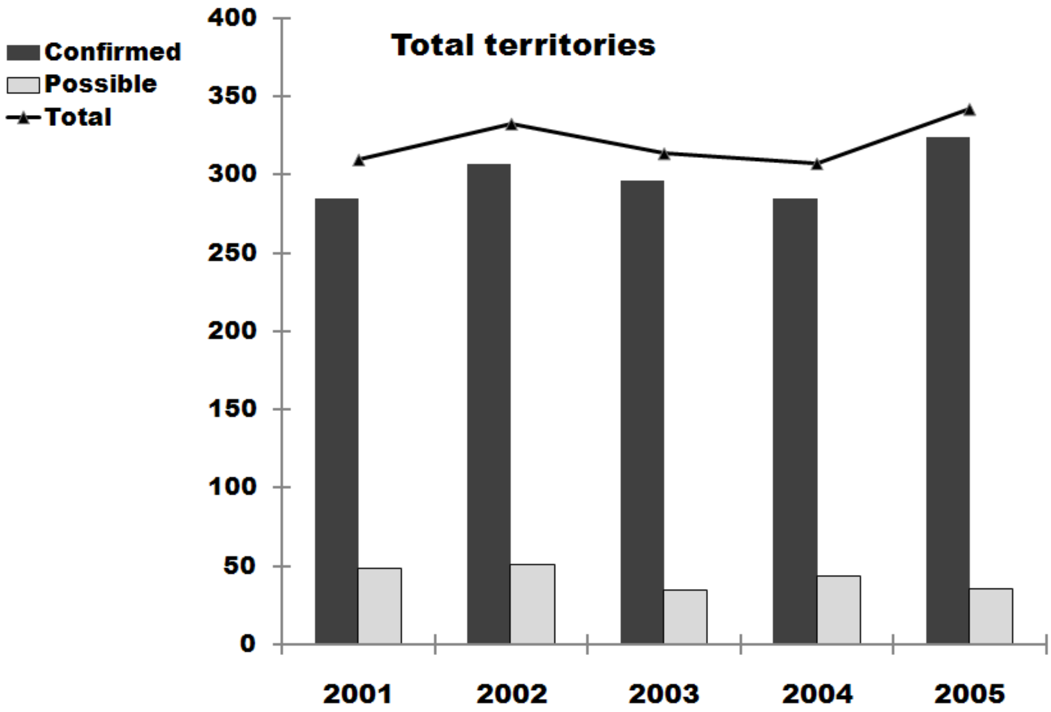

Figure C.4.2. *Changes in the number of territories of raptor species in DNP during 2001 – 2005. Total territories = confirmed plus half of the possible ones.*

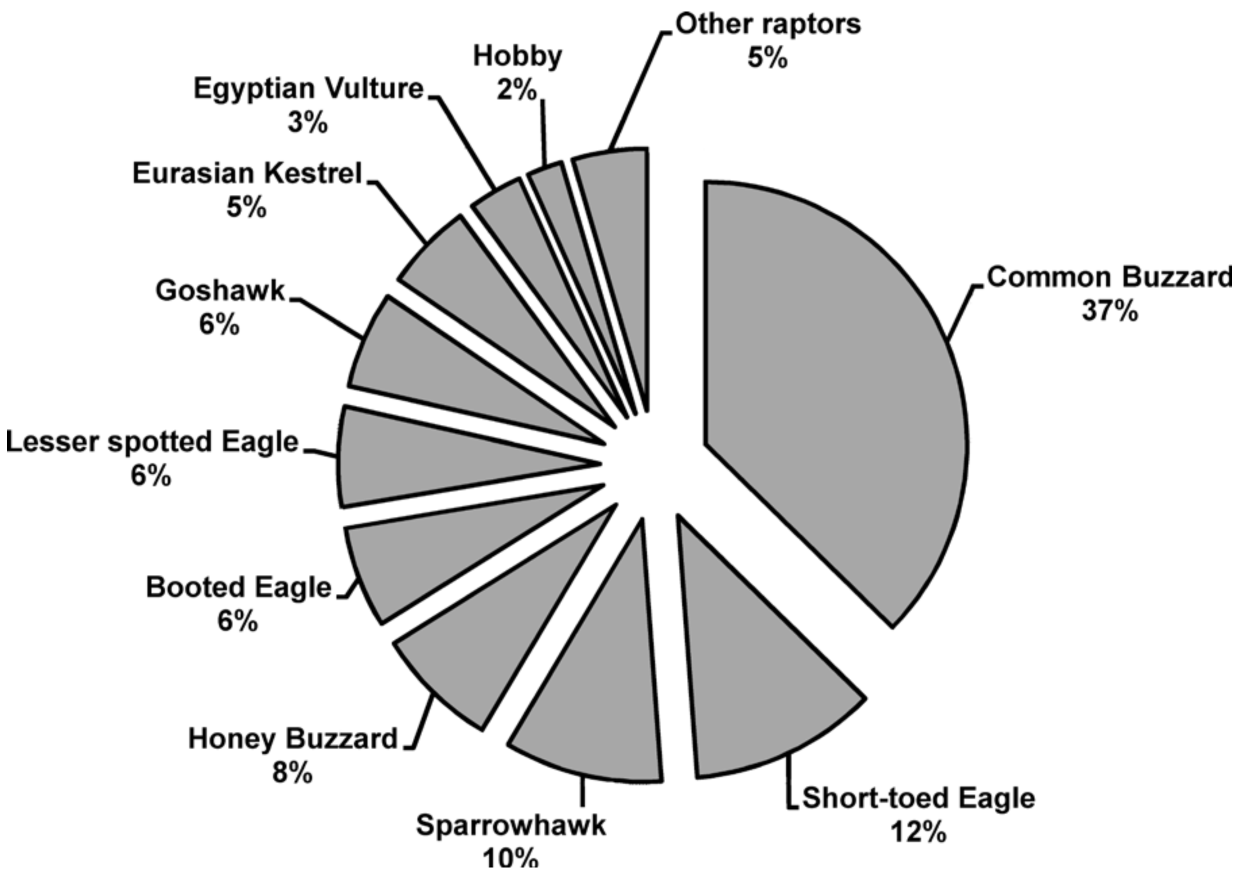

Figure C.4.3. *Each raptor species' average percentage of all raptor territories in DNP during 2001 – 2005.*

Vultures

Three species of vultures breed in the area, namely Black Vulture, Griffon Vulture and Egyptian Vulture. Long-term monitoring data on their population sizes in DNP exist for the first two, while for the third species the available information is scarce (Adamakopoulos et al. 1995; Vlachos et al. 1998).

The Black Vulture is actually the only species in DNP that has shown a significant increase since 1979, due to the protection of nesting sites, supplementary feeding and the reduction of threats, such as poaching and habitat degradation. However, the population has remained stable since 1994 (Skartsi & Poirazidis 2002). The species nests in mature pine trees on steep slopes away of human presence (Poirazidis et al. 2004); a detailed analysis of its population trend and a description of its nesting habitat can be found in Skartsi et al. (2010a).

In contrast, the Griffon Vulture is a colonial, cliff-nesting species. Its numbers increased constantly from 40 individuals in the 1990s to 75–112 at the beginning of the present century. It ceased to breed in 1995 but returned as a breeder in 2007 (for a detailed description of its population trends, see Skartsi et al. (2010b).

With 17 confirmed territories in 1978 (Hallmann 1979), the Egyptian Vulture, another cliff-nesting species, reached 25 territories in 1987 (Vlachos 1989), but thereafter the population declined dramatically to 10–14 pairs in the 1990s (Adamakopoulos et al. 1995). During 2001–2005 the average number of territories was 10.7 ± 1.2 with no significant variation. The estimated density was 2.5 territorries/100 km^2 and the "confirmed" territories (n = 9) were very stable during the monitoring period and constituted the main breeding population in DNP, while the number of "possible" territories varied; these were probably held by non-breeding pairs. The Egyptian Vulture's breeding area in DNP as described in the 1970s (Hallmann 1979) has not changed significantly, yet many of the old nesting sites remain unoccupied (Figure C.4.4).

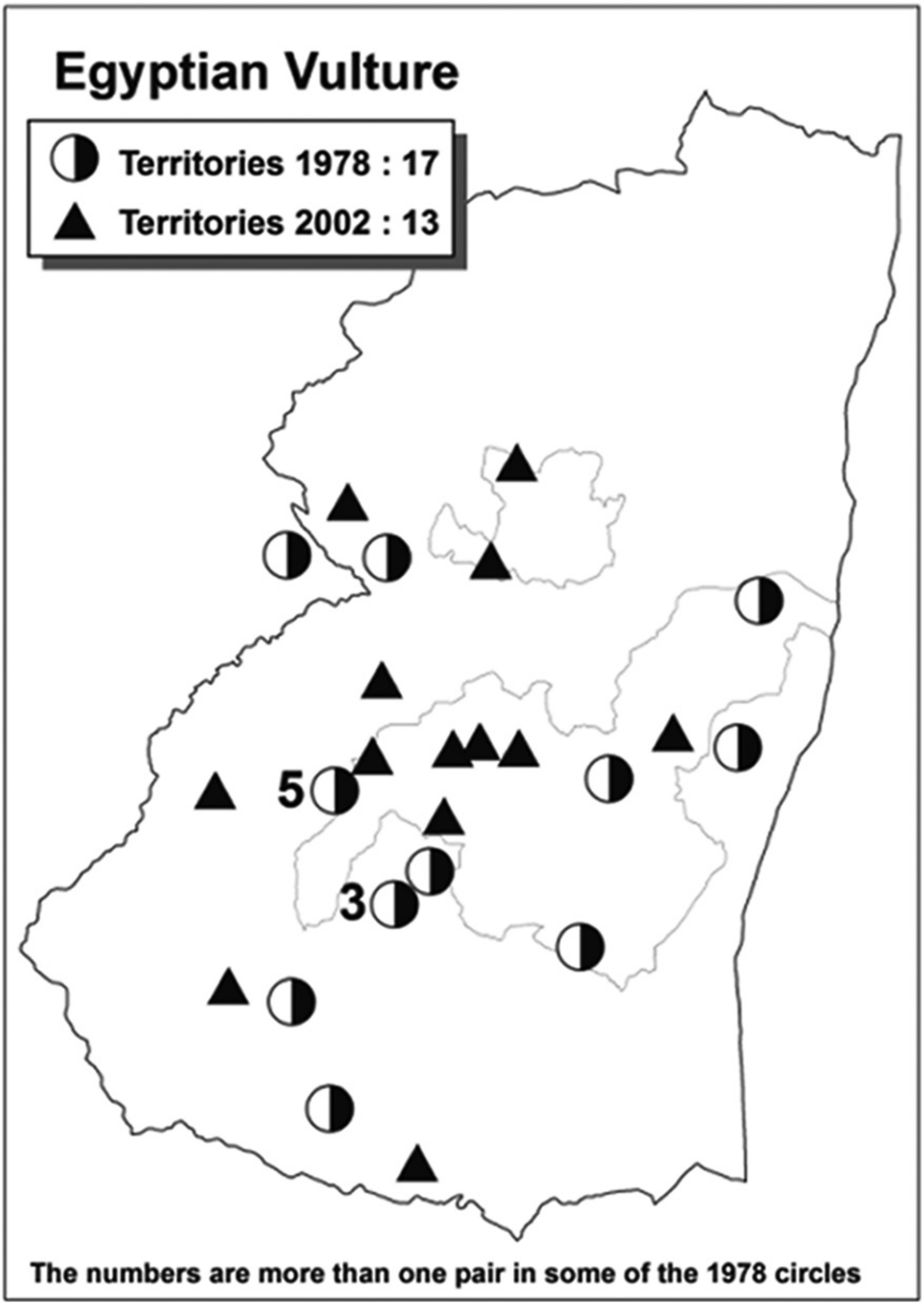

Figure C.4.4. *The number of territories of Egyptian Vulture in 1978 and 2002.*

The operation of the vulture feeding station seems not to have enhanced its population (Vlachos et al. 1995). The factors affecting the breeding population are still unknown and may be associated with the conditions on the wintering grounds in Africa, but this requires further investigation (see also Skartsi et al. 2010b).

Eagles

Six species of eagles breed (or bred formerly) in the area, namely White-tailed Eagle, Imperial Eagle, Golden Eagle *Aquila chrysaetos*, Lesser Spotted Eagle, Short-toed Eagle and Booted Eagle.

The populations of large and disturbance-sensitive raptors, such as the White-tailed Eagle and the Imperial Eagle, have declined during the last 25–28 years. For each of these species only a single territory was present in Dadia during the last years, and they may have disappeared because conditions turned unfavourable. In the late 1970s, the Evros region held 5–7 pairs of Imperial Eagle (Hallmann 1979). In 1986 this figure had decreased to only two pairs (constituting the entire Greek breeding population), with the last confirmed nesting record in the Dadia forest in 1990 (Hallmann 1996). A marked reduction of open and semi-open habitats, which has taken place in the area since the 1950s and which is largely due to land-use changes (Triantakonstantis et al. 2006) affected these large eagles negatively. These changes have occurred for socio-economic reasons and involved land abandonment as well as the decline of free-ranging livestock (see also Liarikos et al. this volume). The Imperial Eagle preferred open areas close to the nest site where it mainly hunted European Glass Lizards *Ophisaurus apodus* and Sousliks *Citellus citellus* (Adamakopoulos et al. 1995). The observed decline of this eagle in DNP, as well as that of the Long-legged Buzzard, followed the progressive disappearance of the Souslik colonies, the last colony of which survived up to 1995 (Adamakopoulos et al. 1995). As only observations of adults and immature birds and no breeding records were made, the Imperial Eagle apparently ceased breeding in the DNP after 1991. However, the recent return of this species in 2000 as a breeding species is a very hopeful message for the effectiveness of the conservation measures of the last 15 years (Figure C.4.5).

One pair of White-tailed Eagle bred until 1990 in the pine forest of the large core area. DNP must be considered as a rather dry ecosystem, at least compared with the breeding habitats normally used by White-tailed Eagles. This species usually forages over water bodies, preying mainly on fish and waterfowl but also feeds on carcasses (Watson et al. 1991). The Dadia pair usually travelled to the Evros delta (40 km away) for foraging. Since 2003 immature and adult individuals have been observed occasionally during spring and summer, and in spring 2005 a sub-adult pair was resident in the traditional breeding territory.

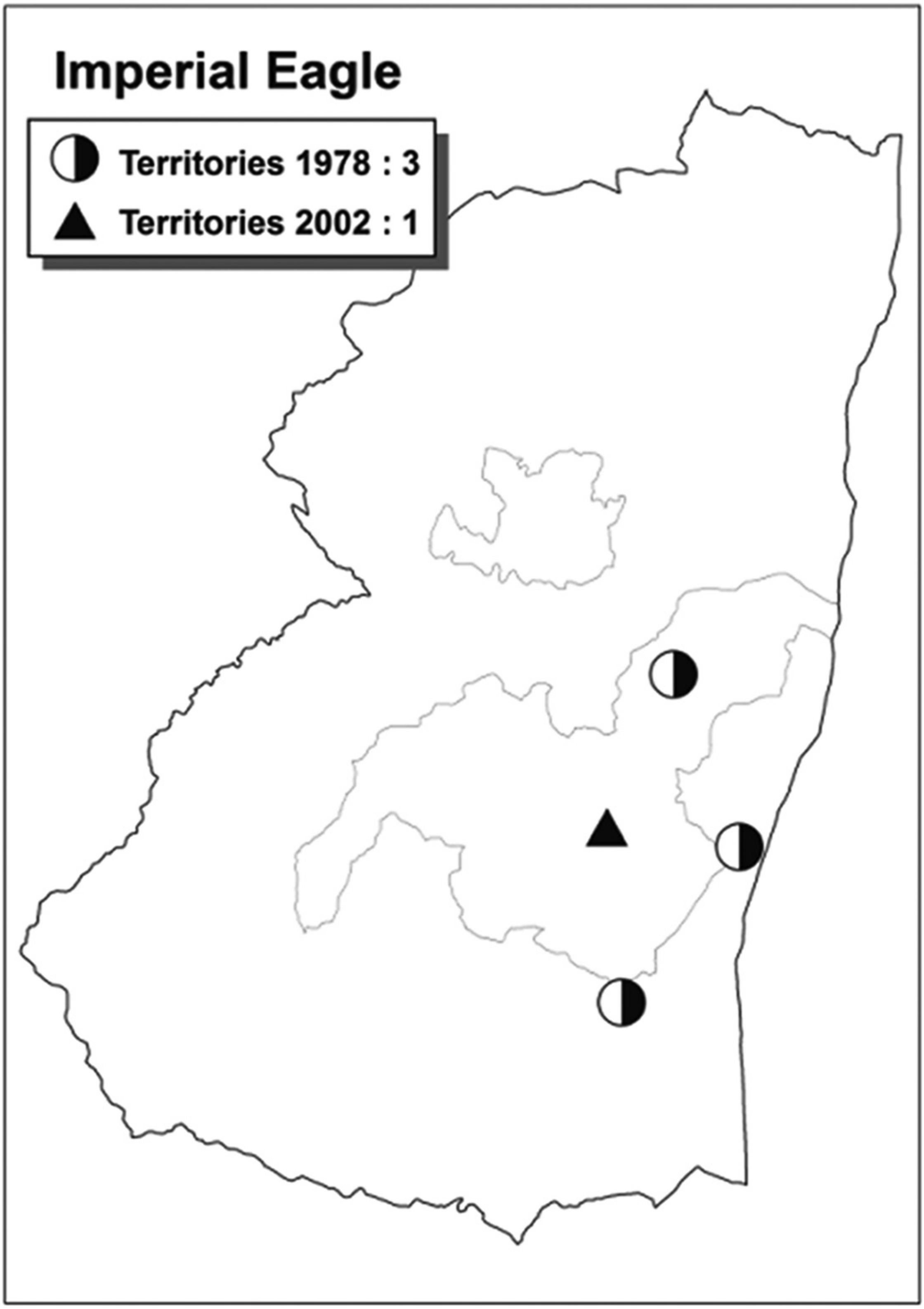

Figure C.4.5. *The number of territories of Imperial Eagle in 1978 and 2002.*

Compared with the number of occupied territories in 1978, in 1995 Golden Eagles had disappeared from some, in agreement with the population trend in all of Greece during that period (Adamakopoulos et al. 1995). For many years the population in the Dadia area was stable at three pairs but during the monitoring period (2001–2005) sub-adult birds were seen flying over unoccupied traditional breeding territories, indicating that new pairs were in the process of establishing themselves. The number of confirmed territories increased from four in 2001 to five in 2005, plus one probable territory. Breeding pairs are strongly territorial and hold extensive territories (mean nearest-neighbour distance (NND) for the years 2002–2005 was 8.9±1.8 km). The nests are either built on rocks or in trees. The main food of Golden Eagles in the DNP during the breeding season is tortoises (Capper 1998), which abound in the forest area (Phokas 2001), while during winter the birds feed mainly on small mammals and carcasses. Although food availability is a potentially limiting factor for this territorial eagle, it is possible that the re-occupation of past territories in recent years might have taken place due to reduced persecution as the environmental awareness of the local people has increased.

The Lesser Spotted Eagle is a priority species for conservation, for which large-scale action was drafted in a recent European Action Plan (Meyburg et al. 2001). The size of the Lesser Spotted Eagle population in DNP seems to have remained stable during the last twenty years. Nineteen pairs were recorded in 1978 (Hallmann 1979), while a population of 16–20 pairs was estimated in 1987 (Vlachos 1989), a number similar to the current population. In DNP, the Lesser Spotted Eagle uses mosaic habitats dominated by forest edges, small portions of mature forests and local streams for nesting (Poirazidis et al. 2007a). Its nesting close to main streams reflects its preference for this particular foraging habitat as indicated by the large proportion of Grass Snakes *Natrix natrix* in its diet (42.3%, Vlachos & Papageorgiou 1996). In Dadia the Lesser Spotted Eagle avoids the north-facing slopes for nesting, although such nest sites would provide protection from the high summer temperatures during its breeding season which extends into July–August. It is possible that the species optimizes its breeding success by avoiding the cold weather conditions that sometimes occur in the early breeding season (Kostrzewa & Kostrzewa 1990). A current analysis of the genetic diversity of this species in Europe found that the Balkan Peninsula acted as a refugium during the last ice age, as the most common

Baltic haplotype was present also in the Dadia population; northern regions were colonized after deglaciation 8000 ± 1500 years ago (Väli et al. 2004a).

Although the population was probably stable during the last 25 years, there was a marked change in the elevations at which the Lesser Spotted Eagles nested. While only 50% of the pairs bred below 100 m in the 1970s (Hallmann 1979), in 2000 this number had risen to 67% (Figure C.4.6).

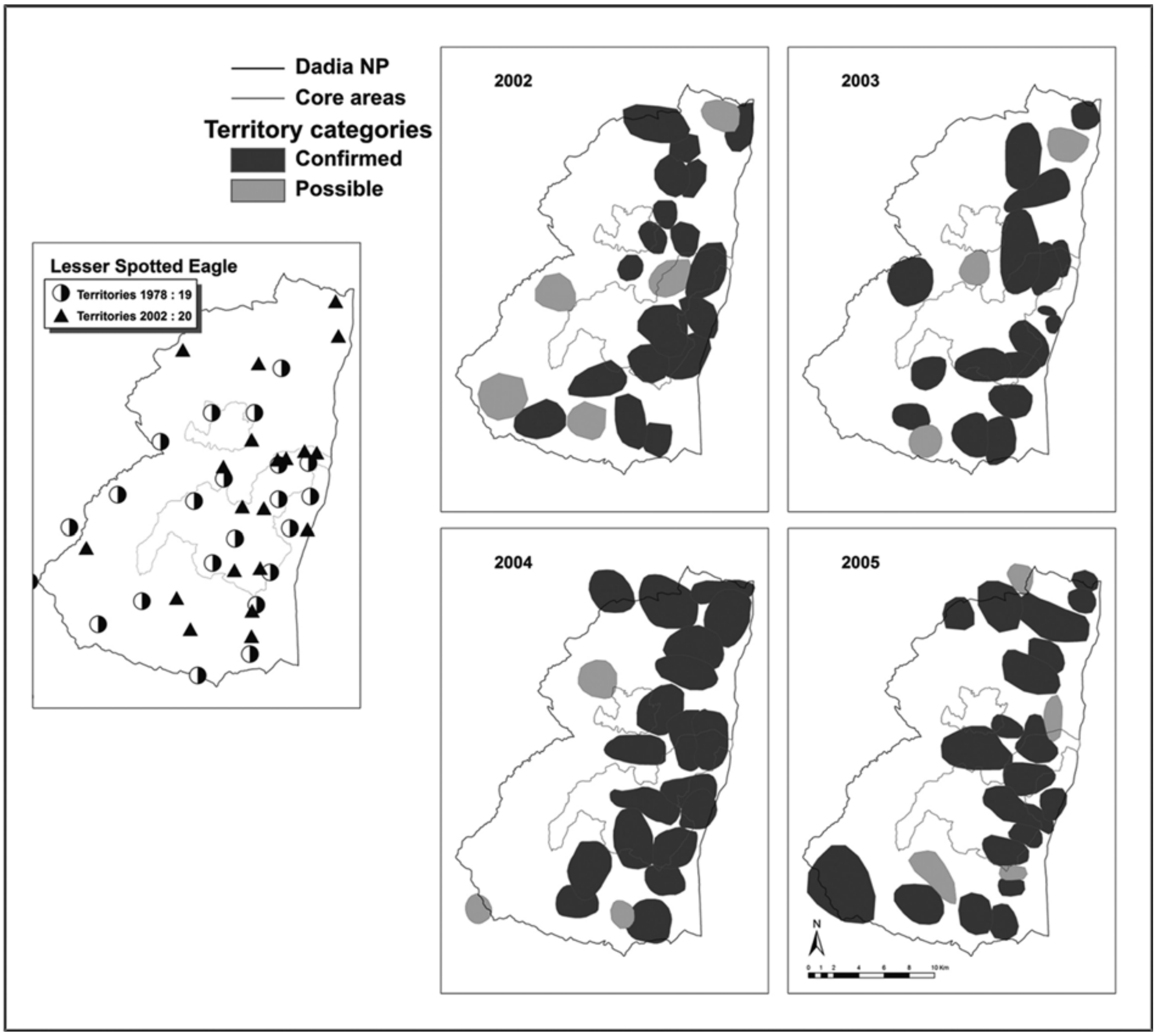

Figure C.4.6. *The number of territories of Lesser Spotted Eagle in 1978 and 2002 and their distribution during the last four years of monitoring (2002 – 2005). Reprinted from Poirazidis et al. (2006).*

Habitat change has been found to affect prey availability for many raptor species negatively (Baker & Brooks 1981, Preston 1990) and the change in the distribution of Lesser Spotted Eagles in Dadia may be related to the reduction of open and semi-open habitats in the interior of the forests that has been recorded since the 1950s.

Such reduction in forest heterogeneity has most likely resulted in a decrease in the density of reptiles and amphibians, important food for the Lesser Spotted Eagle in DNP (Vlachos & Papageorgiou 1996) thus making the population sensitive to further reduction of suitable habitat (Väli et al. 2004b). Although the species is known to be solitary and strictly territorial in other European areas (Cramp & Simmons 1980), in Dadia the concentration of many pairs in a limited area resulted in a clumped nest distribution. Clumped raptor dispersions may arise because of diminished suitability of breeding sites (Solonen 1993). In order to support the isolated (and thus more extinction sensitive) pairs of this species, five small ponds were created by WWF Greece in the core areas of DNP within the framework of a LIFE-Nature project aimed at increasing the abundance of amphibians and other prey taxa (WWF Greece 2006). This action is also expected to affect the breeding population of the Black Stork *Ciconia nigra* positively.

The Short-toed Eagle and the Booted Eagle still maintain their traditional territories within the DNP as recorded in the 1970s, with slight upward trends for both species (Figure C.4.7). Bakaloudis et al. (2005) found 22 active territories of Short-toed Eagle in DNP in 1997, similar to the first population estimate (Hallmann 1979), while data from the 1999–2000 survey showed an important increase (by 83%) (Poirazidis 2003b). For the Booted Eagle Adamakopoulos et al. (1995) found 20 pairs, similar to the current population of 21–25 pairs, and this marks a considerable increase (by 153%) from the first survey in 1979.

