ITALIAN PROTECTED AREAS AND THEIR ROLE IN THE CONSERVATION OF VERTEBRATES

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AUTHORIZATION TO SUBMIT DISSERTATION

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ABSTRACT

In response to a specific request from the Italian Ministry of the Environment and Territorial Protection (Directorate of Nature Conservation) I initiated a study on the Italian existing and proposed protected areas. In particular, I considered the conservation status of terrestrial vertebrates (mammals, regularly breeding birds, amphibians, reptiles) and freshwater fish. For each of the (roughly) 500 species considered, I built a habitat suitability model and, using GAP analysis, irreplaceability analysis, and red list criteria, I evaluated their conservation status inside conservation areas. Moreover, I evaluated the capacity of the existing protected areas to act as buffers against the massive changes in land-use/land-cover that are occurring throughout the Italian peninsula.

More than 11% of the Italian national territory is legally protected, but on the average Italian protected areas are small. Moreover, even considering that the number of GAP species in Italy is relatively low, it must be stressed that most of the species presented a conservation deficit and were not represented enough by the existing protected areas.

The protected areas that are currently proposed for institution in Italy (the so-called Natura2000 network) is an extremely important conservation effort that will raise the percentage of national territory to be protected to almost 20%. However, even with this percentage, the system of conservation areas is not able to preserve into a favorable conservation status the species for which it has been instituted. There are important gaps especially in the Mediterranean islands, exactly where the concentration of species important for conservation is highest.

Finally, given their size, protected areas are not able ,with few exceptions, to arrest or even to slow down the land-use/land-cover change that is ongoing in Italy. This is particularly important in flat areas and coastal plains, where the areas are smaller and most of the changes go towards artificial land-use/land-cover classes.

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PREFACE

In 1999, the Italian Ministry of the Environment and Territorial Protection (Directorate of Nature Conservation) outlined design parameters for the National Ecological Network (REN – Rete Ecologica Nazionale) and defined the structure and principal objectives of the system nationwide. Basically, REN is an integrated program that aims to rebalance socio-economic development trends within a framework of sustainable growth and optimal biodiversity conservation and, as such, it takes the form of a complex network of programs relating to widely varying sectors of the economy, culture, territorial management and, of course, ecology and biodiversity management, particularly with regard to species and types of habitat.

This new course of conservation policy follows the main European Directives on nature conservation, trying to develop a more holistic concept of the land and its natural and human components. It is also integrated with the renewed efforts of the Council of Europe to promote a more comprehensive, less fragmented approach to territorial administration, leading to the adoption of European Landscape Convention. In short, this tendency aims to do more than just emphasize conservation of individual species or protected areas: the focus has shifted onto a systematic policy involving all environmental components, in close connection with European strategies.

In this context, the Italian Ministry of the Environment and Territorial Protection (Directorate of Nature Conservation) has started a series of studies (this thesis is one of them) on the status and the effectiveness of the Italian protected areas.

Altogether, existing protected areas in Italy cover almost 11% of the country, and they can be considered one of the most important components of a potential ecological network dedicated to biodiversity conservation. Nevertheless, because of their size and the criteria by which they were chosen, Italy’s protected areas are probably not sufficient to satisfy the requirements of biodiversity conservation. However, no systematic study has been performed untill now, and a scientifically based evaluation of the Italian protected areas is still lacking.

The aim of this PhD thesis is to verify whether the system of protected areas fully represents the pattern of biodiversity for vertebrates and to determine what sort of action should be taken in order to make the system more efficient in conserving this important component of biodiversity. The study was carried out, in response to a request from the Ministry of the Environment and Territorial Protection, Directorate of Nature Conservation, in collaboration with the Department of Animal and Human Biology of the University of Rome "La Sapienza".

This thesis is organized into 6 independent chapters plus this Preface and a Conclusion. The first three chapters are introductive and set the basis for the development of the thesis. Chapter 1 (Introduction) provides an overview of the field of reservation ecology. Chapter 2 (Descriptive analysis of existing and proposed protected areas in Italy) provides an introductive descriptive analysis of the existing protected areas and Chapter 3 (Geographic distribution of terrestrial vertebrates in Italy) describes the data collected on the ecology and distribution of vertebrate species in Italy and the methodology used to build distribution models for the Italian vertebrates. Chapter 4 (Gap analysis of terrestrial vertebrates in Italy: priorities for conservation planning in a human dominated landscape) is a GAP and irreplaceability analysis of the existing protected areas, and has already been published on Biological Conservation. Chapter 5 (Contribution of the Natura2000 network to biodiversity conservation in Italy) provides a measure of how effectively the Natura2000 network is in providing protection to viable populations of the species for which each area has been established, and it has already been accepted for publication by Conservation Biology with minor revisions. Chapter 6 (Size-dependent resistance of protected areas to land-use change) provides an analysis of the differential efficacy of Italian protected areas in conserving the landscape and the habitats over which they are placed.

CHAPTER 1: INTRODUCTION

The Earth is a different place than it was a century ago. Almost every ecosystem on the earth has seen the influence of human activity and presence, and it is not possible to identify a single ecosystem or a single area in the world that can be considered pristine, except possibly some parts of tropical or temperate forest, Antartica and deep sea ecosystems (William and Turner 1992; Ojima et al. 1994; Vitousek et al. 1997; Sanderson et al. 2002).

About half of the worlds ice-free land surface has been measurably modified by human activities over the last 10,000 years (Lambin et al. 2003), and landscape ecology processes, biogeochemical cycles, and ecosystem functions have been deeply impacted (Houghton 1994; Vitousek et al. 1997; Reid et al. 2000; Lambin et al. 2003).

The causes of all these problems can be related in a more or less direct way to the growth of human population, which now exceeds 6 billion (6,503,539,036 at the date 03/15/06 according to the U.S. Census Bureau), and to the increasing needs for space and resources (Pimm et al. 1995; Pimm and Lawton 1998; Stuart-Chapin III et al. 2000; Pimm et al. 2001; Loh and Wakernagel 2004). As a consequence biodiversity has entered a phase of crisis comparable to those occurring during past geological eras, with species going extinct at a rate estimated to be 100 to 1000 times greater than rates recorded through recent geological time (Jablonski 1985; Smith et al. 1993; Pimm et al. 1995).

The World Conservation Union (IUCN) classified more than 5000 vertebrate species (9% of the species described), almost 2000 invertebrate species (0.17% of the species described), more than 8000 plant species (2.89% of the species described) as being endangered or threatened in the wild; more than 700 animal species and more than 110 plant species are considered extinct (Baillie et al. 2004). Yet these figures represent optimistic and biased underestimates (Smith et al. 1993), because only a small part (from 3% to 15%) of the world's biodiversity has been described (Wilson 1985; Agapow et al. 2004; Mace 2004) and an even smaller part has been assessed for its conservation status (IUCN 2004).

The response of the scientific community to this biodiversity crisis is the development of a new discipline called conservation biology (Soulé 1985). This is supposed to be a crisis discipline that synthesizes and use techniques and results obtained and developed by many other scientific fields (e.g., ecology, population biology, wildlife management, but also sociology and economics); completely particular to conservation biology is the need for integration of science with socio-economics, that often has failed but that is always extremely important (Meffe and Carroll 1997).

Different authors have subdivided conservation biology into sub-fields, commonly reservation ecology and restoration ecology; to these two Rosenzweig (2003) added reconciliation ecology. Of the three sub-fields of conservation biology reservation ecology (the science and the management of protected areas) is probably the most important because of its diffusion and because of its appeal for the lay public (Balmford et al. 1996; Groves 2003).

Protected areas

The importance of protected areas has been widely demonstrated (Bruner et al. 2001; Sinclair et al. 2002; Sànchez-Azofeifa et al. 2003; Chape et al. 2005; but see Hulme 2003; Parmesan and Yohe 2003; Parrish et al. 2003; DeFries et al. 2005; Neptstad et al. 2006). Protected areas are among the most efficient tools to avoid habitat degradation, to stop deforestation and other forms of overexploitation, and to mitigate the effects of logging, hunting, and human activities in general (Bruner et al. 2001).

Extinction of some species (e.g. the white rhino in South Africa, the Alpine ibex in the Italian Alps, the Abruzzi chamois in the Italian Apennines) has been prevented by the establishment of protected areas that, in regions of particularly intense human settlements, are usually the only remaining patches of native vegetation, with species diversity and abundance markedly higher than in surrounding areas (Sinclair et al. 2002; Possingham et al. 2006).

Moreover, besides protecting biodiversity, protected areas often provide other benefits, such as protecting water supplies, providing food protection, protecting cultural values, while providing economic benefits to local communities (Balmford et al. 2002).

Protected areas represent the core conservation strategy for a number of regional, national and international agreements and laws (the Convention on Biological Diversity, the Convention on the Conservation of Migratory Species of Wild Animals, the US Endangered Species Act, the Convention on International Trade in Endangered Species of Wild Flora and Fauna, the Bird European Directive, the Habitat European Directive, the Convention on the Protection of the World Cultural and Natural Heritage, the Convention on Wetlands of International Importance) but the concept of protected area is not a new one. In fact, it can be dated back at least to the Roman Empire and to the sacred groves of Africa and Asia, when some areas in the landscape were "protected" from development and overexploitation in order to generate a potential reserve for game animals, wood (necessary for ships and buildings), and water (Bishop et al. 2004; Phillips 2004). The same situation was common in many European kingdoms, from the Middle Age to the 1800s.

The origin of the modern concept of protected areas can be traced back to the nineteenth century when William Wordsworth described the Lake District as "a sort of national property" and George Catlin expressed the need for "a nation's park, containing man and beast, in all the wild and freshness of their nature's beauty" (Phillips 2004). Yosemite and Yellowstone were designated, respectively in 1864 and in 1872, as the first two national parks in the world (McNeely 1994), followed by Royal National Park (Sydney, Australia) established in 1879, Kruger National Park (South Africa, 1892) and by many others (Phillips 2004; Possingham et al. 2006), especially in the second half of the 20th century (Fig. 1). The first Italian national park (Gran Paradiso National Park) was established in the 1920s, followed soon by other protected areas. Today, according to the official list of the United Nations, approximately 12.65% of the earth's land surface is protected (Chape et al. 2003).

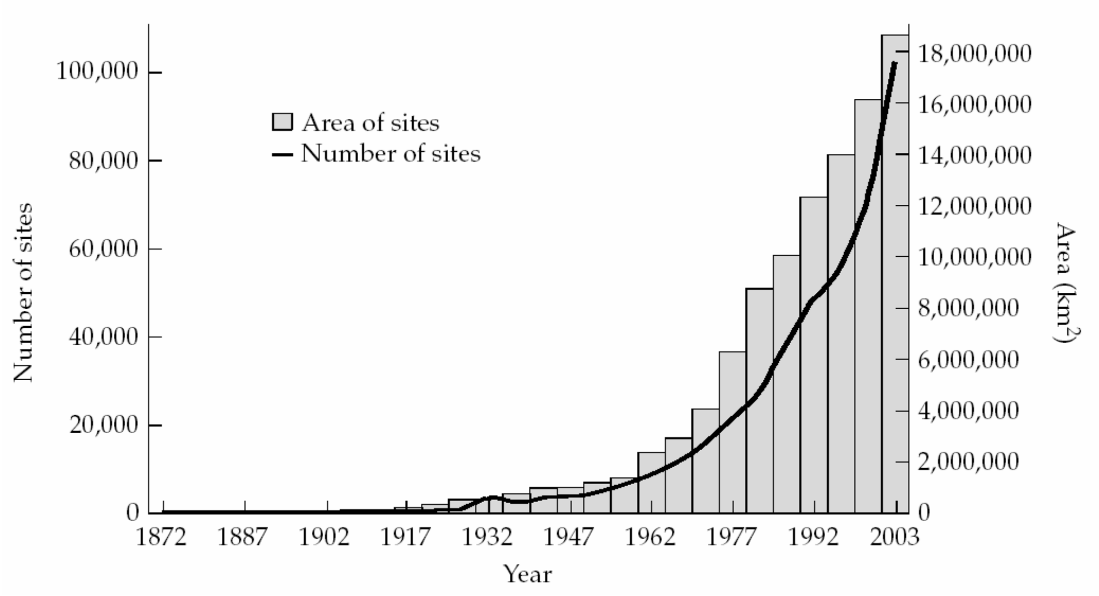

Figure 1. Growth of protected areas over time. Source Possingham et al. 2006.

IUCN classification of protected areas

Protected areas are defined by IUCN as "areas of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means". The same IUCN provides a classification system that groups protected areas according to the different levels of protection that they provide (Bishop et al. 2004; Phillips 2004; Possingham et al. 2006).

Category Ia: Strict nature reserves. Protected areas managed mainly for scientific research. It can be an area of land and/or sea possessing some outstanding or representative ecosystems, geological or physiological features and/or species and it is available primarily for scientific research and/or environmental monitoring.

Category Ib: Wilderness areas. Protected areas managed mainly for wilderness protection. It can be an area of unmodified or slightly modified land and/or sea, retaining its natural character and influence, without permanent or significant habitation, which is protected and managed so as to preserve its natural condition.

The areas classified in Category I cannot allow for recreational uses and their establishment is difficult because they exclude mechanized forms of transportation and extractive use, as well as limiting access.

Category II: National parks. Protected areas managed mainly for ecosystem protection and recreation. It can be a natural area of land and/or sea designated to protect the ecological integrity of one or more ecosystems for present and future generations, exclude exploitation or occupation inimical to the purposes of designation of the area, and provide a foundation for spiritual, scientific, educational, recreational, and visitors opportunities, all of which must be environmentally and culturally compatible.

Category III: Natural monuments. Protected areas managed mainly for conservation of specific natural features. It can be an area containing one or more specific natural or natural and cultural feature that is of outstanding or unique value because of its inherent rarity, representative or aesthetic qualities, or cultural significance. Generally speaking, these areas are more limited in size and scope than area classified in category I or II.

Category IV: Habitat/species management areas. Protected areas managed mainly for conservation through management intervention. It can be an area of land and/or sea subject to active intervention for management purposes to ensure the maintenance of habitat and/or to meet the requirements of specific species. Scientific research and environmental monitoring are often the primary activities undertaken in these areas.

Category V: Protected landscapes/seascapes. Protected areas managed mainly for landscape/seascape conservation and recreation. It can be an area of land (with coast and sea as appropriate) where the interaction of people and nature over time has produced an area of distinct character with significant aesthetic, ecological and/or cultural value, and often with high biological diversity. Safeguarding the integrity of these traditional interactions is vital to the protection, maintenance and evolution of such areas.

Category VI: Managed resource protected areas. Protected areas managed mainly for the sustainable use of ecosystems. It can be an area containing predominantly unmodified natural systems, managed to ensure long term protection and maintenance of biological diversity, at the same time providing a sustainable flow of natural products and services to meet community needs.

The main distinction in the IUCN classification system is among strictly protected areas (categories I to IV) and multiple use areas (categories V and VI). Strictly protected areas are likely to be the most efficient at meeting biodiversity conservation and they have formed the cornerstones of conservation efforts worldwide (Possingham et al. 2006). However, there is growing evidence that conservation cannot be pursued using only strictly protected areas, and in fact 23.3% of the total extent of the world protected areas are classified as Category VI, a percentage similar to the 23.5% devoted to Category II and much greater than the 11% devoted to Category I (Chape et al. 2003).

Managing, designing and planning for protected areas

Naturally, to be effective in conserving biodiversity, protected areas must be managed and designed appropriately. Usually species abundance and diversity is higher within protected areas then in the surrounding landscape (Bruner et al. 2001; Possingham et al. 2006) but many protected areas, especially in developing countries, are inadequately managed. They have limited funding and face many significant threats and challenges, with some protected areas that are not even secure from vegetation clearing (Peres and Terborgh 1995; James 1999; Menon et al. 2001).

The main problem, however, remains planning and design. Single protected areas cannot be considered of adequate size or scope to protect the biodiversity of a region. Certainly, single protected areas may be particularly important for a given species (Possingham et al. 2006) but in most cases it is necessary to develop networks of protected areas. The problem is that historically there has been no comprehensive and coordinated planning effort, and indeed even nowadays a coordinated planning effort is a really rare thing to find (but see Noss et al. 1999; Cowling et al. 2003a; Gelderblom et al. 2003; Groves 2003 for some examples). Historically, protected areas were seen mainly as a tool to preserve single species in a static way, practically isolating the species from what was seen as the threatening process (Sinclair and Byrom 2006). In this context, just setting aside an area represented a reasonable conservation strategy.

However, even the conservation of individual species cannot be successful without understanding ecosystem complexity, that arise from factors like nonlinear biotic interactions, evolutionary history, species assemblages, etc. Modern ecology and conservation biology see nature as a complex and dynamic entity (McCann 2000; Sinclair and Byrom 2006) and recognize that protected areas isolated in a hostile matrix and considered one at time cannot constitute a viable conservation strategy for many reasons (Grumbine 1990; Tilman et al. 1994; Carroll et al. 2004), going from size issues (Diamond 1975; Schwartz 1999; Rodrigues and Gaston 2001; Zhou and Wang 2006), to connectivity (Diamond 1975; Hoctor et al. 2000; Poiani et al. 2000), to environmental and genetic stochasticity (Carroll et al. 2004), to climate change (Peters and Darling 1985; Araujo et al. 2004; Buckley and Roughgarden 2004; Harte et al. 2004; Thomas et al. 2004a; Thomas et al. 2004b; Thuiller et al. 2004; Coulston and Riitters 2005) and so on. Moreover, protected areas are often in conflict with economic activities in a human dominated landscape (Pressey et al. 1993; James et al. 1999, 2001; Marguler and Pressey 2000; Frazee et al. 2003), implying the necessity for strategic planning to account for all the necessary users of natural resources.

Unfortunately this is not what is usually done (Prendergast et al. 1999), and a number of studies have outlined that protected areas are usually not representative of the biodiversity of a region (Pressey et al. 1993; Pressey 1994; Rodrigues et al. 1999; Scott et al. 2001; Andelman and Willig 2003; Rodrigues et al. 2004a). Even today most of the reserves are created as a response to particular situations (Pressey et al 1993; Margules and Pressey 2000), such as a particularly endangered and charismatic species that are present in an area (Simberloff 1998; Sinclair and Byrom 2006; Possingham et al. 2006), or a scenic area that does not have any particular economic conflict or activity (Scott et al. 2001). The result of this process is the selection of a conservation network that is neither biologically effective nor economically efficient (Margules and Pressey 2000). In fact, the worldwide PA network is still far from complete: at least 13% of the species considered (terrestrial vertebrates) are not represented in any PA, and 74% do not achieve their representation targets (Rodrigues et al. 2004a, 2004b). These estimates, obtained using a limited number of taxa, can potentially depict an overly optimistic view of the problem, since other studies estimate that approximately 43% of the world's terrestrial species are not represented in existing protected areas (Ferrier et al. 2004).

Moreover, the species of greatest conservation concern are often those most poorly represented in the existing protected areas (Rodrigues et al. 2004a, 2004b; Brooks et al. 2004a). The global GAP analysis (Rodrigues et al. 2004a, b2004) revealed that 1424 species are not protected in any part of their range, and 804 of these species are threatened, corresponding to 20% of all the threatened species analyzed. These numbers almost double when considering the species that are represented only by very marginal overlaps with existing protected areas. The global GAP analysis also highlighted the skewed distribution of existing protected areas, both geographically and in terms of size. Globally, 46% of the protected areas are found in the tropic where 76% of the species considered are present, and the same pattern has been found in regional studies. Andelman and Willig (2003) found that the median size of strictly protected areas in the New World is only 4.86 km^2 and 57% of the areas are smaller than 10 km^2. Moreover, 35% of the total area of strictly protected areas in the new world can be found in Alaska, where the number of species is lower. Even worse is the situation if we consider marine protected areas: only 0.5% of the surface area of the oceans is protected, corresponding to 9.1% of the area of all protected areas.

Systematic conservation planning

Creating a global reserve system representative of the world's biodiversity would require important economic investments: James et al. (2001) and Balmford et al. (2002) have estimated that we need $3-11 billion per year for the next 30 years. At the same time we are continuing to convert pristine ecosystems to human dominated ones, creating an urgent need for the designation of new protected areas. However we need a way to prioritize the scarce resources available in order to maximize the returns for conservation (Possingham et al. 2006).

The first necessary step is defining the "ideal" properties that a network of protected areas should have. Then we can define a way to come close to the ideal system. Craig Groves proposes the so-called 4-R Framework (Groves 2003), according to which a reserve network should be: representative, restorative, resilient and redundant.

Representative indicates that a reserve network should cover all the biodiversity and environmental features that are naturally present in the region of interest (Margules et al. 1988; Noss 1990; van Jaarsveld et al. 1998a). This criterion is based on the assumption that only in a protected area biodiversity can be considered safe (Soulé 1991; Bruner et al. 2001; Sinclair et al. 2002; Sànchez-Azofeifa et al. 2003; Chape et al. 2005; but see Tilman et al. 1994; Margules and Pressey 2000; Carroll et al. 2004; Sinclair and Byrom 2006). It has been considered a key criterion for protected areas for a long time (Burley 1988; Noss and Cooperrinder 1994).

Redundant indicates that the reserve network should include more than one representation of a given element of the biodiversity or of a given environmental feature. This criterion recognizes that protection of just one occurrence of a given feature is not sufficient to ensure its preservation in the long term (Margules and Pressey 2000; Boyce et al. 2002; Groves 2003). There is not a unique minimum number of occurrences that should be considered, but it depends on what we are trying to protect, on its spatial configuration, on the matrix into which the protected area is, and so on (Fahrig 2001, 2002).

Resilient refers to the fact that features protected in reserve should be resilient to changes (natural or human caused): if we speak of a population we should think of its viability (Carroll et al. 2004), if we consider an ecosystem we should think of the processes that determine its structure (Franklin 1993; Coulston and Riitters 2005).

Restorative refers to the possibility of re-creating a given environment (e.g. Everglades) that has been destroyed or heavily altered by human activities or of restoring population viability that has been compromised by human influence (Sinclair and Byrom 2006).

The 4-R framework is not the only available scheme (see Margules and Pressey 2000; Possingham et al. 2006), but it provides an overall picture of what should be the aims during the design and the selection of a reserve network. Different views of how to best designate a reserve system with its “ideal” properties have been proposed. Here, I will describe in detail the so-called systematic conservation planning approach.