During the monitoring period (2001–2005), the territory density of the Short-toed Eagle was 8.7 territories/100 km^2 and showed no significant changes during the five years, with an average number of territories of 36.9 ± 3.8. The maximum was 40–41 pairs in 2002 and 2005 and the minimum 31 pairs in 2001. In central Italy a lower density (2.05 pairs/100 km^2) was estimated for Short-toed Eagles (Petretti 1988) than the 5.92 pairs/100 km^2 for DNP reported by Bakaloudis et al. (2005) and the 8.7 pairs/100 km^2 found during the monitoring period. Short-toad Eagles select mature pine stands on south-facing slopes, near clearings and in areas with little disturbance (Bakaloudis et al. 2001). They prey exclusively on reptiles, mainly snakes, and seek prey mostly in open habitats where prey availability is higher (Bakaloudis et al. 1998a). DNP is characterized by a high diversity of habitats (Schindler et al. 2008 [= Chapter B.1 of this thesis]) offering this species an optimal landscape for both nesting and foraging.

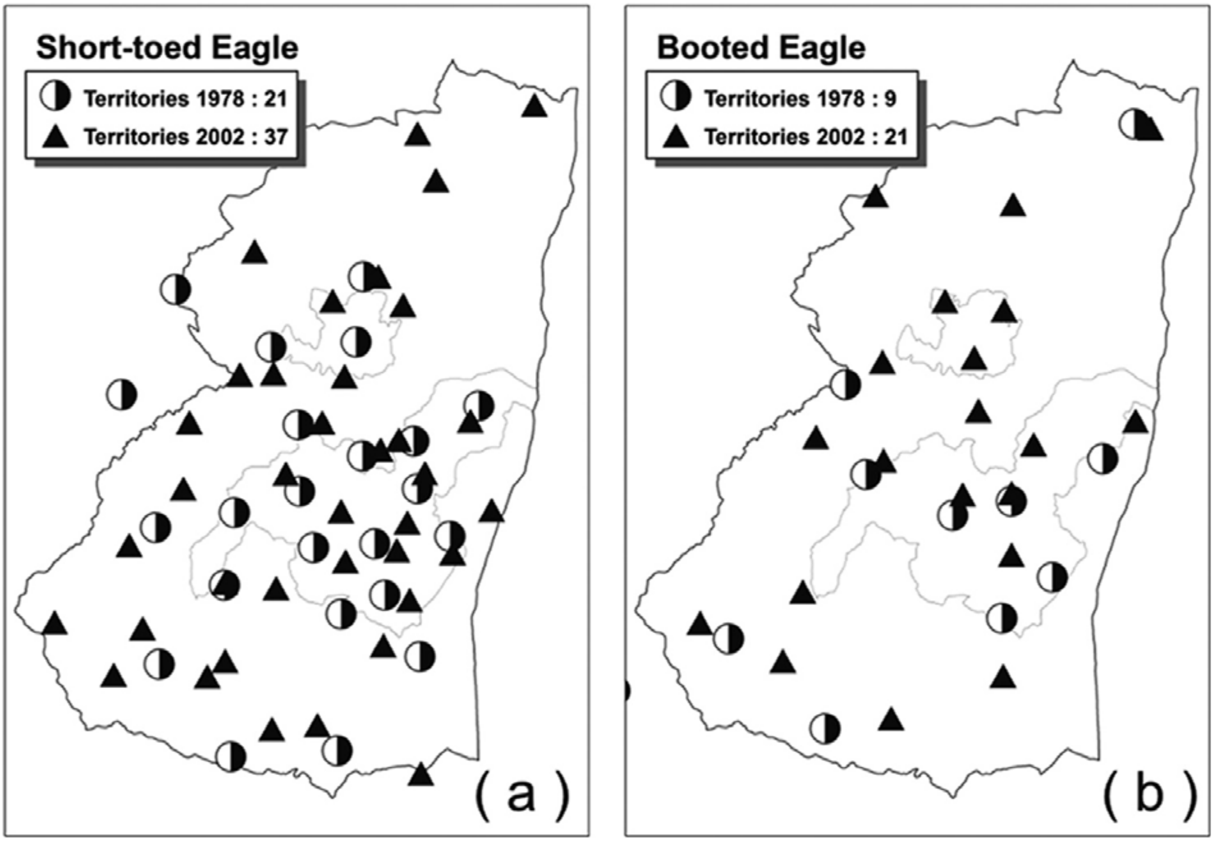

Figure C.4.7. *The number of territories of Short-toed Eagle (a) and Booted Eagle (b) in 1978 and 2002.*

Similarly the Booted Eagle population appeared to be stable during the five years of census (density 4.7 territories/100 km^2). The mean number of territories was 20 ± 1.2, with no significant trend. Although raptor numbers may have been seriously under-estimated in 1979, the increase recorded during the last decade is probably real and is due to improved forest conditions in the DNP created by a more conservation-friendly management. An increase of Booted Eagles has been recorded in Western Europe during the last few decades, which may be attributed to the species' adaptability to changing environments (Carlon 1996). In Doñana National Park (south-western Spain), the Booted Eagle population increased from six pairs in the early 1980s to 150 in 2000 (Suarez et al. 2000).

The Booted Eagle is a generalist raptor (Veiga 1986; Sanchez-Zapata and Calvo 1999) nesting in a variety of areas independently of geomorphology, distance to possible sources of disturbance as well as distance from forest clearings and main streams (Poirazidis 2003a). It also occupies territories in fragmented forests with a high proportion of clearings. Territorial behaviour (average NND 3425 m ± 1230) seems to be one of the main factors determining the location of its nest sites, many of which are located in the mountain zone (Figure C.4.7b). On the other hand, on the micro-scale level this species is very selective with respect to stand structure characteristics, preferring trees of large DBH reflecting the birds' need for mature trees to support their big nest (Cramp & Simmons 1980). In addition, the presence of mature forest around nests was the most important vegetation characteristic, probably because this enables the birds to construct nests in different trees in different years, as also found in Italy by Sergio et al. (2002). Similar findings were made in DNP also for other raptors, such as the Goshawk and the Common/Steppe Buzzard (Poirazidis et al. 2007a). These "forest" raptors preferred to establish nest sites in open forest with high canopy.

Medium-sized raptors

Five species belonging to this category breed in DNP, namely Long-legged Buzzard, Common/Steppe Buzzard, Honey Buzzard *Pernis apivorus*, Black Kite *Milvus migrans* and Marsh Harrier *Circus aeruginosus*.

The Long-legged Buzzard population in Dadia decreased from seven pairs in 1978 (Hallmann 1979) to five in 1990 (Alivizatos 1996) and a stable population of 3–4 pairs presently, which gives a density of 0.9 territories/100 km^2. It has disappeared from most of its traditional forest territories in the highlands, and nowadays nests in the lowlands where a mosaic of habitats exists as a result of human agro-pastoral activities (Figure C.4.8a). Colonies of European Sousliks occurred in 10 of the 16 territories of Long-legged Buzzard found in the Evros region in 1993 (Alivizatos & Goutner 1997). The observed decline of the Long-legged Buzzard in DNP followed the progressive disappearance of the Souslik colonies, the last of which disappeared in 1995 (Adamakopoulos et al. 1995). Considering the large contribution of this small mammal to the diet of the Long-legged Buzzard, it is probable that the decrease in Souslik numbers has affected the distribution of Long-legged Buzzards (Alivizatos & Goutner 1997).

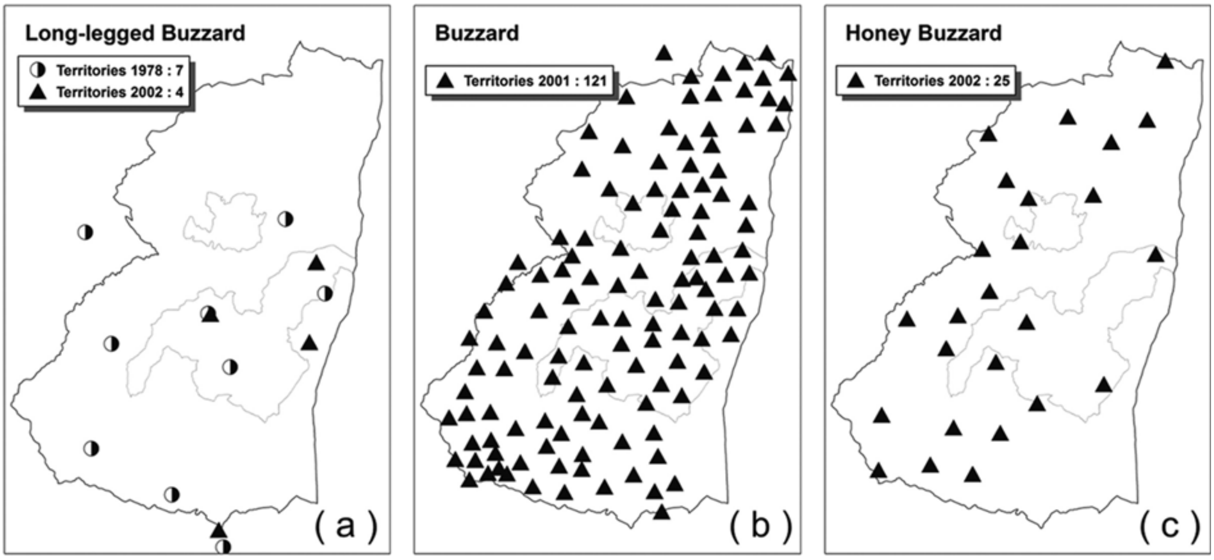

Figure C.4.8. *The number of territories of Long-legged Buzzard (a) in 1978 and 2002, the distribution of the territories of Common/Steppe Buzzard in 2001 (b) and the distribution of Honey Buzzard territories in 2002 (c).*

The Common/Steppe Buzzard is a generalist occurring in almost all available habitats in DNP (Figure C.4.8b) and is very common. Unfortunately, there is no information on its densities in the 1970s or later, so it is impossible to assess its population trend. In DNP, the Common/Steppe Buzzard has a density of 28–30 territories/100 km^2 with a mean NND between the very regularly dispersed nest sites of 1.45 km. This NND is similar to the values found in a study in the UK, where they ranged from 1.53 km to 1.95 km (Dare & Barry 1990). Sergio et al. (2002) found an identical density to that in the DNP in the Italian Pre-Alps (28–31 pairs/100 km^2), but in central Italy populations were less dense with 19.8 pairs/100 km^2 (Cerasoli & Penteriani 1996) and 8.3 pairs/100 km^2 (Penteriani & Faivre 1997b), with a mean distance of 2.5 km between nest sites in the latter study. In DNP, the Common Buzzard population varied during the five years of monitoring between a maximum of 122–128 pairs and a minimum of 110–112 pairs.

In the Italian Alps Common Buzzards shifted nesting sites due to disturbance (Sergio et al. 2002). In DNP they are opportunists regarding their nesting microhabitat and nests regardless of the proximity to human habitations (Poirazidis 2003a), a situation resembling that of the Red-tailed Hawk *Buteo jamaicensis*, a New World species that can nest near human settlements if there is not too much human activity (Bednarz & Dinsmore 1982; Speicer & Bosakowski 1988).

The Honey Buzzard has increased during the last twenty years from 2–4 pairs in 1987 (Vlachos 1989) to 10–12 pairs in 1994 (Adamakopoulos et al. 1994), although this trend is likely to be an effect of underestimates during the early survey years. During the monitoring period (2001–2005), the estimated population averaged 24.3 ± 3.9 pairs, corresponding to a density of 5.7 territories/100 km^2, covering most of the forested area (Figure C.4.8c). The population peaked at 28 pairs in 2001, thereafter declining to only 18 pairs in 2004. In 2005 the population increased again reaching 24 pairs. The density in DNP is low to medium compared with that in other parts of Europe: 11.7 pairs/100 km^2 in southern Finland (Solonen 1993), about 4 pairs/100 km^2 in the German state of Hessen (Schindler 1997) and ranging from 5.0 pairs/100 km^2 to 22.1 pairs/100 km^2 in Austria (Gamauf & Winkler 1991, Gamauf & Herb 1993). Throughout Europe the abundance of Honey Buzzards is highest in broad-leaved and mixed forests on rich soils and in areas with plenty of water bodies. The optimal environments for this raptor seem to occur in areas with higher spring and summer precipitation than DNP.

Black Kites and Marsh Harriers breed in areas adjacent to DNP and use the park temporarily for foraging. The Black Kite has a good breeding population along the riparian forest of the Evros River and in 2003 one pair may have nested in a pine forest close to Dadia village but this was not confirmed. The Marsh Harrier breeds in an extensive reed bed very close to the south-eastern border of the National Park. During 2002–2003, one or two females may have bred inside the south-eastern border of DNP, but this was also not proved.

Hawks

Three species of hawks breed in DNP, Goshawk, Sparrowhawk *Accipiter nisus* and Levant Sparrowhawk *A. brevipes*.

During the last 28 years the overall population of Goshawk has not changed significantly nor has the spatial distribution of its territories varied (Figure C.4.9a, Appendix C.4.2). During the five census years, the Goshawk population was very stable, with 18 to 22 pairs and a mean density of 4.5 territories/100 km^2. The nest spacing was very regular (NND 3061 m ± 1088) indicating a strong territorial behaviour (Poirazidis et al. 2007a). The observed density is similar to that found for other

European populations, such as in Italy, estimated at 5.03 pairs/100 km^2 (Penteriani & Faivre 1997a) and Finland, estimated at 4–6.6 pairs/100 km^2 (Solonen 1993).

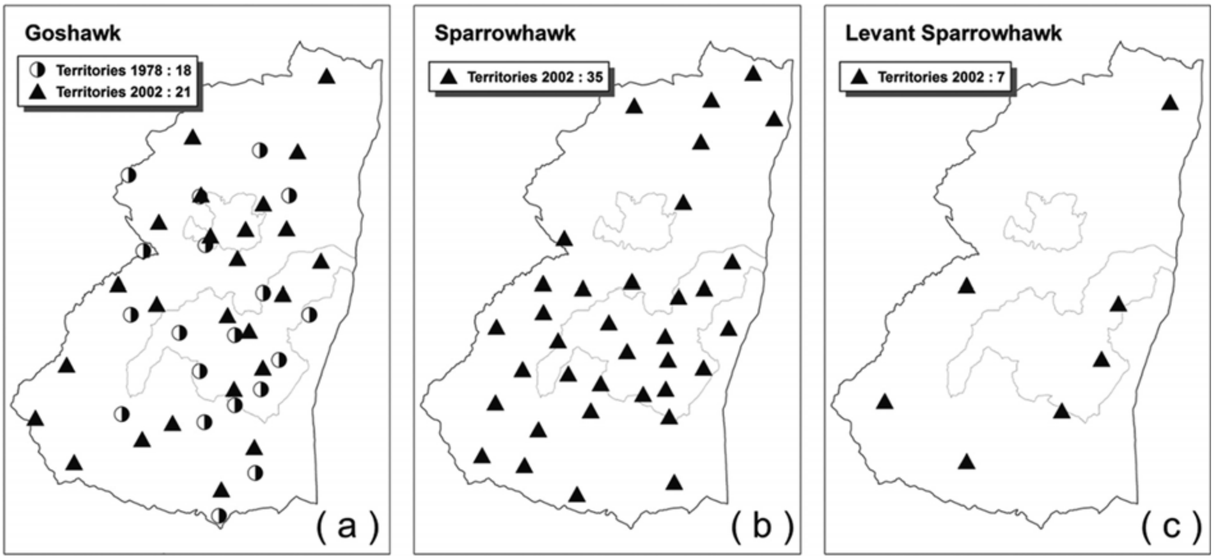

Figure C.4.9. *The number of territories of Goshawk in 1978 and 2002 (a) and territories of Sparrowhawk (b) and Levant Sparrowhawk (c) in 2002.*

The Goshawks' choice of low-elevation sites for nesting (54% of nests below 130 m) – similar to that of the Lesser Spotted Eagle – is probably also related to higher densities of prey in the lowlands (Poirazidis et al. 2006). This results in nests being closer to human habitations than found in other studies (Speiser & Bosakowski 1987, Penteriani & Faivre 1997a). An association between breeding density and main prey distribution has also been reported in Italy, where a higher nest density of Goshawks was found at lower elevations than in the mountain zone (Penteriani & Faivre 1997a), and in Sweden, where food was the main factor determining Goshawks' habitat use (Kenward & Widén 1989).

For Goshawk, an open stand structure is important for pairing and for fledgling activities near the nest before the young birds disperse (Kenward et al. 1993, Penteriani et al. 2001). Nesting in mature forests with an open structure and at great height facilitates the pair's access to the nest, provides good visibility of the surroundings as a protection against predators and facilitates hunting in areas adjacent to the nest (Titus & Mosher 1981; Speicer & Bosakowski 1987; Moorman & Chapman 1996).

The importance of mature forest as a vital parameter in raptors' nesting habitat is suggested by the fact that the variable "number of trees in diameter class 36–80 cm" had the highest loading in a multivariate analysis of four sympatric raptor species in DNP (Poirazidis et al. 2007a). The Goshawk showed the strongest association with this habitat variable among the raptor species in DNP (Bakaloudis et al. 2001; Poirazidis et al. 2007), thus the availability of suitable nesting microhabitats is likely of primary importance for this species, as also found in other studies in Europe and North America (e.g. Reynolds et al. 1982; Crocker-Bedford & Chaney 1988; Lilieholm et al. 1994; Kenward 1996; Penteriani et al. 2001).

The Sparrowhawk population was very stable during the monitoring period with an average number of pairs of 31.2 ± 3.3 at a density of 7.3 territories/100 km^2. These figures are probably underestimates of the population breeding in DNP since this species is secretive and difficult to find with the methods applied and several nest sites no doubt remained undiscovered (Figure C.4.9b). In Scotland nest numbers fluctuated by no more than 15% around the mean level of 34 pairs over a 17-year period, with no overall trend (Newton 1991a).

The Levant Sparrowhawk's main breeding area is along the Evros River, the border to Turkey, where its population is very high (K. Poirazidis pers. obs.). Only a few pairs breed inside the National Park, where a maximum population of seven pairs was observed in 2000 and 2002 (Figure C.4.9c).

Falcons

The DNP is not a suitable area for falcons. Four species of falcon breed in DNP, namely Peregrine *Falco peregrinus*, Lanner *F. biarmicus*, Hobby *F. subbuteo* and Common Kestrel *F. tinnunculus*, but their populations are small. One more falcon, the Lesser Kestrel *F. naumanni*, bred formerly, but no evidence for breeding exists from recent years.

For the last 20 years only one pair of Peregrine has been considered breeding in the study area. However, in 2001 a new territory was verified in the DNP, and in 2003 three pairs were located (Figure C.4.10a). Unfortunately, this increase of the Peregrine Falcon population was followed by the disappearance of the single pair of Lanner Falcon that had bred in the area for more than 20 years (Figure C.4.10b). The observed changes in the status of the big falcons (Peregrine and Lanner) are very difficult to explain, but since both species use similar nesting and foraging habitats, it is possible that inter-specific competition caused the disappearance of the Lanner. It has been observed in similar-sized and powerful raptor species that pairs of one species have sometimes been driven off their former territory by the other (Kostrzewa 1991).

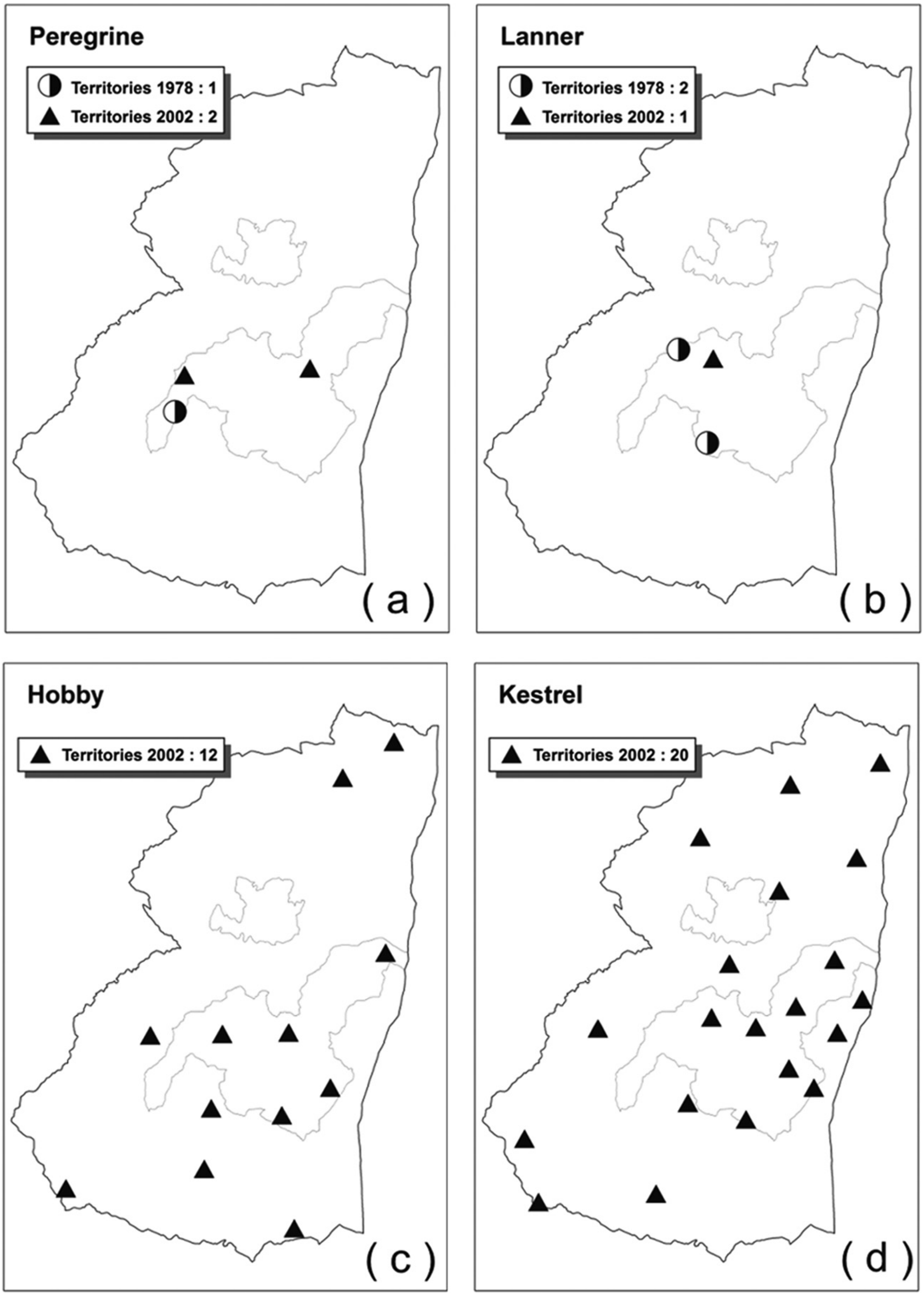

Figure C.4.10. *The number of territories of Peregrine Falcon (a) and Lanner Falcon (b) in 1978 and 2002 and territories of Hobby (c) and Kestrel (d) in 2002.*

The Hobby holds a very stable population in with a maximum number of 12 territories (2.8 territories/100 km^2) estimated during 2002 and a density which is much lower than in other areas. For instance, in northern Italy, Bogliani et al. (1994) estimated a density of 29 nests/100 km^2 in poplar plantations on the Po river plain. During the monitoring period the average number of pairs of this species was 6.9 ± 2.2 with a mean density of 1.6 territories/100 km^2 (Figure C.4.10c). In general, the census methods used are not optimal for the detection of falcon territories. For this reason, the counts may not reflect the true size of the Hobby population and the observed variation may be larger than the true one. In the Evros area the Hobby is a species that mainly nests in poplar plantations along the Evros River (K. Poirazidis pers. obs.) where its densities may be higher than in DNP and comparable to those reported by Bogliani et al. (1994).

With 15 territories in 2001 and 22 in 2005 the Kestrel is probably the only raptor species whose population increased during the five year of systematic raptor monitoring. The average number of territories was 17.4 ± 3.5 (4.1 territories/100 km^2) during 2001–2005 (Figure C.4.10d). The Kestrel is easier to detect than other species of falcon, because it is more active over open ground, which likely results in relatively accurate estimates of its population size.

Conclusions

The assemblage of birds of prey in DNP remains almost as diverse as described 30 years ago and many populations have remained stable since the 1970s. Moreover, some important species that showed population declines during recent decades now show signs of having started to recover. The re-establishment of some old territories of Golden Eagle, the return of the Imperial Eagle and possible return of the White-tailed Eagle, are some of the positive results of the protection and conservation measures implemented in the area during the last 15 years.

Raptor monitoring is a time-intensive and difficult task. In order to estimate raptor population trends, long series and a large amount of data are needed. A systematic monitoring based on GIS methodology could be an efficient tool to deal with data of this kind. The methodology used in DNP is an integrated GIS-based method for the collection and analysis of this huge amount of observations and has provided rather accurate information on which population sizes of typical territorial species, such as most of the eagles, buzzards, hawks and falcons, were estimated. However, a larger amount of data is needed to increase the precision of the population estimates for species that nest at high densities, such as the Common Buzzard. For less territorial species, such as the Short-toed Eagle and the Egyptian Vulture, some difficulties arise. The home-ranges of neighbouring pairs overlap greatly in these species, making the delineation of the different territories difficult. Some other species are very secretive. The key issue for all species, whose population sizes are difficult to estimate, was to obtain more good-quality data (like territorial observations, landings, etc.).

Our findings show that all species have shown more or less stable populations during the five years of intensive monitoring, exhibiting very slight fluctuations, as it is the rule for raptor populations (Newton 1979, 1991b). However, in order to distinguish natural short-term fluctuations from population trends a long-term monitoring programme (of >20 years' duration) must be implemented. In the revised monitoring plan for the DNP, a five-year period between surveys is anticipated instead of annual surveys, in order to minimise costs and to safeguard the surveys' continuity in the future (Poirazidis et al. 2007b). Hopefully, the recently established Management Authority of the National Park will incorporate this monitoring in its future activities.

The investigation of the various raptor species' habitat selection has proceeded in a stepwise fashion, where the various criteria of selection are hierarchically ordered (Penteriani et al. 2001). Geomorphology and distance to foraging areas seem to be the first criteria determining territory segregation in DNP, affected also by the species' territorial behaviour. High habitat diversity resulted in short distances between nest sites. Within their breeding territories the birds were selective with respect to microhabitat, choosing forest structures and nest-tree characteristics that probably maximize breeding success (Poirazidis 2003a).

A multi-layered plan to preserve the remarkable diversity of raptors in DNP must be implemented and certain management measures should be enforced:

(1) In the forest area subjected to management, small groups of mature trees forming open stands must be preserved. Instead of selective loggings where isolated mature trees are kept at a large scale, a management encouraging the formation of

even-aged small forest stands should be followed; this would be the most favourable management for raptors.

(2) Small forest clearings must be retained and/or created in areas of dense forest because such clearings are vital to many bird species occurring in the DNP.

(3) The creation of small wetlands within the forested area would benefit species such as the Lesser Spotted Eagle.

(4) Forests become suitable for nesting to most raptorial birds after 50–60 years. Thus, at any stage of forest management, tree groups of at least this age, in various positions and at least 300–500 m apart must be preserved within the managed stands.

(5) Isolated trees more than 80 years old must be preserved in all stands, especially when occurring in dense forest, because it is the specific features of such trees that are selected by the raptors.

(6) As all of the area is important for the studied species, measures to protect nest-sites should be applied all over the elevation spectrum of the area both in the core zones and in the intensively managed zones.

The DNP is still one of the most important European forests for birds of prey and the integrated monitoring of their populations combined with conservation-oriented management will contribute to safeguarding their future (Poirazidis et al. 2010b [= Chapter D.1 of this thesis]).

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Part D – Conservation Management

The last part of this doctoral thesis presents two case studies regarding conservation management. The first chapter shows an approach of integrating biodiversity conservation into forest management, based on a decision support system. Using systematically collected data of vascular plants, amphibians, small birds, and raptors, we developed niche models for plants and animals, revealing the habitat suitability of each forest stand for the species and higher taxa. The habitat suitability values can be combined with data on timber volume under three socio-economical scenarios, in order to consider both potential timber extraction and biodiversity conservation in forest management plans.

In the second paper of this part, which is the last one of this thesis, we compiled all conservation recommendations for Dadia NP (see Appendix D.2.4.) and for the Bulgarian reserves of the Eastern Rhodopes mountains. We used these compilations to evaluate for each recommendation, if it was well known by local conservation experts, if it was implemented in the area it had been proposed, and if it was evaluated regarding its effectiveness. Beside evaluating differences among the two countries Greece and Bulgaria, we also evaluated differences among taxa, and among categories such as agriculture, forest management, finishing and hunting, legislation, research, etc.

Chapter D.1. Conservation of biodiversity in managed forests: developing an adaptive decision support system

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Running title: Conservation of biodiversity in managed forests

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Lesser Spotted Eagle (Aquila pomarina) - Photo by Torsten Pröhls

Own contribution:

Study design 40%, implementation 20%, writing 60%

Abstract

Forest ecosystems provide several goods and services, but strategies for the conservation of biodiversity are missing in traditional forest management schemes. In this paper we developed a decision support system to optimize the conservation of biodiversity in managed forests, taking as a case study area Dadia National Park, a local Mediterranean hotspot of biodiversity in northeastern Greece. Using environmental niche factor analysis, we produced a series of spatially explicit habitat suitability models for vascular plants, amphibians, small birds and raptors and an overall model for total biodiversity. Further, we produced maps related to timber production and investigated potential conflicts between conservation of biodiversity and wood production. A decision support system based on a conflict assessment was created using three management scenarios. It enables the establishment of integrated management strategies and the assessment of their effects on biodiversity and timber production. Habitat suitability models for selected groups of organisms were found very effective to investigate the impact of the management on forests and wildlife. Further evaluation of key indicator taxa on these models could improve decision support systems and the sustainable management of forests.

Introduction

The increasing exploitation of forests is one of the main reasons of human-induced loss of biodiversity (Lindenmayer et al. 2002; Foley et al. 2005). Although the socio-economic value of biodiversity was underestimated until recently (Costanza et al. 1997; Farber et al. 2002), its maintenance has become a commonly accepted goal of sustainable forestry (United Nations 1992; Kohm & Franklin 1997). The concept of ecosystem services provides a tool for communicating the importance of intact ecosystems for human well-being and a framework for the evaluation of multiple functions of landscapes and forests (Costanza et al. 1997; De Groot et al. 2002; Millennium Ecosystem Assessment 2005; Boyd & Banzhaf 2007; Steffan-Dewenter et al. 2007). In forest ecology, a mayor challenge is finding trade-offs between timber production and conservation of biodiversity (Johns 1997; Putz et al. 2001; Foley et al. 2005; Burke et al. 2008).

Forestry practices can enhance or reduce habitat for particular wildlife species by altering structural features at the stand scale (Burke et al. 2008; Rendón-Carmona et al. 2009). Forest management that enhances the heterogeneity of forests has in general a positive impact on the local biodiversity (Loehle et al. 2005; Gil-Tena et al. 2007; Torras et al. 2008; Kati et al. 2010 [= Chapter B.4 of this thesis]; Poirazidis et al. 2010a [= Chapter A.1 of this thesis]; Schindler et al. 2010 [= Chapter A.2 of this thesis]), but forest management guidelines for the maintenance of biodiversity are mainly valid for site specific conditions and can be rarely used as general directions (Loehle et al. 2005). As it is impossible to measure and monitor the effects of various management practices on the entire ecosystem, indicators are used as surrogates for biodiversity (Lindenmayer et al. 2000). Taxon-based proxies include flagship, umbrella and indicator species (Caro et al. 2004; Roberge & Angelstam 2004; Hess et al. 2006; Cabeza at el. 2008), while structure based ones deal mainly with stand complexity, connectivity and heterogeneity (Lindenmayer et al. 2000; Schindler et al. 2008 [= Chapter B.1 of this thesis]). Many researchers have explored the use of particular taxa, especially vascular plants, arthropods and birds, as surrogates for biodiversity, but a general pattern has not yet emerged (Kati et al. 2004b; Sauberer et al. 2004; Sergio et al. 2005; Billeter et al. 2008; Cabeza et al. 2008; Zografou et al. 2009). The importance of including several guilds of taxa to represent adequately overall biodiversity is currently stressed by several authors (Angelstam et al. 2004; Edenius & Milusinski 2006; Loehle et al. 2006).

In this study, we developed a decision support system with the ultimate goal of providing management guidelines and optimal solutions for the conservation of biodiversity in managed forests. We considered Dadia National Park, a Mediterranean forest mosaic in north-eastern Greece, as a case study. Using available data sets from systematic scientific research in the area, a series of habitat suitability models for groups of indicator species and for overall biodiversity was produced to discover potential conflicts between biodiversity and timber production. Additionally, the effectiveness of different management scenarios was assessed.

Methods

Study Area

This research was conducted within Dadia National Park (hereafter Dadia NP), a sub-mountainous area with a diverse landscape mosaic, dominated by extensive pine (*Pinus brutia, P. nigra*) and oak (*Quercus frainetto, Q. cerris, Q. pubescens*) forest, but containing also a variety of other habitats such as pastures, cultivated land, torrents and stony hills (Schindler et al. 2008 [= Chapter B.1 of this thesis]; Poirazidis et al. 2010a [= Chapter A.1 of this thesis]). Dadia NP covers 43 000 ha in the prefecture of Evros, north-eastern Greece (Figure D.1.1), and was designed to protect the diverse community of birds of prey, including the last breeding colony of the Eurasian black vulture (*Aegypius monachus*) in the Balkan peninsula (Poirazidis et al. 2004, 2010c [= Chapter C.4 of this thesis]; Skartsi et al. 2008). Almost 45% of the National Park is managed mainly for timber production (Zone B1), while it has been recognized during the last years that this specific zone is of great value for many species (Grill & Cleary 2003; Kati et al. 2004a,b,c, 2007; Korakis et al. 2006; Poirazidis et al. 2010a,c [= Chapter A.1, Chapter C.4 of this thesis]).

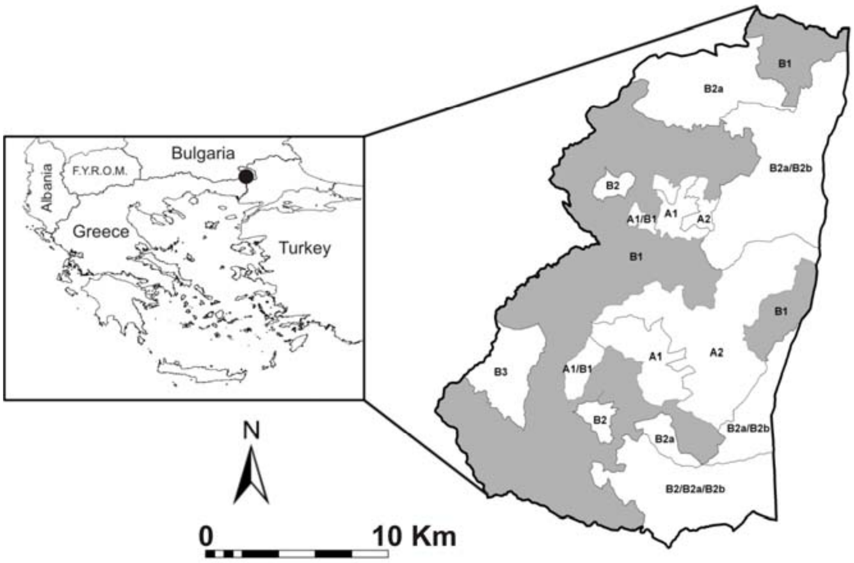

Figure D.1.1. *Location and zoning of Dadia National Park, the case study area in north-eastern Greece. Zone B1 (highlighted in grey) represents the forest management area that was investigated in this study. A1, A2: strictly protected areas, B2: agroforestry area, B3: grazing land, A1/B1: forest management area that changed recently to strictly protected area.*

Species data

We used five datasets of indicator species groups as surrogates for the total biodiversity in Dadia NP, systematically surveyed using appropriate sampling techniques per group. Those comprised woody plants, non-woody vascular plants, amphibians, small birds and birds of prey (Kati & Sekercioglu 2006; Korakis et al. 2006; Poirazidis et al. 2009a [= Chapter C.2 of this thesis]; Kret et al. 2009). For each sampling plot (the number of plots was ranging from 34 to 63 depending on the indicator species group) all present species were evaluated. The survey for vascular plants was based on fieldwork during the years 1999 and 2000, and the 62 sampling plots had been chosen in accordance to the survey for the Nature 2000 Network (Korakis et al. 2006). The sampling scheme for the amphibians was based on the breeding phenology of the species occurring in eastern Greece (Arnold 1978; Helmer & Scholte 1985), and each pond of the study area was visited once per month from February to July during the year 2007. The presence of amphibians was detected through a combination of visual encounter, aural and dip net surveys, during the diurnal transects in the banks of the ponds (Kret et al. 2009). We excluded finally the species *Triturus cristatus* as its presence was verified in two sites, only. Similarly, a sub-set of the existing database for small birds (Kati & Sekercioglu 2006) was used for analysis. As the conservation value was one of the factors under evaluation, we included in our analysis only bird species that are "Species of European Conservation Concern" (SPEC; BirdLife International 2004). These included species with an unfavorable conservation status, either concentrated in Europe (SPEC 2) or not (SPEC 3), as well as species with favorable conservation status, but concentrated in Europe (SPEC 4). Finally, for the small birds, the two species *Dendrocopos syriacus* and *D. medius* were used as a combined dataset due to limited detections of *D. medius*. The survey of birds of prey was based on a systematic monitoring of raptor territories that was conducted from 2001 until 2005 (Poirazidis et al. 2009a, 2010c [= Chapter C.2, Chapter C.4 of this thesis]), and we pooled the data of all five years and plotted the centers of the yearly territories. The Black stork (*Ciconia nigra*), a species of conservation priority in the area (Tsachalidis & Poirazidis 2006), was included in the raptor dataset. A subset of the breeding raptor species was used in this study, and the criterion for selection was the relatively high abundance in order to produce stable habitat suitability models.

Habitat suitability maps and statistical analysis

Habitat suitability maps (HSM) have broad applicability within conservation biology and are of special interest to predict the distributions of wildlife species for geographical areas that have not been extensively surveyed. The methods for modeling habitat suitability can be classified into two groups: those requiring presence-only data and those requiring presence-absence data (Guisan & Zimmerman 2000). Here we prepared HSM using Ecological Niche Factor Analysis (ENFA) provided by the software BIOMAPPER (Hirzel et al. 2002). ENFA is a multivariate approach developed to predict habitat suitability based on the likelihood of occurrence of the species when absence data for the species are not available (Hirzel et al. 2002). Without absence data some limitations on the accuracy of the habitat suitability maps are possible (Hirzel & Le Lay 2008), and we reclassified the predictions into four robust levels (=bins) of suitability to overcome this problem (Hirzel et al. 2006). The suitability is based on functions that define the marginality of the species, i.e. how the species mean differs from the mean of the entire area, and the specialization of the species, i.e. the ratio of the overall variance to the species variance. Marginality lies between 0 and 1, with larger values indicating that the focal species has habitat requirements that differ from the average available conditions. A high specialization value indicates that the focal species has a particular requirement for certain habitat characteristics and occupies a narrow range of variables compared to the overall range of variables within the study area (Hirzel et al. 2002).

We used 23 environmental variables, classified into four groups to derive potentially relevant predictors for species habitat selection (Table D.1.1). This database contained maps stored both in a vectorial and a raster format. All species and habitat information was rasterized into a 50 x 50 m grid cell maps. Topographical data were directly obtained as quantitative variables. Variables quantifying land cover, landscape and potential sources of disturbance were transformed into frequency and distance variables. The forest cover categories were reclassified into pure broadleaves, mixed pine-oak and pure pine forest, but only the first two were used for the models, as the information from the third was redundant. As ENFA does not work with multinomial data, these qualitative maps were converted into several Boolean maps (i.e. one for each variable). Frequency describes the proportion of cells from a given category within a circle around the focal cell and it was derived using a circular moving window. We varied the radius of the moving window to test the performance of three different scales (200 m, 500 m and 1000 m), but finally only the scale of 1000 m was used as it performed better than the others. The topographical descriptors were averaged by means of a similar radius circular moving window. Spatial data analysis was conducted using ArcMap 9.0 and the Spatial Analyst extension.

Table D.1.1 *Environmental variables used in ENFA as predictors to define the species' ecological niche.*

Environmental predictors	Scales (m)
Topography	**-**
1. Altitude	200, 500, 1000
2. 1 SD of altitude	200, 500, 1000
3. Slope	200, 500, 1000
4. Northness aspect	200, 500, 1000
Landscape / Forest attributes	**-**
5. Relative richness index	200, 500, 1000
6. Fragmentation index	200, 500, 1000
7. Frequency of broadleaves	200, 500, 1000
8. Frequency of mixed forest (Pine-Oak)	200, 500, 1000
Other ecological metrics	**-**
9. Frequency of openings	200, 500, 1000
10. Frequency of agricultural lands	200, 500, 1000
11. Frequency of permanent water	200, 500, 1000
12. Frequency of rocky area	200, 500, 1000
13. Distance to openings	-
14. Distance to agricultural lands	-
15. Distance to main river	-
16. Distance to permanent water	-
17. Distance to rocky area	-
Potential disturbance metrics	**-**
18. Frequency of paved roads	200, 500, 1000
19. Frequency of unpaved roads	200, 500, 1000
20. Frequency of urban area	200, 500, 1000
21. Distance to paved roads	-
22. Distance to unpaved roads	-
23. Distance to urban area	-

Correlations between all variables of the initial pool of predictors (Table D.1.1) were calculated prior to the ENFA. When two or more predictors had a correlation coefficient greater than 0.7, only the most proximal was kept (Austin 2002). Topographic and frequency environmental layers were normalized using the 'box–cox' algorithm (Sokal and Rohlf 1981) and distance variables by the 'square root' algorithm. There are different algorithms available in BIOMAPPER to build habitat suitability maps by ENFA (Hirzel et al. 2002) and following Hirzel & Arlettaz (2003) we used the geometric mean algorithm to account for the density of the observations in environmental space.

For the plants, the number of species was used as dependent variable per plot and we created two multiple regression models (one for woody plants and one for non-woody vascular plants) to predict species richness. The resulting models were transformed with the "box-Cox byte" algorithm and combined with equal weight (factor 0.5) to produce the overall "plant HSM". For each of the three groups of fauna, an overall HSM was created combining the specific HSMs by user-defined weight per species (Eastman 2001), which depended on the conservation value (Appendix D.1.1). Finally, all HSMs per organism group were combined into an overall biodiversity HSM applying a new user-defined weight per group. The HSM for breeding Black vulture and Egyptian vulture (*Neophron percnopterus*) – the species with the highest conservation value in the area – were not included in the initial raptor HSM, but were used as Boolean data in a later step (see below) to highlight the priority areas for conservation of these two species.

Timber standing volume

We used the recent forest inventory for wood production of the local Forest Service (2006-2016) to produce quantitative maps of the distribution of standing wood volumes (basal area) (Consorzio Forestale del Ticino 2006). We used the stand level as spatial unit to summarize these data (417 sub-units of the division of managed forest, with an average size of 46.5 ± 18.9 ha). The timber volume was described as pine, oak and total volume (Consorzio Forestale del Ticino 2006). We used only the managed area of Dadia NP (zone B1), excluding the non-managed strictly protected areas (Figure D.1.1).

Establishment of the management scenarios

To obtain spatially explicit management plans at stand level, we re-classified the biodiversity thematic maps into four bins representing habitat suitability as: (1) unsuitable, (2) marginal, (3) suitable and (4) optimal. We also reclassified the timber maps into four bins representing the standing volume as: (1) minimum, (2) medium, (3) large and (4) maximum. We used the Natural Break method (ArcMap) for the biodiversity bin classification, and the four timber volume bins were defined by values of total standing timber volume of <500 m^3, 500-1000 m^3, 1000-2000 m^3 and >2000 m^3 per stand. We finally considered four possible general management actions at the stand level, in order to integrate biodiversity values into the timber management: (1) management without limitations (*free forestry*), (2) management with temporal restrictions, (3) management with temporal and spatial restrictions, and (4) management focussing on the ecological values (*ecological management*).

In this study, we implemented three management scenarios. The "*biodiversity scenario*" focused on the maximization of the biodiversity value (maximum environmental profit) in the managed forest. It was defined by the biodiversity models with each bin of habitat suitability leading to related management actions (Table D.1.2), e.g. biodiversity bin 1 *"unsuitable"* lead to management action 1 *"free forestry"* and biodiversity bin 4 *"optimal"* to management action 4 *"ecological management"*. The "*timber scenario*" focused on the maximization of the economical benefits for the timber production (maximum economical profit) and was defined by the standing volume map with each bin of timber density leading to inverse related management actions (Table D.1.2), e.g. timber volume bin 1 "*minimal*" lead to management action 4 "*ecological management*" or timber volume bin 4 *"maximum"* to management action 1 *"management without limitations"*. The third scenario was the "*trade off scenario*", which attempted to maximize the long-term net benefits for both biodiversity and society. The established trade off matrix considered both biodiversity and timber production at the same level and lead to the final determination of the management action for each stand (Table D.1.2).

We applied each scenario for each biodiversity data set as well as for the overall biodiversity HSM. For each scenario at the last step, we used the suitable and optimal areas for Eurasian Black vulture and Egyptian vulture as Boolean variables as such: suitable and optimal areas for Black vulture were upgraded to the

Management action "4" (ecological management) and for Egyptian vulture to the Management action "3" (temporal and spatial restrictions).

Table D.1.2 *Forest management categories determined by biodiversity and timber production under the scenarios biodiversity, timber and trade off.*

Scenario		**Biodiversity**				**Timber**				**Trade Off**			
Timber bins		**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**
Biodiversity bins	**1**	FF	FF	FF	FF	EM	TSR	TR	FF	FF	FF	FF	FF
	2	TR	TR	TR	TR	EM	TSR	TR	FF	TR	TR	FF	FF
	3	TSR	TSR	TSR	TSR	EM	TSR	TR	FF	TSR	TSR	TR	TR
	4	EM	EM	EM	EM	EM	TSR	TR	FF	EM	EM	TSR	TSR

FF: free forestry, TR: temporal restrictions, TSR: temporal and spatial restrictions, EM: ecological management. Biodiversity bins: 1 unsuitable, 2 marginal, 3 suitable, 4 optimal; timber bins: 1 minimal, 2 medium, 3 large, 4 maximal.

Results

Habitat suitability maps

The species richness of vascular plants (351 plant species in 63 plots) was modeled using the eco-geographical variables as independent variables. The resulting regression model for woody plants was "Y = 4.3 + 2.01 northness – 10.29 frequency of openings + 2.53 frequency of mixed forest + 0.001 frequency of rocks + 0.001 distance to agricultural lands", while for non-woody plants it was "Y = 30.4 + 0.24 slope – 0.23 relative richness index + 5.02 frequency of mixed forest". Both models were significant at the level p=0.05 and were combined equally to the overall HSM for plants (Figure D.1.2a)

Amphibians (10 species in 53 plots) showed a pronounced specialization for certain habitats as their mean global marginality was 0.94 (range 0.63-1.35) and their specialization was 4.37 (range 1.59-12.56). Both groups, small birds and raptors, showed intermediate sensibility and differentiation of habitat use. The mean global marginality of small birds was 0.70 (range 0.35-1.05) and the specialization was 3.23 (range 1.13-6.93). For the raptor HSM, ten species of breeding raptors plus the Black stork had a relative abundance that enabled stable models. The mean global marginality for raptors was 0.63 (range 0.17-1.64) and the specialization was 2.05 (range 1.03-6.05). Finally, a separate HSM was created for each taxon-group of animals (Figure D.1.2b,c,d) using species specific weights (Appendix D.1.1). The combined overall biodiversity HSM resulted (Figure D.1.2e), applying the weights of 0.5 for raptors HSM, 0.25 for amphibians HSM, 0.15 for small birds HSM, and 0.1 for plants HSM.

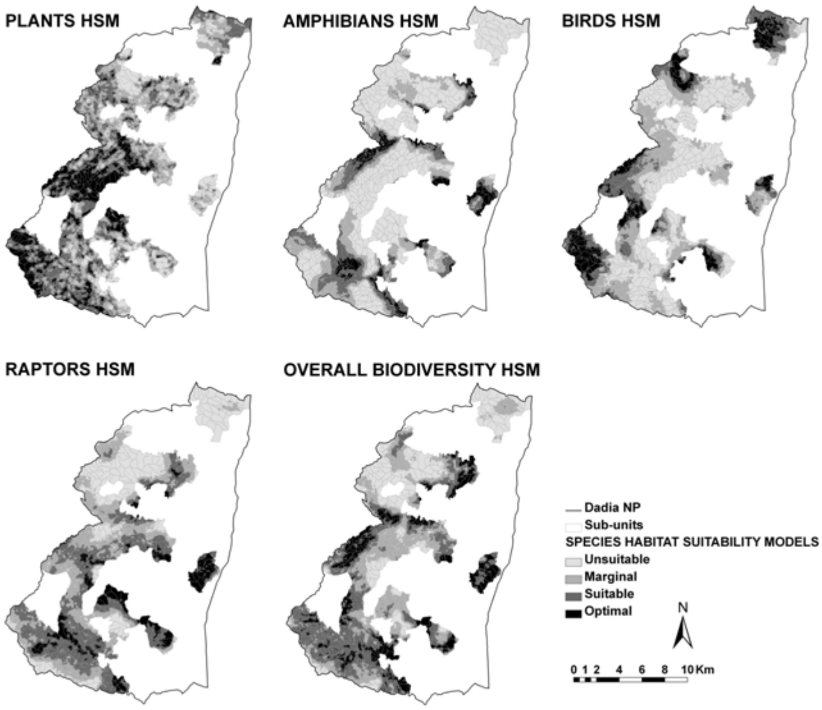

Figure D.1.2. *Habitat suitability maps for (a) plants, (b) amphibians, (c) small birds, (d) raptors and (e) overall biodiversity in Dadia NP.*

Standing volume distribution maps

The mean pine wood volume was 1533.2 m^3 ± 1424.1 (sd) per stand, with a maximum value of 7380.8 m^3 while the mean oak wood volume was 731.5 ± 658.1 m^3 with a maximum value of 4785.3 m^3. The total timber volume ranged from 69 to 8094 m^3 (Figure D.1.3), while the total volume per hectare was 49.2 m^3 ± 26.2 and ranged per forest stand from 2 m^3/ha to 131 m^3/ha.

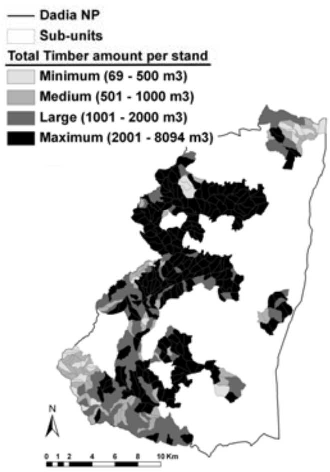

Figure D.1.3 *Total timber standing volume of the managed forest area in Dadia NP.*

Establishment of the management scenarios

We produced three thematic maps of spatially explicit management plans, based on the desired forestry policy in the management area (Figure D.1.4). At the timber scenario, where conservation priorities are considered exclusively in areas without economical value for timber, only 6% of the area was proposed for ecological management and 46% for free forestry. On the other hand, in the biodiversity scenario, where the most suitable areas remain unexploited, 18% of the managed forests were proposed for ecological management and 11% for free forestry. The trade off scenario, taking into account both timber and biodiversity, lies in between, proposing 9% of the area for ecological management and 32% for free forestry.