Margules and Pressey (2000) propose conservation planning as a process in six stages: 1) measure and map biodiversity of the planning region; 2) identify conservation goals for the planning region; 3) review existing protected areas; 4) select additional protected areas; 5) implement conservation actions on the ground; 6) monitor and manage protected areas. The entire process is not unidirectional from step 1 to step 6, but feedbacks are necessary for every step, and decisions can be changed at every moment.

Measuring and mapping biodiversity

The term biodiversity was coined in 1986. Its definition includes the entire biological (from molecules to ecosystems) and taxonomical (from alleles to kingdoms) hierarchies, as well as the diversity of interactions and processes at all levels of organizations (Noss 1990; Humphries et al. 1995; Gaston 2000).

Because of the complexity of biodiversity (Noss 1990; Noss and Cooperrinder 1994; Gaston 2000), measuring and mapping the entire biodiversity of a region is almost an impossible task (Noss 1990; May 1994; Gaston 2000; Sarkar and Margules 2002; Williams et al. 2006), especially considering the temporal and budget constraints of a planning process (Sarkar et al. 2005). Usually, a simplifying proposal is that of using only three levels of the biodiversity continuum, the three that should capture all that is important about biodiversity: genes (if we conserve genetic diversity we take care of the below individual/genotype level), species (if we conserve species we conserve all the higher taxonomic levels), and ecosystems (if we conserve ecosystems we supposedly protect communities and processes) (Sarkar and Margules 2002).

This proposal has been widely accepted but there remain many important problems: in this scheme we are not concerned about the so-called “endangered biological phenomena” (Meffe and Carroll 1997; Sarkar and Margules 2002), and many of the levels of biodiversity may leak out of the scheme (we may lose sub-specific entities for example).

Surrogacy

Realistically, no single measure or level of biodiversity will be capable of capturing all the biological features, and we will remain with two main questions to answer: what is to be measured and are the data that we need to measure obtainable? Sarkar and Margules (2002) name the first issue as the "problem of quantification" and the second as the "problem of estimation". They suggest that the two problems can be solved thought the use of surrogates (Landres et al. 1988; Pearson 1994; Humphries et al. 1995; van Jaarsveld et al. 1998a; Margules et al. 2002; Williams et al. 2002).

Surrogacy is the relationship existing between a parameter that functions as an indicator and a parameter that constitute the "objective" or "target" that, in this case, we hope to conserve (Sarkar and Margules 2002; Sarkar et al. 2005). In particular, it is possible to distinguish among "true surrogates" and "estimator surrogates" (Sarkar and Margules 2002; Sarkar et al. 2005). True surrogates should represent biodiversity in general (i.e. true surrogates have general biodiversity as their target), with the only constraint that they must be quantifiable to be used in ordering processes. Estimator surrogates, in contrast, have true surrogates as targets. For true surrogates, given the indeterminacy of biodiversity, we will always have the problem of empirically proving that what we have chosen as a true surrogate is actually a true surrogate (i.e. that we are representing the full spectrum of biodiversity). On the contrary, for estimator surrogates (once we have chosen our true surrogates and given that this is measurable) we can potentially measure a quantitative relationship with their target, the true surrogates (Landres et al. 1988). In particular, any attempt to assess the adequacy of estimator surrogates should focus on three main points: an estimator surrogate is adequate to represent its target if 1) using the estimator surrogate to prioritize places for conservation we achieve a targeted representation of the true surrogate (adequate representation); 2) the diversity of the estimator surrogate in a region is spatially correlated with the diversity of true surrogates (spatial correlation); 3) the set of areas selected using the estimator surrogate is spatially congruent with the set of areas that would have been selected using the true surrogate set (spatial congruence) (Sarkar et al. 2005). Of the three, spatial congruence is by far the most important, because an estimator surrogate must ensure by definition the full representation of the true surrogate.

Commonly, three candidates have been used as "true surrogates" (Sarkar and Margules 2002): character (or trait) diversity; species diversity; species assemblages (or landscape patterns or life zone diversity).

Character or trait diversity has been suggested as a true surrogate mainly because the evolutionary mechanisms usually influence directly given traits of organisms (Vane-Wright et al. 1991; Humphries et al. 1993). However what we choose as a trait or character depends on what can be studied practically, meaning that trait diversity cannot be used to solve the "quantification problem" practically or theoretically (Sarkar and Margules 2002).

Species diversity is probably the most commonly used true surrogate. In fact measures of species richness and diversity are almost always used in any discussion on biodiversity and species is probably the most well defined category in the biodiversity hierarchy (but see Agapow et al. 2004). However, species diversity has some problems as a true surrogate because we know by the same definition of biodiversity that many levels and many phenomena can potentially be lost focusing on species alone (Sarkar and Margules 2002).

Species assemblages or landscape patterns or life zone diversity have been used in different parts of the world to represent similar things. Their use as true surrogates comes from the consideration that one of the most important aspects of biodiversity is the variety of biotic communities with all their interactions; moreover, it is often assumed that if we focus on communities we necessarily take care of species. Obviously, quantification is also a serious problem in this case: almost any classification of communities involves some arbitrary convention, generating the same type of problems as for trait diversity.

Even considering other possible candidates as true surrogates, there is no unique solution, and we will always need to use some convention or some arbitrary/pragmatic solution. A possibility to consider is that of using a combination of surrogates simultaneously, and in this respect the second and the third are the most promising (Sarkar and Margules 2002).

Coming to estimator surrogates, it is possible to list at least 6 of them: species richness, environmental parameter composition, vegetation class, species diversity, genus (or other taxa higher than species) diversity, and subsets of species composition (Sarkar and Margules 2002).

Species richness has been the most popular (with species diversity being the true surrogate), and Gaston (1996a, 2000) outlined five possible factors playing a role in the choice: 1) species richness can be correlated to many measures of ecological diversity better than many indices like the Shannon-Weaver; 2) species richness in some instances is positively correlated with higher taxa richness; 3) if species richness is high it is correlated with trait diversity; 4) species richness is correlated (sometimes) with measures of the complexity of community webs (probably the most controversial point); 5) high species richness is usually correlated with increasing topographic diversity.

Environmental parameter composition is based on the idea that each point in the “ecological” space constitutes a niche that can be potentially occupied by one (or more) species (Faith and Walker 1996). However, not all the possible niches will be occupied by a species and the correlation between biodiversity and environmental parameters may not be very good (and actually since we cannot measure biodiversity we cannot estimate how good the correlation is). Finally, if environmental parameter composition is used to select areas to protect there is the danger, if the scale is large enough, of loosing ecologically convergent but taxonomically distinct species and/or other levels of the biodiversity continuum.

Vegetation class or types represent a combination of species with their reciprocal interactions, and as such they are assumed to incorporate both ecological processes and a list of species. Moreover, vegetation classes (or better the species that compose a vegetation class) are linked spatially and ecologically to invertebrates, fungi, bacteria and protozoa offering an umbrella effect to species that usually are not considered at all in conservation planning and in conservation in general. However, also in this case the question of the relationship between the empirical estimator and biodiversity remains open.

Species diversity is different from species richness but it carries with it the same problems. Higher taxon diversity may potentially represent also lower level diversity (and thus species diversity), but the correlation between the spatial distribution pattern of higher taxa and that of species should be established and described in detail.

Subsets of species composition, such as birds, plants, butterflies, a combination of these, etc., are the most widely accepted representations of biodiversity for a given region, mainly because it is usually readily available from museums and herbaria datasets. However, these datasets carry with them severe problems of spatial bias (Nelson et al. 1990), and empirical studies have suggested that flagship species and umbrella species do not perform much better as surrogates than species selected randomly from the same dataset (Andelman and Fagan 2000). This suggests that subsets of species in general can perform poorly as surrogates of species diversity.

All the 6 estimator surrogates above can be mapped relatively easy, making them particularly suited for conservation planning. On the contrary, current techniques do not allow for measuring and mapping ecological and evolutionary processes (but see Cowling et al. 1999; Pressey 2004) and conservation planning is, at least in part, a spatial exercise, implying that only biodiversity features that can be mapped are of practical value for it. This is one of the reasons why species richness (or most of the time, subsets of species) and environmental diversity are probably the two most widely used surrogates. Many times the two approaches have been proposed as possible alternatives or have been used in conjunction (Faith and Walker 1996; Davies et al. 1999; Araujo et al. 2001; Sarakinos et al. 2001; Scott et al. 2001; Araujo et al. 2003; Cowling et al. 2003a; Brooks et al. 2004b; Higgins et al. 2004; Pressey 2004; Sarkar et al. 2005). Both have advantages and problems, with trade offs between biological detail and data availability.

Environmental surrogates in conservation planning

Maps of temperature, geology, relief, habitats and vegetation classes are now widely available at fine resolution, particularly in the era of remote sensing (Turner et al. 2003). Data on species distribution, on the other hand, are still limited to the best-known taxa (vertebrates, vascular plants, and some groups of invertebrates [Higgins et al. 2004]), a small fraction of the species diversity worldwide (Gaston 2000; Agapow et al. 2004). Data availability alone would therefore support the use of surrogates based on environmental variables. However, broad scale biodiversity surrogates based on environmental features suffer from important drawbacks. First of all they are usually expressed as abstract and subjective classes partitioning the environmental space: some authors used habitat units derived from a mix of vegetation types, climate, geology, and topography (Lombard et al. 2003; Sarkar et al. 2005), or ecosystem types obtained from satellite imagery (Armenteras et al. 2003), or environmental diversity obtained from multidimensional environmental space (Faith and Walker 1996). All these approaches are obviously highly dependent on the primary variables used to produce them and the cutoffs applied to distinguish among different classes (Brooks et al. 2004c). It is important to remember that discontinuities that are really clear to the human eye are not necessarily important for other species, while we may fail to perceive as important changes that other species perceive as major factors. As an example Brooks et al. (2004b) consider two North American ecoregions (the North Central Rockies Forest and the Northern Short Grassland), that are clearly different from a human perspective, and a South American ecoregion (the Northwestern Andean Montane Forest), that from a human perspective constitutes a single entity. However, from the perspective of amphibians and birds (but not mammals), the northern and southern parts of the Northwestern Andean Montane Forest are more distinct than the two North American ecoregions.

A second problem is that use of environmental surrogates in conservation planning is usually linked to percentage targets, i.e. to a fixed percentage of each environmental attribute (usually 10%) that should be protected by a network of conservation areas. Such targets are usually arbitrary and fail to account for the fact that regions of higher species richness and endemism may require higher representation targets (Rodrigues et al. 2004a). Moreover, the target that we are aiming to potentially depends on the particular classification scheme that we are using. Brooks et al. (2004b) reports that the 10% conservation target has been surpassed for 9 out of 14 major terrestrial biomes if we use Udvardy biome classification system, but if we use Olson biome classification system the 10% target has been surpassed only in 8 biomes out of 16.

Environmental surrogates, even opting for variable targets (Lombard et al. 2003), have also the problem of where to implement the target within a single land class, and it is very well known that it matters where protected areas are located and not just how large is the area occupied (Pressey 1994).

Beyond the theoretical considerations above, some experimental studies do not support the use of environmental surrogates. Lombard et al. (2003) proposed land classes as surrogates for vertebrate species diversity in the Cape Floristic Region and they found really poor performance, suggesting that a better approach would have been integrating land classes data with species distribution. Araujo et al. (2001) explored the value of environmental diversity (ED) as a surrogate for species diversity. The measure of ED (Faith and Walker 1996) for an area is the decrease in the summed distance from all points in the ordination space to the nearest point already selected for conservation. The assumption is that in maximizing the ED value for a conservation network we are also maximizing the true surrogate diversity. Araujo et al. (2001) found that for the European continent, only plants exhibit consistent, non-random positive patterns of representation when areas are selected in order to maximize ED. On the contrary, mammals, breeding birds, amphibians and reptiles are represented less than expected (i.e. the number of species covered by a set of areas selected to maximize ED is smaller than the number of species covered by a random set of areas covering the same surface), indicating that ED constitutes a poor surrogate, at least for terrestrial vertebrates (see Faith [2003] for a different interpretation of the same dataset and Araujo et al. [2003] for a reply). Moreover, the species underrepresented using the ED approach are mainly those with a small Extent of Occurrence (EOO, sensu Gaston 1991), that tend to be more susceptible to extinction (Johnson 1998). This implies that, given the usually right skewed distribution of the sizes of the geographic range within taxonomic assemblages (Gaston 1996b), it is likely that environmental parameter composition, or at least ED, might perform inadequately in many areas and for many taxa. This is going to be particularly probable in Mediterranean-climate areas, where a number of species are endemic and narrowly distributed because of historical and human-related reasons.

Other studies have shown results more encouraging for the use of environmental surrogates (Wessels et al. 1999; MacNally et al. 2002; Oliver et al. 2004). In particular, Sarkar et al. (2005) using dataset from Québec and Queensland applied four methods at seven different spatial scales to assess the extent to which environmental surrogates can represent biodiversity components. They used two species datasets (one for Québec and one for Queensland) as true surrogates and a series of environmental parameters (a combination of climate, elevation, slope, aspect) as estimator surrogates, and found that use of environmental variables as estimator surrogates perform better than random at larger spatial scales (but not at smaller scales). However, problems remain and the same authors underline that their optimistic conclusions should be treated with caution, waiting for further scrutiny on the question (Sarkar et al. 2005).

Moreover, Trakhtenbrot and Kadmon (2005) provided a test for the efficacy of the environmental cluster analysis (ECA) as a surrogate for biodiversity. In particular, they built a map of the environmental diversity using ECA on three variables, precipitation, temperature and lithology, that should be important for biodiversity. They obtained that ECA perform better as a surrogate than a random set of sites and also than a selection of sites based on a floristic map. In particular, using ECA the authors were able to select areas with higher species richness and with higher rare species richness. Yet, the study was focused on a single region and on a single taxon. Further studies are needed to evaluate the efficacy of ECA in different regions (particularly with lower environmental diversity) as well as with more mobile organisms. Moreover, the efficacy of ECA in relation to variation in clustering algorithms and weights given to environmental variables should be investigated.

Species surrogates in conservation planning

The results and the considerations presented above indicate that species distribution data are essential in conservation planning (Brooks et al. 2004c), first of all because species are usually considered the fundamental units (and the "most natural" unit) of biodiversity (Wilson 1992). Obviously, species datasets have many limitations, the most important of which is certainly data availability (Ferrier et al. 2004). Distribution data for species worldwide is available only for a limited fraction of the species that are known (Gaston 2003; Higgins et al. 2004) but many initiatives are now underway to chart biodiversity, both at regional levels and worldwide (Stattersfield et al. 1998; Boitani et al. 1999; Rodrigues et al. 2003; Brooks et al. 2004c; Rodrigues et al. 2004a; IUCN et al. 2004), and new techniques to overcome these limitation have been proposed (Ferrier et al. 2004). However, the bias towards vertebrates is still very large, while we know very little about the distribution of plants and invertebrates, species that are critical to maintain structures, functions and services of ecosystems (Higgins et al. 2004; Pressey 2004).

Data on the distribution patterns (or more rarely on the abundance) of species can be compiled from collections of field records or they can be gathered from new surveys specifically designed. Acquiring new data, especially if the acquisition is done using properly designed surveys (Gillison and Brewer 1985; Wessels et al. 1998), is highly desirable but rarely feasible given time and cost constraints (Balmford and Gaston 2001). For this reason, existing data collections are usually used, even if they have not been specifically collected for conservation. Data on species distribution are available in the form of points of presence or as EOs, and both the approaches present problems and biases.

One important point is that the details of sampling methods are often unrecoverable and each collection set can have some particular spatial and/or temporal bias. Many field records are taken from places that the collector already knows the species is present, or they are sampled opportunistically. Examples are records of koalas in Australia (Margules and Austin 1994) and tree species in the Yucatan peninsula (Fig. 2 - Williams et al. 2002) that clearly map road networks, or tree records in the Amazon that map river networks (Williams et al. 1996), and many other examples are present in the literature documenting the associations of points of presence with cities, rivers, air stations, and street networks (Freitag et al. 1996; Lawes and Piper 1998; van Jaarsveld et al. 1998b; Maddock and du Plessis 1999; Reddy and Davalos 2003).

Figure 2. Field records of tree species clearly depict street network in the Yucatan peninsula. Modified from Williams et al. 2002.

Spatial consistency (i.e. a spatially homogeneous sampling) is a particularly important problem (Williams et al. 2002; Margules et al. 2002). In fact, conservation planning is essentially a problem of area comparison, and valid comparisons cannot be made unless the same relationships between sample and populations can be assumed to hold for all areas being compared.

A second important point is that the sampling effort changes with different data collections and this can represent another important source of bias. Ideally, sampling effort should be spatially uniform (an assumption rarely met in existing datasets [Williams et al. 2002]), so that variations in distribution and/or abundance depict real pattern and are not an artifact of variation in sampling effort. This is particularly important for species that are not easily recorded during surveys, or if the sampling effort is limited. In fact, the relationship between the number of species recorded and the sampling effort (the so-called species-discovery curve) generally increases steeply at first, but with larger samples the number of "new" species discovered become smaller.

The problem is even worse if we consider that species’ ranges are not static entities, but shift continuously following individuals and populations movements (especially true for animals) so that there may be no local fixed number of species. Grinnell (reported in Williams et al. [2002]) calculated that a sampling period of 410 years limited to California would allow for at least one record of presence for every North American bird species.

A third important limitation is that many existing datasets have been recorded over one limited time period. When these dataset are pulled together to build a conservation plan for a region it is more than probable that different time periods are implicitly considered in the same way. The importance of historical presence data for conservation planning in not questioned (Gaston et al. 2002), but the immediate need for priority area selection is to estimate the current distribution patterns. Historical data can confuse or even hide the pattern we are trying to measure.

EOs share many of the problems presented above with points of presence, because they are often based on points of presence. An EO identifies the outer most limits to the occurrence of a species (Gaston 2003) and, usually, it is built as a subjective abstraction, including sampling biases, personal interpretations and extrapolations from known localities to unsampled areas (Kodric-Brown and Brown 1993). Implicit in the same definition of EOO there is also a relatively high rate of commission errors, because within EOs many factors can influence the distribution of species and the outer most boundaries will naturally include areas where the species is not actually present (Mackey & Lindenmayer 2001).

The risks associated with use of incomplete and/or biased datasets in conservation planning has been investigated by modifying existing datasets to assess the effects of missing sites, missing taxa, and missing records (Freitag and van Jaarsveld 1998). The results showed that data deletions corresponded to increased variation in the network of areas selected and in a lower percentage of the initial target achieved, especially if missing data are concentrated in particular sites and or in particular taxa. The important conclusion is that species data should be distributed as broadly and as uniformly as possible both among sites and among taxa.

Some of the problems outlined above for EOs and points of presence can be solved using habitat suitability models to map the distribution of species (Nicholls 1989; Corsi et al. 2000; Guisan and Zimmerman 2000; Scott et al. 2002; Newbold and Eadie 2004). In particular, habitat suitability models can be used to move from the concept of EOO towards that of area of occupancy (AO) that can be defined as the area inside the EOO that is effectively occupied by the species, excluding cases of vagrancies (Gaston 2003). AO is the ultimate parameter that we want to measure and map if we use species distribution data in conservation planning, because having an unbiased map of the AO for each species would imply having no commission and/or omission error in the analysis (Rondinini et al. 2005; Wilson et al. 2005a). However, even habitat suitability models have their problems (Scott et al. 2002): accuracy can vary with localities; some species cannot be reliably modeled because of insufficient records, inadequate data on explanatory variables, or both; validation of the models with field data is seldom performed, especially at smaller scale, or it is performed only for a limited number of species.

Finally, if we focus on species data, whether we are using points data, EOs or habitat suitability models, we still have the problem of choosing (or having) a subset of species that is representative of the biodiversity of a region (Landres et al. 1988). In fact, no single species or taxon can be expected to adequately represent or indicate patterns for all other species or taxa, and even less for biodiversity in general (Pearson

1994). The studies on this aspect of biodiversity surrogacy have varied widely in geographic scale, study regions, and methods used to measure the association between the distribution of groups of taxa. Vane-Write (reported in Margules et al. [2002]) pointed out that, despite the theoretical co-evolution that should be ongoing among butterflies and plants, raw measures of plant diversity are poor predictors of butterfly diversity on a global scale; Williams and Gaston (1994) noted poor correlations among taxonomic groups within continents; Gaston et al. (1995) showed weak correlations among families worldwide; Majer (1983) showed that variation in plant diversity account for only a minor part of the variation in ant diversity in Western Australia; Yen (reported in Margules et al. 2002) found no correlation between the number of vertebrate species and the number of beetle species, and also no correlation among beetle communities or vertebrate communities and plant communities in Australia; Prendergast et al. (1993), Prendergast and Eversham (1997), and Williams and Gaston (1998) found only partial correspondence between richness hotspots of dragonflies, butterflies and breeding birds in the UK; van Jaarsveld et al. (1998a) found a weak correspondence in areas chosen to represent many taxonomic groups in South Africa; Andelman and Fagan (2000) found that umbrella and flagship species used as estimator surrogates do not perform significantly better than a random place selection procedure.

However, the issue should not be considered completely solved, since some studies obtained opposite results. Howard et al. (1998) showed that areas selected for one group of species in Uganda where often, but not always, good at representing species in other groups; Garson et al. (2002) showed that birds in Québec can be used as estimator surrogates; Schulze et al. (2004) found that patterns of species richness for trees, understory plants, birds, butterflies and dung beetles in Sulawesi where significantly correlated; Fleishman et al. (2005) found that it is possible to obtain a small group of indicator species that can be used as a proxy for species richness. Rondinini and Boitani (2006) found that amphibians and mammals in Africa effectively acted as an umbrella for high proportion of species in other taxa. Lamoreux et al. (2006), using global datasets, demonstrated that global patterns of richness among amphibians, reptiles, birds and mammals are highly correlated, and the same is true for endemics. They were also able to demonstrate that, even though the correlation among endemics and general species richness is low, selecting ecoregions for conservation using endemic richness capture significantly more species than expected by chance.

Further work is clearly necessary before the true value of using sub-sets of taxa in conservation planning can be considered fully tested.

Choosing the best biodiversity surrogate

The only conclusion that can be drawn from the consideration above is that there is no best surrogate (Margules and Pressey 2000; Possingham et al. 2006). The decision on which to use will depend on many factors including what data are available, what resources are available for the analysis of the data, and on the scale of the analysis. However, following Brooks et al. (2004b) and even considering all the limitations outlined above, I argue that species data must remain central to conservation planning. Restoration of habitats, ecosystems, processes is always really difficult, but sometimes it is possible (Dobson et al. 1997); on the contrary species extinction is irreversible. Using species data, even if we have distribution data for all the known taxa, in conservation planning does not guarantee that we are covering all of biodiversity (Higgins et al. 2004; Sinclair and Byrom 2006) but at least it would guarantee coverage for all known taxa. Moreover, even though species data have spatial biases and many false negative records

(Pressey 2004), these errors are far less serious in conservation planning than false positives introduced by environmental data (Loiselle et al. 2003; Rondinini et al. 2005).

Identifying conservation targets

The overall goals for a network of conservation areas can be summarized with two words: representation and persistence (Margules and Pressey 2000; Groves 2003). The primary purpose in setting a conservation goal is to estimate the effort that will be necessary to sustain “biodiversity” into the future. Naturally, in order to be of practical use for conservation planning these goals have to be translated into specific, quantitative targets (Margules and Pressey 2000; Pressey et al. 2003; Desmet and Cowling 2004). But how much is enough? How many populations are necessary to ensure viability? What proportion of a habitat (or of a species) needs to be protected? It is important to recognize that no single, objective answer exists and that there will always be a high degree of uncertainty (Soulé and Sanjayan 1998; Svancara et al. 2005).

Conservation targets can be divided into two broad categories based on the scale of biodiversity surrogates that are being targeted (Groves 2003; Pressey et al. 2003): coarse filter approaches set targets for features such as vegetation types, ecosystems and land classes; fine filter approaches use species or populations as targets for conservation. It is important to say that these are all targets for representing biodiversity patterns, while targets for ecological processes are much more problematic (Margules and Pressey 2000): conservation planning is a spatial exercise and even processes must be based on their spatial surrogates rather than on the processes themselves.

Areal targets

Percentages of land that should be set aside for conservation have been usually proposed as targets for conservation area networks. In 1982 the World Park Congress held in Bali recommended that countries set aside 10% of their land surface for conservation; the target was clearly arbitrary but with some foundations in the species-area relationships of island biogeography (Desmet and Cowling 2004). Subsequently, in 1987, the Brundtland Commission recommended that each nation set aside 12% of their land (Groves 2003). The 10% and 12% figures have been widely used as targets to assess existing conservation areas and to establish new areas, considering both land types (or ecoregions) and populations (Wright et al. 1994; Maddock and Benn 2000; Sierra et al. 2002; Rondinini et al. 2005).

While the two figures are still widely used, especially in political contexts where decisions are primarily based on socio-economic bases, they are no longer considered biologically adequate (Pressey et al. 2003; Solomon et al. 2003; Svancara et al. 2005). These conservation targets have been highly criticized because they can be easily obtained focusing on the less productive landscapes (Scott et al. 2001), they can become "de facto ceilings of protection" leading the lay public to believe that limited conservation action is adequate (Soulé and Sanjayan 1998), they were primarily politically based with few or no scientific inputs (Soulé and Sanjayan 1998; Groves 2003; but see Desmet and Cowling 2004); they are too small to effectively protect biodiversity. In fact, percentages required to protect the biodiversity of a region vary widely even within a single region (Soulé and Sanjayan 1998; Rodrigues and Gaston 2001), but many are substantially larger than 10/12%. Noss et al. (1999) indicated 65% as target for the Klamath-Siskiyou ecoregion; Noss et al. (2002) indicated 70% as target for the Greater Yellowstone Ecosystem; Cowling et al. (2003) indicated 52% for the

Cape Floristic region. Pressey et al. (2003), going further and using the concept of extinction debt (Tilman et al. 1994; Brooks et al. 1999), proposed that, in regions threatened by agriculture, urbanization, and habitat fragmentation in general, any target smaller than the remaining area of natural vegetations (that is any target smaller than 100%) is effectively a target for further loss of biodiversity.

All the criticisms reported above are mainly focused on how conservation targets have been defined and not on the idea of the target itself. Conservation targets, if wisely devised, are important because they have potentially many advantages (Gaston et al. 2002; Groves 2003; Pressey et al. 2003): they allow for a clear evaluation of the effectiveness of proposed conservation areas, they help managers and planners in decisions about tradeoffs with economic activities, they influence the number and extent of conservation areas in a region, they provide a vision for future conservation efforts (Margules and Pressey 2000; Groves 2003). Accordingly, conservation targets should be an integral part of the political processes governing a region and as such they should pass through periodic reviews.

New ways of defining conservation targets

Rodrigues et et. (2003), basing their reasoning on the "double jeopardy" concept (Lawton 1993), have suggested a more detailed approach in setting conservation targets, establishing a different representation target for each species: in particular they suggested establishing higher conservation targets for species with smaller ranges. In fact, setting a constant representation target (e.g. 10%) would favor wide ranging species in comparison to narrow endemic species. The assumption is that narrow endemic species (or better species with smaller EOs) tend to be rare not only in terms of range size but also in term of local abundance (Brown 1984; Gaston et al. 1997).

Therefore, protecting 10% of the range of a narrow endemic species would imply protecting a disproportionably smaller number of individuals than the number that would be protected by 10% of the range of a widespread species. Moreover, there is a well-known negative relationship between species’ range size and extinction risks (Purvis et al. 2000) linked to a higher vulnerability to stochastic events and anthropogenic activities. Therefore, Rodrigues et et. (2003) proposed a representation target of 10% for widespread species (those with ranges larger than 250,000 km^2 on a global scale) and of 100% for narrow endemic species (those with ranges smaller than 1,000 km^2 on a global scale). For species with ranges in between the representation target was interpolated between the two extremes using a logarithmic transformation. A similar strategy was proposed also by Rodrigues et al. (2000) and by Lombard et al. (1997).

It is clear that we cannot devise a single and universally accepted target (or better a single and universally accepted way to define a target), but we can use ecological and evolutionary biological principles to provide guidelines (Margules and Pressey 2000; Groves 2003).

Traditionally, biogeographical theory, and in particular the equilibrium theory of island biogeography, has been used to help in setting conservation targets (usually qualitative targets), especially for size, shape and spatial configuration of reserves. Biogeographical theory has been used to suggest that bigger reserves are better than smaller ones, that less fragmented are better than more, and that corridors can help conservation. However, in real world applications the opportunity to apply such guidelines must consider many different factors. If the area devoted to conservation is limited, we have to make a choice among establishing a few large reserves (that favor the persistence of particular species) and many small ones (that favor the full representation of the biodiversity of a region, but are not effective for insuring persistence). Moreover, the equilibrium theory of island biogeography considers islands (or better reserves in our case) as if they have no internal structure, and each island is considered to be equivalent to all the others. Naturally this is not the case, and there is evidence that increasing isolation, at least for certain species, decreases the probability of persistence (Davies et al. 2000) as predicted by island biogeography theory.

Also, metapopulation dynamics has been considered an important factor in driving reserve design. In particular, metapopulation theory calls for targets that consider reservation of species throughout their entire range, decreasing the extinction probability linked to stochastic events. It also calls for the retention of habitat connectivity to promote dispersal, and for the protection of suitable patches of unoccupied habitat to promote recolonization.

Sometimes seen as a particular type of metapopulation dynamic, source-sink dynamics represent another important point. The theory obviously calls for reservation of source habitat patches as well as for corridors and connectivity (and warns against the reservation of sink habitat).

Ecological theory can contribute to target definition with the concept of ecological succession. Most regions contain areas at different stages of a given succession and all the stages of the succession might be needed for conservation. Also in this case, large reserves are better than smaller ones, because they can accommodate more than one stage and they allow for natural disturbances.

Particularly important are the spatial autoecological requirements of species to be conserved. Most reserves contain one or more species that would not persist if the reserve becomes isolated, unless it is complemented by unreserved habitat that can support specific life stages of species. Thus the persistence in the long term of species depends on factors that are spread out in the entire landscape. A proposal in this respect is that of focal species (Lambeck 1997; Lindenmayer et al. 2002), that is an attempt to integrate patterns and processes by identifying those species in a landscape that are more demanding of resources and then targeting them for management. The assumption is that the conservation of focal species would ensure the conservation of all the other species in the region.

Targets to off reserve conservation are particularly important in fragmented landscapes, where reserves are small and often isolated in a heavily-modified matrix. In fact, if reserves remain isolated the persistence of species can be compromised (Tilman 1994; Loehle and Li 1996; Hanski and Ovaskainen 2002; Carroll et al. 2004) and thus habitat modification should be carefully considered when establishing conservation targets.

Finally, conservation targets should consider carefully the importance of evolutionary theory. Species in conservation should not be considered as static units (Rojas 1992) and this implies that areas where taxa are undergoing active phylogenetic radiations should be targeted for conservation (Vane-Wright et al. 1991).

The different aspects of ecology and evolutionary biology described above should not be considered as separate aspects, but they should be integrated in conservation planning if representation and persistence are to be achieved (Margules and Pressey 2000). An example, indeed not really common, of the entire process of establishing a representation target for conservation planning can be found in Pressey et al. (2003).

Groves (2003) adds one final statement to the considerations above: observe the precautionary principle. The precautionary principle is linked to the fact that usually we cannot prove with certainty that a given development or action will cause a given negative result (e.g. climate change). Under the principle, the burden of proof is shifted to those that propose the action to prove that their action is not harmful.

In our context the precautionary principle calls for relatively high conservation targets, to anticipate activities and developments that will cause further loss of biodiversity.

Review existing protected areas

Determining the degree to which the conservation target has already been achieved in the existing reserves is one of the most important steps in the conservation planning process (Margules and Pressey 2000; Groves 2003). Systematic reviews of the existing reserves have a long history and are the conceptual basis for the GAP analysis program in the US (Scott et al. 1993; Jennings 2000). Burley (1988) described systematically the process to identify conservation gaps. He started with identification of the various elements of biodiversity and went on with examination of the existing system of protected areas: this process would have allowed for the identification of those elements that are not represented (or that are poorly represented) in the existing reserve system, and for the clear setting of future conservation priorities.

The first application of a gap analysis can be traced back to the conservation of Hawaiian bird, with a project developed in the 1980s by J. Michael Scott (http://www.gap.uidaho.edu; Scott et al. 1987). Later, a research group lead by the same J.M. Scott initiated the Idaho GAP as the first pilot study under the USFWS (Scott et al. 1993). From that point on, gap analyses has been conducted throughout the entire USA (Caicco et al. 1995; Strittholt and Boerner 1995; Kiester et al. 1996; Jennings 2000; Scott et al. 2001; Dietz and Czech 2005) as well as in other part of the world (Bojorquez-Tapia et al. 1995; Fearnside and Ferraz 1995; Araujo 1999; Rodrigues et al. 1999; Powell et al. 2000; Smith and Gillet 2002; De Klerk et al. 2004; Fjeldså et al.

2004; Oldfield et al. 2004; Rodrigues et al. 2003; Rodrigues et al. 2004b; Yip et al. 2004).

The basic steps in a gap analysis program typically involve developing a vegetation map to identify the location of the major vegetation or land cover types of a region; using that map as the basis of habitat relationship models to describe predicted distribution of terrestrial vertebrates, overlapping information on distribution of vegetation cover types and vertebrate species with mapped information on land ownership and stewardship status (Jennings 2000; Groves 2003). In this way gaps in the network of conservation areas can be identified for both vegetation types (or land cover classes) and vertebrate species. Such an approach can be expanded to include almost any type of conservation target that can be mapped at the appropriate spatial and temporal scale.

Gap analysis was originally developed as a simple tool to evaluate gaps in the coverage of reserve networks but it is increasingly being utilized in conjunction with systematic planning methods to identify candidate sites to fill the existing gaps (Kiester et al. 1996; Margules and Pressey 2000; Groves 2003; Rodrigues et al. 2004b).

Problems and limitations in Gap analyses

Gap analyses traditionally have been concentrated on which features are represented or not represented and to what extent. Other aspects have received little attention and some important limitations still remain (Flather et al. 1997). Probably the most important is the likelihood of species or habitats becoming extinct without conservation actions (Margules and Pressey 2000). In fact, features that are underrepresented according to the established representation target can have very different degrees of vulnerability to threatening processes, and thus, some of the gaps are more important than others (Stoms 2000).

A second important point that usually is not explicitly considered in a gap analysis is related to natural processes and biodiversity persistence. Examples measuring gaps in processes and persistence are few (Peres and Terborgh 1995; Cowling et al. 1999; Noss 1999; Allen et al. 2001; Carroll et al. 2003; Solomon et al. 2003; Salomon et al. 2006) and a comprehensive, unique set of criteria for measuring gaps in processes is still far to come (Margules and Pressey 2000).

Moreover, gap analysis focuses on the current distribution of conservation targets and typically does not reveal anything about historical distribution and historical losses of targets (Groves 2003). If the element to protect has already lost 70% of its previous distribution and the gap analysis shows a 10% representation in conservation areas today, actually only 3% of the element's historical distribution is represented (Jennings 2000).

Finally, gap analyses are usually not assessed using independent data-sets. Jennings (2000) reported that approximately 4% of ovenbird (*Seiurus aurocuillus*) habitat is represented in conservation lands in Arkansas. However, the accuracy of this statement has not been evaluated. In this context, scale remains an issue, especially for smaller features.

Selecting additional protected areas

The results of GAP analyses usually demonstrate that protected areas are not representative of the biodiversity of a region (Pressey et al. 1993; Scott et al. 1993; Rodrigues et al. 1999; Margules and Pressey 2000; Rodrigues et al. 2004a). Then, following review of existing reserves, it is often necessary to propose new protected areas that will complement the existing network towards the conservation targets that has been set for the study area. In this stage existing reserves are recognized not only for their contribution to the target but also because they can be considered spatial constraints around which new reserves can be located (Margules and Pressey 2000).

However, the "traditional" way of selecting areas for conservation, based on socio-economic, aesthetic and political criteria, cannot be considered efficient for conservation (Pressey 1994; Scott et al. 2001) and thus it is important to devise a set of explicit criteria to help in the prioritization of sites (Margules and Pressey 2000; Justus and Sarkar 2002) and to extract the maximum value from both biological data and available funding (Pressey and Cowling 2001).

History and evolution of systematic reserve selection methods

Justus and Sarkar (2002) give a clear sketch of the history of some systematic reserve selection methods, and the following discussion refers mainly to their paper. The origin dates back to 1970, when during the symposium "Conservation and Productivity" D.A. Ratcliffe described a set of criteria for the comparative evaluation and selection of sites to be targeted for conservation. In particular, he described reserve selection as a process in three steps: 1) field surveys; 2) application of explicitly agreed criteria for site prioritization; 3) final choice of a set of high quality sites. In 1977 the same Ratcliffe identified 994 sites in Britain as worthy of conservation attention based on the set of criteria proposed in 1970. In 1977 Ratcliffe used 10 criteria to perform the selection of sites: 1) size: larger sites should be preferred to smaller ones; 2) diversity (intended as richness of community and species); 3) naturalness: habitat which appears to be unmodified by human influence; 4) rarity: the presence of even one rare species on a site gives it higher value than other comparable sites with no rarities; 5) fragility (intended as the sensitivity of habitats, community and species to environmental changes); 6) typicallness: all species (or other surrogate) of a region should be represented; 7) recorded history: a well-kept scientific record makes a site valuable; 8) position in and ecological/geographic unit: places emphasis on the contiguity of one site with another one; 9) potential value: possibility that good management practices may add to the value of a site; 10) intrinsic value: different species do not have the same value because of bias in human interest (but the same author explicitly emphasized the importance of conserving less charismatic species). The 10 criteria were not considered all at the same level but there was a hierarchy among them: intrinsic value was less important than rarity, fragility was considered for sites that are important for other criteria, and so on. It is also important to note that, despite the historical importance of Ratcliffe's contribution to systematic reserve selection, the criteria he proposed were a mixture of biological consideration and socio-political ones, representing just a way of making systematic and explicit the traditional (and often implicit) way of selecting areas for conservation (Justus and Sarkar 2002).

Ratcliffe's scheme was the most comprehensive, but it was not the only one proposed in the 1970s. In the same time period there were a number of studies proposing reserve selection on the basis of richness and/or diversity, rarity, area, threats from human impact, naturalness, representativeness, scientific value, educational value, recorded history, uniqueness, etc. (see Margules and Usher [1981] and Justus and Sarkar [2002] for a review). However, most of the studies used political and scientific criteria in the same way, without any distinction. Margules and Usher (1981) were the first to clearly distinguish the two, saying that scientific criteria should be considered in the process of selecting potential sites for conservation, while political criteria, sometimes even more important that ecological ones, should be considered only for the final decision on conserving a site, a decision that is usually taken at a political level, and not by scientists.

Margules and Usher (1981) arranged the scientific criteria into three categories: criteria that can be assessed in one site visit (diversity and area); criteria that can be assessed with extensive surveys (rarity, naturalness and representativeness); criteria that can be assessed using case histories of sites (recorder history, potential value, ecological fragility).

Scoring methods and complementarity

During the 1980s, following Margules and Usher's (1981) proposal, many studies used only scientific criteria (the so called scoring methods) for prioritizing areas. The process was greatly enhanced by development of micro-computers and in particular of GIS (Justus and Sarkar 2002), and many scoring methods used a single criterion (such as species richness) or a combination of several criteria (Pressey and Nicholls 1989). For example Rapoport et al. (1986) developed a scoring method to asses the conservation value of an area on the basis of all species inhabiting that area. In their proposal each species contribute to the conservation value of each area according to the following criteria: 1) how widespread the species is in the planning region; 2) the geographic range of the species; 3) the mean density of the species.

Scoring methods are simple to use, but they have some limitations, the most important of which is their inability to recognize the way in which sites can complement each other (Kirkpatrick 1983). Likewise, there is no guarantee that all species will be included in a set of reserves chosen using a non-iterative scoring method, and other methods have been demonstrated to be more efficient (Pressey and Nicholls 1989; Virolainen et al. 2001).

In the 1980s there were two main goals for systematic reserve selection: representing in reserves the maximum number of species and maintaining representative samples of a region (Justus and Sarkar 2002). The two goals are apparently similar but they gave two different outcomes: the first one led to species richness as a criterion, the second led to complementarity. Kirkpatrick (1983) noted that use of species richness led to inefficient reserve networks. In fact, two areas (site A and B) in the same region may have extremely high number of species, but contain almost the same set of species; a third site (C) may contain a lower number of species, but may be extremely different from the other two for species composition. Following the criterion of species richness, site A and B would be selected giving a reserve network that does not cover the entire biodiversity of the region. However, if maximizing the total number of species protected is the goal, site A (or B) and site C would be selected. This is the concept of complementarity: selecting areas that add the most species that have not already been represented (Justus and Sarkar 2002).

The first to use the concept of complementarity, even though implicitly, was Kirkpatrick (1983) in Tasmania. He noted that a non-iterative procedure (that is a scoring process that use a single application formula), and in particular species richness, is inefficient because it does not achieve adequate representation for biodiversity in the smallest possible set of conservation sites. The author followed an easy argument: once species richness (whatever formula has been used) has been calculated, the first area (the one with the highest rank) is selected and preserved; then species that have high values because of their poor preservation status, and that are in the first area selected for conservation, should not have the same weighting in the selection of the second reserve. As an example, consider that a given species, totally unrepresented in the existing reserves, adds 100 to the value of a particular site at the beginning of the process. Then the site is selected to be reserved and the species now is worth only 50, because it is found now at least in one reserve. If we go on with the process the value of the species goes down to 0. The important point is that site scores should be adjusted at each step of the analyses, changing the weightings of the attributes on the assumption that higher value sites are reserved (Kirkpatrick 1983). The author gave also two reasons why his approach should be considered superior to the non-iterative scoring system: 1) non-iterative methods lead to overrepresentation of some species and to non-representation of other; 2) the iterative method provides the maximum nature conservation value per unit area preserved.

The process proposed by Kirkpatrick (1983) does not reject richness, but weights each species with a scoring system that decreases the weight as the representation of each species increases. Even though the author does not make any clear reference to the concept of complementarity, the scoring system that he proposes is a clear application of it (Justus and Sarkar 2002).

Margules et al. (1988) gave a second, completely independent from the first one, application of the complementarity concept to select a subset of wetlands for conservation in Australia. They actually presented two algorithms, both including complementarity. The first algorithm was a procedure to ensure representation of all the native plant species in the study region using a combination of complementarity and rarity in 4 steps. 1) Select all sites with any species that occurs only once. 2) Start with the rarest species in the data matrix that is not yet represented and select from all the sites where the species occurs the one contributing the maximum number of still unrepresented species. 3) If two or more wetlands contribute equally then select the one with the group of species having the smallest sum of frequencies of occurrence in the remaining unselected wetlands. 4) If still two or more sites are contributing in the same way select the first one encountered. Step 2 in this algorithm is a clear application of the complementarity approach, even if rarity has the precedence in the first step. The second algorithm (3 steps) was intended to represent each of the wetland type and each of the plant species at least once. 1) Select the wetland from each habitat type that hosts the highest number of species; if all species are included then stop. 2) Select a second wetland from each habitat type that adds the most new species; if all species are included then stop; if no wetland in a particular habitat type add new species then pass over the habitat type. 3) Continue with the process until all the species are represented. Also in this case step 2 is a clear application of the complementarity process.

Margules et al. (1988) noted also that using Kirkpatrick's (1981) approach (i.e. a combination of species richness and complementarity) sites with unique species tend to be added late in the selection process, adding a higher number of sites to the reserve network in order to cover all the species (in other words the richness/complementarity algorithm is not efficient). Margules et al. (1988) algorithms were subsequently modified mainly to improve efficiency (Pressey and Nicholls 1989) and to incorporate other criteria, among which it is important to remember adjacency, which puts a premium on a site being next to one already selected, all other things being equal (Nicholls and Margules 1993). These modified algorithms have been the basis for the creation of software packages like C-Plan (Justus and Sarkar 2002).

Complementarity was independently rediscovered for the third time in South Africa (Rebelo and Siegfried 1990) but only in 1991 Vane-Wright et al. introduced the term complementarity. One of the reasons why the complementarity approach has been so widely accepted is that complementarity-based algorithms outperform richness-based algorithms in minimizing the number of sites selected to achieve a required level of representation for species or other biological surrogates (Williams et al. 1996; Csuti et al. 1997).