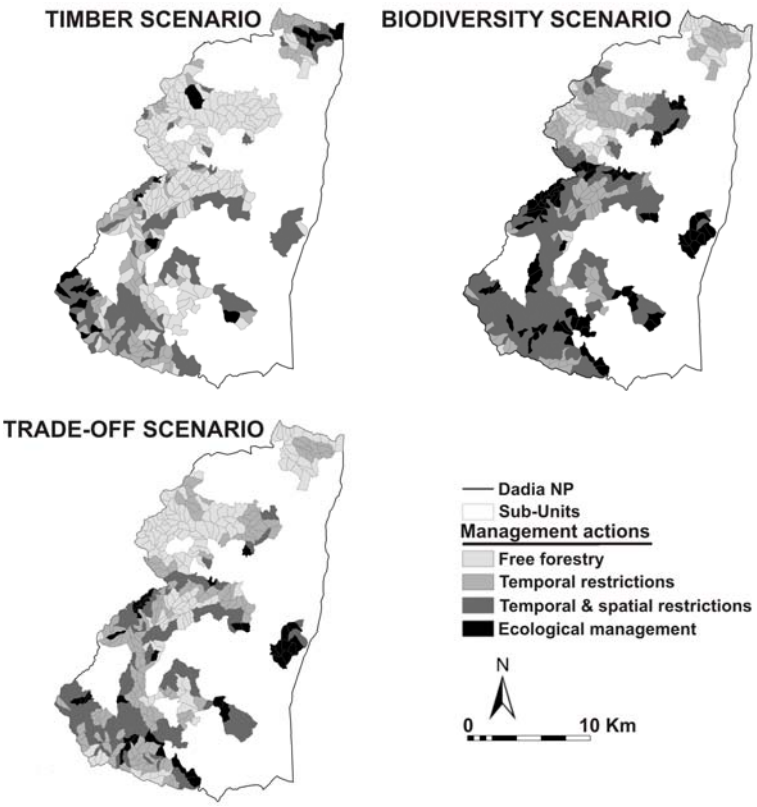

Figure D.1.4. *Spatial forest management plans, presenting the distribution of the four forest management categories under the timber, trade off and biodiversity scenario.*

The trade-off scenario served both ecosystem services, biodiversity values and timber production (Figure D.1.5). In this scenario, 91% of the area with low suitability for biodiversity (*bins unsuitable and marginal*) was covered by the management category "free forestry", while the areas of high suitability for biodiversity (*bins suitable and optimal*) were intensively covered by the management categories "temporal and spatial restrictions" (47%) and "ecological management" (25%). For comparison, in the timber scenario, only 60% of the low biodiversity area was dedicated to free forestry and more importantly only 42% and 4% of the high biodiversity areas were classified as "temporal and spatial restrictions" and "ecological management", respectively (Figure D.1.5).

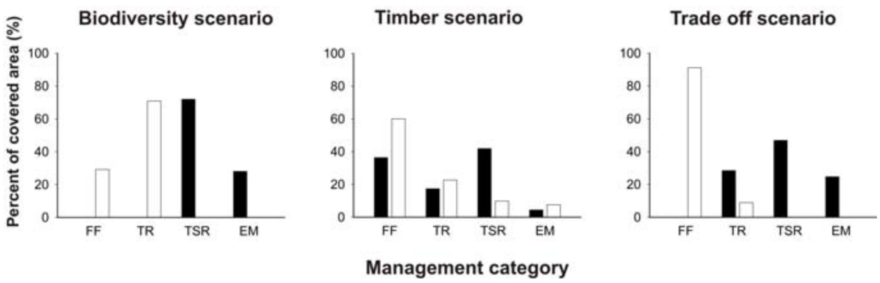

Figure D.1.5. *Management and conservation of areas of differing suitability of biodiversity under the scenarios "Biodiversity", "Trade off", and "Timber". Black bars: forest stands of high suitability for biodiversity (bins suitable and optimal), white bars: forest stands of low suitability for biodiversity (bins unsuitable and marginal); FF: free forestry, TR: temporal restrictions, TSR: temporal and spatial restrictions, EM: ecological management.*

Discussion

Integrating biodiversity into forest management

New environmental policies call for increased attention to biodiversity issues in forest management planning, given that the loss and fragmentation of mature forest together with the structural diversity decline have threatened forest-dependent species (Andrén 1994; Siitonen 2001; Thompson et al. 2003; Angelstam et al. 2004; Poirazidis et al. 2004). Sustainable forestry and deadwood supply have recently emerged as two of the twenty six headline indicators towards halting further biodiversity loss in Europe (European Environmental Agency 2007a). In this frame, the approach developed in this study provides a useful tool for forest managers. We established biodiversity priority areas into the managed areas, providing a guideline for effective management strategies. We also developed habitat suitability models based on environmental features and we identified habitat associations that provide an important source of information for general habitat management issues. These models quantifying relationships between species and their habitats are considering nowadays one of the most efficient tools for forest management (Edenuis & Mikusinsky 2006). Sustainable forest management should be efficient, satisfying on one hand conservation goals while minimizing on the other hand socio-economic costs and the area removed from timber production (Pressey et al. 1997; Montigny & McLean 2005).

Species selection and multi-taxa indicator species

We modeled in this research habitat suitability for several groups of organisms, using totally 351 taxa of vascular plants, 10 species of amphibians and 23 species of birds for the assessment. For a successful use of habitat suitability models in forest biodiversity management an appropriate selection of species is required and multi-taxa bio-indication has several advantages (King et al. 1998, Angelstam et al. 2004; Rempel et al. 2004; Wrbka et al. 2008). Ecologically different taxa can show different pattern of biodiversity and it is assumed that even several species of one single taxa or guild are not enough for being representative (Schulze et al. 2004; Billeter et al. 2008; Cabeza et al. 2008). Also Edenius & Mikuszinski (2006) stress the need for multispecies selection procedures in their recent review on the use of HSM in forest management. They have found only one study (out of 55 reviewed ones) that followed a multi-taxa approach, and only five papers of the review (9%) could be attributed to indicator species in the species selection procedure.

The indicator species approach has been criticized on conceptual grounds, such that no species share the same ecological niche, as well as on empirical grounds, i.e. untested or unverified relationships between the indicator and the species or species groups that the indicator supposedly covers (Lindenmayer et al. 2000; Rempel et al. 2004, Roberge & Angelstam 2004; Edenius & Mikuszinski 2006). In our study we used vascular plants, amphibians, small birds and raptors as indicator groups in habitat suitability models. Recent research confirmed that plants and birds are well performing surrogate taxa for overall biodiversity in Dadia NP (Kati et al. 2004b; see also Sauberer et al. 2004 for a Central European case study). Amphibians, due to their very specific habitat needs and life cycle, are important for being complementary and good indicators of habitat matrix permeability (Ray et al. 2002; Kati et al. 2004a, 2007; Cabeza et al. 2008). Raptors are top predators; requiring enough prey, large areas and limited disturbance, they indicate ecosystem health and perform well as indicators of biodiversity (Sergio et al. 2005; Sekercioglu 2006; but see also Cabeza et al. 2008). Raptors are also focal species of conservation efforts in the reserve, as their populations in Dadia NP are of regional importance (Poirazidis et al. 2004, 2007a, 2010c [= Chapter C.4. of this thesis]; Skartsi et al. 2008).

Decision Support Systems and comparison of scenarios

Concerning limited funding and limited data sources, adaptive management is a useful tool for fast implementations (Angelstam et al. 2004, Duff et al. 2009). Ideally, an active adaptive management approach with iterated assessment and corrective action should be applied through continuous mutual learning by scientists, policymakers, managers and other actors until the targets are reached (Simberloff 1999; Brown et al. 2001; Angelstam et al. 2004; Steffan-Dewenter et al. 2007; Duff et al. 2009). The three scenarios presented in this case study, are adaptive in terms of their main objectives and regarding their simplicity. The timber scenario is a simple approach to integrate conservation of biodiversity into forest management when timber production has the main priority. In this scenario more restrictive conservation management will be done only in forest stands with little timber. The biodiversity scenario can be followed when conservation is the key issue. Restrictions are proposed, where habitat suitability reaches maximum values, the performance regarding conservation is optimal, but the socio-economic benefits remain totally unused in forest stands with a high level of biodiversity. The trade off scenario as an alternative solution, proofed very useful to integrate timber extraction and nature conservation and an optimization of the benefits for society and biodiversity could be achieved. Compared with the timber scenario, free forestry is encouraged were habitat suitability is lower but forest stands of high biodiversity have more restrictions. A decision support system can be an effective mechanism to support technological and managerial decision making (Malczewski 2006) as it can combine multiple sources of information (models and data) into a single system that provides a tool to manipulate the information. With these capabilities, it supports decision makers in cognitive tasks that involve choices, judgment and decisions, in recognizing needs and identifying objectives, as well as in formulating and evaluating different courses of action (Garcia & Armbruster 1997). In the case of sustainable forest management, these actions are forest management scenarios, i.e.

collections of rules and strategies regarding harvest scheduling and forest regeneration (Van Damme et al. 2003).

Timber harvesting and conservation of biodiversity are not necessarily mutually exclusive and some rules of temporal and spatial restrictions can optimize their coexistence (Lõhmus 2005; Brown et al. 2007). Integrating different data sources to a decision support system for spatial forest management planning can increase clearly the sustainability of forest management. Viable populations of indicators species and a high level of biodiversity can be maintained, without loosing the socio-economic benefits of professional timber production. At the local scale, a selective targeting approach that identifies forest stands of potential high biodiversity and nature conservation value is essential. Once identified, these areas can be highlighted for inclusion in future local targets and management prescriptions altered accordingly (Bayliss et al. 2005). As maps of habitat suitability were initially created for individual species, our approach provides also a further resource for species specific conservation management. We recommend applying habitat suitability modeling for selected groups of indicator organisms to develop spatial management plans for managed forests. This enhances the sustainability of the management and promotes monitoring and evaluation of its effects on wildlife. The inclusion of further taxa as indicators of overall biodiversity into the existing decision support system is a prerequisite for continuous improvements of a sustainable forest management.

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Chapter D.2. From research to implementation: nature conservation in the Eastern Rhodopes mountains (European Green Belt)

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Running title: From research to implementation

Keywords: Bulgaria; Conservation in practice; Conservation management; Conservation recommendations; Dadia National Park; Evidence-based conservation; Greece; Protected areas

Illustration by Tuisku Sarrala

Own contribution:

Study design 85%, implementation 50%, writing 65%

Abstract

Nature conservation should ideally build on the scientific recommendations that are concluded from applied conservation research, as well as on monitoring schemes that evaluate the effectiveness of recommendations. We considered as a case study a system of six protected areas located in the Eastern Rhodopes Mountains in the southern part of the European Green Belt (EGB). To investigate nature conservation effectiveness, we reviewed 196 articles from scientific journals and books, 8 doctoral and Master theses, and 39 scientific reports regarding the Greek (one protected area, 428 km^2) and the Bulgarian (five protected areas, 904 km^2) part of the study area. We extracted 743 conservation recommendations, and through questionnaires completed by ten local experts, we found that 74% of the recommendations were familiar for the experts. However, only 52% and 16% of the recommendations were implemented, and an even smaller proportion of 29% and 19% of the above were evaluated in Greece (GR) and Bulgaria (BG) respectively. According to the experts, the main reasons for non-implementation and non-evaluation were absence or incompetence of the responsible authorities. Some recommendations obtained a remarkable low rate of implementation, such as those regarding agriculture and livestock rearing practices (GR: 29%, BG: 16%) or mammal conservation (GR: 0%, BG: 16%). Some other recommendations obtained higher rates at least for Greece, such as hunting and fishing (GR: 88%, BG: 10%) and bird conservation (GR: 57%, BG: 11%). We found that researchers and conservation managers at both sides of the Greek-Bulgarian border face similar implementation problems, related often to the lack of political will for nature conservation and establishment of competent authorities. The role of the EGB is crucial in enhancing the established cross-boarder collaborations between stakeholders involved in nature conservation.

Introduction

Although nature conservation activities have increased substantially over the last decades and environmental change and biodiversity conservation are currently highly ranked in the political agendas worldwide (Pullin & Knight 2001, 2009), it is widely recognized that the 2010 target of halting the loss of biodiversity is not virtually achieved (European Environmental Agency 2007b; Fisher 2009). Considerable time and financial resources have been spent in addressing conservation recommendations all over the world, but it is still a common situation that only few of them have been efficiently implemented (Mauerhofer 2010). Moreover, a large amount of conservation efforts have never been evaluated or monitored, leaving a gap in our knowledge about whether the conservation objectives have been achieved (Pullin & Knight 2001, 2005). Therefore, a recognized challenge in the post 2010 era is to directly link scientific knowledge with policy-making and *in situ* effective implementation of conservation actions (European Platform for Biodiversity Research Strategies 2009; Pressey & Botrill 2009; Pullin et al. 2009).

Such conservation actions and policies are focused mainly in networks of protected areas, given that they constitute the cornerstone of global conservation effort (Chape et al. 2005; Jones-Walters 2007; Jongman 1995; Jongman et al. 2004). The literature abounds of evidence about the importance and inadequacies of protected areas and other conservation strategies (Chape et al. 2005; Mas 2004; Gaston et al. 2008 a,b; Nikolov 2009; Wrbka et al. 2008). However, even within ecological networks, conservation targets can not be achieved without assessments on whether the proposed conservation recommendations have been implemented and once implemented, if they operate effectively (Pullin & Knight 2005). In this study we attempted for the first time to evaluate in a systematic way the real implementation value of scientific conservation recommendations, taking as a case study the area of Eastern Rhodopes Mountains, laying in Greece and Bulgaria.

The Eastern Rhodopes are a part of two biodiversity hotspots, the Mediterranean Basin and the Balkans (Griffiths et al. 2004; Myers et al. 2000; Temple & Terry 2007), and a cornerstone of the existing national and international ecological networks, i.e. IBAs, Natura 2000 and the European Green Belt (Kostadinova and Gramatikov 2007;

Stoychev & Petrova 2003). Situated around the EGB border between Greece and Bulgaria, they maintain a particularly high level of biodiversity (Beron & Popov 2004; Catsadorakis & Källander 2010; Schlumprecht 2010). The area has been comparatively well studied (Beron & Popov 2004; Catsadorakis & Källander 2010) and it should be expected that many conservation recommendations have been developed. Administrative isolation has led to no integrated management of protected areas to date from both sites of the border, although such recommendation exists (Beron & Popov 2004; Vasilakis et al. 2008). Since Bulgaria recently joined the EU, it is crucial to analyze and compare management strategies between Bulgaria and Greece to increase the effectiveness and integration of the existing ecological networks.

The opinion of both Greek and Bulgarian governmental authorities on the status of protected areas is positive, but proposed policy measures are rarely implemented in practice (Apostolopoulou & Pantis 2009; Dimitrakopoulos et al. 2004; Liarikos 2006; Mateeva 2009; Papageorgiou & Vogiatzakis 2006). Although NGOs and environmental scientific organizations have put a lot of effort lobbying for the correct implementation of the elaborated conservation recommendations (e.g. Catsadorakis 2010; Catsadorakis et al. 2010; Kostadinova & Gramatikov 2007; Mateeva 2009), a systematic analysis about effectiveness and flaws of nature conservation measures have not been undertaken so far in any of the two countries. The main objectives of the present study were to summarize conservation recommendations for several model sites in the Eastern Rhodopes Mountains, and to analyze for the Greek and the Bulgarian part the level of implementation and the reasons for their non-implementation. Further aims were to assess the degree of evaluation of effectiveness, the reasons for non-evaluation, and the sources of information uptake of local conservation managers.

Methods

Study area

The Eastern Rhodopes mountains (Figure D.2.1) occupy about 6000 km^2 shared between Greece (1800 km^2) and Bulgaria (4200 km^2) (Beron & Popov 2004). They are characterized by Continental-Mediterranean climate and a hilly and low mountainous landscape with altitudes ranging from 0 to 1483 m (Beron & Popov

2004). Specific natural and cultural values occur commonly in both countries, including traditional pastoralism and the resulting landscape heterogeneity, old-growth pine and oak forests and a few other rare habitats, high level of endemic and rare plant and animal species, high diversity and population density of raptorial birds, and finally a variety of geological and cultural monuments (Beron & Popov 2004; Poirazidis et al. 2002).

In the Greek part of Eastern Rhodopes we limited our study to one large protected area, Dadia National Park (Dadia NP), covering 428 km^2, including two strictly protected core areas (78 km^2). We did not consider the three established protected areas Treis Vryses (99 km^2), Oreinos Evros (489 km^2) and Potamos Filouris (21 km^2), (Schlumprecht & Ludwig 2007), as we were not aware of any publications regarding these areas. Dadia NP is a hilly area (altitudes ranging from 20 to 645 m), covered by extensive pine and oak forest and characterized by a heterogeneous landscape (Kati et al. 2010 [= Chapter B.4 of this thesis]; Poirazidis 2003a; Schindler et al. 2008, 2010 [= Chapter B.1, Chapter A.2 of this thesis]). It is an essential refuge for breeding populations of a unique assemblage of raptors (Poirazidis et al. 2009a, 2010c [= Chapter C.2, Chapter C.4 of this thesis]), containing the only remaining Black Vulture (*Aegypius monachus*) breeding colony in the Balkan Peninsula (Poirazidis et al. 2004; Skartsi et al. 2008), and a high diversity of passerines (Kati & Sekercioglu 2006), amphibians and reptiles (Kati et al. 2007), butterflies (Grill & Cleary 2003), grasshoppers (Kati et al. 2004), and vascular plants (Kati et al. 2000; Korakis et al. 2006).

The Bulgarian part of Eastern Rhodopes includes five protected areas of totally 904 km^2 (Figure D.2.1): the important bird areas (IBAs) Arda Bridge (150 km^2), Byala Reka (446 km^2), Krumovitza (112 km^2), Madzharovo (36 km^2) and Studen Kladenets (160 km^2). The whole area is characterized by exceptional biodiversity, including about 50% of Bulgarian flora, 70% of Bulgarian herpetofauna, and 70% of Bulgarian avifauna (Beron & Popov 2004). Furthermore, the National Strategy for Conservation of Biodiversity (NSCB) considers the Bulgarian part of Eastern Rhodopes as a priority area for the creation of new protected areas at a national scale, because of its great importance concerning species diversity, endemism and rarity.

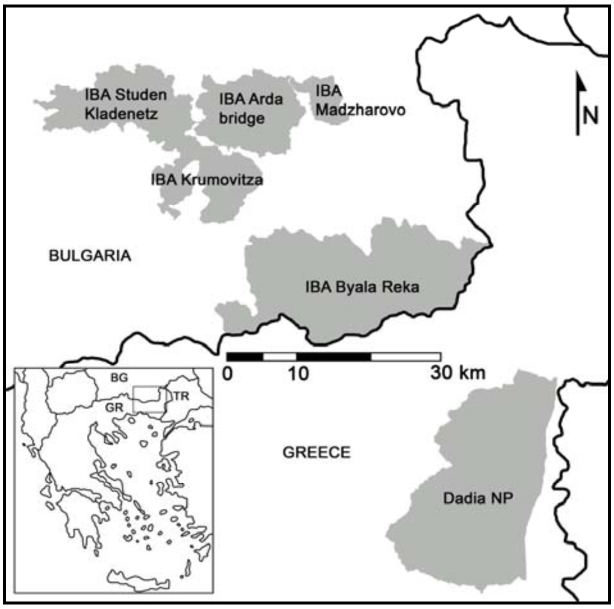

Figure D.2.1. *The Eastern Rhodopes Mountains, located in north-eastern Greece and southern Bulgaria, and the six protected areas, for which conservation recommendations were detected, extracted and analyzed in the frame of this study.*

Literature Review

To collect scientific literature, we used the SCOPUS online search machine and the following terms for title, abstract & keywords: "(Eastern Rhodopes OR southern Bulgaria OR north-eastern Greece) AND (biodiversity OR conservation)". We additionally considered the literature compiled in recent scientific books about the Eastern Rhodopes (Stoychev & Petrova 2003; Beron & Popov 2004; Kostadionva & Gramatikov 2007; Catsadorakis & Källander 2010). We reviewed the collected articles that included peer reviewed ones such as journal articles, short communications, book chapters, conference proceedings, and papers in local journals as well as not necessarily peer-reviewed ones such as unpublished reports and doctoral and master theses. We thoroughly reviewed and included in our analyses all peer reviewed literature, but only the local and grey literature that was assessed as relevant and not repetitive to scientific publications. We inventoried all conservation recommendations from the literature, and extracted for each recommendation, its conservation goal, and the name of the specific area and taxonomic group it concerned (see Appendix D.2.4).

Evaluation of recommendation implementation

We distributed a questionnaire to five local conservation experts in each country to assess the implementation of the recommendations (Appendix D.2.1). Several questions and categories were adapted from a survey of management-plan compilers in the United Kingdom and Australia (Pullin & Knight 2005). The experts were selected based on their great experience and overview about the conservation management in the area and had published at least one scientific paper on this issue. The recommendations were randomly and equally distributed to the experts for evaluation. The experts had to answer for each recommendation (a) whether they ever heard about it and from which source, (b) whether the recommendation is or not implemented and why, and in case of implementation, whether (c) justification was given for the implementation and (d) whether the effectiveness of the recommendation was evaluated.

Data analysis

In our analysis, we grouped the recommendations in the following eleven categories: legislation, administration, research, monitoring, landscape conservation, forest management, agriculture and livestock rearing, wildlife management, hunting and fishing, tourism and environmental education, and sustainable development. We assessed the level of awareness of the recommendations by the local conservation experts, the experts' sources of information uptake, and the degree of implementation of recommendations as proportion of those implemented from the total recommendations proposed for each taxonomic group and for each of the above categories (Appendix D.2.4). In the same way, we assessed the degree of evaluation per taxon and per category.

Results

Reviewed literature

We reviewed 119 and 124 articles respectively for the Greek (GR) and the Bulgarian (BG) part of the Eastern Rhodopes (Table D.2.1). Thereof 101 articles had as their main topic nature conservation, and a total of 743 recommendations were extracted from 105 articles (Table D.2.1, see Appendix D.2.3 for the articles, and Appendix D.2.4

for the recommendations). Particularly in Bulgaria, there is a lack of conservation recommendations in scientific papers. In the 31 reviewed journal papers, only one recommendation was detected, while 416 recommendations were detected in the reviewed book chapters and unpublished reports (Table D.2.1).

Table D.2.1. *Reviewed articles and extracted recommendations for the Greek (GR) and the Bulgarian (BG) part of the Eastern Rhodopes.*

Source	**Detected articles**		**Reviewed articles**		**with topic nature conservation**		**Articles with conservation recommendations**		**Number of conservation recommendations**	
	GR	**BG**	**GR**	**BG**	**GR**	**BG**	**GR**	**BG**	**GR**	**BG**
Per reviewed literature										
Journal papers	70	42	55	31	20	3	29	1	99	1
Short communications	21	8	17	7	4	0	4	0	11	0
Book chapters	15	54	12	53	8	15	9	13	34	198
Conference proceedings	21	9	19	2	8	0	10	0	46	0
Papers in local journals	87	19	0	0	0	0	0	0	0	0
Ph.D./M.Sc. theses	32	3	6	2	6	1	5	1	23	2
Reports	86	31	10	29	6	30	9	24	111	218
TOTAL per country	332	166	119	124	52	49	66	39	324	419
TOTAL in GR & BG	487		243		101		105		743	

Implementation, justification and evaluation

The local conservation experts were aware of most of the recommendations (79% GR, 70% BG): in the Greek part mainly from existing management plans (42%), in Bulgaria mainly from unpublished reports (43%) and books (33%) (Figure D.2.2). More than half (52%) of the Greek but only 16% of the Bulgarian recommendations were already implemented. In both countries, the main reasons given for non-implementation were the lack of sufficient competence in the responsible authorities (44% GR, 17% BG), the absence of responsible authorities (28% GR, 10 % BG), and costly implementation of the recommendation (23% GR, 10% BG) (Figure D.2.2). In Bulgaria, most of the reasons for the non-implementation of proposed recommendations were not listed in our questionnaire ("other") and were related to the "lack of political will".

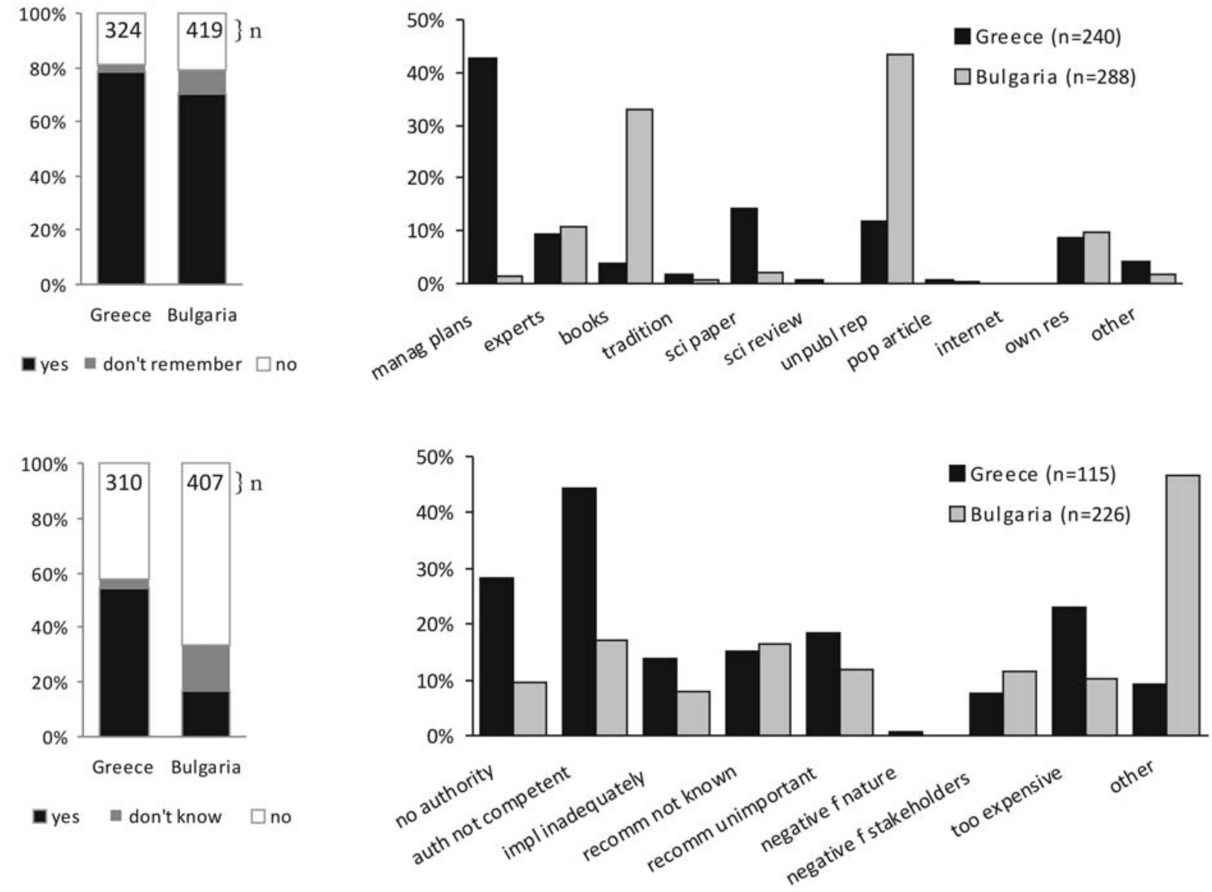

Figure D.2.2. *Sources of information uptake of conservation experts and rates of implementation of published conservation recommendations for the Greek and the Bulgarian part of the Eastern Rhodopes Mountains. Answers to the questions "Have you ever heard before about this recommendation, and if yes, from which source?" (upper panels) and "Is this recommendation implemented in your area and if not, why not?" (lower panels). N-values show the number of recommendations for which an answer was provided by the experts. For the exact formulation of the chosen options, see Appendix D.2.1.*

Justification was given for 79% of the Greek implementations (mostly in the form of management plans, 49%) but for only 31% of the Bulgarian ones (mostly in the form of unpublished reports and books, 43% and 33%, respectively) (Figure D.2.3). For only a minority of recommendations not currently implemented, experts could confirm that they will be implemented soon (8% GR, 1.5% BG). A very small proportion of the implemented recommendations has been evaluated regarding their effectiveness (as proportion from all recommendations: 15% GR, 3.1% BG; as proportion from the implemented recommendations: 29% GR, 19% BG). Main reasons were the insufficient competence of the responsible authorities (GR 46%, BG 38%), and the absence of such authorities (GR 29%, BG 31%)" (Figure D.2.3).