Operation research and conservation planning

The complementarity-based algorithms described till now are heuristic (they work step by step to find the solution), and as such they do not guarantee by definition that the selected set of sites is the most efficient that is possible to find, the so called global optimum (Underhill 1994). If the target for representation is 1 (i.e. 1 occurrence for each species) it can be shown that the problem of obtaining the global optimum is equivalent to the "maximum covering" problem of the theory of algorithms that can be solved using integer linear programming techniques (Cocks and Baird 1989; Church et al. 1996).

The mathematical solution to the "maximum covering" problem has its origins after the Second World War in the field of operations research (Kingsland 2002; Rodrigues and Gaston 2002). Operations research was developed as a method of providing a quantitative basis for making decisions in a military context, and after the war the method was extended to many different domains, like industry and economy. In the case of reserve selection, given that the amount of land that can be reserved for conservation is limited, one optimization problem would be to find the minimal set of sites containing all features that are considered important to conserve. These techniques were used for the first time at the end of the 1980s in the Eyre peninsula (South Australia) to exemplify how mathematical programming can be used to find the best sets of sites to protect (Cocks and Baird 1989). The technique has been subsequently refined and used in site selection studies in Europe (Sætersdal et al. 1993; Stokland 1997; Rodrigues et al. 2000, Cabeza et al. 2004; Juutinen et al. 2004), North and South

America (Camm et al. 1996; Church et al. 1996; Haight et al. 2000; Kelley et al. 2002; Nalle et al. 2002; Sarkar et al. 2004), Australia (Pressey et al. 1999), and Africa (Moore et al. 2003).

If we let m represent the total number of sites that compose our study region, n as the number of different species (or vegetation types or any other attribute that we can use) to be represented in the protected areas network, and we want 1 representation of each species in the network, then we can formulate the maximum covering problem using the notation below (Arthur et al. 1997; Possingham et al. 2000). Let **A** be the site-by-species matrix (m X n) whose elements a_{ij} are

$$a_{ij} = \begin{cases} 1 \ \mathit{if\ \ species\ \ j\ \ occurs\ \ in\ \ site\ \ i} \\ 0 \ \mathit{otherwise} \end{cases} \qquad \text{for } i=1,\ldots,m \text{ and } j=1,\ldots,n$$

We can define a control variable that determines whether or not a site is included in the reserve as the vector **X** with dimension m and elements x_i given by

$$x_i = \begin{cases} 1 \ \mathit{if\ \ site\ \ i\ \ is\ \ included\ \ in\ \ the\ \ reserve} \\ 0 \ \mathit{otherwise} \end{cases} \qquad \text{for } i=1,\ldots,m$$

Using these definitions the problem is

$\min\sum_{1=1}^{m} x_1$ [minimize the number of sites in the reserve system]

$\mathit{subject\ to}\sum_{i=1}^{m} a_{ij}x_i \geq 1$ for $j=1,\ldots,n$ [subject to each species being represented at least once]

The formulation of the problem can be easily modified to include a cost for each site being selected, as well as multiple representations of species in the reserve system.

However, the solution is not so simple. One possibility is to use a complete enumeration of possibilities, i.e. to evaluate one by one all possible combinations. The number of such evaluations would be 2^m (where *m* has the same definition given above, i.e. the number of possible sites). Consequently, if m=10 the number of possible combinations is 1024; if m=50 the number of possible combinations is 1,125,899,906,842,624. Let's say the computer time needed to evaluate a single combination is 10^{-6} seconds; then the problem with 10 sites would be solved in 0.001024 second, but the problem with 50 sites would take approximately 36 years. Naturally most of the reserve selection exercises are performed on numbers much greater than 50 sites, making it impossible to find the mathematical solution in a convenient time frame even using the most powerful available computers. One solution has been found in the "branch & bound" algorithm, that repeatedly divides the original problem into smaller sub-problems: when a solution that meets the original integer requirements is found in a sub-problem this solution is adopted for the original problem. Still, different authors have pointed out that time is a constraint, even using the "branch & bound" algorithm (Pressey et al. 1996a; Cabeza and Molainen 2001). On the contrary, Rodrigues and Gaston (2002) demonstrated that using modern computer and standard optimization software, such as LINDO or C-PLEX, most of the reserve selection problems can be solved relatively quickly. The main obstacle remains the accessibility of these software packages to conservation biologists (Moore et al. 2003).

Simulated annealing

A third possibility for reserve selection is simulated annealing, that is part of the global heuristic algorithms (Kirkpatrick et al. 1983; Possingham et al. 2000; Leslie et al. 2003; Cook and Auster 2005; Possingham et al. 2006). Simulated annealing is a minimization method that begins by generating a completely random reserve system and then iteratively explores different solutions by making random changes to the initial system. It can include in the system a random site that was not yet included, or it can delete from the system a random site. At each step the new solution is compared with the previous one on the basis of an evaluation function (called objective function) and the best one is accepted (Possingham et al. 2000). In details, the algorithm works as follows: 1) set input parameters and the maximum number of iterations; 2) generate an initial reserve system consisting of sites selected at random and compute the objective function; 3) randomly select a site to add to the system or to delete from the system; 4) evaluate the resulting change in the objective function: if $e^{\left(\frac{-change}{Accep\tan ce\ Level}\right)} < Random\ Number$ then accept the change, otherwise reject it; 5) decrease the acceptance level and repeat steps 3-5 for the given number of iterations.

The acceptance level determines what size change will be accepted. Negative changes (those that decrease the objective function) will always be accepted. When the acceptance level approaches 0 the only acceptable changes are those that reduce the objective function. The use of the exponential in the evaluation of the reserve systems means that the algorithm spends relatively little time accepting bad changes, and much more resolving small differences.

Possingham et al. (2000) demonstrated the use of this method for a region in Australia with 1885 sites and 248 species of conservation concern. For the same dataset, the classical heuristic algorithms gave as the best results 57 sites, while simulated annealing obtained a minimum of 54 sites, but with a longer running time.

PROs and CONs of different algorithms

All the methods outlined above have pros and cons and they have been investigated in many different studies. Heuristic algorithms have been criticized manly because of their sub-optimality (Underhill 1994; Rodrigues et al. 2000). The degree of sub-optimality of heuristic algorithms has been investigated in several studies with varying results (Sætersdal et al. 1993; Pressey et al. 1996a; Csuti et al. 1997; Stockland 1997; Pressey et al. 1999; Moore et al. 2003). The optimality of the different algorithms was found to be a function of the dataset considered. The number of rare features is an important factor in explaining the differences between iterative heuristics and integer linear programming: the larger the number of rare features, the more sites are needed and the closer are the two solutions. Other important factors are the size of the dataset and the size of the selection units (Pressey et al. 1999). Csuti et al. (1997) compared several heuristic algorithms (4 richness-based algorithms, 13 rarity-based algorithms, 1 simulated annealing) with the optimal solution obtained using integer linear programming. They found that several simple heuristic algorithms provided near-optimal solutions for their dataset and they suggest that using heuristic algorithms can be preferable given speed and simplicity of these ones compared to integer linear programming. Moore et al. (2003) found that heuristic methods needed 2-10% more sites to achieve the conservation goal compared to the optimal solution. Pressey et al. (1996a) compared two different heuristic algorithms with the optimal solution and they found a sub-optimality for the heuristic methods of 50% and 6% for the number of sites; and of 40% and 11% for the area to be reserved. The authors suggested also that the gap between the result obtained using a heurist algorithm and those obtained using integer linear programming can be narrowed if a large proportion of the features to be represented in a reserve system are rare features. In fact, this will force the algorithms to select a very similar set of sites.

Rodrigues et al. (2000) and Rodrigues and Gaston (2002) reviewed a number of studies and compared the results obtained using heuristic algorithms with those obtained through integer linear programming. The authors outlined that most of the studies using heuristic algorithms reported only the best results obtained after many runs and thus the comparison with the results obtained using integer linear programming are probably underestimates of the problem that potentially represents an important cost: Juutinen et al. (2004) estimated that some 160-320 million US dollars could have been saved selecting the Finnish conservation areas with an optimality increase of 9-19%.

However, to evaluate the relative merit of different algorithms, we can use schemes that do not consider exclusively efficiency (the value of reserve selection algorithms is primarily indicative and not prescriptive, so why bother so much with absolute efficiency?), but also other parameters that are important in conservation planning on the ground (Pressey et al. 1996a; Pressey et al. 1997; Cabeza and Moilanen 2001). In fact, in practical applications heuristics can still be used to quickly identify minimum reserve requirements, and they can be easily adapted to solve problems that are too complex for integer linear programming (Polasky et al. 2000; Williams and Araujo 2000).

Simulated annealing represents something between classical heuristic algorithms and integer linear programming algorithms, in that it is able to find solutions more efficiently than heuristic approaches but not necessarily the best solution existing. At the same time simulated annealing is quicker than integer linear programming but slower than heuristics. A peculiarity of simulated annealing is the possibility of finding multiple alternative solutions from the same dataset, a characteristic that has been considered both a limitation and an advantage of the algorithm (Possingham et al. 2000; Rodrigues and Gaston 2002).

Theoretically, integer linear programming techniques should be preferred because they guarantee optimality, and they are usually used for theoretical exercises and when the size of the dataset and the conservation goals are appropriate (Arthur et al. 1997; Cabeza and Molainen 2001). Heuristics and simulated annealing are used when solutions are needed in seconds or minutes and when it is necessary to interact with decision makers (Margules and Pressey 2000; Cabeza and Molainen 2001).

An important point to consider is the availability of software to perform the analysis. There exist a lot of free available programs that can be used in reserve selection (C-Plan, Conservation Action Planning Toolkit, CCP GIS Tools, CLUZ, CODA, MARXAN, PANDA, ResNet, SITES, SPEXAN, Worldmap), and most of them can perform many different types of heuristic algorithms. Some of them can perform simulated annealing, but none can be used for integer linear programming. This is probably the single most important reason why heuristic and simulated annealing is so widely used, while operations research is almost exclusively an academic exercise (at least in conservation planning).

All the algorithms described have an important limitation: they provide no information on the potential contribution to the conservation target of the unselected areas in the study region (Pressey et al. 1993; Pressey et al. 1994; Pressey 1999a; Ferrier et al. 2000). Consequently, most of them do not indicate optional replacements for the areas that have been selected and that can become unsuitable or unavailable for conservation management before being effectively preserved (Pressey 1999a; Ferrier et al. 2000; but see Williams et al. 1996). The alternative sets of areas that can achieve the conservation goal in a region can number in the hundreds or even in the hundreds of thousands (Pressey 1999a), increasing with the number of potential conservation areas in the region (Pressey et al. 1994) and decreasing when conservation targets become larger (Pressey 1999a) or the features to be conserved are narrowly distributed (Ferrier et al. 2000).

Irreplaceability in conservation planning

It is not feasible to consider all the possible reserve systems, but information on optional conservation areas can be extremely valuable when dealing with local constraints. This information can be obtained from a map of the irreplaceability of each potential conservation area in a region. Like all the algorithms described above, any analysis of irreplaceability is driven by a quantitative target for the features (species, habitats, etc.) in the region. Once the target has been defined clearly, irreplaceability can be defined in two different ways: 1) the likelihood that the area will be required as part of a conservation system that achieves the set of targets; 2) the extent to which the option for achieving the set of targets are reduced if the area is unavailable for conservation (Pressey et al. 1994).

Irreplaceability can be measured with values ranging from 100% to 0% (Pressey 1999a; Ferrier et al. 2000). Areas where irreplaceability is 100% are totally irreplaceable and must be included in the system of protected areas to achieve the conservation target; likewise if these areas lose their conservation values because of development or overuse, one or more of the conservation targets for the study area will become unachievable. Areas with progressively lower irreplaceability values have progressively more replacements in the region, and they are less likely to be required as part of a system of conservation areas; likewise if these areas are destroyed or made unavailable for conservation, the impact on the achievement of the target would be lower. Areas where irreplaceability is 0 are those that contain features that have already met their target in the existing reserves. Areas with total or high irreplaceability can be considered the nodes of an expanded system of conservation areas, around which other areas can be grouped. Choices between areas with lower irreplaceability can be resolved according to location, size, condition, cost and other factors (Pressey 1999a; Ferrier et al. 2000).

Irreplaceability can be related to measures of rarity (low abundance or restricted distributions) and endemism (number of features unique to an area). In fact, endemic features will always confer total irreplaceability, at least if conservation targets are one occurrence of each feature; if targets are multiple occurrences the relationship will be poorer (Pressey 1999a; Ferrier et al. 2000). Moreover, irreplaceability is also linked to complementarity in three ways: 1) irreplaceability for an area is calculated in the context of the features it contains, the regional target for the features, and the distribution of the feature in other areas within the region; 2) irreplaceability values are calculated after the contribution of any existing reserve is taken into account; 3) irreplaceability can be recalculated quickly to account for new reserves being established and/or modeled (Pressey 1999a; Ferrier et al. 2000).

How to measure irreplaceability

The concept of irreplaceability is pretty straightforward but its measurement is not, since no simple arithmetic index can be used. As an example (following Ferrier et al. 2000), suppose we want to select a system of protected areas and we have set a quantitative target for each feature that we want to protect. The study region can divided into units (hereafter called sites), and for each site we have calculated the amount (area, numbers, etc.) of each feature. A possible measure of irreplaceability could be obtained dividing the area of each feature occurring in a site by the target for that feature and then summing these proportions across all features in the site. Such an index can give an indication of the potential contribution of a site to achieve the target, but it does not tell us anything about the irreplaceability of that site. Ferrier et al. (2000) provided the following example: consider two sites in a region of several hundred sites, both hosting a single feature: site 1 contains 50 ha of vegetation type A, site 2 contains 50 ha of vegetation type B. The protection target for both vegetation types is 100 ha, but the two vegetation types have a different total extent: vegetation type A has a total area of 150 ha while type B has a total area of 2000 ha. The arithmetic approach proposed above would assign equal priority to these two sites because they each contribute equally to achievement of targets (50%). However, the irreplaceability of site 1 should be viewed as much higher than that of site 2 because the contribution of the latter can be more easily replaced by protecting other sites in the region.

The arithmetic approach can be considered only in special cases (sites of equal sizes, no more than 1 feature in each site) but in general a simple arithmetic approach would neither be able to consider the extent to which a site's contribution can be replaced by protection of other sites in the study area, nor to measure the extent to which options for achieving targets are reduced if this site is not protected (Ferrier et al. 2000).

Pressey et al. (1994) proposed measuring irreplaceability as the percentage of representative combinations of areas (i.e. alternative systems of areas that will achieve conservation targets for all features) in which each area occurs (Kiester et al. 1996; Pressey 1999a; Ferrier et al. 2000). If the system of protected areas is selected with a combination of n sites out of a total of t sites, the total possible combinations of size n that can be constructed is given by the classical binomial coefficient:

$$C = \frac{t!}{n!(t-n)!}$$

Of all the *C* possible combinations only some will met the conservation target (set A in Fig. 3), while all others will fail to meet targets for one or more features. If we consider a single site *x*, the set of representative combinations can be further subdivided into two subsets, one containing all combinations that include site *x* (set B in Fig. 3) and the other containing combinations that do not include site x. Pressey et al. (1994) measured irreplaceability of site *x* by dividing the number of representative combinations that include site *x* (set B in Fig. 3) by the total number of representative combinations (set A in Fig. 3).

However, they noted that the set of representative combinations that includes site *x* (set B in Fig. 3) can be further subdivided into those combinations that would no longer be representative if site *x* were removed from the combination (set C in Fig. 3) and those combinations that would still be representative after removal of site *x* (i.e. all the combinations in which site x is redundant). The inclusion of site *x* in a representative combination affects the calculation of irreplaceability even if that site is not making a critical contribution to achieving targets (Pressey et al. 1994). The effect of this redundancy on the original irreplaceability measure increases with increasing combination size. The problem is removed by measuring the proportion of representative combinations for which site *x* plays a critical role, that is dividing the number of representative combinations that include site *x* but would no longer be representative if site *x* were removed (set C in Fig. 3) by the total number of representative combinations (set A in Fig. 3).

With this measure of irreplaceability a totally irreplaceable area would occur in all the representative combinations either because it contains one or more unique features or because it contains sufficiently large occurrences of one or more features that the conservation targets for those features cannot be met without it (Pressey 1999a). However, the problem is well beyond the capabilities of even the most powerful computers; Ferrier et al. (2000) considered as an example a case study with t=359 and n=33 (numbers much smaller than real world applications) that would involve inspecting $5.25*10^{46}$ combinations. However, the combinatorial approach was used by Pressey et al. (1994) for a really small dataset and by Kiester et al. (1996) with t=389 and n=5.

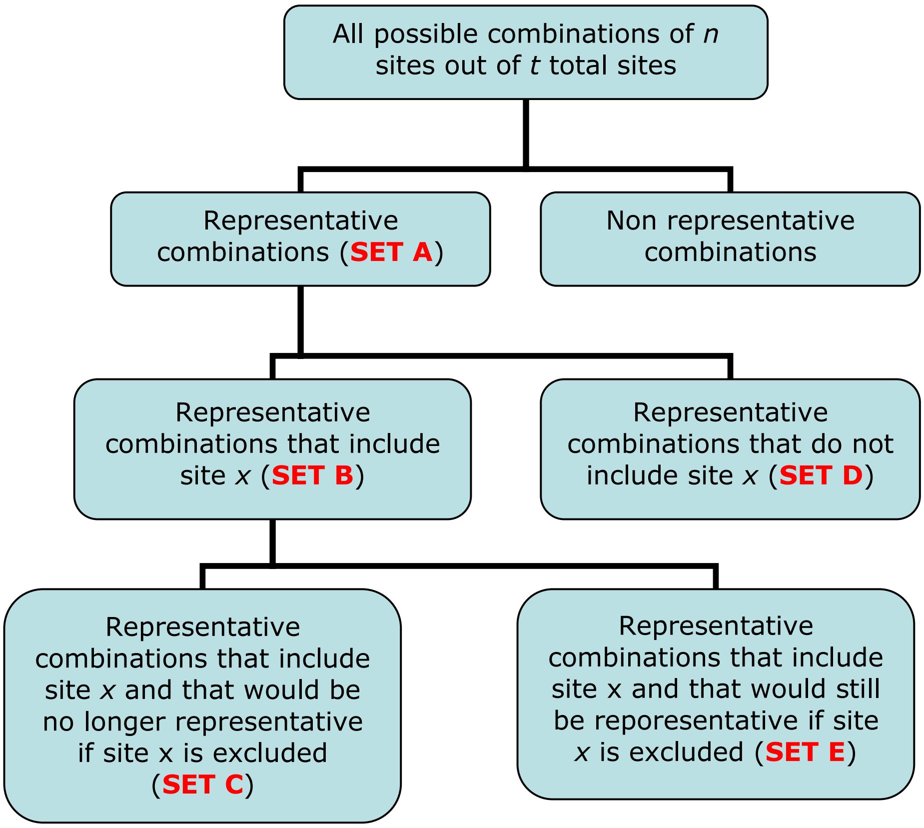

Figure 3. Site combinations used to measure irreplaceability for site *x*. Modified from Ferrier et al. 2000.

A major step forward in the exact measurement of irreplaceability was provided by Csuti et al. (1997). They applied branch and bound optimizing algorithms to a dataset with 441 areas and 426 vertebrate species to find all the minimum-sized sets of areas that would achieve the conservation targets. Then they measured irreplaceability by counting the frequency of each area in the 144 sets of 23 areas that represented each species at least once. The method has now been improved and made quicker (Ferrier et al. 2000) but it is too slow to be useful in real-time calculations.

Given that no exact measure is possible, the only possible approaches to measure irreplaceability are predictive approaches (Ferrier et al. 2000). Three such approaches are currently available. The first one is "effective maximum rarity" that was proposed by Pressey et al. (1994) and is based on the frequency of unprotected features in the dataset; it can produces reasonably accurate results for targets of one occurrence of each feature and in small datasets. The second approach is the "multiple minimum sets" that was proposed by Rebelo and Siegfired (1992) and modified by Freitag et al. (1998). They applied the conservation algorithm to their data set many times, starting each application with a set of randomly chosen "seed" areas, and then they approximated the irreplaceability of each area by calculating the percentage of all sets in which the area occurred. The third approach is a statistical measure of irreplaceability proposed by Ferrier et al. (2000).

A probabilistic approach to irreplaceability

Ferrier et al. (2000), starting from the revised calculation of irreplaceability (set C divided by set A), proposed a re-expression of the problem. Let $R_{x\_included}$ be the number of representative combinations that include site *x* (set B in Fig. 3), $R_{x\_excluded}$ be the number of representative combinations that do not include site *x* (set D in Fig. 3) and

$R_{x\_removed}$ be the number of representative combinations that include site *x* and would still be representative if site *x* were removed (the combinations in which site *x* is redundant; set E in Fig. 3). The calculation of irreplaceability for site *x* can then be expressed as:

$$Irr_x = \frac{\left(R_{x\_included} - R_{x\_removed}\right)}{\left(R_{x\_included} + R_{x\_excluded}\right)} \text{ (Equation 1)}$$

Consider a case in which we have a single feature (*i*) to conserve and we know the area occupied by this feature within each of the *t* sites in the study area. We can consider the *t* sites as a finite population, and each combination of *n* sites selected from this population can be considered as a sample (without replacement). Then, based on the central limit theorem, the total area of *i* protected by a randomly selected combination of *n* sites is expected to have a distribution approaching a normal distribution with mean $\overline{Y}_i n$ (Equation 2) and variance $S_i^2 n(1 - n/t)$ (Equation 3) where $\overline{Y}_i$ and S_i^2 are the population mean and variance of the area of feature *i* across all *t* sites. This distribution can be used to estimate the proportion of possible combinations that will protect an area equal to or greater than the protection target, and then an estimate of the number of representative combinations can be obtained by multiplying the estimated proportion of combinations that will achieve the target by the total number of possible combinations. Following this approach we can estimate $R_{x\_included}$, $R_{x\_excluded}$, and $R_{x\_removed}$ using 4 steps (see discussion below on how to calculate irreplaceability following Ferrier et al. 2000 and NSW-NPWS 2001; Fig. 4).

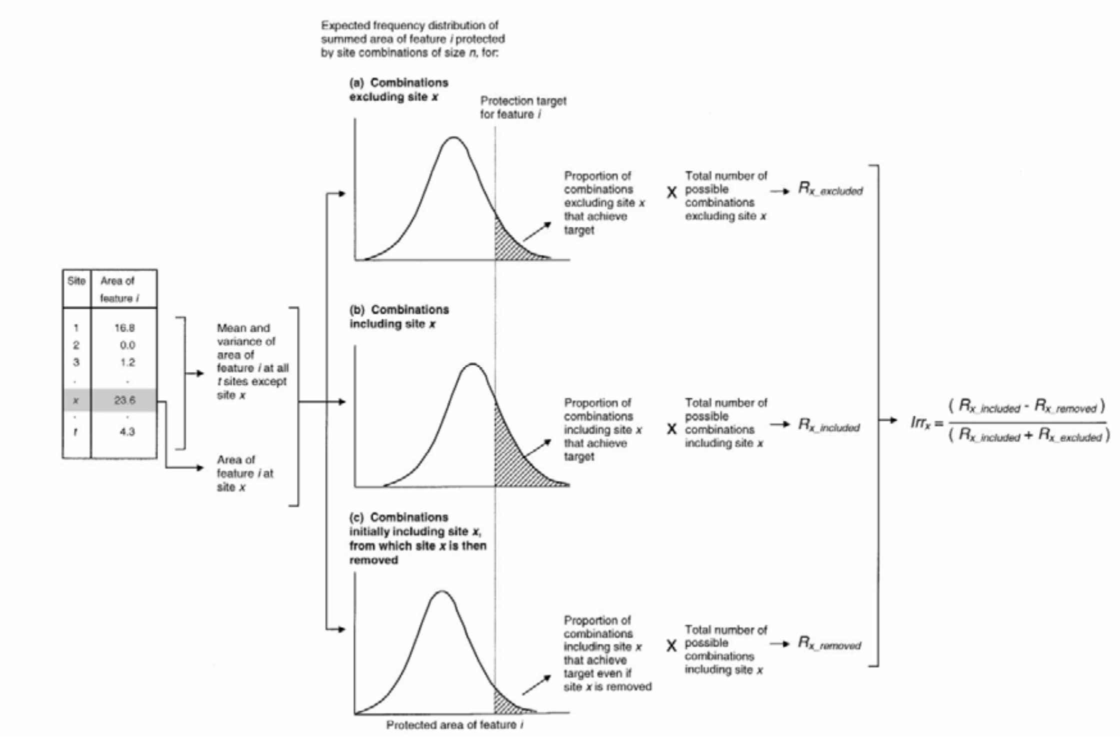

Figure 4. Steps involved in the statistical calculation of irreplaceability. See the text below for a detailed description of each step. Modified from Ferrier et al. 2000.

Step 1. The mean $\overline{Y_i}$ and variance S_i^2 of the feature's area are calculated considering all sites except for site *x* (the one for which irreplaceability is being calculated).

Step 2. Using the results from step 1 it is possible to estimate the mean and variance of normal distributions describing the expected distribution of the summed area of feature *i* protected by a randomly selected combination of size *n*. In particular, three different normal distributions will be estimated, one for combinations excluding site *x*, one for combinations including site *x*, and one for combinations including site *x* from which site *x* is then removed.

To estimate mean and variance for $R_{x\_excluded}$ (the number of representative combinations that do not include site *x*) we need to calculate the mean and variance of a normal distribution describing the expected distribution of the summed area of feature *i* in a randomly selected combination of *n* sites drawn from all *t* sites except site *x*. The expected mean of this distribution can be calculated using Equation 2 and $\overline{Y_i}$ obtained from Step 1. The expected variance of the distribution is calculated using S_i^2 from Step 1 and the following modified version of Equation 3: $S_i^2 n(1-\frac{n}{t-1})$. Replacing *t* in Equation 3 with *(t – 1)* we can account for the fact that each random combination of *n* sites is not drawn from the total set of *t* sites but rather from a reduced set of *(t – 1)* sites (site *x* has been excluded).

To obtain the mean and variance for $R_{w\_included}$ we need to calculate the parameters describing the expected distribution of the summed area of feature *i* in a combination of *n* sites that includes site *x* and *(n – 1)* other sites selected at random. The expected mean of this distribution is calculated by summing y_{ix} (the area of *i* at *x*) and $\overline{Y}_i\left(n-1\right)$ (the mean summed area of feature *i* expected for the *(n – 1)* sites selected at random.). The expected variance of the distribution is calculated using the following modified version of Equation 3: $S_i^2\left(n-1\right)(1-\frac{n-1}{t-1})$. *t* is replaced by *(t – 1)* to account for the fact that the randomly selected sites are drawn from a reduced set of sites (excluding site *x*). *n* is replaced by *(n – 1)* to account for the fact that only *(n – 1)* members of the combination are free to vary. Site *x* is always included in the combination and therefore does not contribute to the variance of the summed area.

The expected mean for $R_{x\_removed}$ (the number of representative combinations that include site *x* and would still be representative if site *x* were removed) is $\overline{Y}_i(n-1)$, which is the mean used for estimating $R_{x\_included}$ minus y_{ix} to account for the removal of site *x* from the initial combination. The expected variance for $R_{x\_removed}$ is identical to that for $R_{x\_included}$ because removal of site x has no effect on the variance of the summed area of feature *i* for the remaining sites.

Step 3. The parameters that we have calculated in Step 2 can be used to estimate the probability that the summed area of feature *i* protected by a randomly selected combination will be equal to or greater than the feature's protection target. This probability gives also an estimate of the proportion of combinations that will be representative (i.e. achieve the target for feature *i*). To estimate this proportion for a given normal distribution, the protection target for feature *i* (called T_i) is first transformed to a standard normal deviate following the formula: $z_T = (T_i - \mu)/\sigma$, where μ and σ are the mean and standard deviation of the distribution considered. Then, the proportion of combinations for which the summed area of feature *i* will be equal to or greater than the feature's protection target is the proportion of the area under the normal curve to the right of T_i (Fig. 4) and it can then be estimated using the normal integral:

$$P(z \geq z_T) = \int_{z_T}^{\infty} \frac{1}{\sqrt{2\pi}} e^{-z^2/2} dz$$

This can be applied to each of the three normal distributions for which mean and variance parameters have been estimated in Step 2. Three proportions are therefore generated: 1) the proportion of combinations of *n* sites that exclude site *x* that are predicted to satisfy the protection target for feature *i* (that will be used to estimate

$R_{x\_excluded}$); 2) the proportion of combinations of *n* sites that include site *x* that are predicted to satisfy the protection target for feature *i* (that will be used to estimate $R_{x\_included}$); 3) the proportion of combinations of *n* sites that include site *x* that are predicted to satisfy the protection target for feature *i* even if site *x* is removed from the combination (that will be used to estimate $R_{x\_removed}$).

Step 4. $R_{x\_excluded}$, $R_{x\_included}$ and $R_{x\_removed}$ are estimated by multiplying each of the proportions derived in step 3 by the total number of possible combinations that can be obtained in each of the three scenarios. In particular, the total number of possible combinations of size *n* that include site x (used in estimating $R_{x\_included}$ and $R_{x\_removed}$) is $\frac{nC}{t}$ and the total number of possible combinations that do not include site x (used in estimating $R_{x\_excluded}$) is $\frac{(t-n)C}{t}$. In both cases C is calculated using the binomial coefficient.

To be useful for real-world application the technique described above must be extended to situations in which multiple features are assigned protection targets and two or more of these features can occur within any given site in a region. An approximate approach to estimate $R_{x\_excluded}$, $R_{x\_included}$ and $R_{x\_removed}$ for multiple features is to assume that these features are distributed independently across sites in the region. Then the proportion of combinations in a set that achieve targets for all features can be estimated by multiplying the proportions of combinations that achieve targets for individual features. For example, if in calculating $R_{x\_included}$ for a dataset containing three features it is estimated that the proportion of combinations including site *x* that achieve the target is 0.4 for feature 1, 0.5 for feature 2 and 0.2 for feature 3 then the estimated proportion of combinations that achieve all three targets is 0.4 * 0.5 * 0.2 = 0.04. $R_{x\_included}$ is then estimated by multiplying this combined proportion by the number of possible combinations that include site *x*.

There are two potential problems with this approach. The first problem relates to the assumption that features are distributed independently across sites within a region. It is assumed that for any pair of features A and B there is no correlation between the distributions of these features. The assumption is not realistic: for example, features A and B might occupy similar environmental niches and, therefore, sites containing a relatively large area of feature A will contain, on average, a larger area of feature B than sites containing a small area of feature A. However, given that real data sets are likely to contain a mix of positively correlated and negative correlated features (especially if the number of features is high) it is probable that the effects of these two types of correlation on irreplaceability predictions are cancelled.

The second problem is related to the measure itself. What we are estimating is the likelihood that a given site will need to be protected to achieve targets for all features under consideration. An irreplaceability value of 1 for a site indicates that, unless that site is protected, one or more features will fail to achieve the specified target. The problem is that this measure tells us nothing about how many features will fail to meet their targets as a result of not protecting a site. Multiple feature irreplaceability, on its own, does not provide an adequate basis for prioritising these sites. The measure does not discriminate between sites that are highly irreplaceable for one or many features.

To address these problems Ferrier et al. (2000) have developed alternative approaches, named summed irreplaceability (the sum of the irreplaceability values obtained for each feature separately) and weighted irreplaceability (in which features can be weighted according to their conservation value). Other possible algorithms have been suggested (Tsuji and Tsubaki 2004) but they have had few practical applications.

Irreplaceability has been used extensively, especially in Australia (reported in Ferrier et al. 2000; NSW-NPWS 2001; Justus and Sarkar 2002) but also in the USA (Noss et al. 2002), in south Africa (Freitag et al. 1996; Lombard et al. 1997; Lombard et al. 1999; Cowling et al. 2003a) and worldwide (Rodrigues et al. 2003, 2004a), and it is now an important component of reserve selection algorithms.

Expert-driven vs systematic conservation planning

In spite of their increasing popularity reserve selection algorithms have their critics (Prendergast et al. 1999; Cabeza and Moilanen 2001; Sarakinos et al. 2001; Justus and Sarkar 2002; Cowling et al. 2003b) and they have been contrasted with the traditional ad hoc approach. Both approaches have pros and cons.

The pros of the expert-driven approach are that it draws on expert judgments about biodiversity persistence and pragmatic management and implementation issues (for example it can consider the need for rationalisation of reserve boundaries as well as socioeconomic constraints on implementation) not normally included in biodiversity feature-site data matrices (Hannah et al. 1998; Prendergast et al. 1999; Maddock and Samways 2000); moreover, it can be used to create consensus and build capacity for conservation planning (Moote et al. 1997; Rodriguez and Young 2000). The main con of this approach is that it involves biases associated with experts' knowledge of regions and taxa (Kress et al. 1998; Maddock and Samways 2000).

Using reserve selection algorithms it is possible to perform a transparent analysis of more-or-less consistent data, it is possible to use explicit targets, it is possible to be flexible in the analyses, changing data and targets, and assessing different options for achieving those targets (Pressey et al. 1993; Pressey1999b; Pressey and Cowling 2001; Cowling et al. 2003b). The major disadvantages of the systematic approaches are: the widespread ignorance among managers of these approaches (Prendergast et al. 1999), the absence in the data matrix of important undocumented information on biodiversity as well as implementation opportunities and constraints (Prendergast et al. 1999), and their inability to consider issues of biodiversity persistence (but see Williams and Araujo 2000; Cowling and Pressey 2001; Cowling et al. 2003a; Wilson et al. 2005b).

Moreover, Prendergast et al. (1999) stated that the quality of the data that these sophisticated algorithms need is much higher than what most managers may ever expect to have. They stated also that in the absence of high quality data the only possible option is that of acquiring this data, with costs that, in their opinion, would have been used better acquiring land for conservation. However, ad hoc reservation using no data or no analysis on incomplete data has led to inefficient reserve networks with a misuse of the few existing resources (Pressey 1994; Pressey and Cowling 2001; Cowling et al. 2003b). In addition, reserve selection algorithms can work with very simple datasets (especially if using environmental surrogates and not species data) and in this way they can provide at least a guide to the decision on where and how to spend the limited resources available for conservation (Justus and Sarkar 2002).

Implementing conservation actions on the ground

There is obviously a huge difference between performing theoretical applications of the process to select conservation areas and implementing the process on the ground. In fact, implementation on the ground has to do with a lot of social, political and economic problems that can make conservation of one area impossible.

On the contrary current methods for conservation planning treat both biodiversity and human economic systems as static. They use a snapshot in time of the distribution and abundance of biodiversity and assume that once a reserve network is identified it can be implemented immediately with all its areas (Meir et al. 2004). Naturally this assumption does not hold in the real world, where implementation of a reserve network can be represented better as a sequential process evolving along decades (James et al. 2001; Pimm et al. 2001; Balmford et al. 2002).

While the reservation process goes on, habitat alteration, loss of species, and landscape modifications continue to change and to limit conservation options. Meir et al. (2004) found that, given the rate of habitat loss reported in the literature, conservation plans should be updated yearly, something that is highly unrealistic. It is important then that we schedule the implementation of conservation plans using the relative vulnerability of the sites considered in order to minimize the extent to which conservation objectives are compromised (Possingham et al. 2006).

In particular, the more vulnerable areas should receive higher priority, especially if there are few or no alternative areas available to protect the features they contain (Pressey and Taffs 2001; Noss et al. 2002; Lawler et al. 2003). The approach is analogous to that used to prioritize regions or countries for conservation (Myers et al. 2000) and it allows for a minimization of the extent to which targets are compromised by threatening processes (Pressey et al. 2004).

In conservation planning, vulnerability has been defined as the likelihood or imminence of biodiversity loss to current or impending threatening processes (Pressey et al. 1996b). Wilson et al. (2005b) extended this definition distinguishing three dimensions of vulnerability: exposure, intensity and impact. Exposure can be measured either as the probability of a threatening process affecting an area over a specified time or as the expected time until an area is affected. Exposure is therefore a measure of risk and it has been measured both categorically (Pressey and Taffs 2001) and on a continuous scale (Serneels and Lambin 2001).

Intensity of a threat in biological conservation can be measured in many different ways (examples may be cubic meters of timber extracted, density of livestock, presence of invasive species), including measures of magnitude, frequency and duration (Wilson et al. 2005b).

Impact refers to the effect of a threatening process on particular features and could indicate effects on the distribution of species, on their abundance, or on their likelihood of persistence (Wilson et al. 2005b).

Areas of particular concern for conservation are usually those with high exposure to highly intense threatening processes. Features of concern should be those occurring in such areas and experiencing strongly negative impacts.

One of the most important steps in a conservation plan is that of producing spatially explicit data on these three dimensions of vulnerability. A map of exposure requires a spatial prediction of the future distribution of the threatening processes: for example, to create a map of future vegetation clearings one can consider variables like agricultural suitability, terrain, proximity to infrastructure, proximity to towns, etc. (Mertens and Lambin 1997; Serneels and Lambin 2001). Once obtained, the map of exposure can then be used to obtain the vulnerability of areas or species (Pressey and Taffs 2001).

Spatial predictions of intensity are less common than predictions of exposure, especially because it is difficult to rank one area in relation to all the others. For many threats, areas are considered either as interested or not (Mertens and Lambin 1997) and the primary concern falls back to exposure. Sometimes, however, it is possible to link a particular threat to some factors that allow one to map gradients of intensity: a good example is grazing, that is known to vary according to factors such as proximity to watering points, distance from population centers, socioeconomic status of human communities, etc. (Pringle and Landsberg 2004).

Of the three components, impact is probably the most difficult to map. In fact, without considering all the other necessary information, producing a map of the impact of a given threat often requires feature-specific information on the effect of different intensities on the different features, effects that may also change from area to area (Wilson et al. 2005b).

Wilson et al. (2005b) have reviewed the methods used to assess vulnerability and divided them into 4 groups on the basis of the type of data employed. All methods they reviewed estimate exposure, but some deal also with intensity and impact. Group 1 considers the methods based on tenure and land use; it can infer the vulnerability of features from their relative amount within conservation areas (exposure), or from permitted or projected land uses (exposure and intensity). Group 2 considers the methods based on environmental or spatial variables; it can infer the vulnerability of features using the past impacts of threatening processes to give a value to areas presently unaffected that contain the same features (exposure), it can use the same information in qualitative and quantitative analysis and models (exposure and intensity). Group 3 measures vulnerability on the basis of the number of threatened species (exposure, intensity and impact). Group 4 considers expert opinion to evaluate vulnerability (exposure, intensity and impact). For a complete review of these methods see Wilson et al. (2005b).

To be useful in the practical implementation of a conservation plan, vulnerability should be used in conjunction with irreplaceability. Margules and Pressey (2000) proposed a framework that uses these two measures to define different management prescriptions for areas with different measures of vulnerability and irreplaceability (Fig. 5). Quadrant 1 (Fig. 5) groups the areas most likely to be lost (high vulnerability) and with fewest replacements (high irreplaceability). In this case protection is urgent or the conservation targets will be lost. However, it is most probable that reservation will not be the most appropriate strategy for all these areas, and off-reserve management can be considered. Quadrant 2 (Fig. 5) groups the areas with high vulnerability but with more replacements with respect to quadrant 1, either because features are relatively common or because targets have been partly met in existing reserves. Some conservation measure (mainly off-reserve management, but also reservation) is necessary to prevent loss of some areas that will cause others to move upwards into quadrant 1. Quadrant 3 groups the areas with low vulnerability but with high irreplaceability. Protection is less urgent than for quadrants 1 and 2 because of the lower vulnerability. Quadrant 4 groups the areas that have low vulnerability and low irreplaceability. These areas are of lower conservation concern and at the same time they are relatively stable. However, they should be monitored to avoid a change to a different quadrant.

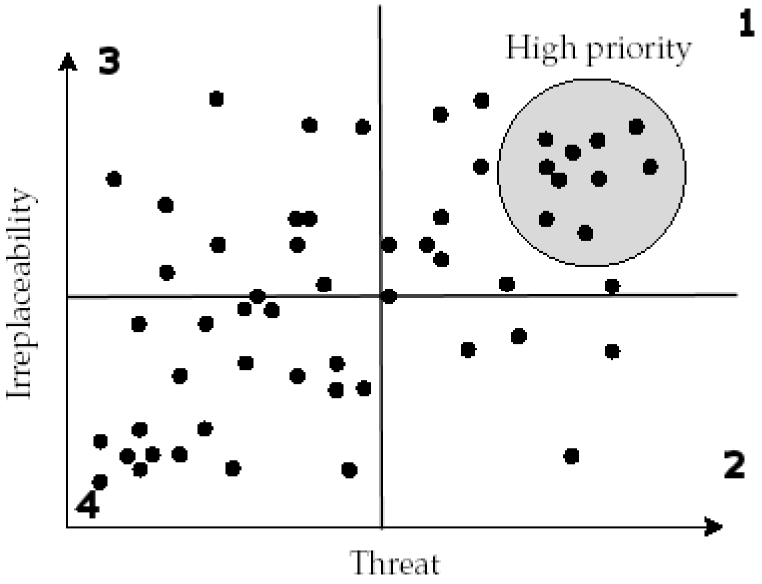

Figure 5. Site evaluation according to irreplaceability and vulnerability. Modified from Rodrigues et al. 2004a.

Management and monitoring of reserves

Once a reserve system has been identified following a pre-defined process, single reserves (usually one at a time) are established, creating the need for a process that is at least as demanding as the planning process and for sure much longer: management and monitoring (Margules and Pressey 2000). A reserve should be managed to ensure that its biological value (basically the reason why the reserve has been established) is retained, even in the face of internal dynamics, external pressures, and human uses. However, it is not uncommon to have limited resources dedicated to the management of protected areas, that are also often threatened by illegal activities, going to the extreme of having protected areas only on paper (Southworth et al. 2006).

A proper and effective management strategy must involve a series of subsequent steps similar to what I have described for the planning process: it requires information on the biodiversity of each reserve (step 1); it should be based on explicit targets(step 2); on the basis of the extent to which management targets have already been achieved (step 3), prescriptions, zoning and other management activities should be reviewed and action plans should be prepared as appropriate (stage 4); key interest groups should be considered and actively involved in the implementation of the management plans in order to avoid problems and minimize potential conflicts (stage 5).

Adaptive management can represent an important tool at this stage, not only to follow the status of biodiversity elements but also to assess the adequacy of resources being used, and the defensibility of the results being obtained (Hockings and Phillips 1999).

Systematic conservation planning and the real world

Systematic conservation planning is an evolving discipline with research and development still ongoing (Pressey 1999b; Possingham et al. 2006). Many new studies, for example, are focused on the use of multiple objectives for conservation planning (Arthur et al. 2004; Moffett et al. 2006; Wilson et al. in press). However, the entire field of conservation planning is still riddled with uncertainty, a characteristic that can be lessened but that will never be eliminated (Margules and Pressey 2000). There is uncertainty in the use of biodiversity surrogates, in the conservation targets, in the review of the existing protected areas, in the implementation process. However, some points should be considered as more important than others.

There is certainly a need for a greater accuracy and precision in the measurement of biodiversity across regions and biomes. Part of this can be accomplished allocating more resources to the collection of field data. However, we will never have more than samples of the existing biodiversity, and we will always need to build models of wider spatial distribution patterns for the features of interest. This clearly calls for sound ecological and statistical bases behind data collection.

A second important point is linked to threats to biodiversity. Mapping and monitoring of these threats has not been one of the priorities in current conservation planning, and yet such threats are the phenomenon to which conservation biologists are responding.

A third point regards management and monitoring of existing protected areas. We have enough ecological information to build a biologically sound management plan only for a tiny part of the existing biodiversity, mainly for large vertebrates. Clearly, there is a need for more resources dedicated to basic ecological studies.

Finally, and probably most importantly, “professional” conservation biologists and ecologists should participate more in real world planning. This is the only way available to incorporate social, economical, and political constraints into conservation planning.