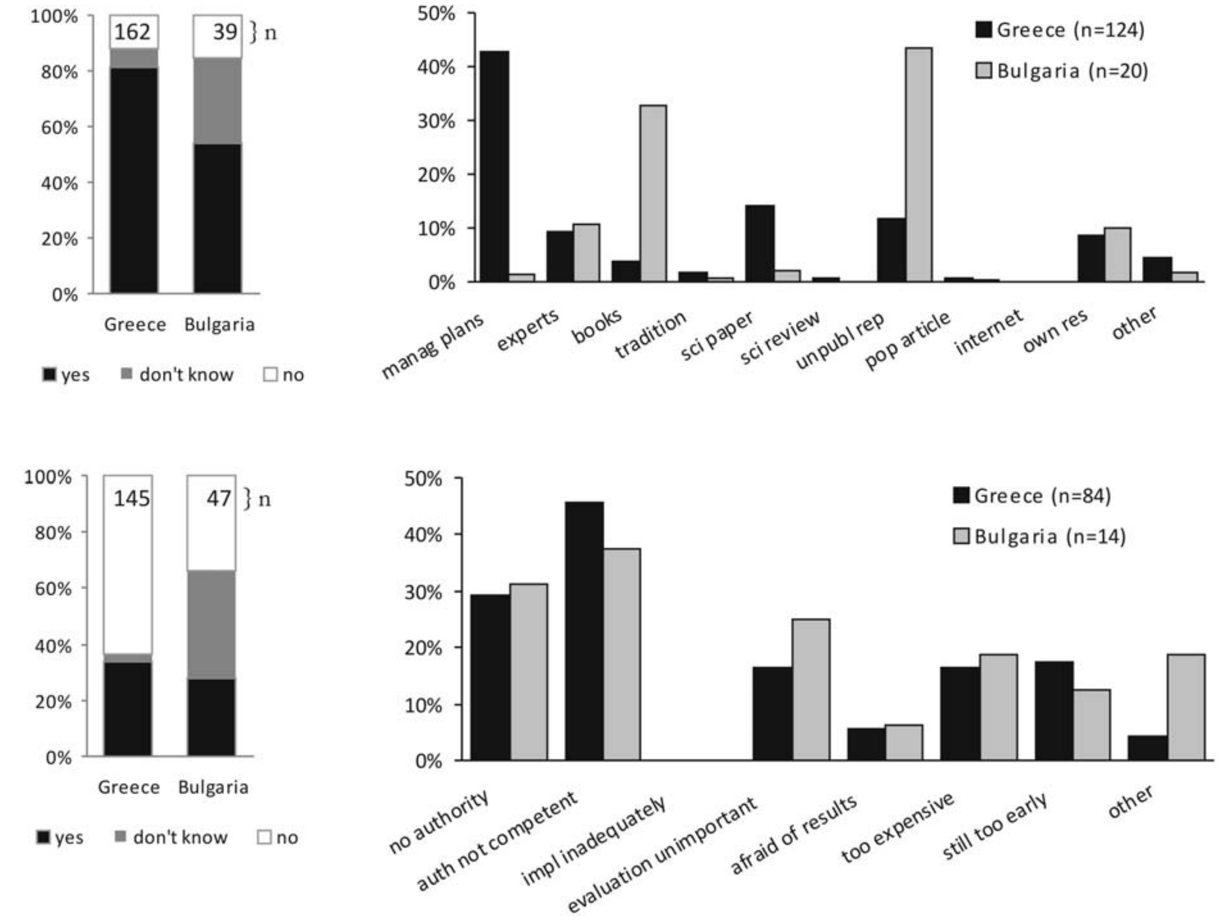

Figure D.2.3. *Rates of justification, and degree of evaluation of implemented conservation recommendations for the Greek and the Bulgarian part of the Eastern Rhodopes Mountains. Answers to the questions "Has justification been given for its implementation, if yes in form of" (upper panels) and "If implemented, is the effectiveness of the implemented recommendation evaluated in your area, and if not, why not?" (lower panels). N-values show the number of recommendations for which an answer was provided by the experts. For the exact formulation of the chosen options, see Appendix D.2.1.*

Taxa and categories of recommendations

Scientific research focused mostly on birds (54), plants (33) and invertebrates (31) in Greece, and on invertebrates (29), reptiles (13) and birds (10) in Bulgaria, while the least studied groups were generally amphibians and particularly fish (Appendix D.2.2). In Greece, the recommendations regarding bird conservation were implemented at a rather high rate (57%) particularly for black vulture (76%) and other raptors (47%) (Appendix D.2.2). Conversely, in Bulgaria the overall rate of implementation of recommendations concerning birds was very low (10%). On the other hand, in Bulgaria conservation recommendations were well implemented for mammals (30%), but no implementation exists in Greece for this group (Figure D.2.4). Few recommendations were implemented for herpetofauna and fish, for invertebrates the rates were 42% (GR) and 17% (BG), and for plants 7% (GR) and 15% (BG). Rates of evaluations were low throughout all taxa, for most of them evaluations were not performed. The highest value (16%) was obtained for birds in Greece (Figure D.2.4).

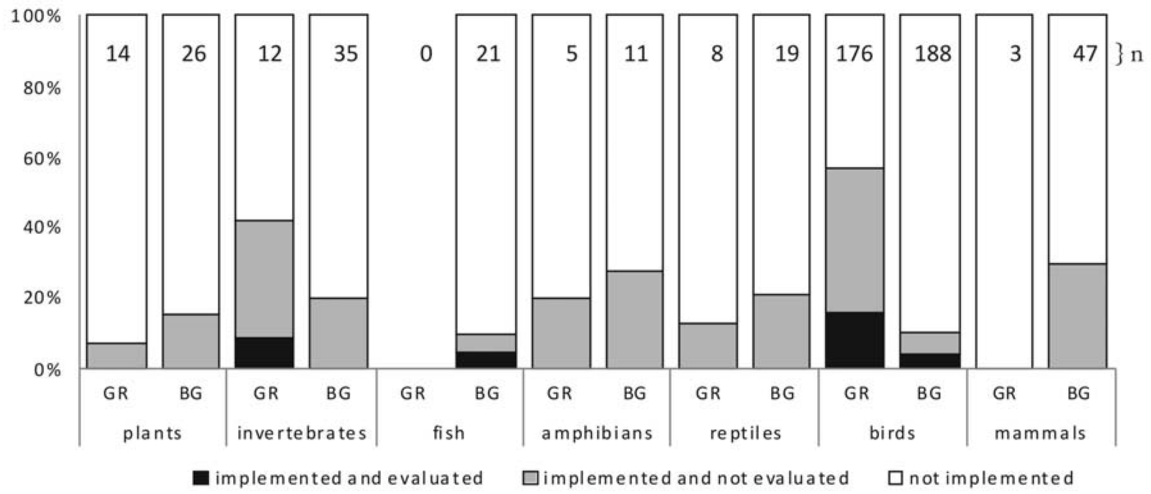

Figure D.2.4. *Taxon specific implementation and evaluation rate of conservation recommendations in the Greek (GR) and Bulgarian (BG) part of the Eastern Rhodopes mountains. n = number of detected and considered recommendations per taxa.*

Most of the recommendations dealt with forest management, administration, legislation (especially in BG), wildlife management, and tourism and environmental education (Figure D.2.5). The highest proportions of implementation were obtained for recommendations regarding hunting & fishing (GR: 88%, BG: 11%), tourism & environmental education (GR: 57%, BG: 42%), and administration (GR: 66%, BG: 17%), while the weakest implementation rates were obtained for recommendations regarding legislation (GR: 14%, BG: 7%), and agriculture & livestock rearing (GR: 29%, BG: 16%).

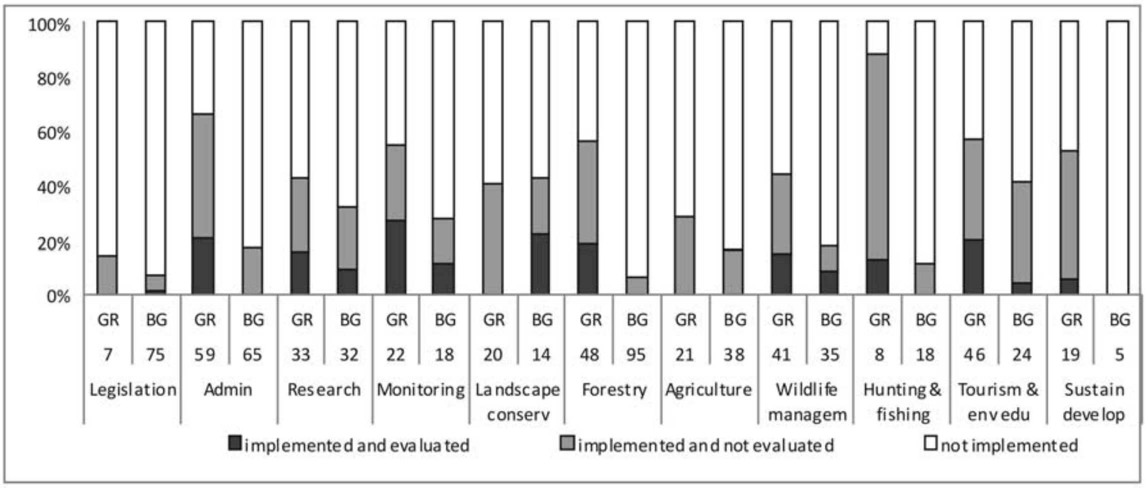

Figure D.2.5. *Proportion of implemented and evaluated recommendations in Greece (GR) and Bulgaria (BG). The number of recommendations is presented below the country codes. The recommendations had been assigned to the categories legislation, administration, research, monitoring, landscape conservation, forest management, agriculture and livestock rearing, wildlife management, hunting and fishing, tourism and environmental education, and sustainable development.*

Discussion

Constraints of evidence-based conservation

In the light of continuous biodiversity loss and global environmental change, there is an urgent need not only for further research and better understanding of our natural world, but even more for concrete synthesis and implementation of the current state of knowledge in practice. We have a scrappy knowledge of biodiversity patterns and natural processes, introducing uncertainty as an inherent characteristic to any conservation decision (Hey et al. 2003; Meffe & Carroll 1994; Regan et al. 2005). Furthermore, we lack a strong mechanism to synthesize, evaluate and communicate the current state of our knowledge as concrete evidence-based recommendations for nature conservation worldwide, accessible and available to decision-makers and managers (Fisher 2009; Loreau & Oteng-Yeboah 2006; Pullin & Knight 2009). Conservationists cannot benefit from predefined and universal prescriptions to solve environmental problems, differentiating between good and harmful management practices and human interventions. Taking as a case study an ecologically important area in the SE European Green-Belt, this study proved furthermore that even in cases where high-standard scientific research is available and solutions are provided through precise conservation recommendations, there is a weakness in implementing them in practice and an even greater weakness in evaluating their effectiveness. A lack of evaluations of effectiveness is particularly problematic, because chances get missed for both, essential improvements in conservation measures, and convincing arguments for decision makers in favour of urgent implementations of conservation measures (Pullin & Knight 2005).

One of the main problems of evidence-based conservation is that scientific research rarely reaches local conservation managers (Pullin & Knight 2001, 2005). This problem can be confirmed by our results. Additionally, at least for Bulgaria, scientific papers very rarely contain any conservation recommendation. Thus, even the conservation experts chosen for this study, all persons who read and publish scientific papers, rarely obtained their information from primary scientific literature.

Differences between Bulgaria and Greece

A main element enhancing conservation is the management plan (Anderson et al. 2002). In this study, the implementation rate was more than three times higher in Greece than in Bulgaria, and also the main sources of information uptake and for justifications for implementation differed between the countries. In Greece, mainly management plans were used, while for the Bulgarian protected areas, management plans are only obligatory for National Parks and Natural Parks, and are often missing in other reserves (Kostadinova & Gramatikov 2007), and the main source were books and unpublished reports. Obviously, a good management plan is not a panacea but an important source of information for conservation managers (Pullin & Knight 2005) and an important step towards effective nature conservation. Further, the majority of Bulgarian scientific literature sources related to biodiversity issues did not discuss any conservation problems and recommendations, while the main part of the unpublished reports (made by nature-protective NGOs) was focused on this. These facts can partly be caused by values and style of scientific writing in Bulgaria, where priority is given to the pure descriptive studies (i.e. faunistics and floristics), while nature-protective NGOs are often constrained by other priorities thus inhibiting the publication of their concepts and studies in scientific journals. This probably influenced the "quality" of the recommendations as most of the Bulgarian ones resulted from experts' knowledge, while only few of them were evidence-based.

Another difference among old and new EU member states is the stage of development of the Natura 2000 process. Although in Greece the system is far from functioning well in practice (Apostolopoulou & Pantis 2009; WWF Greece 2004, 2007), Bulgaria joined the EU and its mechanisms recently and is thus still limping behind in the implementation of the conservation directives. In Greece the designation of 359 Greek Sites of Community Importance (2006/613/EU) is finalized, and 27 management agencies were established in 61 Greek Natura 2000 sites (Apostolopoulou & Pantis 2009), including Dadia NP in the Eastern Rhodopes. In Bulgaria, the draft list for the Natura 2000 network included 551 provisional sites covering about 34% of the national territory (without marine sites) (WWF 2006). The implementation of the network was retarded by the lack of specific budget lines established for the implementation of Natura 2000 and by postponing of 26 SPAs (WWF 2006). Currently, a total of 114 SPAs (based on the existing IBAs) and 225 SCIs were established, covering a total of 33.8% of the state territory (about 24% and 30%, respectively), but very few of them have management plans and agencies (Kostadinova & Gramatikov 2007; WWF 2006).

Greece and Bulgaria differ in the development of their nature conservation activities, especially in those from the public sector. The main drawback in Bulgaria seems to be that authorities that should deal with conservation issue don’t exist yet, and that there is little political will for creating such authorities. In Greece, authorities do exist, but their competence seems to be insufficient, which can be caused by the absence of the adequate mechanism to employ high permanent quality scientific and administrative staff. Further reasons for the inadequacies in both countries should be sought in the lack of vertical and horizontal coordination among state services, the huge overlaps and gaps of responsibilities, the perplexed legal systems, the poor spatial planning systems and, ultimately, the almost complete absence of political commitment to conservation coupled with economic interests related to the territory of the potential protected areas, and a high level of beaurocracy (Apostolopoulou & Pantis 2009; Liarikos 2006; Mateeva 2009; Papageorgiou & Vogiatzakis 2006).

The importance of the conservation history

Pullin & Knight (2005) detected in their survey among UK and Australian management plan compilers that the differing conservation history was a main reason for the different results between the two countries. Similarly, the conservation history differs in our case between the Greek and the Bulgarian part of the Eastern Rhodopes mountains. In Dadia National Park, Greece, raptors and vultures were identified from the beginning of its designation as a Nature Reserve as the main conservation value of the area. As this large forested area was very sparsely populated, the dedicated involvement of WWF Greece in the area allowed the achievement of the minimum necessary conditions, alliances and partnerships with the local authorities, to enable the implementation of many of the necessary conservation measures. The comparatively high implementation and evaluation rates should be attributed to the catalytic, continuous long term presence of the environmental NGO WWF Greece, which had a rightly focused conservation strategy, helped to prepare a well informed management plan, lobbied, made alliances and pressed for implementation of basic conservation measures, and did the monitoring for the evaluation of management on its own resources. Emerging ecotourism helped to raise the awareness of the local stakeholders and to create socio-economic benefits from nature conservation. However, from the taxonomic point of view, the focus was on birds of prey, while cultural and financial restrictions did not permit an extensive implementation of measures for other taxa.

Also the Bulgarian share of the study area was loosely populated and maintained almost undisturbed biodiversity. Unfortunately, during the period between the end of the old regime and the implementation of the pan-european environmental conservation measures, a significant part of the wild habitats and species suffered from tremendous reduction. For instance, between 1992–2000, 70% of the riverain forests along the Maritza River (the biggest river in Southern Bulgaria) was cut down and consequently, the local populations of colonially breeding birds decreased by 70% (Green Balkans, pers. obs.). Although a “National Strategy on the Biodiversity” was developed in 1993–1994, all European Conventions in the field of environmental protection were ratified in 1990–1997, and nine national laws related to nature conservation were announced in 1997–1999, environmental protection remains ineffective. The main reasons for biodiversity loss in the area could be accounted to poverty and to land privatization of 1996. Many nature-protective NGOs (Bulgarian Society for the Protection of Birds, Green Balkans, WWF, Bulgarian Biodiversity Foundation, etc.) fight against these negative environmental processes, but still major problems remain including the lack of developed capacity of environmental policy makers, especially at the administrative level. Only 5% of the personnel in the

Bulgarian Ministry of Environment and Waters (MEW) works in the field of nature conservation and only one to three biodiversity experts work in each of the 15 regional inspections of the MEW, therefore one expert is responsible for the environmental monitoring, threat control and for the effectiveness of seven international conventions and 19 national legislative acts of an area of 5000 km^2.

Implementation rates of different categories of recommendations

Regarding the categories of implementation, Greece has generally higher implementation values than Bulgaria. Comparatively high values in both countries are achieved for tourism and environmental education, while very low values were achieved in both countries for recommendations regarding legislation. Greece obtained the highest rates for the categories hunting & fishing, and administration. Although the Greek rates are still considered as unsatisfactory, they may be attributed to the relative effectiveness of the Greek Forest Service in certain issues, a quality which however declines recently as a result of political decisions and financial restrictions. The categories where Bulgaria obtained comparatively good rates were landscape conservation and tourism and environmental education. The reason could be socio-economical as landscape conservation is directly related to ecotourism development, an issue considered as an important factor for the progress of the local economy (Gerasimov & Stoeva 1997). Landscape conservation also might be a closer concept to the public than the protection of specific taxa.

Conclusions and recommendations

Science has long identified many problems. What is missing is the political will to improve environmental conservation and create a decent national protected areas system in the countries, capable of coping with site-specific issues. NGOs and independent researchers who have been working in the area for long and are able to suggest a number of interconnected priority issues at national, regional and local level in most cases necessitating a "horizontal-type" arrangement, the resolution of which will create the necessary framework for a satisfactory conservation of biodiversity hot spots in the Eastern Rhodopes (Catsadorakis et al. 2010)

We have to conclude that scientists should shift their research focus from purely descriptive to applied ecological and conservation science (Pullin et al. 2009;

Poirazidis et al. 2010b [= Chapter D.1 of this thesis]). They should further be encouraged to communicate their main research findings to local authorities through native language texts. Authorities should provide incentives for increasing the access to scientific literature e.g. by promoting open access journals or by covering the costs of access to standard journals for conservation authorities, and they should promote participatory approaches and effective communication strategies (European Platform for Biodiversity Research Strategies 2009; Papageorgiou & Vogiatzakis 2006) such as web pages in local languages, which organize and prioritize solutions without complex scientific argumentation.

We can further conclude from our research that for successful nature conservation, there is an urgent need for high standard conservation relevant research for many taxa (including rare and rarely studied ones), an increase of quality and quantity of evaluations regarding the effectiveness of conservation measures, a sound interpretation to the local language for conservation managers, establishment, funding and staffing of public authorities, a better access to relevant literature, an increase in collaboration among scientific and nature-protective NGO communities, regions and countries, and cross-border conservation and management activities such as joint projects and conservation initiatives. The European Green Belt is a very important instrument to achieve these prerequisites as it is serving as originator and promoter for several of the recommended activities (Terry et al. 2006; Ullrich & Riecken 2010; Zmelik et al. 2010), and not least for initiating and enhancing collaborations for nature conservation on the scattered political map of Southeastern Europe.

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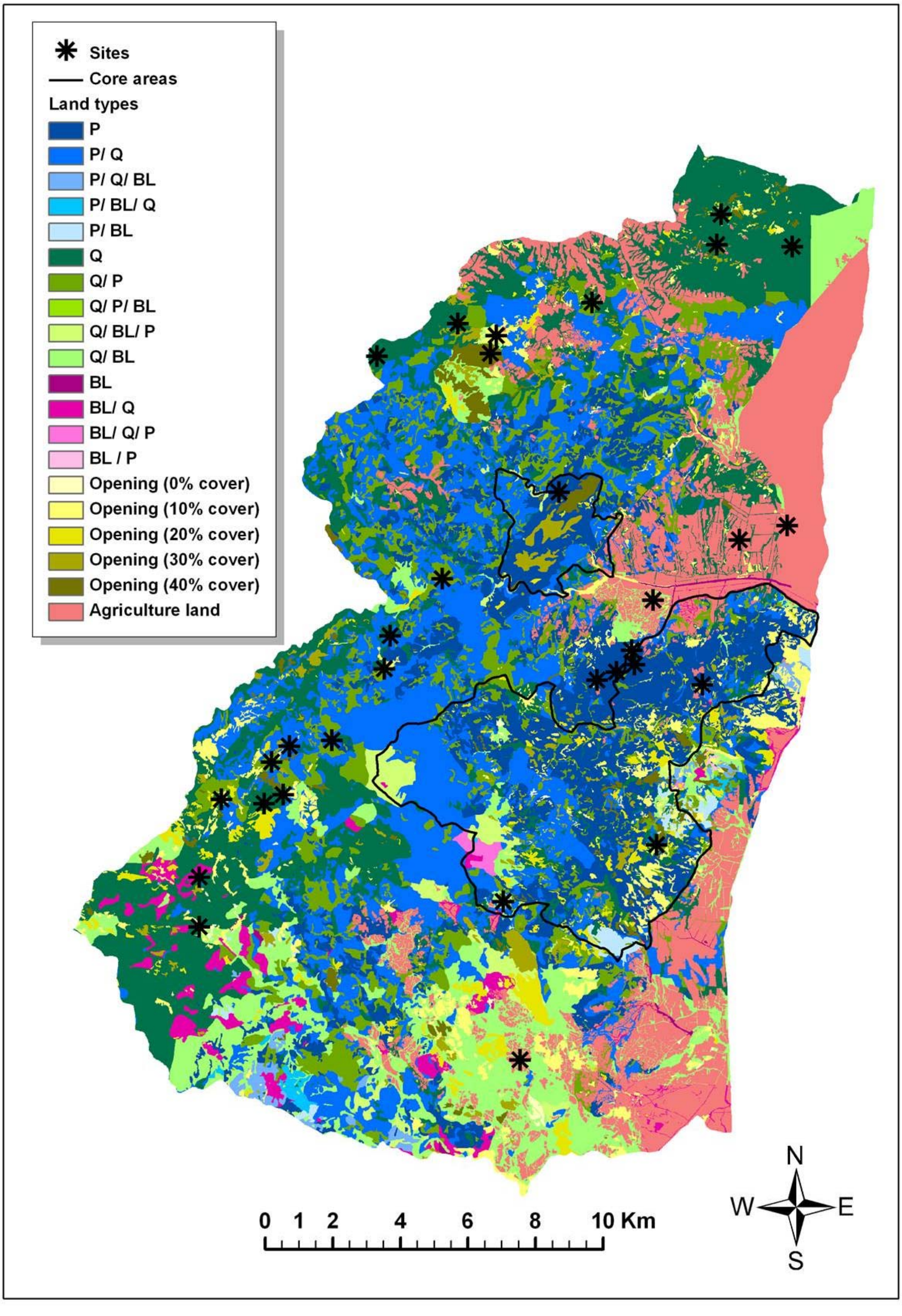
Appendices

Appendix B.4.1. *Vegetation map of the 20 landcover types in Dadia National Park and sites sampled. Forest types: P - pine , Q - oak, BL - broadleaved*



Appendix B.4.2. *Sites sampled in the Dadia National Park. Habitat codes refer to Annex I of the Directive 92/43/EEC (European Commission 2003). Codes in parenthesis refer to the additional Hellenic habitat types (Dafis et al 2001).*