Protected areas in the era of climate change

Something that is still lacking in my review on protected ares is probably one of the most widely debated argument in the last months, on which I will touch briefly here: the effect of climate change. Alread in 1985 Peters and Darling stated that “global warming would diminisch biological diversity by causing extinctions among reserve species….Conservation plans should reflect knowledge of climatic effects as soon as it becomes available”. However, their suggestion has not been followed and, with the exception of studies evaluating the resilience of existing protected areas towards climate change (Araujo et al. 2004; Thomas et al. 2004a, 2004b), the examples of the integration among conservation planning and climate change are really few (Hannah et al. 2002; Coulston and Riitters 2005), and climate change has not yet been routinely integrated in protocols for reserve selections (Araujo et al. 2004).

Yet, climate change over the past 30 years have already produced many different shifts in the distribution and abundance of species (Parmesan and Yohe 2003; Root et al. 2003), and at least one species has been driven to extinction by climate change (Pounds et al. 1999). From the available studies it is clear that the existing protected areas will not be sufficient to ensure biodiversity conservation if the current trend of climate change is maintained (Halpin 1997; Araujo et al. 2004, 2006; Thomas et al. 2004a; Coulston and Riitters 2005; Gritti et al. 2006) but a single solution is still far to come.

One of the main approaches that is followed is that of using climatic envelope models to predict species distribution in response to possible climate change scenarios

(Araujo et al. 2006). However, Halpin (1997) criticized the blind extrapolation of the observed species distribution in relation to present climate to project future responses, unless such extrapolations are based over physiological tolerances, competition and dispersal mechanisms that characterize the species considered. The solution that he proposes passes through the understanding of local disturbancec regimes, through a better knowledge of how landscape fragmentation interacts with population mobility and dynamics, and through a critical evaluation of management interventions with regards to ecological viability.

A different approach is that proposed by Hannah et al. (2002). The authors propose a collaboration involving biogeography, ecology and applied conservation, and the resulting framework, the so-called Climate Change-integrated Conservation Strategies, should apply available tools to respond to the conservation challenges posed by climate change. Collaboration across disciplines is necessary to plan conservation responses to climate change adequately. Biogeography and ecology should provide insights into the effects of climate change on biodiversity that have not yet been fully integrated into conservation biology and applied conservation management. The Climate Change-integrated Conservation Strategies provide a framework in which biogeographers, ecologists and conservation managers can collaborate to address this need.

Clearly, the studies proposed by Araujo et al. (2006) and by Coulston and Riitters (2005) together with many others fall under the umbrella of the Climate Change-integrated Conservation Strategies. However, many problems still remain, mainly linked to the uncertainty that is naturally present in this studies (once the author provide an evaluation of the stability of their results how is it possible to validate these models?) and to the different interpretations that is possible to give at the same results (see for example Thomas et al. 2004a and all the replyies to the paper published by Nature).

However, even considering all the inherent limitations, it is not possible to wait for a better understanding and for better data. The IPCC 4th Assessmemt Report on climate change for 2007 (available from www.ipcc.ch) clearly demonstrate that we have no more time to waste but that we must take action now. In this framework it is clear that the Climate Change-integrated Conservation Strategies is probably the best way to go that we have.

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CHAPTER 2: DESCRIPTIVE ANALYSIS OF EXISTING AND PROPOSED PROTECTED AREAS IN ITALY

The Italian peninsula (Fig. 1), with its long and complicated biological and geological history (Blondel and Aronson 1999), can be divided into 6 main zoo-geographical regions: the Alps, the Apennines, the Apulian region, the Sicilian region and the Sardinian region. Its fauna is the richest among the European countries, and the number of known species is continuously increasing, even among mammals (Minelli et al. 2002).

The species of animals known to be present in Italy are 55,600, 82% of which are arthropods (just considering the insects the species counted are 37,000 corresponding to 67% of the Italian species), while only 2% are vertebrates. The number of endemic species is relatively high (10% of all the species are present only in Italy), being as high as 28% for some taxa.

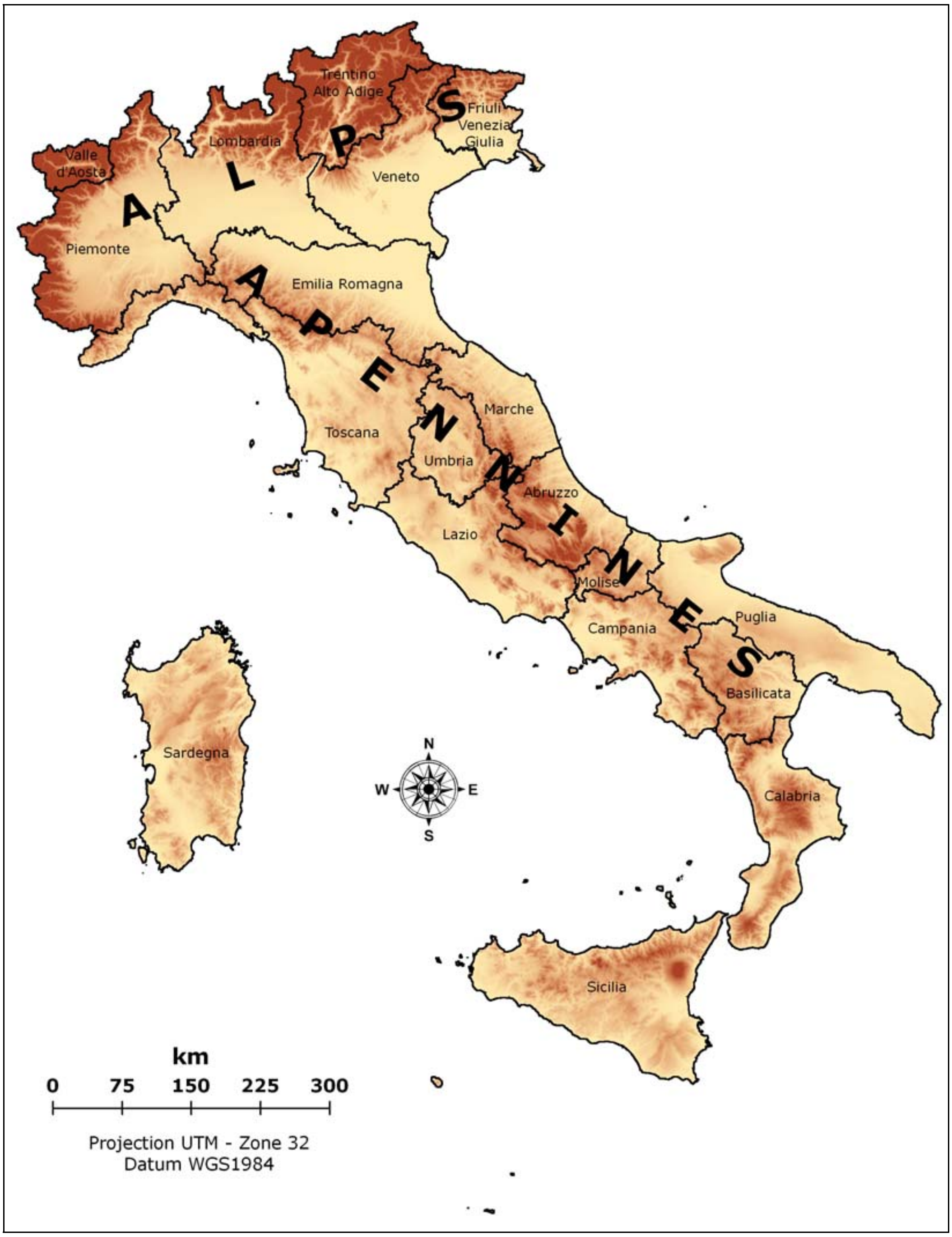

Figure 1. Map of the Italian peninsula.

To protect this rich biodiversity 1,004 protected areas have been established in Italy (Gambino and Negrini 2001). Altogether, I considered 777 protected areas for the analysis (Fig. 2). I omitted 17 protected areas (16 “Protected Marine Natural Areas and Marine Reserves” and 1 “Other National Protected Natural Area”) and the marine part of three National Parks (those of the Maddalena Archipelago, the island of Asinara and the Tuscan Archipelago) because the data available do not cover the marine realm, plus all the terrestrial protected areas for which it was not possible to find a paper and/or a digital map. The 777 protected areas accounts for 96.6% of the total area of the Italian protected areas system.

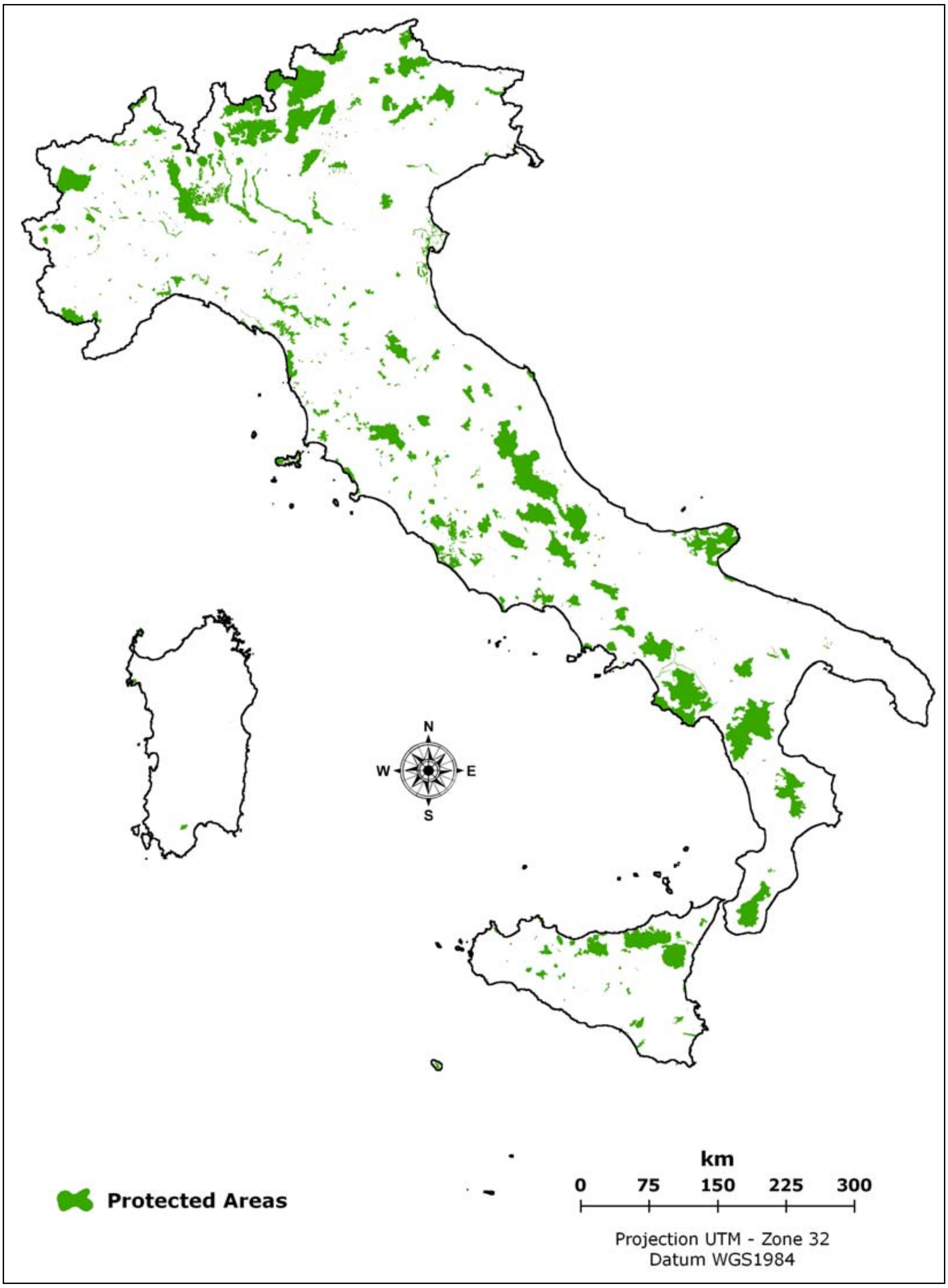

Figure 2. Protected areas in Italy.

The 777 protected areas that I considered cover more than 3.37 million hectares (area calculated using ArcGIS, thus there is not an exact correspondence with the areas declared by the Official List drawn up by the Ministry of the Environment), corresponding to more than 11% of the national territory. They are distributed extremely irregularly among the Italian administrative regions (Fig. 3): in some regions, 25% or more of the territory is protected (e.g. Abruzzo, Lombardia, Autonomous Province of Bolzano, Campania), whereas in others less than 5% of the territory is protected, as in the extreme cases of Molise (where 1.5% of the territory is protected), and Sardinia (less than 1% protected).

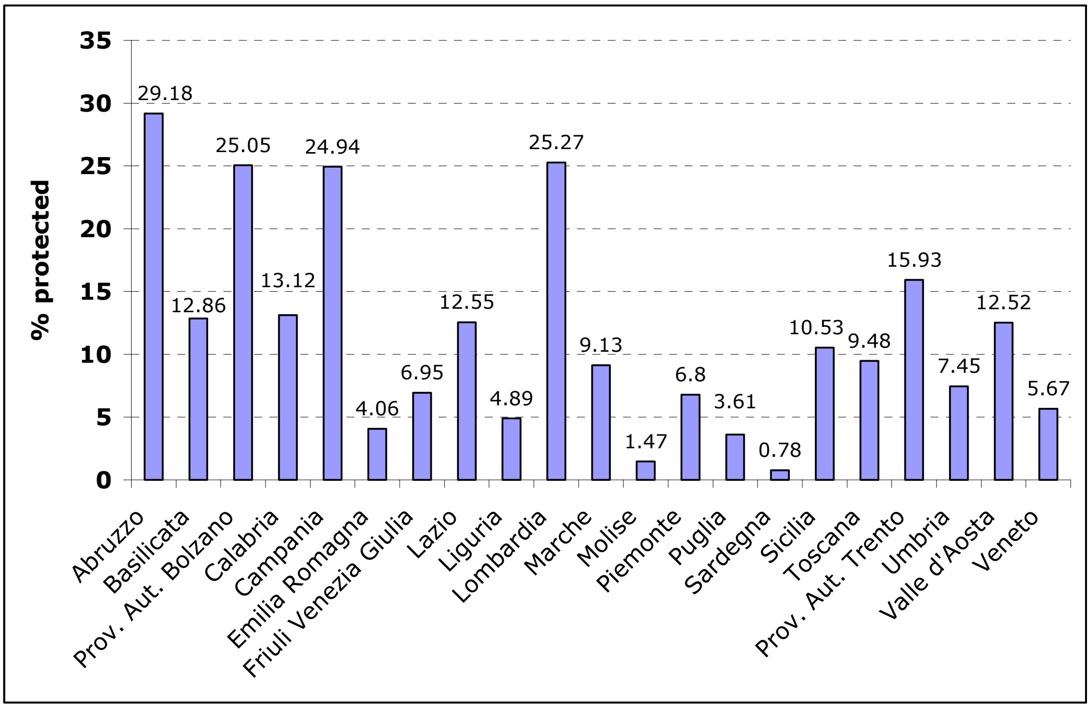

Figure 3. Percent coverage of the existing protected areas in the Italian administrative regions.

Protected areas have a mean size of 4352.5 ha, but the median size is only 265.4 ha. The discrepancy between mean and median area is due essentially to the considerable size of five National Parks, all with an area of over 100,000 ha. The biggest protected areas is the Pollino National Park that covers more than 190,000 ha (it should be noted that, according to the Official List drawn up the Ministry of the Environment, the largest park is the Cilento and Vallo di Diano National Park), while the smallest is the Sasso di Preguda Regional Natural Monument (Lombardia Region), which covers 0.05 ha. The entire distribution of sizes for protected areas is skewed towards smaller areas (Fig. 4): 70% of the areas cover less than 1,000 ha, 60% less than 500 ha, 33% less than 100 ha, and 9% less than 10 ha.

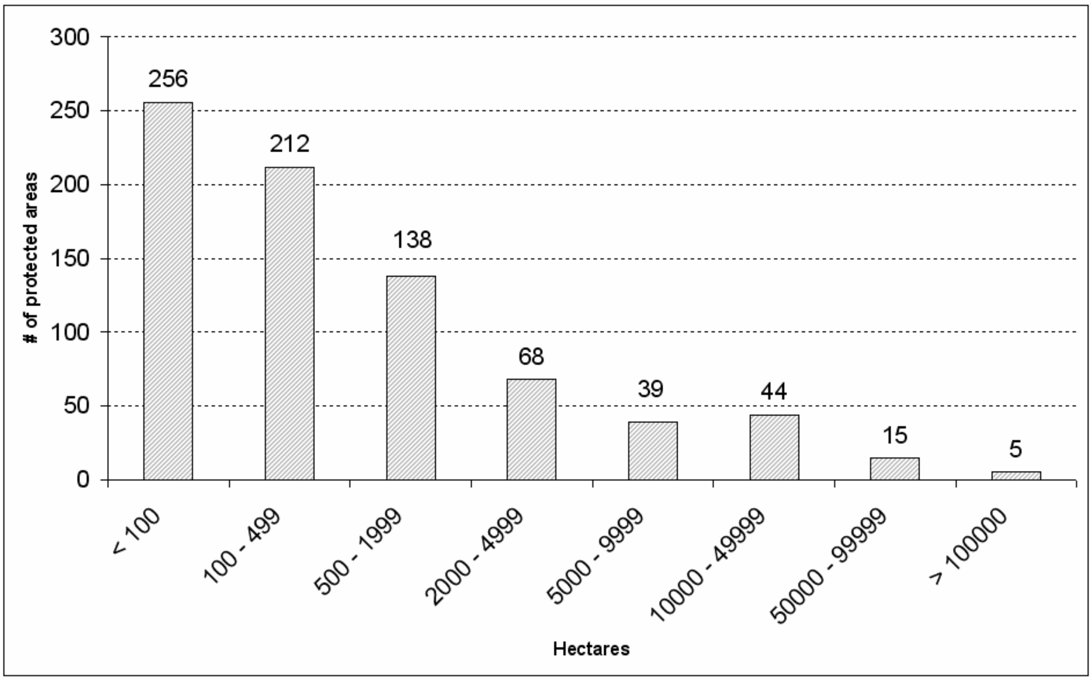

Figure 4. Number of protected areas per class area.

If we compare the elevation of the protected areas with the overall situation in Italy (Fig. 5), it can be seen that, in general, the elevation of most of the PA (median = 902 meters; mean = 1,017 meters) is greater than the mean value for Italy as a whole (median = 337 meters; mean = 535 meters), particularly for National Parks (median = 1,043 meters; mean = 1,157 meters) and Regional Parks (median = 971 meters; mean = 1,054 meters).

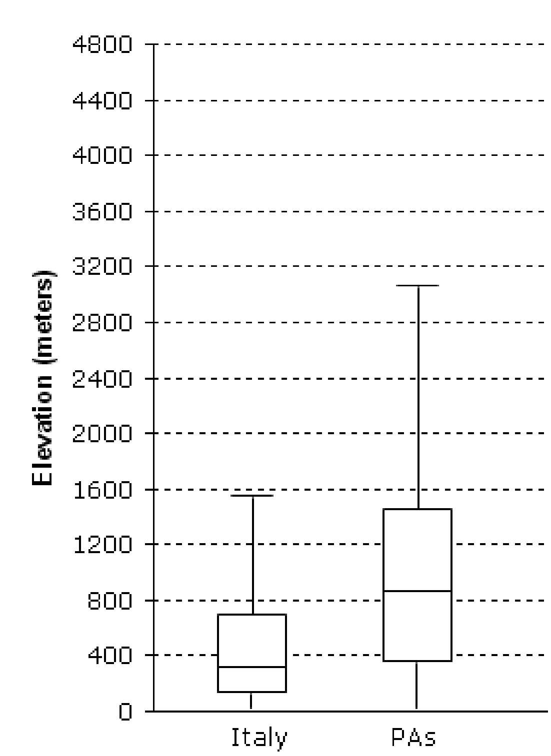

Figure 5. Distribution of the elevation values in Italy and in the protected areas.

Protected areas in Italy have a long history, going back at least to the Roman Empire, when some forests were protected for economic and/or religious reasons. The first "modern" protected area, the Gran Paradiso National Park, was established in Italy in 1922, followed in 1923 by the Abruzzo, Lazio and Molise National Park. Both the number and the surface protected (Fig. 6 and 7 respectively) remained mostly unchanged until the mid 1970s, when the number of protected areas started to increase more or less constantly.

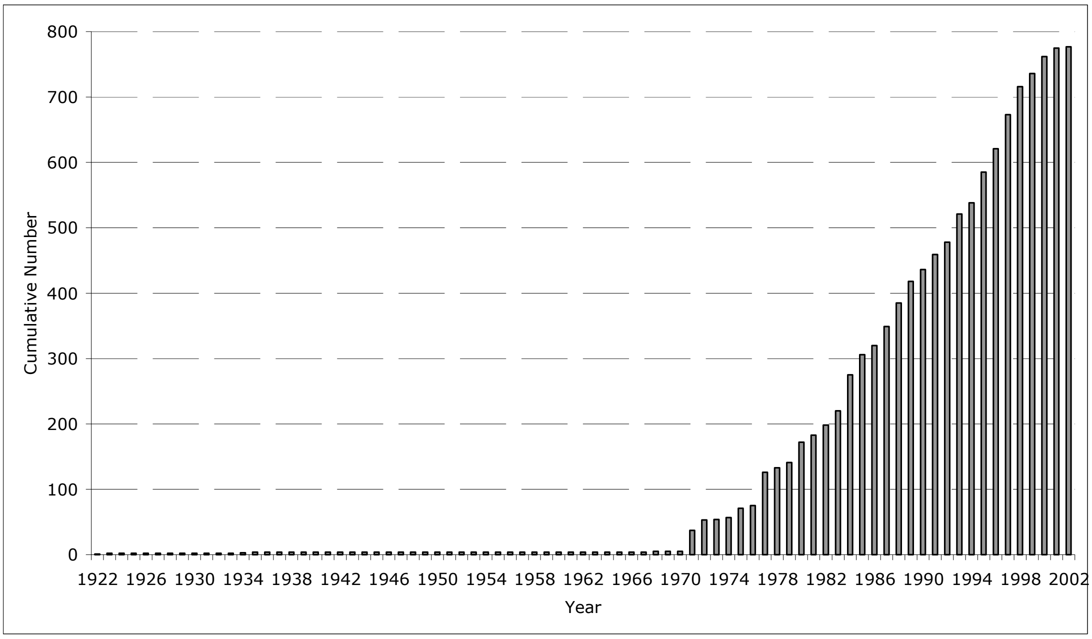

Figure 6. Cumulative growth (in number) of the terrestrial protected area system in Italy.

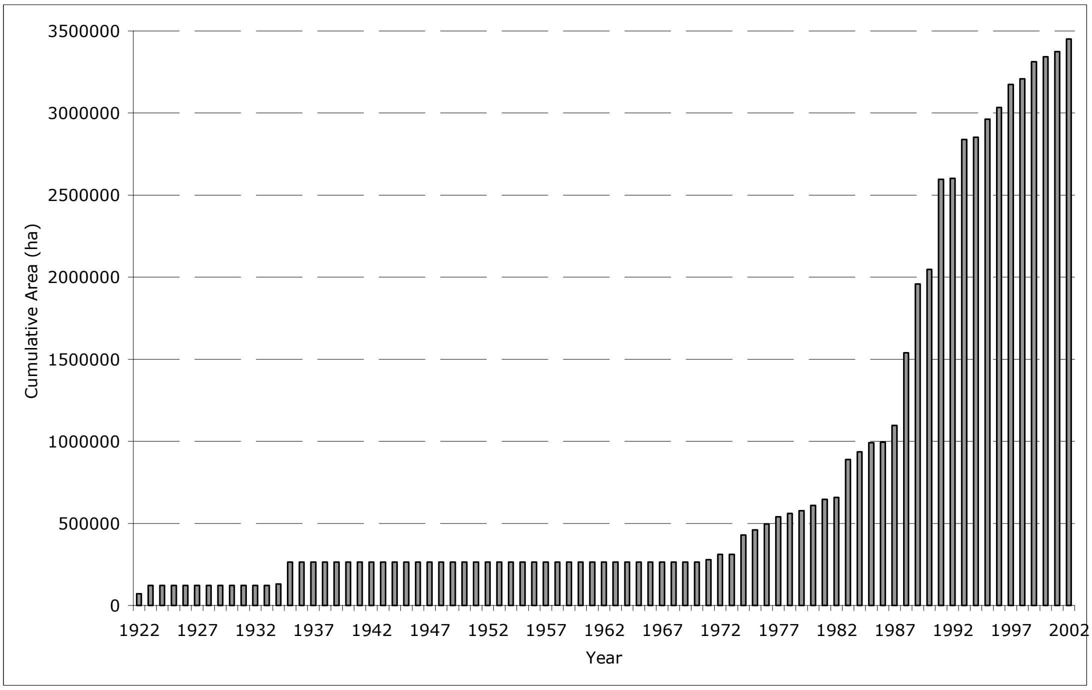

Figure 7. Cumulative growth (in hectares) of the terrestrial protected area system in Italy.

Besides the existing protected areas, the Italian government has established a Nature2000 Network of areas. The Nature2000 Network, as described by the Directive 92/43/CEE issued in 1992, is intended to be a series of areas with the function of protecting biodiversity in the territory of the European Community. When the network will be completed, through agreements to be reached between the European Union and the member countries, it will consist of two kinds of areas: SCI (Sites of European Community Importance) and SPA (Special Protection Areas). Both types are currently undergoing final verification by the European Commission (EC), and if they are approved they will represent the potential future for the growth of the system of protected areas in Italy. For the moment, the status of these areas is that of sites proposed and they are subject to transitory provisions providing a legal protection against alteration.

In Italy, the Nature2000 Network currently consists of 559 SPA and 2,255 SCI (data obtained from the Italian Ministry of the Environment, site accessed on 29 March 2006). The EC list in Italy 2286 SCIs (total area=44,979 km2) and 566 SPAs (with respectivelya total area of 44,979 km^2 and of 34,683 km^2; list available at: http://ec.europa.eu/environment/nature/nature_conservation/useful_info/barometer/index_en.htm, accessed on December 2006). At a regional level the process of individuation of new areas to be included in the network is still ongoing, and I obtained data on 554 SPAs (Fig. 8) and 2,255 SCIs (Fig. 9). For some of the areas it was not possible to obtain digital and/or paper data, while the areas located in the sea have not been considered.

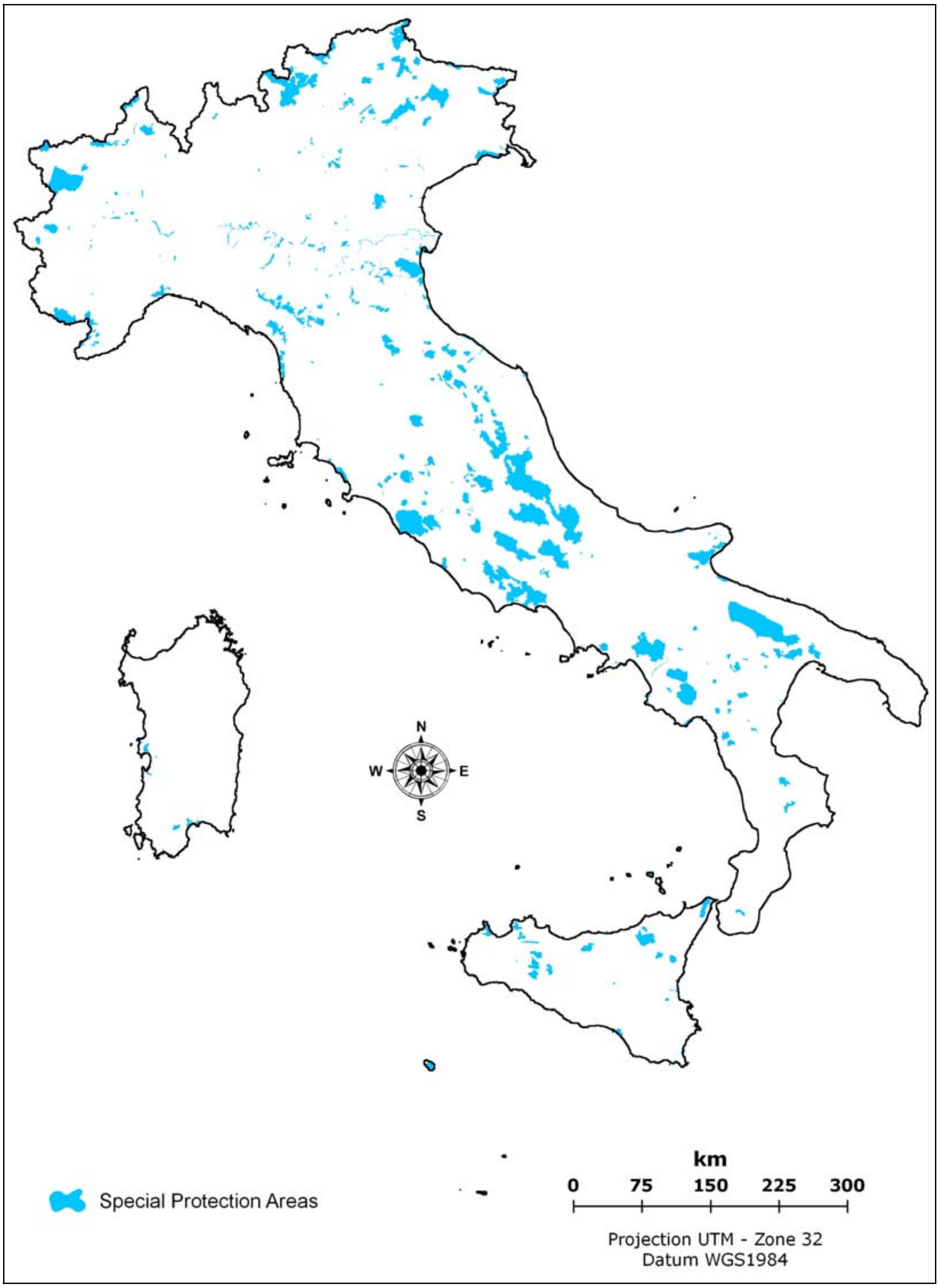

Figure 8. Special Protection Areas in Italy.

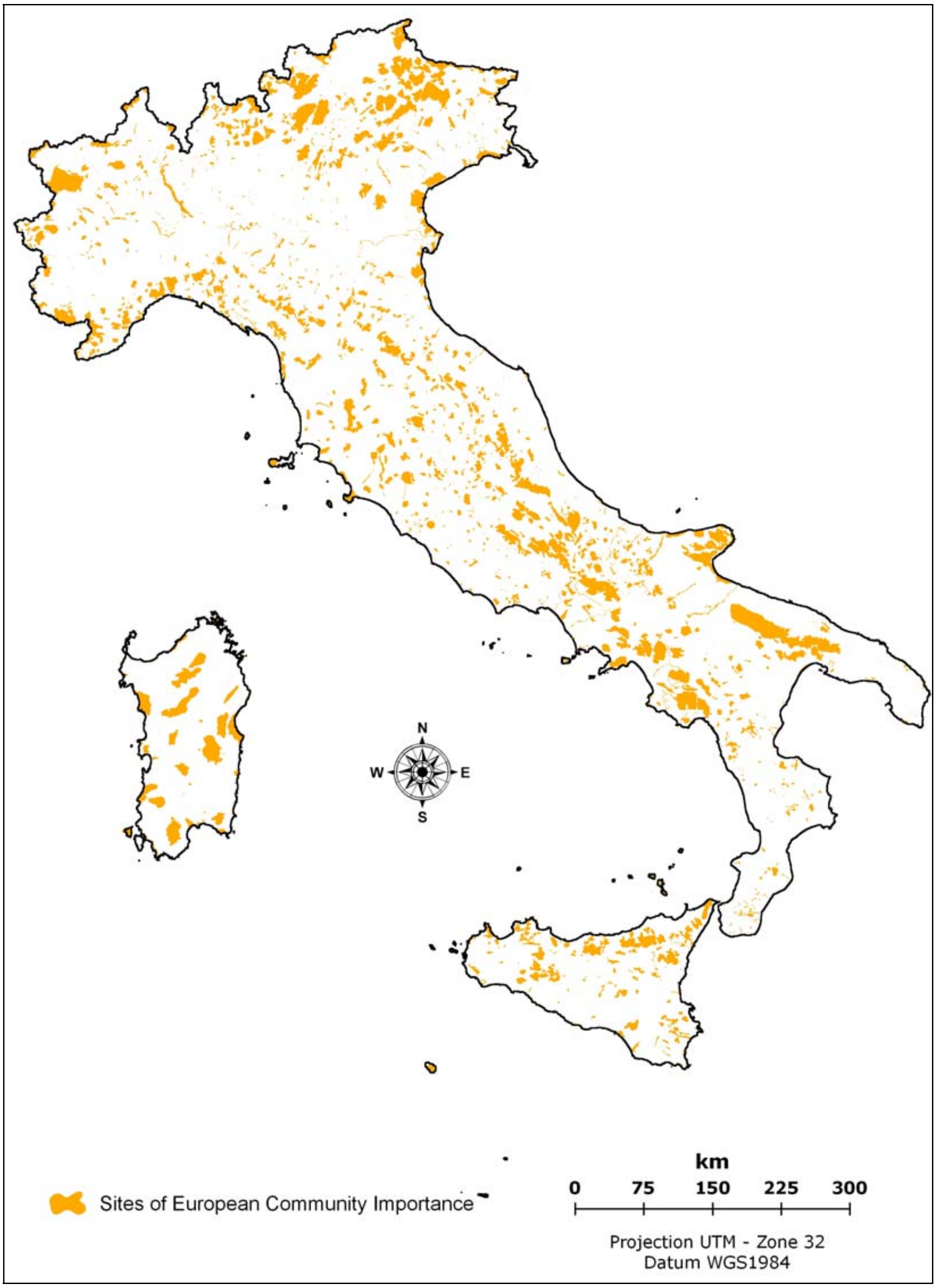

Figure 9. Sites of European Community Importance in Italy.

The 554 SPAs considered cover 31,442 km^2 (97.9% of the Italian SPAs), corresponding to 10.4% of the Italian peninsula. The 2,242 SCI considered cover 44,105 km^2 (98.6% of the Italian SCIs), corresponding to more than 14.6% of the national territory.

The mean area for the SPA is 5,420 ha, while the median area is 1,229 ha. The biggest SPA is the Murgia Alta Special Protection Area in southern Italy with more than 141,000 hectares, while the smallest cover just over 4 ha (Fig. 10). The mean area for SCI is 1,889 hectares and the median is 520 hectares. The biggest SCI is the Murgia Alta SCI (in the same area as the Murgia Alta SPA) with more than 127,000 hectares; the smallest is less than 1 hectare (Fig. 10).

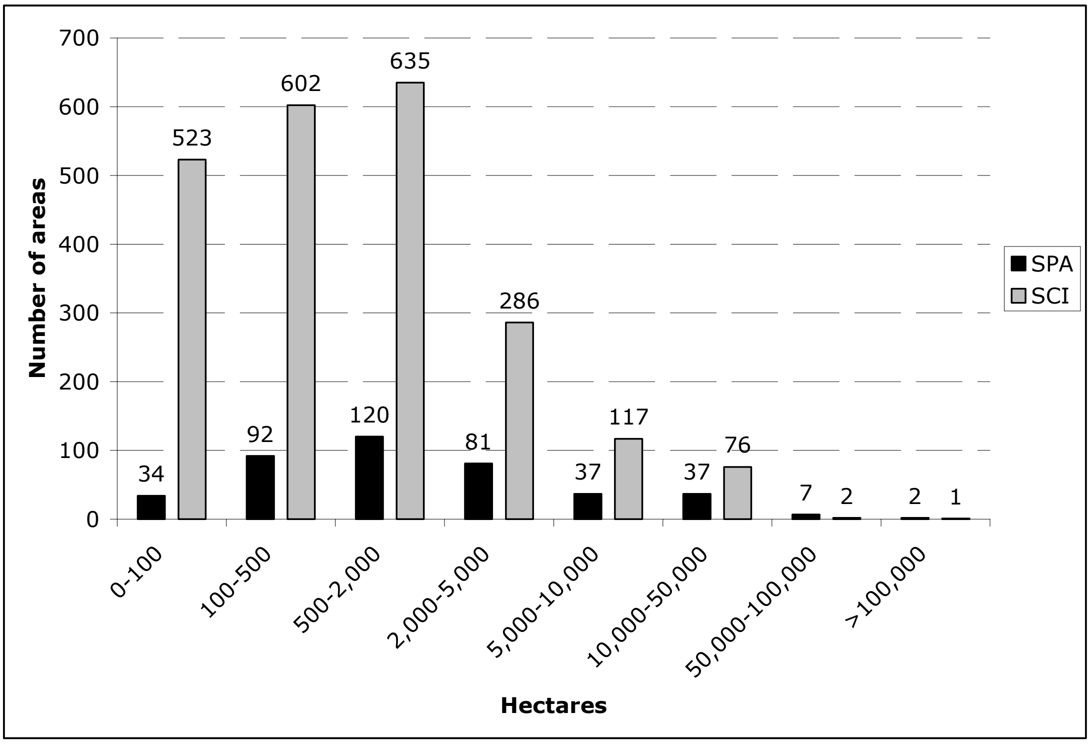

Figure 10. Number of SPA and SCI per class area.

Both SCI and SPA are distributed irregularly: in Emilia Romagna there are 75 SPA while in Molise the number of SPA is 2; in Sicily there are 205 SCI, while in Valle d'Aosta there are 26 SCI (Fig. 11). Also the area coverage of SCI and SPA is irregularly distributed: SPA cover more than 20% of the regional territory in Abruzzo and less than 1% in Molise; SCI cover more than 32% in Valle d'Aosta and less than 5% in Calabria (Fig. 12).

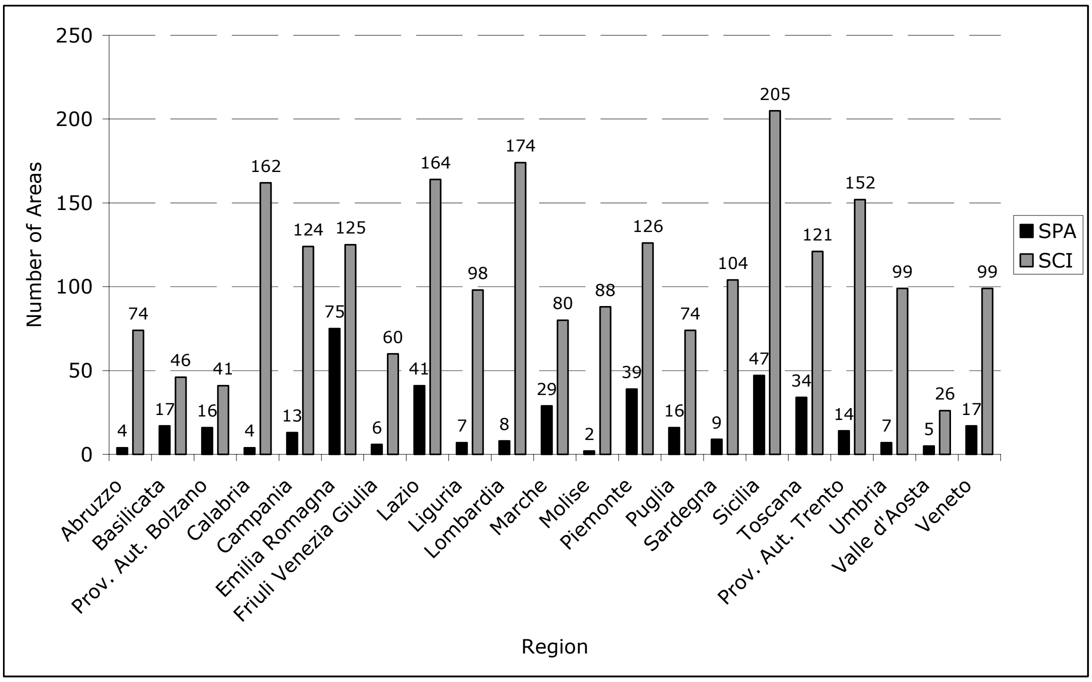

Figure 11. Number of SPA and SCI per administrative region.

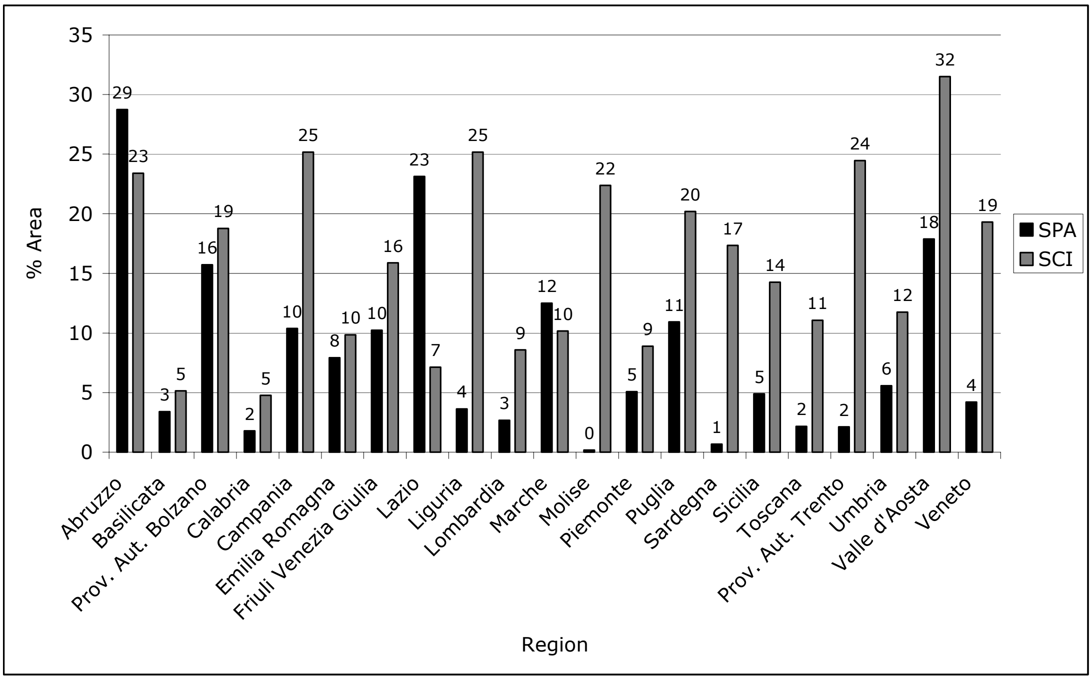

Figure 12. Percentage of area occupied by SPA and SCI per administrative region.

On average, the elevation of the Natura2000 network is higher than the mean for the peninsula (median = 337 meters; mean = 535 meters); SCI have a mean elevation of 948 meters (median = 787 meters) while SPA have a mean elevation of 1028 meters (median = 915 meters).

Even with their original differences (SPA have been identified specifically for the protection of bird species while SCI have been identified for the conservation of habitat and species in general) SCI and SPA cannot be considered separated. Often the same area has been identified as a SCI and as a SPA: 70% of the SPA area has also been designated as a SCI, and 36% of the SCI area has also been designated as a SPA. Considering the two types of area together, the Natura2000 Network covers more than 4.9 millions hectares, with a mean area of 2,598 hectares. The Natura2000 network is also extensively overlapped with the existing protected areas but, potentially, it would add 2.8 millions hectares of protected areas, making more than 20% of the Italian territory reserved for conservation.

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CHAPTER 3: GEOGRAPHIC DISTRIBUTION OF TERRESTRIAL VERTEBRATES IN ITALY

Introduction

Data on the distribution of terrestrial vertebrates represent the basic information that will be used and analyzed throughout this thesis. The most common type of distribution data that is usually available is in the form of extents of occurrence. However, extents of occurrence are often not available, incomplete, spatially biased towards areas of easier accessibility, or taxonomically biased towards flagship species or taxa (Polasky et al. 2000; Margules et al. 2002; Williams et al. 2002; Wilson et al. 2005). Moreover, species do not completely occupy their extent of occurrence (Gaston 1991), making them particularly prone to commission errors that can be particularly important in conservation planning (Fielding and Bell 1997; Loiselle et al. 2003; Rondinini et al. 2005; Wilson et al. 2005).

Besides these general problems, in my case study it was also impossible to obtain detailed distribution data for all the species considered: the extents of occurrence that were available for breeding birds and for some of the mammals were extremely more detailed than what was available for rodents, bats, amphibians, reptiles and freshwater fish. So I used distribution models to overcome some of these problems (Corsi et al. 2000; Guisan and Zimmerman 2000; Scott et al. 2002).