	Site code	**Site description**	**Habitat type code**	**Site area (ha)**
Pinewoods	P1	(Sub-) Mediterranean pine forests (*Pinus nigra*)	*9530	5
	P2	Mediterranean pine forests (*P. brutia*)	9540	20
	P3a	Mediterranean pine forests (*P. brutia*)	9540	15
	P3b	with scrub undergrowth		5
Pine-oak	PQa	Thermophilous pine-oak forests	9540 X (924A)	20
forests	PQb			20
Oak forests	Q1	Thermophilous oak woodlands (*Q.frainetto/cerris)*	(924A)	20
	Q2a	Thermophilous oak woodlands	(924A)	20
	Q2b	(*Q.pubescens)*		20
	Q3a	Thermophilous oak woodlands	(924A)	20
	Q3b	(*Q.pubescens)* with scrub undergrowth		20
	Q4	*Quercus pubescens* open woodlands	(924A)	20
Broad-leaved	BL1a	Alluvial forests (*Alnus glutinosa)*	*91E0	10
woods	BL1b			10
	BL2a	Scrubs and woodlands with *Arbutus*	9340	15
	BL2b	*unedo.*		15
	BL3	Thermo-Mediterranean and pre-desert scrub (*Ph. latifolia)* X pseudosteppe with grasses and annuals.	(934A) X *6220	10
Heaths	Ha	Garrigues of Eastern Mediterranean	(5340)	10
	Hb	(*Erica arborea)*		10
Grasslands	G1	Pseudomaquis X Mediterranean tall humid grasslands	(5350) X 6420	10
	G2a	Pseudo-steppe with grasses and annuals	*6220	5
	G2b			10
	G3	Semi-natural dry grasslands and scrubland facies on calcareous substrates	6210	5
Agricultural	A1a	Field crops	(1020)	20
Land	A1b		(1020)	20
	A2a	Rural mosaics (field with hedges)	(1020)	20
	A2b		(1020)	20
Mosaics	M1a	Mosaic: pseudomaquis X lowland hay	(5350) X 6510	20
	M1b	meadows		20
	M2	Mosaic: pseudomaquis x lowland hay meadows X Mediterranean humid grasslands	(5350) X 6510 X 6420	10
Total	30		16	445

**Priority habitat type of Annex I of the Dir. 92/43/EEC.*

Appendix B.4.3. *Database of the 189 species of the six biological groups used in data analysis (see Kati et al. 2004b)*

Woody plants
Acer monspessulanum
Alnus glutinosa
Arbutus andrachne
Asphodelus aestivus
Carpinus orientalis
Cistus salviaefolius
Clematis vitalba
Clematis viticella
Colutera arborescens
Cornus mas
Cornus sanguinea
Crataegus monogyna
Cytisus villosus
Erica arborea
Ficus carica
Fraxinus ornus
Jasminus fruticans
Juniperus oxycedrus
Ligustrum vulgare
Lonicera sp.
Loranthus sp.
Malus sp
Morus alba
Paliurus spina-christi
Phyllirea latifolia
Pinus brutia
Pinus nigra
Populus nigra
Prunus dulcis
Prunus persica
Prunus spinosa
Pyrus amygdaliformis
Pyrus communis
Pyrus sp.
Quercus cerris
Quercus frainetto
Quercus pubescens
Rosa glutinosa
Rubus sp.
Rubus ulmifolius
Salix cinerea
Salix fragilis
Sambucus nigra
Sorbus domestica
Sorbus torminalis
Tamus communis
Ulmus sp.
Vitis vinifera

Orchids
Anacamptis pyramidalis
Cephalanthera epipactoides
Cephalanthera longifolia
Cephalanthera rubra
Dactylorhiza romana
Epipactis helleborine
Himantoglossum caprinum
Limodorum abortivum
Ophrys mammosa
Orchis coriophora
Orchis fragans
Orchis laxiflora
Orchis mascula
Orchis morio group
Orchis purpurea
Orchis tridentata
Orchis ustulata
Platanthera clorantha
Serapias vomeracea

Orthoptera
Acrida ungarica
Acrometopa servillea
Acrotylus insubricus
Acrotylus patruelis
Aiolopus strepens
Bucephaloptera bucephala
Calliptamus barbarus
Chorthippus bornhalmi
Chorthippus parallelus
Conocephalus hastatus
Decticus verrucivorus
Dociostaurus maroccanus
Euchorthippus declivus
Gryllus campestris
Locusta migratoria
Melanogryllus desertus
Metrioptera oblongicollis
Oecanthus pellucens
Oedaleus decorus
Oedipoda caerulescens
Oedipoda germanica
Oedipoda miniata
Omocestus minutus
Omocestus rufipes
Paracaloptenus caloptenoides
Paranocarodes chopardi
Pezotettix giornae
Pholidoptera aptera
Platycleis escalerai
Platycleis incerta
Platycleis intermedia
Platycleis sepium
Poecilimon brunneri
Poecilimon zwicki
Pterolepis germanica
Sphingonotus caerulans
Tettigonia viridissima
Tylopsis lilifolia

Aq. herpetofauna
Bombina variegata
Bufo bufo
Pseudepidalea viridis
Emys orbicularis
Hyla arborea
Mauremys rivulata
Rana dalmatina
Pelophylax ridibundus
Lissotriton vulgaris

Ter. herpetofauna
Ablepharus kitaibelii
Lacerta viridis/trilineata
Pseudopus apodus
Ophisops elegans
Podarcis erhardii
Podarcis muralis
Podarcis taurica
Testudo graeca
Eurotestudo hermanni

Birds
Aegithalos caudatus
Alauda arvensis
Anthus campestris
Calandrella brachydactyla
Carduelis carduelis
Carduelis chloris
Certhia brachydactyla
Cettia cetti
Coccothraustes coccothraustes
Corvus corax
Corvus corone
Delichon urbica
Dendrocopos major
Dendrocopos medius
Dendrocopos minor
Dendrocopos syriacus
Emberiza cirlus
Emberiza hortulana
Emberiza melanocephala
Erithacus rubecula
Fringilla coelebs
Galerida cristata
Garrulus glandarius
Hippolais olivetorum
Hippolais pallida
Hirundo rustica
Lanius collurio
Lanius senator
Lullula arborea
Luscinia megarhynchos
Melanocorypha calandra
Milaria calandra
Motacilla alba
Motacilla cinerea
Muscicapa striata
Oenathe oenathe
Oriolus oriolus
Parus caeruleus
Parus lugubris
Parus major
Parus palustris
Passer domesticus
Phoenicurus phoenicurus
Phylloscopus bonelli
Phylloscopus collybita
Picus viridis
Regulus ignicapillus
Riparia riparia
Saxicola rubetra
Saxicola torquata
Serinus serinus
Sitta europaea
Streptopelia decaocto
Streptopelia turtur
Sturnus vulgaris
Sylvia atricapilla
Sylvia cantillans
Sylvia communis
Sylvia curruca
Sylvia hortensis
Sylvia melanocephala
Troglodytes troglodytes
Turdus merula
Turdus philomelos
Turdus viscivorus
Upupa epops

Appendix B.4.4. *Ecological heterogeneity indices and species richness for each site sampled*

Ecological Heterogeneity Indices						**Species richness**						
Site code	***SIDI***	***ECON***	***SHAPE***	***NL***	***1/D***	**Total**	**Woody plants**	**Orchids**	**Orthoptera**	**Aquatic herpetofauna**	**Terrestrial herpetofauna**	**Birds**
P1	0.00	0.00	1.33	2.00	1.50	29	1	2	7	1	3	15
P2	0.00	0.00	1.38	4.00	2.80	25	2	1	2	0	2	18
P3a	0.01	18.53	1.29	4.00	5.26	39	8	1	8	0	2	20
P3b	0.00	0.00	1.53	2.00	2.25	32	6	1	8	0	3	14
PQa	0.60	13.52	2.01	5.00	4.59	55	14	6	12	0	4	19
PQb	0.35	9.51	1.68	5.00	3.27	61	13	4	12	4	4	24
Q1	0.00	0.00	1.30	2.00	1.50	39	3	4	9	0	2	21
Q2a	0.00	0.00	1.50	2.00	1.67	41	2	0	17	0	5	17
Q2b	0.12	0.00	1.91	2.00	1.50	32	2	0	8	0	3	19
Q3a	0.29	7.89	1.58	4.00	5.26	49	5	1	11	0	7	25
Q3b	0.34	0.00	2.92	3.00	4.00	56	7	0	15	0	7	27
Q4	0.54	15.75	1.62	5.00	5.99	64	9	6	19	0	3	27
BL1a	0.30	9.39	2.29	3.00	2.10	48	12	0	3	3	2	28
BL1b	0.59	2.53	1.84	4.00	2.80	31	2	0	3	3	2	21
BL2a	0.30	6.78	1.21	3.00	2.10	40	7	3	6	0	1	23
BL2b	0.00	0.00	1.15	4.00	2.80	38	7	2	6	0	0	23
BL3	0.49	8.88	2.38	1.00	1.00	39	5	0	10	0	2	22
Ha	0.08	0.00	1.28	2.00	1.50	42	3	0	11	0	6	22
Hb	0.00	0.00	1.25	2.00	1.50	36	4	0	10	0	4	18
G1	0.60	3.31	1.84	3.00	10.00	59	7	5	15	3	3	26
G2a	0.04	0.00	1.35	3.00	5.99	36	6	0	7	0	3	20
G2b	0.38	0.00	2.13	3.00	5.99	42	6	0	9	2	4	21
G3	0.16	0.00	1.57	2.00	3.00	40	4	0	13	0	2	21
A1a	0.00	0.00	1.11	0.00	0.00	37	1	0	15	4	2	15
A1b	0.16	17.67	3.07	0.00	0.00	41	2	0	13	5	2	19
A2a	0.42	86.09	4.69	4.00	10.00	72	16	0	16	4	2	34
A2b	0.48	69.70	4.80	4.00	7.52	74	19	0	13	6	2	34
M1a	0.66	0.00	2.63	5.00	9.35	80	21	8	11	2	2	36
M1b	0.73	7.17	1.98	4.00	5.26	58	9	0	12	3	3	31
M2	0.67	5.56	2.23	5.00	10.53	77	10	7	17	5	5	33

SIDI: Simpson's Diversity Index of landcover types, ECON: mean edge contrast index, SHAPE: area-weighted mean patch shape index, NL: Number of vegetation layers, 1/D: Simpson's Diversity Index of vertical structure.

Appendix B.4.5. *Average proportion (%) of the species richness that is maintained in the reserve networks of λ_{max} number of sites, designed after the four approaches.*

Biological group	**S_{tot}**	**λ_{max}**	**Reserve design approach**				
			Complementary	**Scoring**	**EH**	**CEH**	**Random**
woody plants	48	8	83	79	64	48	38
orchids	19	5	79	72	62	45	24
orthoptera	38	5	82	75	64	66	47
aquatic herpetofauna	9	3	85	81	67	74	26
terrestrial herpetofauna	9	3	89	85	56	63	42
birds	66	9	85	80	75	69	62
all groups	**189**	**33**	84	79	65	61	40

Appendix C.2.1. *Summary of the specific problems and advantages of the application of the methodology described in this paper for the estimation of the raptor territories.*

Species	***Problems due to weak territoriality of the species***	***Problems with few data due to secretiveness or late arrival***	***Raised accuracy due to many records of important data***	***Total usefulness of the GIS based methodology***
White-tailed Eagle	*medium*	*very few*	*medium*	*medium*
Golden Eagle	*not any*	*very few*	*medium*	*very high*
Imperial Eagle	*very few*	*high*	*medium*	*medium*
Lesser spotted Eagle	*few*	*very few*	*high*	*very high*
Short-toed Eagle	*high*	*not any*	*medium*	*medium*
Booted Eagle	*very few*	*few*	*medium*	*high*
Egyptian Vulture	*very high*	*few*	*very high*	*high*
Common Buzzard	*very few*	*not any*	*high*	*very high*
Long-legged Buzzard	*very few*	*few*	*high*	*very high*
Honey Buzzard	*very few*	*high*	*medium*	*medium*
Black Kite	*very high*	*few*	*very few*	*low*
Marsh Harrier	*high*	*very few*	*very few*	*low*
Goshawk	*not any*	*medium*	*few*	*medium*
Levant Sparrowhawk	*very few*	*very high*	*few*	*low*
Sparrowhawk	*very few*	*medium*	*few*	*medium*
Peregrine Falcon	*few*	*very few*	*high*	*very high*
Lanner Falcon	*very few*	*very few*	*high*	*very high*
Hobby	*not any*	*high*	*few*	*medium*
Eurasian Kestrel	*not any*	*very few*	*medium*	*very high*
Black Stork	*very high*	*not any*	*medium*	*medium*

Appendix C.4.1. *Birds of prey observed in the DNP. B: Breeding; M: Migrating; R: Resident; W: Wintering; S: Summer visitor; BF: Bred formerly.*

	Species		Present status
1	Honey Buzzard	*Pernis apivorus*	B, M
2	Black Kite	*Milvus migrans*	M
3	Red Kite	*Milvus milvus*	M
4	White-tailed Eagle	*Haliaaetus albicilla*	BF
5	Bearded Vulture	*Gypaetus barbatus*	BF
6	Egyptian Vulture	*Neophron percnopterus*	B, M
7	Griffon Vulture	*Gyps fulvus*	R, M
8	Black Vulture	*Aegypius monachus*	R
9	Short-toed Eagle	*Circaetus gallicus*	B, M
10	Marsh Harrier	*Circus aeruginosus*	M, W
11	Hen Harrier	*Circus cyaneus*	M, W
12	Pallid Harrier	*Circus macrourus*	M
13	Montagu's Harrier	*Circus pygargus*	M
14	Goshawk	*Accipiter gentilis*	R, M, W
15	Sparrowhawk	*Accipiter nisus*	R, M, W
16	Levant Sparrowhawk	*Accipiter brevipes*	B, M
17	Common and Steppe Buzzard	*Buteo b. buteo, B. b. vulpinus*	B, M, W
18	Rough-legged Buzzard	*Buteo lagopus*	W
19	Long-legged Buzzard	*Buteo rufinus*	B, M, W
20	Steppe Eagle	*Aquila nipalensis*	M, W
21	Lesser Spotted Eagle	*Aquila pomarina*	B, M
22	Greater Spotted Eagle	*Aquila clanga*	W
23	Imperial Eagle	*Aquila heliaca*	R, W
24	Golden Eagle	*Aquila chrysaetos*	R
25	Booted Eagle	*Hieraaetus pennatus*	B, M
26	Bonelli's Eagle	*Hieraaetus fasciatus*	FB, M
27	Osprey	*Pandion haliaetus*	M
28	Lesser Kestrel	*Falco naumanni*	B, M
29	Common Kestrel	*Falco tinnunculus*	B, M, W
30	Red-footed Falcon	*Falco vespertinus*	M
31	Merlin	*Falco columbarius*	W
32	Hobby	*Falco subbuteo*	B, M
33	Eleonora's Falcon	*Falco eleonorae*	S
34	Lanner Falcon	*Falco biarmicus*	R
35	Saker Falcon	*Falco cherrug*	W
36	Peregrine Falcon	*Falco peregrinus*	B, M, W

Appendix C.4.2. *Historical data on number of territories of birds of prey in the DNP.*

Survey period	**1979 Hallman (1979)**	**1987 Vlachos (1989)**	**1993–1994 Adamakopoulos et al. (1995)**	**1999–2000 Poirazidis (2003)**
Vultures				
Gypaetus barbatus	No data	1 ind.	1 ind.	0
Aegypius monachus	5	12–15	20	20
Gyps fulvus	0	8–10	8–12	0
Neophron percopterus	17	20–25	10–14	13–14
Eagles				
Haliaaetus albicilla	1	1	0	0
Aquila chrysaetos	5	4–5	3–4	4
Aquila heliaca	3	1	0	1
Aquila pomarina	19	16–20	14–17	20
Circaetus gallicus	21	13–16	20–23	37–40
Hieraaetus pennatus	9	8–10	20	21–25
Medium-sized raptors				
Buteo buteo	No data	15–20	16–20	120–130
Buteo rufinus	7	5–10	7–9	4
Pernis apivorus	No data	2–4	10–12	25–30
Hawks				
Accipiter gentilis	18	10–15	10–12	21
Accipiter nisus	No data	5–10	8–10	35
Accipiter brevipes	No data	No data	8–12	7
Falcons				
Falco tinnunculus	No data	No data	5–10	12
Falco subbuteo	No data	?	3–5	12
Falco peregrinus	1	No data	1	2–3
Falco biarmicus	2	1	1	1–2

Appendix D.1.1. *Selected species used for the habitat suitability models for amphibians, small birds and raptors, and user defined weights (adding up to the value of 1 per group. SPEC values for avian "Species of European Conservation Concern" (BirdLife International 2004): 2 - "concentrated in Europe and with an unfavorable conservation status"; 3 - "not concentrated in Europe, but with an unfavorable conservation status"; 4 - "concentrated in Europe, but with a favorable conservation status". For the list of the 351 plant species, used for this analysis see Korakis et al. (2006), available by the authors.*

Species	**-**	**SPEC**	**Weight factor**
Amphibians			
Fire Salamander	*Salamandra salamandra*	-	0.2
Yellow-bellied Toad	*Bombina variegata*	-	0.15
Common Toad	*Bufo bufo*	-	0.1
European Green Toad	*Bufo viridis*	-	0.1
Common Spadefoot	*Pelobates fuscus*	-	0.1
Smooth Newt	*Triturus vulgaris*	-	0.1
European Tree Frog	*Hyla arborea*	-	0.1
Marsh Frog	*Rana ridibunda*	-	0.05
Balkan Stream Frog	*Rana graeca*	-	0.05
Agile Frog	*Rana dalmatina*	-	0.05
Small birds			
Woodchat Shrike	*Lanius senator*	2	0.1
Ortolan Bunting	*Emberiza hortulana*	2	0.1
Black-headed Bunting	*Emberiza melanocephala*	2	0.1
Woodlark	*Lullula arborea*	2	0.1
Corn Bunting	*Milandra calandra*	2	0.1
Bonelli's Warbler	*Phylloscopus bonelli*	2	0.1
Green Woodpecker	*Picus viridis*	2	0.1
Olivaceous Warbler	*Hippolais pallida*	3	0.05
European Bee-eater	*Merops apiaster*	3	0.05
Orphean Warbler	*Sylvia hortensis*	3	0.05
Red-backed Shrike	*Lanius collurio*	3	0.05
Middle Spotted Woodpecker	*Dendrocopos medius*	4	0.05
Syrian Woodpecker	*Dendrocopos syriacus*	4	0.05
Raptors			
Eurasian Black Vulture	*Aegypius monachus*	1	Special category
Egyptian Vulture	*Neophron percnopterus*	3	Special category
Golden Eagle	*Aquila chrysaetos*	3	0.3
Lesser Spotted Eagle	*Aquila pomarina*	2	0.2
Booted Eagle	*Hierraetus pennatus*	3	0.2
Black Stork	*Ciconia nigra*	2	0.1
Short-toed Eagle	*Circaetus gallicus*	3	0.1
Goshawk	*Accipiter gentilis*	-	0.05
Honey Buzzard	*Pernis apivorus*	-	0.03
Common Buzzard	*Buteo buteo*	-	0.01
Sparrowhawk	*Accipiter nisus*	-	0.01

Appendix D.2.1. *Questionnaire filled by local conservation experts assessing the implementation of conservation recommendations in the Eastern Rhodopes Mountains.*

1. Have you ever heard before about this recommendation? (yes/no/don't remeber) ____

1.1 If YES, please choose from where you were informed for the first time (1 option)

- a) existing management plans ☐
- b) expert opinion from expert(s) outside your team ☐
- c) books, handbooks ☐
- d) documentation or personal accounts of traditional management practices ☐
- e) scientific publications ☐
- f) reviews in scientific journals ☐
- g) unpublished reports ☐
- h) popular articles (incl. media such as TV) ☐
- i) internet ☐
- j) own research ☐
- k) other ☐

2. Is this recommendation implemented in your area? (yes/no/don't know) ____

2.1 If you awnsered **NO**:

2.1.1 Why not? (mark all options which are "true")

- a) There is no such authority responsible to implement this recommendation ☐
- b) The responsible authority is not competent enough to implement such recommendation ☐
- c) The responbile authority implemented inadequately this recommendation ☐
- d) The recommendation is not known in my area ☐
- e) The recommendation is not considered as important for my area ☐
- f) The authority does not implement it as it is considered to have negative side effect on other aspects of nature/ecosystems/biodiversity ☐
- g) The authority does not implement it because is is supposed to have negative efects on human activities in the area (hunting, agriculture, forestry...) ☐
- h) The implementation of the recommendation is expensive ☐
- i) other: ☐

2.1.2 Will it be implemented in the near future? (yes/no/don't know) ____

2.2 If you awnsered **YES**:

2.2.1 By whom? ____

2.2.2 When (year) ____

2.2.3 Has justification been given for its implementation? (yes/no/don't know) ____

2.2.3.1 If **yes,** in the form of: (1 option only)

- a) existing management plans ☐
- b) expert opinion from expert(s) outside your team ☐
- c) books, handbooks ☐
- d) documentation or personal accounts of traditional management practices ☐
- e) scientific publications ☐
- f) reviews in scientific journals ☐
- g) unpublished reports ☐
- h) popular articles (incl. media such as TV) ☐
- i) internet ☐
- j) own research ☐
- k) other: ☐

2.2.4 Is this action a continuation of traditional practices? ____

2.2.5 Is the effectiveness of the implemented recommendation evaluated in your area? (yes/no/don't know)

2.2.5.1 If NO, why not? (mark all options which are "true")

- a) There is no such authority responsible to evaluate this implementation ☐
- b) The responsible authority is not competent enough to evaluate such implementation ☐
- c) The responbile authority evaluated inadequately this recommendation ☐
- d) The evaluation is not considered as important for this implemented recommendation ☐
- e) The authority does not evaluate this implemented recommendation, because it is afraid of results that would suggest ineffectivness ☐
- f) The evaluation of the effectivness of the implemented recommendation is expensive ☐
- g) Potential effects of the implemented recommendation will be detectable only after many years, and it is still too early to evaluated them ☐
- h) other: ☐

2.2.5.2 Will it be evaluated in the near future? (yes/no/don't know) ____

Appendix D.2.2. *Number of articles,number of conservation recommendations, and implemention rate per taxon, for the Greek (GR) and the Bulgarian (BG) part of the Eastern Rhodopes Mountains.*

Taxon	**Number of articles**		**Number of articles containing recommendations**		**Number of recommendations**		**% of implemention**	
	GR	**BG**	**GR**	**BG**	**GR**	**BG**	**GR**	**BG**
Animalia								
Birds	51	10	34	10	173	188	56.8	10.1
Black Vulture	15	-	10	-	59	-	76.3	-
Short-toed Eagle	6	-	5	-	16	-	37.5	-
Lesser-spotted eagle	3	-	1	-	12	-	33.3	-
Imperial Eagle	-	1	-	1	-	9	-	22.2
Lesser Kestrel	-	1	-	1	-	2	-	0.0
Other birds of prey	8	-	6	-	8	-	14.3	-
Vulture diversity	1		1		17		58.8	
Bird of prey diversity	8		7		51		60.8	
Black Stork	2	-	2	-	10	-	40.0	-
Shrikes	1	-	1	-	2	-	0.0	-
Landbirds / Passerines	7	-	1	-	3	-	0.0	-
various spp./ bird diversity	3	8	0	8	0	177	na	9.6
Mammals	9	7	3	6	3	47	0.0	29.8
Bats	5	3	3	2	3	19	0.0	26.3
Wolfs	-	1	-	1	-	12	-	16.7
large mammals	-	1	-	1	-	8	-	62.5
small mammals	2	2	0	2	0	8	na	25.0
Other	1	-	0	-	0	-	na	-
Amphibians	9	5	2	3	5	11	20.0	27.3
Reptiles	12	11	4	6	8	19	12.5	21.1
Fish	0	2	-	2	-	21	-	9.5
Invertebrates	13	44	3	7	12	35	41.6	17.3
Orthoptera	9	-	2	-	7	-	57.1	-
Butterflies & moths	3	4	1	2	5	18	20.0	11.1
Dragonflies	-	2	-	1	-	3	-	33.3
Other insects	1	23	0	1	0	8	na	25.0
Spiders	-	3	-	1	-	2	-	50.0
Other invertebrates	1	15	0	2	4		na	25.0
Plantae	27	5	6	1	14	26	6.7	15.4
Orchids	7	-	2	-	10	-	10.0	-
other flowering plants	3	-	1	-	1	-	0.0	-
Trees and Shrubs / Woody vegetation	19	-	4	-	4	-	0.0	-
Plant diversity	4	5	0	1	0	26	0.0	15.4

Appendix D.2.3. *List of articles of the Greek part of the Eastern Rhodopes Mountains that contained conservation recommendations. Year of publication and number of extracted recommendations (R).*

Year	R	Reference
1971	22	Hoffmann, L., Bauer, W., & Müller, G. (1971). Proposals for Nature Conservation in Northern Greece. IUCN.
1979	30	Hallmann, B. (1979). Guidelines for the conservation of Birds of Prey in Evros. Morges: IUCN & WWF.
1984	5	Χανδρινός, Γ., & Hallmann, B. (1984). Οικοανάπτυξη στο Νομό Έβρου: Δέλτα Έβρου – Δάσος Δαδιάς (Σουφλίου). Athens: Υφυπουργείο Νέας Γενιάς και Αθλητισμού.
1985	2	Helmer, W., & Scholte, P. (1985). Herpetological research in Evros, Greece. Proposal for a biogenetic reserve. Arnhem/Nijmegen: Societas Europaea Herpetologica.
1989	9	Blachos, C. (1989). Ecology of Lesser Spotted Eagle in Dadias Forest of Evros Prefecture. Ph.D. thesis, Aristotle University of Thessaloniki.
1989	7	Dennis, R. (1989). The Conservation and Management of Birds of Prey and their Habitats in Evros; Greece. Munlochy: The Royal Society for the Protection of Birds.
1991	5	Spiropoulou, S. (1991). Black Vulture Conservation and Forest Management in Evros, Greece. MSc. dissertation, University College London.
1995	2	Kalopissis, J. (1995). Cephalanthera epipactoides Fisch & C. A. Meyer. In: D. Phitos, A. Strid, S. Snogerup & W. Greuter (Eds.), The Red Data Book of Rare and Threatened plants of Greece pp. 176-177. Athens: WWF Greece.
1995	18	Adamakopoulos, T., Gatzoyannis, S., & Poirazidis, K. (1995). Specific environmental study of the Forest of Dadia. Part C. Athens: Ministry for the Environment, Physical Planning and Public works , Ministry of Agriculture & WWF Greece.
1995	1	Christensen, K. I. (1995). Eriolobus trilobatus (Poiret) M. Roemer Rosaceae. In: D. Phitos, A. Strid, S. Snogerup & W. Greuter (Eds.), The Red Data Book of Rare and Threatened plants of Greece pp. 254-255. Athens: WWF Greece.
1995	1	Kamari, G. (1995). Minuartia greuteriana Kamari, Caryophyllaceae. In: D. Phitos, A. Strid, S. Snogerup & W. Greuter (Eds.), The Red Data Book of Rare and Threatened plants of Greece pp. 362-363. Athens: WWF Greece.
1996	3	Poirazidis, K., Skartsi, Th., Pistolas, K., & Babakas, P. (1996). Nesting habitat of raptors in Dadia reserve, NE Greece. In: J. Muntaner & J. Mayol (Eds.), Biology and Conservation of Mediterranean Raptors,1994. pp. 327-333. SEO/Birdlife.
1996	17	Skartsi, Th. (1996). An integrated management project in Dadia Forest Reserve implemented by WWF Greece. In: P. Regato (Ed.), SILVA Handbook 1996. A WWF course about Mediterranean forests pp. 107-114. Rome: WWF Mediterranean.
1996	9	Vlachos, C., & Papageorgiou, N. (1996). Breeding biology and feeding of the lesser-spotted eagle (*Aquila pomarina*) in Dadia Forest, north-eastern Greece. In: B.-U. Meyburg & R. D. Chancellor (Eds.), Eagle Studies pp. 337-347. Berlin: World Working Group on Birds of Prey and Owls.
1998	2	Bakaloudis, D. E., Vlachos, C.G., & Holloway, G. J. (1998). Habitat use by short-toed eagles (*Circaetus gallicus*) and their reptilian prey during the breeding season in Dadia Forest (north-eastern Greece). Journal of Applied Ecology, 35, 821-828.
1998	3	Capper, S. (1998). The predation of Testudo spp. by Golden Eagles *Aquila chrysaetos* in the Dadia forest reserve, NE Greece. MSc. dissertation, Reading University.
1998	6	Hallmann, B. (1998). The Black Vultures of Greece. In: E. Tewes, J. J. Sanchez, B. Heredia & M. Bijleveld van Lexmond (Eds.), International Symposium on the Black Vulture in South Eastern Europe and Adjacent regions pp. 27-32. Palma de Mallorca: Black Vulture Conservation Foundation & Frankfurt Zoological Society.

Year	R	Reference
1998	17	Katsadorakis, G., Poirazidis, K., Gatzogiannis, S., Adamakopoulos, T., Tsekouras, G., & Matsouka, P. (1998). The management of the vulture population and habitat in Dadia Forest Reserve (Greece): a conceptual framework. In: E. Tewes, J. J. Sanchez, B. Heredia & M. Bijleveld van Lexmond (Eds.), International Symposium on the Black Vulture in South Eastern Europe and Adjacent regions pp. 11-18. Palma de Mallorca: Black Vulture Conservation Foundation & Frankfurt Zoological Society.
1998	5	Spyropoulou, S. (1998). Black Vulture Conservation in the Dadia forest Reserve. Actions taken up to 1992. In: E. Tewes, J. J. Sanchez, B. Heredia & M. Bijleveld van Lexmond (Eds.), International Symposium on the Black Vulture in South Eastern Europe and Adjacent regions pp. 33-38. Palma de Mallorca: Black Vulture Conservation Foundation & Frankfurt Zoological Society.
1998	4	Valaoras, G. (1998). Alternative development and biodiversity conservation: Two case studies from Greece. The Mountain Forum Online Library, http://www.mtnforum.org/rs/ol.cfm.
1998	1	Vlachos, C., Papageorgiou, N., & Bakaloudis, D. (1998). Effects of the feeding station establishment on the Egyptian vulture *Neophron percnopterus* in Dadia forest, North Eastern Greece. In: R. D.Chancellor, B.-U. Meyburg & J. J. Ferrero (Eds.), Holartic Birds of Prey pp. 197-207. Badajoz: World Working Group on Birds of Prey and Owls & ADENEX.
2000	3	Bakaloudis, D. E. (2000). The ecology of Short-toed Eagle (*Circaetus gallicus*, Gm.) in Dadia-Lefkimi-Soufli Forest complex, Thrace, Greece. Ph.D. thesis, Reading University.
2000	1	Ivanova, Th. (2000). New data on bats (Mammalia: Chiroptera) from the Eastern Rhodopes, Greece (Thrace, Evros). Historia naturalis bulgarica, 11, 117-125.
2000	8	Kati, V., Lebrun, P., Devillers, P., & Papaioannou, H. (2000). Les Orchidées de la réserve de Dadia (Grèce), leurs habitats et leur conservation. Les Naturalistes Belges, 81, 269-282.
2000	3	Valaoras, G. (2000). Conservation and Development in Greek Mountain Areas. In: P. M. Godde, M. F. Price & F. M. Zimmermann (Eds.), Tourism and Development in Mountain Regions, pp. 69-83. CAB International 2000.
2001	1	Bakaloudis, D. E., Vlachos, C., Papageorgiou, N., & Holloway, G. (2001). Nest site habitat selected by Short-toed Eagle (*Circaetus gallicus*) in Dadia-Lefkimi-Soufli forest complex, North-eastern Greece. Ibis, 143, 391-401.
2001	2	Kati, V. (2001). Maximizing biodiversity conservation in a forest ecosystem by identifying zones of hotspot overlap. In: K. Radoglou (Ed.), Proceedings of the International Conference Forest Research: A challenge for an Integrated European Approach pp. 581-585. Thessaloniki.
2001	1	Kati, V., & Papaioannou, H. I. (2001). Identifying biodiversity indicators in a forest ecosystem. In: K. Radoglou (Ed.), Proceedings of the International Conference Forest Research: A challenge for an Integrated European Approach pp. 501-506. Thessaloniki.
2002	3	Stavropoulos, N. (2002). Tourism in protected areas in Greece. MSc. dissertation, University of Greenwich.
2002	2	Moskat, C., & Fuisz, T. I. (2002). Habitat segregation among the woodchat shrike, *Lanius senator*, the red-backed shrike, *Lanius collurio* and the masked shrike, *Lanius nubicus*, in NE Greece. Folia Zoologica, 51, 103-111.
2002	17	Skartsi T., & Poirazidis K., (2002). Management plan for the black vulture (*Aegypius monachus*) in the Nature Reserve of Dadia-Lefkimi-Soufli Forest. Athens: WWF Greece.
2002	5	Valaoras, G., Pistolas, K., & Sotiropoulou, H. Y. (2002). Ecotourism revives rural communities. The case of the Dadia Forest Reserve, Evros, Greece. Mountain Research and Development, 22, 123-127.
2003	5	Grill, A., & Cleary, D. F. R. (2003). Diversity pattern in butterfly communities of the Greek nature reserve Dadia. Biological Conservation, 114, 427-436.
2003	3	Kati, V. (2003). Orthopterans and vultures, living together in the Mediterranean. Ecologia Mediterranea, 29, 107-108.

Year	R	Reference
2004	2	Kati, V., Devillers, P., Dufrêne, M., Legakis, A., Vokou, D., & Lebrun, Ph. (2004). Testing the value of six taxonomic groups as biodiversity indicators at local scale. Conservation Biology, 18, 667-675.
2004	10	Poirazidis, K., Goutner, V., Skartsi, Th., & Stamou, G. (2004). Modelling nesting habitat as a conservation tool for the Eurasian Black Vulture (*Aegypius monachus*) in Dadia Nature Reserve, northeastern Greece. Biological Conservation, 118, 235-48.
2004	2	Kati, V., Devillers, P., Dufrêne, M., Legakis, A., Vokou, D., & Lebrun, Ph. (2004). Hotspots, complementarity or representativeness? Designing optimal small-scale reserves for biodiversity conservation. Biological Conservation, 120, 471-480.
2004	4	Kati, V., Dufrêne, M., Legakis, A., Grill, A., & Lebrun, Ph. (2004). Conservation management for the Orthoptera in the Dadia reserve, Greece. Biological Conservation, 115, 33-44.
2004	1	Väli, Ü., Treinys, R., & Poirazidis, K. (2004). Genetic structure of Greater (*Aquila clanga*) and Lesser Spotted Eagle (A. pomarina) populations: Implications for phylogeography and conservation. In: R. D. Chancellor and B.-U. Meyburg (Eds.), Raptors Worldwide pp. 473-482. World Working Group on Birds of Prey and Owls & MME/Bird Life Hungary.
2005	6	Svoronou, E., & Holden, A. (2005). Ecotourism as a tool for nature conservation: the role of WWF Greece in the Dadia-Lefkimi-Soufli Forest Reserve in Greece. Journal of Sustainable Tourism, 13, 456-467.
2006	2	Hovardas, T., & Stamou, G. P. (2006). Structural and Narrative Reconstruction of Representations of "Environment", "Nature" and "Ecotourism". Society & Natural Resources, 19, 225-237.
2006	4	Kati, V., & Sekercioglou, C.H. (2006). Diversity, ecological structure, and conservation of the landbird community of Dadia reserve, Greece. Diversity and Distributions, 12, 620-629.
2006	1	Bakaloudis, D.E., Vlachos, C.G., & Holloway, G.J. (2006). Nest spacing and breeding performance in Short-toed Eagle Circaetus gallicus in northeast Greece. Bird Study, 52, 330-338.
2006	3	Ministry of Rural Development and Food (2006). Conservation report and management of public forest Dadia-Lefkimi-Soufli, Forest Service of Soufli, Evros Prefecture.
2006	4	Hovardas, T., & Poirazidis, K. (2006). Evaluation of the Environmentalist Dimension of Ecotourism at the Dadia Forest Reserve (Greece). Environmental Management, 38, 810–822.
2006	2	Hovardas, T., & Stamou, G. P. (2006). Structural and narrative reconstruction of rural residents' representations of 'nature', 'wildlife', and 'landscape'. Biodiversity and Conservation, 15, 1745-1770.
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Appendix D.2.4 *List of conservation recommendations extracted in this study, conservation goal and reference area.*

Recommendation	Goal	Area
Legislation		
Establishment of legal/administrative regulations	Integrated conservation management	Buffer zone
Greece should sign the international *Agreement on the Conservation on Bats in Europe* (EUROBATS)	Bat conservation	Everywhere
Seasonal restrictions for logging, hunting and other human activities	Integrated conservation management	Whole DNP
Implementation of relevant laws to stop poisoning events (the use of poisoned baits)	Reduce poisoning events on vultures	Whole DNP
Secure protection, implement conservation measures	Integrated conservation management	Whole DNP
Progress in legal protection	Improvement of the functioning of the reserve	Whole DNP
Avoidance of disturbance during breeding period of Lesser spotted eagle: forbiddance of military activities	Lesser spotted eagle conservation	Whole DNP
Administration		
Research centre defines and/or proposes new regulations	Research for conservation	"Evros Park"
Submit projects to intergovernmental organizations	International collaboration	"Evros Park"
Have well-informed experts as custodians of the park	Administration and wardening	"Evros Park"
Establishment of a research station in the Park	Research for conservation	"Evros Park"
Stimulate interchange of knowledge and work between Greek and foreign experts in research, management, legislation, education and public relations	International collaboration	"Evros Park"
Establishment of a research station in the Park	Research for conservation	"Evros Park"
Second buffer zone towards the east until Evros	Maintain valuable natural resources	"Wider area"
Enforce and broaden the transborder collaboration in bat conservation	Large-scale, integrated bat conservation	"Wider area"
Establish a communication network	Integrated conservation management	"Wider area" 1500km^2
Only temporal forest tracks should be allowed to be built in the buffer area	Maintain valuable natural resources	Buffer zone
Felling projects and road tracing in buffer zone should be studied and approved by supervisor	Maintain valuable natural resources	Buffer zone
Road access in significant parts of the buffer zone must be controlled using bars managed by the Forest Service.	Black vulture conservation	Buffer zone
Delineation of the buffer zone	Integrated conservation management	Buffer zone
Establishment of a National Park	Maintain valuable natural resources	Buffer zone
Completed management plan	Integrated conservation management	Buffer zone
Conservation of a “sensibility zone” area	Raptor conservation	Buffer zone
Establishment of a central Reserve Authority	Integrated conservation management	Buffer zone
Research staff should be member of conservation committee	Maintain valuable natural resources	Core area
Controlling and guarding of the core area by Forest Service	Integrated conservation management	Core area
Forbiddance of creation of new roads in the core area	Integrated conservation management	Core area
Preservation and control of road system in the core area	Black vulture conservation	Core area
Strict control of access to the core area to avoid disturbance to Black vulture and other species	Black vulture conservation	Core area
Creation and implementation of general regulations and restrictions	Integrated conservation management	Core area

Recommendation	Goal	Area
Organization of a fire-prevention protection plan	Integrated conservation management	Core area
Maintenance and control of the road network	Integrated conservation management	Core area
Forbiddance of hunting and creation of new roads	Black vulture conservation	Core area
Forbiddance of hunting in the core area	Integrated conservation management	Core area
Determination of high sensibility area	Black vulture conservation	Core area
Wardening the core zones	Vulture conservation	Core area
Practical and legal support for wardens	Maintain valuable natural resources	Core area
Determination of high sensitivity area	Integrated conservation management	Core area
Organization of fire-prevention plan	Integrated conservation management	Core area
Protection of nest sites in the core area	Black vulture conservation	Core area
Intensify protection, training of wardens	Increase protection of vulture nests	Core area
Superintendence and staff should manage the reserve	Maintain valuable natural resources	Core area
Forbiddance of any activity in the core area from 1st March until 15 September	Black vulture conservation	Core area
Protected area managers have to avoid value contradictions while defining or communicating management priorities as well as during the preparation of environmental awareness campaigns and environmental education projects.	Integrated conservation management	Everywhere
Fundraising	Integrated conservation management	Everywhere
Lobbying for further protection	integrated conservation management	Everywhere
Enforce and broaden the transborder collaboration in vulture conservation	Vulture conservation	Evros
Enforce and broaden the transborder collaboration in vulture conservation	Black vulture conservation	Greece, Bulgaria
Enforce and broaden the transborder collaboration in vulture conservation	Black vulture conservation	Greece, Bulgaria, Balkans
Presence of a permanent scientific authority in the Park	Enhance butterfly diversity	Whole DNP
Promotion of a systematic anti-poisoning strategy, at local and national level	Reduce poisoning events on vultures	Whole DNP
Collaboration of WWF and management body	Black vulture conservation	Whole DNP
Establishment of a central Reserve Authority	Integrated conservation management	Whole DNP
Cluster model of 2 core areas and 1 buffer zone	Maintain valuable natural resources	Whole DNP
Protection of nest sites	Black vulture conservation	Whole DNP
Organization of a fire-prevention protection plan	Black vulture conservation	Whole DNP
Establishment of local conservation agency	Improvement of the functioning of the reserve	Whole DNP
Establishment of a protected area (National Park)	Maintain valuable natural resources	Whole DNP
Park provided with staff for management and research	Maintain valuable natural resources	Whole DNP
Establishment of a central Reserve Authority	Improvement of the functioning of the reserve	Whole DNP
Research centre defines and/or proposes new regulations	Maintain valuable natural resources	Whole DNP
Establish a committee for conservation issues for the whole prefecture	Maintain valuable natural resources	Whole DNP
Collaboration between local community, state authorities and NGO (WWF Greece)	Ecotourism and NP management	Whole DNP
Creation of integrated management plan	Integrated conservation management	Whole DNP

Recommendation	Goal	Area
Establishment of a protected area (National Park)	Maintain valuable natural resources	Whole DNP
Creation of a biogenetic reserve for birds of prey	Raptor conservation	Whole DNP
Research		
Analyze landscape change, land management and its effects	Landscape monitoring & evaluation of measures	Buffer zone
Research on thresholds of forest openings size that do not harm forest species and regeneration	Sustainable forest management	Core area
Assess demographics and gene flow of 2 Spotted eagle species	Spotted eagle conservation	E Europe
Use of complementarity approach in reserve design	High quality reserve design	Everywhere
Use of complementarity approach in reserve design	High quality reserve design	Everywhere
Use of different approaches in reserve design	High quality reserve design	Everywhere
Apply the theoretical methodology developed by Papageorgiou et al. (2006)	Conserve forest genetic diversity	Everywhere
Experimental introduction of European Suslik in selected areas	Long-legged Buzzard conservation	Evros
Habitat suitability modelling for further indicator organisms	Sustainable forest management	Managed forests
Inclusion of further taxa into habitat suitability modelling	Sustainable forest management	Managed forests
Pesticide impact on raptors	Raptor conservation	Whole DNP
Habitat suitability modelling for selected groups of organisms to develop management scenarios for managed forests	Sustainable forest management	Whole DNP
Future study for long term movement pattern of the tortoises' females	Reptile conservation	Whole DNP
Further research on butterfly species of European Conservation Concern	Conservation of endangered species	Whole DNP
To investigate where Egyptian vultures find their tortoise prey in the National Park	Egyptian vulture conservation	Whole DNP
Study possible pesticide impact on Short-toed eagle	Short-toed eagle conservation	Whole DNP
Studies of food requirements and availability for Black vulture should be initiated	Vulture conservation	Whole DNP
Further examine new requirements and the potentials for ecotourism that are derived by the designation of the Protected Area as a NP	Knowledge on potential of ecotourism	Whole DNP
Research on availability of suitable Golden eagle nest site	Raptor conservation	Whole DNP
Assess the role of fire on the region's ecosystems	Orchid conservation	Whole DNP
Further research on causes and circumstances of butterfly decline in the DNP	Butterfly conservation	Whole DNP
Further research on biodiversity indicators	Sustainable forest management	Whole DNP
Specification and mapping of favourably area for breeding of Lesser spotted eagle	Raptor conservation	Whole DNP
Assess availability of suitable raptors’ nesting sites	Raptor conservation	Whole DNP
Assess important areas for breeding raptors	Raptor conservation	Whole DNP
Nest site preference of Black stork outside the core area	Black stork conservation	Whole DNP
Creation of a specific environmental study	High quality environmental management	Whole DNP
Recognition of environmental variables which are necessary for Black vulture's reproduction in the area	Vulture conservation	Whole DNP
Investigation of genetic diversity of Black vulture population	Vulture conservation	Whole DNP
Configuration of methodology for the systematic monitoring of landscape and biotopes	Systematic monitoring of landscape and biotopes	Whole DNP
Configuration of methodology for the systematic monitoring of Black vulture	Systematic monitoring of Black vulture	Whole DNP

Recommendation	Goal	Area
Regular checking of endangered plant's conservation status	Conservation of endangered species	Whole DNP
Configuration of methodology for the systematic monitoring of raptors	Systematic monitoring of raptors	Whole DNP
Monitoring		
Long-term landscape monitoring	Integrated conservation management	"Wider area" 1500km²
Monitoring in buffer zone	Integrated conservation management	Buffer zone
Suitable Black vulture habitat monitoring following systematic monitoring plan	Vulture conservation	Buffer zone
Creation of a monitoring plan for the core area	Integrated conservation management	Core area
Scientific evaluation of wildlife and the evolution of forest vegetation after the implementation of the proposal measurements	Evidence based conservation management	Core area
The gradual canopy closure of the forest around Black vulture nest sites must be monitored periodically	Vulture conservation	Core area
Long-term landscape monitoring	Landscape conservation	Whole DNP
Monitoring of poisoning events	Vulture conservation	Whole DNP
Establishment of a monitoring scheme on biological and environmental factors	Integrated conservation management	Whole DNP
Systematic monitoring of raptor populations	Raptor conservation	Whole DNP
Monitoring plan of Black vulture	Vulture conservation	Whole DNP
Marking of Black vulture	Vulture conservation	Whole DNP
More effort in monitoring Black vulture nest sites and productivity	Vulture conservation	Whole DNP
Long-term monitoring programme for the Short-toed eagle population	Short-toed eagle conservation	Whole DNP
Telemetry of Black vulture	Vulture conservation	Whole DNP
Landbird community included as indicator species in monitoring programme	High quality biodiversity monitoring	Whole DNP
Systematic monitoring of raptor populations	Raptor conservation	Whole DNP
Evaluation of management effects on forest and wildlife	Evidence based conservation management	Whole DNP
Use of woody plants as indicators in biodiversity monitoring	High quality biodiversity monitoring	Whole DNP
Herpetofauna should be integrated per se in the management and monitoring scheme of NP	Herpetofauna conservation	Whole DNP
Long-term landscape monitoring	Landscape conservation	Whole DNP
the use of 9 Orthoptera indicator species and *Paranocarodes chopardi* in the reserve monitoring program	High quality biodiversity monitoring	Whole DNP
Landscape conservation		
Preservation of landscape heterogeneity	Black vulture conservation	Buffer zone
Maintenance of forest openings in the buffer zone	Orthoptera conservation	Buffer zone
Maintain rural mosaics and forest openings in the buffer zone	Bird conservation	Buffer zone
Improvement of the water conditions of the forest	Integrated conservation management	Core area
Improvement of hydrous condition for forest (for example, ponds)	Integrated conservation management	Core area
Maintenance of forest openings in the core area	Orthoptera conservation	Core area
Management of rural areas	Integrated conservation management	Core area
Support (reintroduce) wild ungulates	Maintenance of forest openings for Short-toed eagle conservation	Grasslands in DNP

Recommendation	Goal	Area
Maintenance of rural mosaic landscape	Bird conservation	Mediterranean
Maintain human activities that support landscape mosaic	Short-toed eagle conservation	Whole DNP
Conserve the present gradient from open vegetation to more closed woodland	Raptor conservation	Whole DNP
Maintenance of forest heterogeneity	Orthoptera conservation	Whole DNP
Maintenance of forest openings with high vegetation diversity and structural complexity	Maximizing biodiversity conservation	Whole DNP
Maintenance of forest heterogeneity	Orthoptera conservation	Whole DNP
Maintain rural mosaics and forest openings in the whole park	Maximizing biodiversity conservation	Whole DNP
Controlled burning in the winter time	Conservation of reptiles and raptors	Whole DNP
Maintenance of forest openings with high vegetation diversity and structural complexity	Maintenance of vegetation diversity	Whole DNP
Support (reintroduce) wild ungulates	Maintenance of forest openings for orchid conservation	Whole DNP
Creation and/or restoration of small forest openings in dense forest areas, through low-intensity livestock grazing and re-introduction of herd grazing	Maintenance of forest openings for raptor conservation)	Whole DNP
Maintenance of habitat heterogeneity	Conservation of biodiversity	Whole DNP
Agriculture and livestock rearing		
Control new agricultural development	Sustainable agriculture	"Evros Park"
Support of traditional and alternative forms of agriculture and livestock raining	Integrated conservation management	"Wider area" $1500km^2$
Enhancement of stock farming	Integrated conservation management	Core area
Maintain traditional grazing system in the core area	Black vulture conservation	Core area
Encourage grazing in the core area	Integrated conservation management	Core area
Maintain traditional grazing system in the core area	Integrated conservation management	Core area
Permission for livestock grazing, agriculture, research in the core area	Integrated conservation management	Core area
Management of agricultural areas in the core area	Sustainable agriculture	Core area
Prevent rural depopulation and abandonment of traditional stock-raising practices in Evros	vulture conservation	Evros
Improvement of traditional agriculture practice at small scale	Herpetofauna conservation	Evros
Low-intensity farming systems supported through an active EU agricultural policy.	Short-toed eagle conservation	Farmlands in DNP
Minimize the use of agrochemicals in the intensively cultivated land	Short-toed eagle conservation	Farmlands in DNP
Grazing concentrated on dense shrublands and, to a lesser extent, open areas.	Short-toed Eagle conservation	Grasslands in DNP
Encourage grazing in the whole NP	Raptor conservation	Whole DNP
Encourage grazing in the whole NP	Short-toed eagle conservation	Whole DNP
Adopt a pluri-annual rotational grazing system	Maintain open areas for orchid conservation	Whole DNP
Favour the use of goats and sheep instead of cows	Maintain open areas for orchid conservation	Whole DNP
Encouragement of traditional agricultural practices in areas surrounded by forest and low-intensity grazing	Butterfly conservation	Whole DNP
Encourage grazing in the whole NP	Lesser spotted eagle conservation	Whole DNP
Enhancement of periodical livestock grazing	Orthoptera conservation	Whole DNP
Enhancement of periodical livestock grazing	Preservation of landscape heterogeneity for Orthoptera conservation	Whole DNP

Recommendation	Goal	Area
Forest management		
Forestry practices that maintain value of woodland for scenic beauty and wildlife	Sustainable forest management	"Evros Park"
Systematic rehabilitation of more degraded woodlands	Sustainable forest management	"Evros Park"
Forestry based on indigenous species	Sustainable forest management	"Evros Park"
No logging of trees in neighborhood of raptor/Black stork nests	Raptor and Black stork conservation	"Evros Park"
Logging activities and other disturbance in buffer zone must be restricted to the autumn period	Black vulture conservation	Buffer zone
Long term restoration of afforested areas to mixed self-sustaining forests including high oak forest	Raptor conservation	Buffer zone
Measures and limitations for forest exploitation	Integrated conservation management	Buffer zone
Felling in buffer zone should stay away 50 m from nest sites	Conservation of endangered birds	Buffer zone
Regulations for forest and logging exploitation in the buffer zone	Integrated conservation management	Buffer zone
Preservation of suitable nest trees for Black vulture in the buffer area	Black vulture Conservation	Buffer zone
Reforestation prohibited in the buffer zone	Conservation of endangered birds	Buffer zone
Creation of an annual logging catalogue for the buffer area	Sustainable forest management	Buffer zone
Preservation of all remaining old tree stands in the core area	Preservation of natural forests	Core area
Forest exploitation forbidden in the core area	Black vulture conservation	Core area
Seasonal restrictions for logging	Sustainable forest management	Core area
Management of stands and of partially wooded areas in the core area	Sustainable forest management	Core area
Use "soft-forestry" methods; pay workers by the hour and not per wood quantity	Preservation of natural forests	Core area
Setting aside the previously untouched woodland in the core area	Conservation of endangered birds	Core area
Organization of the collection, transportation and distribution of timber products in the core area	Sustainable forest management	Core area
Stop plantation forestry in the core area	Preservation of natural forests	Core area
Forest exploitation forbidden in the core area	Conservation of endangered birds	Core area
Spatial and temporal organization of interventions	Sustainable forest management	Core area
Preservation of all remaining old tree stands in the core area	Black vulture Conservation	Core area
Maintenance of ecologically acceptable programme of selective felling/natural regeneration in managed woods	Raptor conservation	Core area
Measurements of vegetation management (logging plan)	Integrated conservation management	Core area
Prohibition of mature tree cutting in the core area	Black vulture conservation	Core area
Preservation of the old oak forest in the low and high mountain	Herpetofauna conservation	Evros
Mitigation of afforestation in marginal fields	Short-toed eagle conservation	Farmlands in DNP
Spatial forest management planning (aimed at sustainability)	Sustainable forest management	Managed forests
Preservation of all remaining old tree stands in the whole park	Raptor conservation	Whole DNP
Preservation of small groups of mature forest with loose density and sparse intermediate canopy	Black stork conservation	Whole DNP
Protection of mature pine trees around 60-70 years old	Lesser spotted eagle conservation	Whole DNP
Supply foresters with information & directions that will assist them in the proper planning of forest works	Raptor conservation	Whole DNP
Increase of forest vegetation with implementation of suitable forest measurements (regulation of grazing and reforestation)	Sustainable forest management	Whole DNP

Recommendation	Goal	Area
Preservation of all remaining old tree stands in the whole park	Black stork conservation	Whole DNP
Preservation of isolated groups of mature trees in each forest stand	Black stork conservation	Whole DNP
Preservation of isolated groups of mature trees in each forest stand	Black stork conservation	Whole DNP
Maintenance of the open forest structure	Herpetofauna conservation	Whole DNP
Forestry operations should be restricted during the breeding season in an area of 800m radius around existing nests	Black vulture conservation	Whole DNP
Preservation of mature pines close to water	Black stork conservation	Whole DNP
Logging activities applied in the end of the Black stork breeding season and/or after the young fledged from nests	Black stork conservation	Whole DNP
Preservation of tall trees on steep mountain slopes	Black vulture conservation	Whole DNP
Preservation of all remaining old tree stands in the whole park	Goshawk conservation	Whole DNP
Strictly avoid any forestry activity in the vicinity of Black vulture nests and during breeding season	Black vulture conservation	Whole DNP
Logging of small tree groups in homogenous forest	Black stork conservation	Whole DNP
Maintain or reduce rate and duration of timber extraction in the Park	Short-toed eagle conservation	Whole DNP
Collaboration between foresters and conservationists	Black vulture conservation	Whole DNP
Preservation of all remaining old tree stands in the whole park	Raptor conservation	Whole DNP
Wildlife management		
Cultivation of *Eriolobus trilobatu*s in botanical gardens	Tree species conservation	Botanical gardens
Urgent actions are needed in the buffer zone	Integrated conservation management	Buffer zone
Creation of an action plan for Black vulture conservation	Vulture conservation	Buffer zone
Protection of the active vulture nesting sites	Vulture conservation	Core area
Prevent demolition of disused mine's entrances	Bat conservation	Core area
Operation of the feeding system for vultures	Vulture conservation	Core area
Application of the management plan in the core area	Effective management plan	Core area
Reintroduction / reinforcement plan for ungulates (deer) and of mid-sized birds (partridges) in the core area	Fauna conservation	Core area
Reintroduction and demographic enhancement of *Lepus europaeus, Perdix perdix, Alectoris chukar, Dama dama, Capreolus capreolus*	Fauna conservation	Core area
Creation of artificial feeding places for vultures	Vulture conservation	Core area
Avoid Black vulture restocking	Vulture conservation	Europe
Consider genetic issues in the implementation of European conservation strategies for Black vulture	Vulture conservation	Europe
Promoting hedges containing shrubs and trees in case of land reallotment	Maintain/improve habitat for Short-toed eagle	Farmlands in DNP
Water spring cultivation or the construction of small ponds near openings	Improve habitat for Short-toed eagle & herpetofauna	Grasslands in DNP
Water management to favour water concentrations in grasslands	Improve habitat for Short-toed eagle & herpetofauna	Grasslands in DNP
Total protection of the natural habitat of *Cephalanthera epipactoides* (rare orchid)	Orchid conservation	Greece
Fencing of one or two areas of high abundance of *Cephalanthera epipactoides* (rare orchid) away from tourists	Orchid conservation	Greece
Assure that grazing only takes place after the orchids flowering season (after may)	Orchid conservation	Whole DNP
Prevent construction of forest roads near existing Black vulture nest sites	Vulture conservation	Whole DNP

Recommendation	Goal	Area
Creation of small wetlands in the forested area	Black stork conservation	Whole DNP
Preservation of scrubland in pine forest for protection of small fauna (reptiles, rodents)	Lesser Spotted eagle conservation	Whole DNP
Network of artificial feeding places for Black vultures	Vulture conservation	Whole DNP
Increasing of natural food sources for Black vultures	Vulture conservation	Whole DNP
Conservation management for birds of prey in Dadia	Raptor conservation	Whole DNP
Dispersion of suitable Black stork nesting habitat	Black stork conservation	Whole DNP
Preservation of old olive, almond and walnut groves, and old *Platanus* along the streams	Masked-shrike conservation	Whole DNP
Sustain a suitable number of nesting sites for Lesser spotted eagle	Lesser spotted eagle conservation	Whole DNP
Detection of all trees with Lesser spotted eagle nests and forbiddance of any human activities around 100m of nest	Lesser spotted eagle conservation	Whole DNP
Creation of new nest sites within the current limits of the Black vulture colony	Vulture conservation	Whole DNP
Management of Short-toed eagle nesting area	Short-toed eagle conservation	Whole DNP
Management of Short-toed eagle foraging areas	Short-toed eagle conservation	Whole DNP
Operation of 2 feeding sites for vultures	Vulture conservation	Whole DNP
Firebreaks as a conservation tool for tortoises' nest site and eagles' hunting areas	Fauna conservation	Whole DNP
Delimitation and protection of European ground squirrel areas	Lesser spotted eagle conservation	Whole DNP
Introduction of *Alectoris chukar* (Chukar partridge)	Lesser spotted eagle conservation	Whole DNP
Management for a wide range of species	Effective management plan	Whole DNP
Maintain sheep and goat grazing in the Park	Woodchat shrike conservation	Whole DNP
Habitat management to maintain a high density of reptiles	Reptiles + Lesser-spotted eagle conservation	Whole DNP
Make Black vulture nests more sturdy for storms + construction of artificial eyres in the corners of former distribution area	Vulture conservation	Whole DNP
Operation of 3 feeding sites for vultures	Vulture conservation	Whole DNP
Hunting & fishing		
Sport hunting and fishing in carefully chosen and well-marked zones, away from tourist itineraries	High quality zoning of NP	"Evros Park"
Complete ban on hunting of birds of prey and larger carnivores (exception for problematic animals)	Raptor and Large carnivores conservation	"Evros Park"
Prohibition of hunting in the core area	Avoid negative human impact in core area	Core area
Stop the use of poisoned baits	Vulture conservation	Everywhere
Stop the use of poisoned baits	Black vulture conservation	Everywhere
Stop the use of poisoned baits	Vulture conservation	Evros
Prohibition of hunting in important raptor areas	Raptor conservation	Whole DNP
Stop the use of poisoned baits	Raptor conservation	Whole DNP
Tourism & environmental education		
Road building should maintain scenic beauty and wildlife, and create accesses to tourist	High quality regional planning	"Evros Park"
Tourist facilities in locations that do not damage scenic beauty or wildlife	High quality regional planning	"Evros Park"
Research Centre with own research programme and coordinate activities of visitors	High quality NP information and research center	"Evros Park"
Build access and facilities for tourists guided tours	High quality ecotourism	"Evros Park"
Creation of screens and hides for wildlife viewing	High quality ecotourism	"Evros Park"
Prepare and distribute informative material (e.g. guide books and pamphlets)	High quality ecotourism & environmental education	"Evros Park"
Development of soft recreational activities	Sustainable rural development	"Wider area" 1500km^2

Recommendation	Goal	Area
Regulation in infrastructure and recreation areas	Decrease human impact and human disturbance	Buffer zone
General public should not leave roads, stay overnight, collect organisms, damage nature	Avoid negative human impact	Core area
Organization and control of tours and sight-seeing schemes in the core area	Controlled tourism in core area	Core area
Organization and control of normal and conducted tours	High quality ecotourism & avoid negative impact of unsustainable and uncontrolled tourism	Core area
Limit tourist numbers in the core area	Orchid conservation	Core area
Development of tourism should not be encouraged as long as there is no well managed National Park	Avoid negative impact of unsustainable and uncontrolled tourism	Core area
Organization and control of tours and sight-seeing schemes	Avoid negative impact of unsustainable and uncontrolled tourism	Core area
Creation of infrastructure for tourism in Dadia	Increment of benefits for locals from ecotourism	Dadia village
Combination of tourism and conservation	Sustainable ecotourism as conservation tool	Everywhere
Raise the environmental awareness of rural residents	Environmental education of locals as conservation tool	Everywhere
Thoughtful conservation measures in ecotourism	Sustainable ecotourism	Everywhere
Attraction of visitors outside the area has to provide additional income which otherwise would not be generated	Increment of benefits for locals from ecotourism	Everywhere
Ecotourism should focus on the interplay between society and nature	Public awareness & environmental education	Everywhere
High quality management of ecotourism	Involvement of locals in ecobusiness & sustainable tourism	Everywhere
Basic infrastructure to give incentives to private entrepreneurs	Increment of benefits for locals from ecotourism	Everywhere
Involvement in ecotourism and participation in environmental education programs could not suffice to enhance environmental conservation or quality of life issues within rural communities living in protected areas	Enhance environmental conservation or quality of life issues within rural communities living in protected areas	Everywhere
Visitor information and environmental education not confined to mere descriptions of biodiversity and conservation measures applied within protected areas	High quality ecotourism & environmental education	Everywhere
Establishment of an ecotourism infrastructure, data collection and management system, training of local people	High quality ecotourism & environmental education	Everywhere
High quality of the visitor experience	High quality ecotourism	Everywhere
Environmental education	Increased public awareness as conservation tool	Everywhere
Limit tourist numbers in the whole NP	Butterfly conservation	Whole DNP
Accounting for visitors prior knowledge in developing education methods	High quality ecotourism	Whole DNP
Develop public awareness activities related to Black vultures	Black vulture conservation	Whole DNP
Environmental education of local community	Increased public awareness as conservation tool	Whole DNP
Involvement of local communities	Public awareness & environmental education	Whole DNP
Training of eco-guides	High quality ecotourism	Whole DNP
Presentations and promotional material at national and international meetings	Public awareness & marketing	Whole DNP
Visitor group size compatible with natural and social carrying capacity levels	High quality ecotourism	Whole DNP

Recommendation	**Goal**	**Area**
Suitable ecotourism measurements	Sustainable ecotourism	Whole DNP
Establish limited areas for tourist visits	Orchid conservation	Whole DNP
Balanced increase of ecotourism	Sustainable ecotourism & increment of benefits for locals from ecotourism	Whole DNP
Involvement and sensitization of stakeholders to stop poisoning events	Vulture conservation	Whole DNP
Create accommodation for visitors	Improved ecotourism	Whole DNP
Prepare and distribute informative material (e.g. guide books and pamphlets)	High quality ecotourism & environmental education	Whole DNP
Education and public awareness	Environmental education as conservation tool	Whole DNP
Avoid mass tourism in the whole NP	Orchid conservation	Whole DNP
Sales of local products and local meals	Involvement of locals in ecobusiness	Whole DNP
Content of on-site interpretation favouring interconnections between natural and human features of protected areas	High quality ecotourism	Whole DNP
Educational programmes structured as to take advantage of the site's attraction to visitors	High quality ecotourism	Whole DNP
Sustainable development		
Building of private houses restricted to villages	Avoidance of human activities with significant negative impact on nature	"Evros Park"
No industrial construction should be permitted inside the NP	Avoidance of human activities with significant negative impact on nature	"Evros Park"
Sustainable use of resources in the "wider area"	Sustainable rural development	"Wider area" 1500km^2
Support of crafts and small industries	Support traditional lifestyle	"Wider area" 1500km^2
Regulation in the rural exploitation areas	Avoidance of human activities with significant negative impact on nature	Buffer zone
Large scale changes in land use only with proper impact assessment	Controlled and sustainable land use & land use change	Buffer zone
Prohibition of the execution of earthwork altering geomorhological features and natural beauties	Controlled and sustainable land use & land use change	Core area
Environmental control of infrastructure projects in the core area	Conservation of core areas	Core area
Prohibition of excavation of mines and quarries	Controlled and sustainable land use & land use change	Core area
No industrial construction should be permitted inside the core area	Conservation of core areas	Core area
Changes in land use controlled by reserve staff	Controlled and sustainable land use & land use change	Core area
Prohibition of installation of houses and huts in the core area without the permission of the superintendent	Conservation of core areas	Core area
Involvement women in women's cooperative	Sustainable rural development	Everywhere
Evolution of local community enterprises	Sustainable rural development	Everywhere
Successful partnership between the private and public sectors	Sustainable rural development	Everywhere
Collaboration of private and public bodies, women, local community leaders and conservationists	Sustainable and balanced development beneficial for all stakeholders	Everywhere
Promote rural development plans that safeguard vulture conservation and expansion	Black vulture conservation	Evros

Recommendation	Goal	Area
Restrictions of land use changes in the Park	Controlled and sustainable land use & land use change	Whole DNP
Promote rural development	Sustainable and balanced development beneficial for all stakeholders	Whole DNP

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Main research interests

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Studies and working experience

1989 – 1994: Federal High School for Chemistry and Chemical Industry, Department Biochemistry, Biotechnology and Genetics, Vienna, Austria.

1995 – 2002: Zoology, Ecology and Spanish at the **University of Vienna**, Austria. Graduation with **MSc thesis** on "Territoriality and habitat-use of wintering Common Buzzards (*Buteo buteo*) in Schleswig-Holstein, Germany".

Since 2001: **free lance biologist**, working on several ornithological studies and impact assessments, (mainly regarding wind parks and power lines). Examples of employers are: "SOS Storch" (2001), "Technical Office for Biology Mag. Rainer Raab" (2002, 2003, 2006), "BIOME – Technical Office for Biology, Ecology and Nature Conservation, Mag. Dr. Andreas Traxler" (2006-2009), "Technisches Büro für Biologie, Dr. Hans Peter Kollar" (2007), "NP Neusiedler See – Seewinkel" (2002, 2006-2009).

2003 – 2005: **WWF Greece** - **Dadia project,** working for the LIFE-Nature project "Conservation of Birds of Prey and their habitat in the Dadia Forest Reserve, Greece (LIFENAT02/GR/8497). Main responsibilities: Systematic raptor monitoring, Landscape structure analysis, Error assessment for the telemetry study of Black Vulture (*Aegypius monachus*), Impact Assessment of wind farms on raptors, Development of a capture-recapture methodology for population analysis of Black Vulture, Teaching of colleagues and volunteers (mainly raptor identification and GIS), Systematic capturing of Black and Griffon Vultures (*Gyps fulvus*), Black Vulture breeding monitoring and telemetry, etc.

Since 2006: **PhD at the University of Vienna**, Austria. Dptm. Population Ecology and Dptm. Conservation Biology, Vegetation & Landscape Ecology. Working title: "Landscape structure, biodiversity and birds of prey of Dadia National Park, Greece."

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Teaching experience

- Teaching at the University of Vienna. 1 ECTS. "Naturschutzrelevante Methoden der Sozial-, Wirtschafts- und Kulturwissenschaften": 2010.
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- University of Vienna. Assistant at the field course "Biotope Mapping": 2008.
- 1.5 hour lecture at the Seminar of the Dptm. for Conservation Biology, Univ. of Bern: Oct 2006.
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Awards and Grants

- University of Vienna (Förderungsstipendium, Research Grant for traveling expenses):
  2006: € 1985; 2008: € 980; 2009: € 1600.
- Austrian Research Community (ÖFG): Grants for presentations at international conferences:
  2007: € 400; 2008: € 730; 2009: € 700; 2010: € 400.
- Conference Grants from Scientific Societies
  2007: IALE (International Association of Landscape Ecology): Student Grant for attending the 7th IALE World Congress in Wageningen, Netherlands. € 500.
  2008: IUFRO (International Union of Forest Research Organizations): Student Grant for attending the IUFRO Landscape Ecology Conference, Chengdu, P.R. China. $ 330.
- Awards for best presentations:
  2009: RFF (Raptor Research Foundation): Best Student Poster Award at Annual Meeting: $ 100.

International scientific collaborations (examples)

- University of Ioannina (**Greece**): Dr. Vassiliki Kati
- Demokrit University of Orestiada (**Greece**) : Dr. Kostas Poirazidis
- WWF Greece (**Greece**): Dimitris Vasilakis, researcher
- Bulgarian Academy of Science, Sofia (**Bulgaria**): Dr. Stoyan Nikolov
- Universität Halle (**Germany**): Henrik von Wehrden, PhD candidate
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Memberships in Scientific Societies

IALE (international association for landscape ecology),
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Computer Skills

- Experienced user of GIS and several extensions and of other GIS based software: ArcView 3.2; ArcGIS 9.0 package; LOAS – Location Of A Signal
- Landscape Structure Analysis: FRAGSTATS 3.3
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Native: German

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Editorial Work

Co-editor, *Rural Landscape, Biodiversity and Society* (from 2009): http://rurallandscape.eu/

Guest Editor *Journal for Nature Conservation*, Special Issue: "Landscape and conservation research along the European Green Belt". Planned publication: 2010

Organization of Scientific Meetings

Schindler S, Uebl Ch, Hermann A, Zmelik K, Wrbka T 2008. Initialization-Workshop Platform Biodiversity Research Austria (BDFA), 14-15 Apr 2008, Hardegg, Austria.

Schindler S, Wrbka T, Marold B, Kuttner M, Bohner A 2010 2. Jahrestagung Plattform Biodiversität Forschung Austria (BDFA), 22-23 Feb 2010, Gumpenstein, Austria.

Schindler S, Gallardo A. 2010. Symposium “Measures of landscape structure as ecological indicators and tools for conservation planning and forest management” at the International Conference Forest Landscapes and Global Change: New Frontiers in Management, Conservation and Restoration, 21-27 Sep 2010, Bragança, Portugal.

Participation in International Scientific Meetings

1. EPBRS - Meeting, 13-15 April 2010, Palma de Mallorca, Spain.
2. 18th Conference of the European Bird Census Council, 22-26 Mar 2010, Cáceres, Spain.
3. Workshop “Der Zwischenstaatliche Wissenschaftsrat für Biodiversität und Ökosystemdienstleistungen (IPBES) unter der Lupe”, 1 Dec 2009, Leipzig, Germany.
4. EPBRS Strategy beyond 2009, EPBRS Steering Committee Meeting, 24-25 Nov 2009, Brussels.
5. 2nd DIVERSITAS Open Science Conference - Biodiversity and society: understanding connections, adapting to change. 13-16 Oct 2009, Cape Town, South Africa.
6. 1st Latin American IALE conference 4-7 Oct 2009, Campos de Jordão, Brazil.
7. RRF (Raptor Research Foundation) 2009 Annual Conference, 29 Sep-4 Oct 2009, Pitlochry, Scotland.
8. Infoday FP7-ENV-2010 Environment (including climate change), 17 Sep 2009, Brussels.
9. 2nd European Congress of Conservation Biology, 1-5 Sep 2009, Prague, Czech Republic.
10. 7^{th} EOU Conference 2009, 21-26 Aug 2009, Zurich, CH.
11. European IALE Conference 2009, 12-16 Jul 2009, Salzburg, Austria.
12. Jahrestagung des Umweltfachverbandes, 19-20 Jun 2009, Gaming, Austria.
13. EPBRS - Meeting, 19-22 May 2009, Prague, Czech Republic.
14. 1^{st} Adriatic Flyway Conference, 14-17 Apr 2009, Ulcinj, Montenegro.
15. BOU Annual conference 2009 Lowland farmland birds III, 31 Mar-2 Apr 2009 Leicester, UK.
16. EPBRS - Meeting, 17-21 Nov 2008, Paris, France.
17. Geoscape 2008: Living Landscapes: Memory, Transformation & Future Scenarios, 10-11 Nov 2008, Usti nad Labem, Czech Republic.
18. IUFRO Landscape Ecology Conference, 16-18 Sep 2008, Chengdu, Sichuan, P.R. China.
19. EPBRS Workshop on a "strategy for European biodiversity research", 17 Jun 2008, London.
20. Impact Assessment of Land Use Change, 6-9 Apr 2008, Berlin, Germany.
21. EPBRS - Meeting, 15-18 Jan 2008, Brdo, Slovenia.
22. The White-tailed Eagle in the heart of Europe. 17-18 Nov 2007, Illmitz, Austria.
23. 37^{th} Annual Conference of the Ecological Society of Germany, Austria and Switzerland (GfÖ), 10-14 Sept 2007 Marburg, Germany.
24. Monitoring the effectiveness of nature conservation. International conference at the Swiss Federal Research Institute WSL, 3-6 Sept 2007, Zurich, Switzerland.
25. 7^{th} IALE World Congress. 8-12 Jul 2007, Wageningen, The Netherlands.

26. 17th Conference of the European Bird Census Council. 17-22 Apr 2007, Chiavenna, Italy.
27. Annual meeting RSP (Raptor Protection Slovakia), 21 Oct 2006, Varin, Slovakia.
28. 14th Internat. Symposium on Landscape Ecology Research, 4-7 Oct 2006, Stara Lesna, SLK.
29. 1st European Congress of Conservation Biology, 22-26 Aug 2006, Eger, Hungary.
30. International Conference for Conservation and Management of Vulture Populations. 14-16 Nov 2005, Thessaloniki, Greece.
31. Medforex Annual Meeting 2005. 14-16 Apr 2005, Orestiada, Greece
32. Workshop of the Balcan Network for the Conservation of the Bearded Vulture (*Gypaetus barbartus*). 28-30 Mar 2004, Dadia, Greece.
33. 6th South-Eastern European Bird Migration Network (SEEN) Workshop, 6-8 Feb 2004, Istanbul, Turkey.
34. 7th CMS (Convention on the Conservation of Migratory Species of Wild Animals) meeting. 19-20 Sep 2002, Bonn, Germany.

Scientific Reviews

2007: ARDEA - Netherlands Ornithologists' Union (***IF 2008: 1.33***).

2008: Conservation Biology (***IF 2008: 4.71***); Landscape Online (IALE-D)

2009: Landscape Ecology (***IF 2008: 2.45***); Land Use Policy (***IF 2007: 1.82***); 2 x Journal of Environmental Management (***IF 2008: 1.74***); 2 x Forest Ecology and Management (***IF 2008: 2.11***); Studies in Avian Biology - Cooper Ornithological Society; Czech Science Foundation, Hungarian Science Foundation, Centre for evidence based conservation.

2010: Journal of Environmental Management (***IF 2008: 1.74***)

Publication list (2010.04.30)

theses

1. **SCHINDLER S** ***(2002)*** Territoriality and habitat-use of wintering Common Buzzards (Buteo buteo) in Schleswig-Holstein, Germany. Unpubl. Master Thesis. University of Vienna. 34 pp.
2. **SCHINDLER S** ***(2010)*** Dadia National Park, Greece – an Integrated Study on Landscape, Biodiversity, Raptor Populations and Conservation Management. Doctoral Thesis. University of Vienna. 318 pp.

publications in international peer-reviewed journals

3. Zmelik K, **SCHINDLER S**, Wrbka Th ***(under revision)*** The European Green Belt: targeted research and large scale conservation. Journal for Nature Conservation (IF 2008: 0.94)
4. **SCHINDLER S**, von Wehrden H, Poirazidis K, Wrbka T, Kati V ***(under revision)*** Multiscale performance of landscape metrics as indicators of species richness of plants, insects and vertebrates. Ecological Indicators (IF 2008: 1.98).
5. Poirazidis K, **SCHINDLER S**, Kakalis E, Ruiz C, Bakaloudis D, Scandolara C, Eastham C, Hristov H, Catsadorakis G ***(under revision)*** Population trends in the diverse raptor assemblage of Dadia National Park, Greece. Ardeola (IF 2008: 0.46).
6. **SCHINDLER S**, Curado N, Nikolov S, Kret E., Cárcamo B, Poirazidis K, Catsadorakis G, Kati V ***(under revision)*** From research to implementation: nature conservation in the Eastern Rhodopes mountains (European Green Belt) Journal for Nature Conservation (IF 2008: 0.94)
7. Renetzeder C, Kuttner M, **SCHINDLER S**, Wrbka Th ***(under revision)*** Landscape structure of the European Green Belt. Journal for Nature Conservation (IF 2008: 0.94)
8. Kati V, Poirazidis K, Dufrêne M, Halley JM, Korakis G, **SCHINDLER S**, Dimopoulos P ***(2010)*** Toward the use of ecological heterogeneity to design reserve networks: a case study from Dadia National Park, Greece. Biodiversity and Conservation 19(6), 1585-1597. (IF 2008: 1.47).
9. Renetzeder C, **SCHINDLER S**, Peterseil J, Prinz MA, Mücher S, Wrbka T ***(2010)*** Can we measure ecological sustainability? Landscape pattern as indicator for naturalness and land use intensity at regional, national and European level. Ecological Indicators 10: 39-48. (IF 2008: 1.98).
10. Poirazidis K, **SCHINDLER S**, Ruiz C, Scandolara C ***(2009)*** Monitoring raptor populations – a proposed methodology using repeatable methods and GIS. Avocetta 33, in press.
11. **SCHINDLER S**, Poirazidis K, Wrbka T ***(2008)*** Towards a core set of landscape metrics for biodiversity assessments: a case study from Dadia National Park, Greece. Ecological Indicators 8(5): 502-514. (IF 2008: 1.98)
12. Wrbka T, **SCHINDLER S**, Pollheimer M, Schmitzberger I, Peterseil J ***(2008)*** Impact of the Austrian Agri-Environmental Scheme on diversity of landscape, plants and birds. Community Ecology 9(2): 217-227. (IF 2008: 0.90)

book chapters in peer-reviewed international books

13. POIRAZIDIS K, **SCHINDLER S**, KATI V, MARTINIS A, KALIVAS D, KASIMIADIS D, WRBKA T, PAPAGEORGIOU AC ***(2010)*** Conservation of biodiversity in managed forests: developing an adaptive decision support system. In: Li C, Lafortezza R, Chen J (eds), Landscape ecology and forest management: challenges and solutions in a changing globe. Higher Education Press – Springer. In press
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