In this chapter I will describe the data collected on the distribution and ecology of vertebrate species in Italy and the procedure used to produce and validate the habitat suitability models that I will use. This chapter provides the detailed methodology; a shorter version is presented in the subsequent chapter that has been submitted to Biological Conservation.

Material and methods

In order to build the list of species that will be considered throughout this thesis, I obtained the complete list of terrestrial vertebrate species from the National Ecological Network project (Boitani et al. 2003a). Overall, I considered 102 mammals (Amori et al. 1999), 244 regularly breeding birds (Brichetti and Massa 1998), 43 reptiles (Societas Herpetologica Italica 1996), 34 amphibians (Societas Herpetologica Italica 1996) and 82 freshwater fish (Kottelat 1997; Bianco 1998), for a total 504 species. From these I excluded 8 freshwater fish species because their distribution is completely unknown and 35 species that have recently been introduced in the Italian peninsula (28 freshwater fish, 5 mammals, 1 breeding bird, and 1 amphibian). I retained in the analyses all the species introduced in historical times and now part of the naturally occurring Italian fauna (an example is the fallow deer *Dama dama* that was probably introduced by the Romans more than 2000 years ago [Boitani et al. 2003b]). This list is composed by 462 species (91.7% of the total 504): 97 mammals, 243 birds, 43 reptiles, 32 amphibians, and 47 freshwater fish, and it was used for the analyses described in Chapter 4.

In early 2006 the Atlas of Italian Amphibians and Reptiles (Sindaco et al. 2006) was published, updating all the information that has been previously collected for reptiles and amphibians. In particular, the species of amphibians that I considered following the new publications are 36, with the "new" species *Speleomantes strinatii*, *Bombina pachipus*, *Hyla intermedia*, *Rana bergeri et hispanica* COMPLEX (mainly resulting from splitting of old species). The species of reptiles that I considered following Sindaco et al. (2006) are 44, with new species *Podarcis raffonei*, *Tesdudo marginata*, and *Chalcides striatus*, and with the disappearance from the italian territory of the species *Hierophis gemonensis* and *Ophisaurus apodus*. This final list (available in Appendix I at the end of this chapter) is considered in all the analyses except for Chapter 4, and it is composed of

468 species: 98 mammals, 243 birds, 44 reptiles, 36 amphibians, and 47 freshwater fish.

For each species considered I obtained from the National Ecological Network project (Boitani et al. 2003a) the extent of occurrence (sensu Gaston 1991) plus habitat preferences. The original data were obtained from published literature and expert's opinions and the entire database is freely available at http://www.gisbau.uniroma1.it/ren.php. For Amphibians and Reptiles I modified both the EOO and the habitat preferences following Sindaco et al. (2006). I used the modified data in all the analyses except for those described in Chapter 4, where I used the original data.

To build the habitat suitability models I considered 4 environmental layers: land cover, elevation, distance to water and distance to roads. I chose the 4 layers considering the data on species-habitat relationships that were available and the availability of digital maps covering the entire national territory with the necessary spatial detail.

In particular, I used the CORINE 2000 Land Cover map (obtained in ArcInfo vector format from http://terestrial.eionet.eu.int/CLC2000), a digital elevation model (obtained from the Nature Conservation Directorate – Italian Ministry of the Environmtent in ArcInfo raster format with pixel size 75 meters), a street network (obtained from the Nature Conservation Directorate – Italian Ministry of the Environmtent in ArcInfo vector format, original scale 1:200,000), a hydrological network (obtained from the Nature Conservation Directorate – Italian Ministry of the Environmtent in ArcInfo vector format, original scale 1:25,000).

For subsequent analyses all the layers were transformed (if necessary) from vector to raster with a pixel size of 100 meters (the lowest possible resolution for the CORINE

Land Cover map) and an extent corresponding to that of the CORINE Land Cover; the DEM, already in raster format, was re-sampled to obtain a pixel size of 100 meters.

I used the species-habitat relationships and the available environmental layers to build deductive species distribution models (sensu Corsi et al. 2000) for 442 vertebrate species. I did not build a DM for 7 amphibians which live mainly in subterranean habitats that are not mapped on a national level, for 6 birds that are primarily marine and/or breeds on small islands, for 2 birds linked to habitats poorly mapped on a national level, for 4 mammals whose ecology is poorly known, and for 1 reptile, endemic of a small island. These 20 species were included in the analysis using their full EOs (see Appendix I for the species-specific details).

The species distribution modeling procedure involved 2 main steps: reclassification of the environmental layers into species-specific suitability ranks, and combination of the layers to obtain the final DM. In the first step, for each taxa, with the exception of fish, the CORINE Land Cover map was reclassified into 4 classes (land cover types non suitable for the presence of the species, land cover classes with low suitability, medium suitability, high suitability), the DEM was reclassified into 3 classes (elevation values where the species is not present, elevation values where the species can be present, elevation values that represent the optimum for the species), the distance to water was reclassified into 2 classes (1st class: from the water body or stream to the maximum distance to water tolerable for the species; 2nd class beyond the maximum distance to water tolerable for the species); the street network was reclassified into 2 classes (1st class: from the street to the minimum distance to street tolerable for the species; 2nd class: all values beyond the first class). The process was different for fish: the land cover was not considered but the different types of internal waters were reclassified into 3 suitability values and the DEM was reclassified into 3 classes.

In the second step, the 4 (2 for fish) reclassified layers were combined to produce the final DM. I used three different reclassification schemes: one for birds, for which the influence of the elevation has been considered smaller than for the other taxa, one for mammals, amphibians and reptiles, and one for fish.

The distance to streets has been introduced in the models on a species specific basis: the presence of roads in a pixel of the model decreased the suitability for a species negatively influenced by roads but it was non-important for the other species. Also distance to water was considered on a species-specific basis: DMs for water dependent species have been built only within a given distance to water.

All the DMs have been clipped using the species' EOs, allowing us to take into account historical constraints, complex disturbance regimes, and other non-ecological factors that influence species distribution (Morrison et al. 1998).

We tested the predictive power of 303 DMs (73.2% of the models) using points of presence (from 20 to 4392 points per species) independently collected during different projects (Fornasari et al. 2001; Ruffo & Stoch 2005; Abruzzo-Lazio-Molise National Park unpublished data; M.Masi & P.Ciucci unpublished data; A.Mortelliti unpublished data; G.Puddu unpublished data; W.Reggioni unpublished data; S.Sarrocco unpublished data). For each point of presence (and for each species tested) we built a circular buffer with a radius corresponding to the location error associated with each point (200m-3km). The points of presence and the DMs were considered to agree if a given percentage of area inside the buffer was classified as medium suitability or high suitability. To avoid the subjectivity of a pre-defined percentage, we considered a range of possibilities going from 1 cell inside the buffer classified as medium or high suitability to 20% of the area inside the buffer classified as medium or high suitability. Thus we calculated a range of percentages of agreement for each species tested.

We used a permutation test to test the significance of the agreement among points of presence and the DMs. We compared the percentage of agreement calculated for the point of presence with that obtained with 1000 sets of random points sharing the same characteristics as the set of points of presence (same number of points, same distribution of buffer sizes). If the percentage of agreement calculated for the points of presence was in the top 5% of the agreements obtained from the random samples, the percentage of agreement was significant and the model was considered as validated. We performed the same test for all the different percentages of high and medium suitability inside the buffer.

All 468 species were weighted for the analysis using their conservation value and their degree of vulnerability. I used 12 rules of international and national conventions, treaties and laws, and 6 published conservation-related indexes (Tab. 1) to assign species-specific weights. Conventions and laws were used to score each species on the number of times the species was listed and the total score was scaled between 0 and 100, 0 being assigned to species not considered in any convention or law and 100 being assigned to species considered in all the conventions and laws relative to each taxonomic group. The same operation was performed with the conservation-related indexes. The two scores were then summed and the total was rescaled from 0 to 100. The 118 species ranked with a final score greater or equal to 50 were considered to be of conservation interest.

	National Law 157/92 art. 1$^{(1)}$	National Law 157/92	EU Bird Directive 79/409/EEC Ann. 1	Berna Convention 1979 Ann. 2	Berna Convention 1979 Ann. 3	Washington Convention (CITES) App. A	Washington Convention (CITES) App. B	Bonn Convention 1979 Ann. 1	Bonn Convention 1979 Ann. 2	EU Habitat Directive 92/43/eec Ann. 2	EU Habitat Directive 92/43/EEC Ann. 4	Barcellona Convention App. 2	Endemic Taxon$^{(2)}$	Intrinsic Value$^{(3)}$	% of population/range$^{(4)}$	Trend in Italy$^{(4)}$	Threat Category $^{(5)}$	SPEC $^{(6)}$
Mammals	X	X		X	X	X	X		X	X	X		X		X	X	X	
Birds	X	X	X	X	X	X	X	X	X					X			X	X
Reptiles				X	X	X	X			X	X		X		X	X	X	
Amphibians				X	X	X	X			X	X		X		X	X	X	
Fish				X	X	X	X			X	X	X	X		X	X	X	

Table 1. List of variables considered for the classification into conservation categories of each taxa. (1) Special protected species; (2) Bulgarini et al. 1998, Amori et al. 1999, Societas Herpetologica Italiana 1996 , Gandolfi et al. 1991; (3) Brichetti and Gariboldi 1997; (4) Bulgarini et al. 1998; (5) Amori et al. 1999, Baillie et al. 2004, Bulgarini et al. 1998; (6) Birdlife 2004, Birds in Europe, available online at http://www.birdlife.org/action/science/species/birds_in_europe/index.html; Tucker and Heath 1994 has been used instead for the analyses described in Chapter 4.

Results and Discussion

Considering all taxa, we measured a concordance among points of presence and DMs that was greater than the average concordance among random points and DMs for a minimum of 89% and a maximum of 93% of the DMs. For most of the DMs (from 69% to 79%) the difference was significant at the 0.05 level.

Each taxon showed a relatively high percentage of DMs agreeing with presence points better than random, with a minimum of 83.3% for mammals and a maximum of 100% for amphibians (Tab. 2). However, the percentage of DMs with a significant difference varied from a minimum of 51.4% for mammals (the taxon with the lowest number of presence points available for validation) to a maximum of 85.9% for birds (the taxon with the highest number of presence points available for validation).

Taxon	**Better than random**	**Significant**	**# points**
Mammals	83.3 - 88.9	51.4 - 69.5	76 (168)
Birds	93.6 - 94.9	78.9 - 85.9	145 (517)
Reptiles	92.0 - 96.0	64.0 - 72.0	105 (236)
Amphibians	90.5 - 100	57.1 - 80.9	123 (233)
Fish	86.2 - 89.7	68.9 - 79.3	127 (299)

Table 2. Results of the validation of DMs for individual taxonomic groups. "Better than random" reports the percentages (minimum – maximum) of DMs that showed an agreement with points of presence which was higher than that obtained with random points. "Significant" reports the percentage (minimum – maximum) of DMs for which the agreement with points of presence was significantly better than random. "# points" reports the median number of points of presence together with the interquartile range in parentheses.

I weighted each species with a conservation value obtained using biological and socio-political criteria and the final ranking seems to correctly reflect the relative conservation value of each species. Endemic species such as *Pelobates fuscus*, *Euproctus platycephalus*, *Podarcis wagleriana*, *Vipera ursinii*, *Lampetra zanandreai*, *Rupicapra pyrenaica ornata* had the highest conservation values and species like *Anguilla anguilla*, *Cyprinus carpio*, and *Rattus norvegicus* had the lowest. However, some species appeared in the ranking with a relative value which was higher or lower than expected. The wild-boar (*Sus scrofa*), usually considered a pest, was classified in the central part of the ranking because there is the possibility of an endemic sub-species (*Sus scrofa mediterraneus*) still surviving in Sardinia (Boitani et al. 2003b). Conversely species such as *Lepus corsicanus*, which was thought to be extinct and which has been rediscovered only very recently (Pierpaoli et al. 1999; Riga et al. 2001), are not considered in many conservation treaties or in the IUCN red lists, making this a low-ranked species. For the detailed results on the index of conservation interest see the Appendix I.

In all the analyses I used a combination of distribution models and of extents of occurrence. To my knowledge, all studies on reserve selection and gap analysis that focus on species distribution as a surrogate for the biodiversity of a region have used raw distribution data in the form of points of presence and/or extents of occurrence (van Jaarsveld et al. 1998; Araujo 1999; Rodrigues et al. 1999; Polasky et al. 2000; Dimitrakopoulos et al. 2004; Rodrigues et al. 2004a, b; Yip et al. 2004; Solymos and Fehér 2005) or distribution models (Scott et al. 1993; Clark and Slusher 2000; Allen et al. 2001; Loiselle et al. 2003; Rondinini et al. 2005; Wilson et al. 2005).

However, both distribution models and extents of occurrence have intrinsic limitations and they cannot be used for all the species at the same way. Some species have small and well known distribution ranges that models cannot improve: for example, *Larus audouinii* in Italy breeds only on a few small islands and even the exact number of birds breeding on each island is known (Serra et al. 2001). Other species have a narrow distribution and are limited to particular habitat types that are not mapped on traditional land-use maps (e.g., *Proteus anguinus* is limited to caves in North-eastern Italy).

In the same way distribution models cannot be developped with success for some species, especially for those whose presence is constrained by micro-habitat characteristics that are not mappable on a small scale. For these species (an example in Italy is *Sorex araneus*) there is no way of building a distribution map over large areas and a model that poorly reflects reality may result in misleading and/or harmful conservation and management actions (Loiselle et al. 2003; Wilson et al. 2005).

Despite these problems, conservation planning cannot be delayed till complete biodiversity surveys or data to model distribution of a given species are available, or options for conservation would be dramatically reduced (Margules and Pressey 2000). Therefore, I argue that distribution models and extents of occurrence can be integrated successfully: the former should be thought of as a finer resolution version of the latter, practically representing a way of moving from extent of occurrence towards areas of occupancies (Gaston 1991).

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APPENDIX I: list of the Italian terrestrial vertebrates considered in the analyses.

In the following table I report all the 504 species of terrestrial vertebrates in Italy. Those species excluded from the analyses have 0 in the field "Model/EOO". Those species included in the analyses using their habitat suitability model have 1 in the field "Model/EOO". Those species included in the analyses using their extent of occurrence have 2 in the field "Model/EOO". The index of Conservation Interest goes from 0 (no conservation interest) to 100 (highest conservation interest); for the details of how it has been calculated see the chapter above.

CLASS CEPHALASPIDOMORPHA

Scientific name	Model/EOO	Conservation Interest
Petromyzon marinus	0	--
Lampetra planeri	1	48.38
Lethenteron zanandreai	1	81.69
Lampetra fluviatilis	0	--

CLASS ACTINOPTERYGII

Scientific name	Model/EOO	Conservation Interest
Acipenser sturio	0	--
Acipenser naccarii	1	67.59
Huso huso	0	--
Alosa agone (migratory eco-phenotype)	1	34.67
Alosa agone (landlocked eco-phenotype)	1	76.00
Salmo trutta	1	18.00
Salmo marmoratus	1	55.52
Salmo carpio	1	80.00
Salmo fibreni	1	80.00
Oncorhynchus mykiss	0	--
Salvelinus alpinus	1	39.33
Salvelinus fontinalis	0	--
Thymallus thymallus	1	38.19
Coregonus fera	0	--
Coregonus macrophthalmus	0	--
Esox lucius	1	31.33

Rutilus aula	1	0.00
Rutilus rubilio	1	73.69
Rutilus pigus	1	47.71
Rutilus rutilus	0	--
Pachychilon pictum	0	--
Abramis brama	0	--
Abramis bjoerkna	0	--
Leuciscus cephalus	1	0.00
Leuciscus lucumonis	1	33.71
Telestes muticellus	1	50.36
Telestes agassii	0	--
Scardinius scardafa	1	40.00
Scardinius erythrophthalmus	1	0.00
Chondrostoma genei	1	77.69
Chondrostoma soetta	1	73.71
Chondrostoma nasus	0	--
Alburnus arborella	1	0.00
Alburnus albidus	1	77.69
Phoxinus phoxinus	1	35.33
Rhodeus amarus	0	--
Gobio benacensis	1	0.00
Gobio gobio	0	--
Pseudorasbora parva	0	--
Barbus plebejus	1	50.38
Barbus tyberinus	1	8.00
Barbus caninus	1	44.38
Barbus barbus	0	--
Barbus comizo	0	--
Cyprinus carpio	1	0.00
Carassius auratus	0	--
Tinca tinca	1	0.00
Cobitis bilineata	1	37.02
Sabanejewia larvata	1	77.71
Barbatula barbatula	1	0.00
Misgurnus fossilis	0	--
Ictalurus melas	0	--
Ameiurus nebulosus	0	--
Ictalurus punctatus	0	--
Silurus glanis	0	--
Anguilla anguilla	1	0.00
Lota lota	0	--
Gasterosteus aculeatus	1	35.33
Atherina boyeri	1	0.00
Odonthestes bonariensis	0	--
Aphanius fasciatus	1	50.57
Gambusia holbrooki	0	--
Perca fluviatilis	1	0.00
Gymnocephalus cernuus	0	--

Sander lucioperca	0	--
Micropterus salmoides	0	--
Lepomis gibbosus	0	--
Salaria fluviatilis	1	45.33
Padogobius bonelli	1	51.46
Padogobius nigricans	1	78.86
Knipowitschia panizzae	1	27.46
Knipowitschia punctatissima	1	76.00
Cottus gobio	1	38.19
Salmo cettii	1	0.00
Pomatoschistus canestrinii	1	53.71
Ctenopharyngodon idellus	0	--
Hypophthalmichthys molitrix	0	--
Aristichthys nobilis	0	--
Clarias sp.	0	--

CLASS AMPHIBIA

Scientific name	Model/EOO	Conservation Interest
Proteus anguinus	2	49.31
Salamandra salamandra	1	33.33
Salamandra atra	1	67.93
Salamandra lanzai	1	55.33
Salamandrina terdigitata	1	70.67
Triturus carnifex	1	18.00
Triturus alpestris	1	33.33
Triturus vulgaris	1	20.67
Triturus italicus	1	70.67
Euproctus platycephalus	1	86.64
Speleomantes italicus	2	70.67
Speleomantes ambrosii	2	78.00
Speleomantes genei	2	78.00
Speleomantes strinatii	2	70.67
Speleomantes imperialis	2	78.00
Speleomantes flavus	2	78.00
Speleomantes supramontis	2	78.00
Discoglossus pictus	1	51.31
Discoglossus sardus	1	47.98
Bombina variegata	1	57.91
Bombina pachipus	1	67.91
Pelobates fuscus	1	93.33
Pelodytes punctatus	1	46.67
Bufo bufo	1	3.33
Bufo viridis	1	6.67
Hyla arborea	1	21.33
Hyla intermedia	1	41.33
Hyla meridionalis	1	42.00
Hyla sarda	1	28.00
Rana lessonae et *esculenta* COMPLEX	1	6.67

Rana bergeri et hispanica COMPLEX	1	26.67
Rana ridibunda	1	38.67
Rana temporaria	1	11.33
Rana dalmatina	1	6.67
Rana latastei	1	62.64
Rana italica	1	70.67
Rana catesbeiana	0	--

CLASS REPTILIA

Scientific name	Model/EOO	Conservation Interest
Emys orbicularis	1	44.67
Testudo hermanni	1	55.98
Testudo marginata	1	55.98
Tarentola mauritanica	1	3.33
Cyrtodactylus kotschyi	1	22.00
Hemidactylus turcicus	1	3.33
Phyllodactylus europaeus	1	51.98
Archaeolacerta bedriagae	1	48.67
Lacerta horvathi	1	18.00
Timon lepidus	1	42.67
Lacerta agilis	1	46.00
Lacerta viridis + bilineata	1	6.67
Zootoca vivipara	1	34.00
Podarcis melisellensis	1	18.00
Podarcis muralis	1	6.67
Podarcis sicula	1	36.67
Podarcis wagleriana	1	86.64
Podarcis raffonei	1	86.64
Podarcis tiliguerta	1	6.67
Podarcis filfolensis	1	66.67
Algyroides fitzingeri	1	61.98
Algyroides nigropunctatus	1	18.00
Psammodromus algirus	2	42.67
Chalcides chalcides	1	3.33
Chalcides striatus	1	3.33
Chalcides ocellatus	1	6.67
Anguis fragilis	1	3.33
Coluber hippocrepis	1	50.00
Coluber viridiflavus	1	6.67
Coronella austriaca	1	6.67
Coronella girondica	1	34.00
Elaphe longissima	1	6.67
Elaphe quatuorlineata	1	24.67
Elaphe situla	1	44.67
Natrix maura	1	3.33
Natrix natrix	1	36.67
Natrix tessellata	1	6.67
Macroprotodon cucullatus	1	42.67

Malpolon monspessulanus	1	23.33
Telescopus fallax	1	42.00
Vipera ammodytes	1	45.33
Vipera aspis	1	3.33
Vipera berus	1	3.33
Vipera ursinii	1	85.33

CLASS AVES

Scientific name	Model/EOO	Conservation Interest
Accipiter gentilis	1	49.52
Accipiter nisus	1	39.27
Acrocephalus arundinaceus	2	21.72
Acrocephalus melanopogon	1	41.19
Acrocephalus palustris	2	25.53
Acrocephalus schoenobaenus	1	58.14
Acrocephalus scirpaceus	1	26.47
Actitis hypoleucos	1	51.34
Aegithalos caudatus	1	27.02
Aegolius funereus	1	39.30
Alauda arvensis	1	27.64
Alcedo atthis	2	45.68
Alectoris barbara	1	52.89
Alectoris graeca	1	55.47
Alectoris rufa	1	47.98
Anas clypeata	1	56.57
Anas crecca	1	43.69
Anas platyrhynchos	1	19.18
Anas querquedula	1	50.91
Anas strepera	1	64.01
Anthus campestris	1	34.46
Anthus spinoletta	1	18.53
Anthus trivialis	1	16.27
Apus apus	2	17.02
Apus melba	1	29.28
Apus pallidus	1	29.47
Aquila chrysaetos	1	57.18
Ardea cinerea	1	29.56
Ardea purpurea	2	51.04
Ardeola ralloides	1	53.60
Asio otus	1	34.74
Athene noctua	1	36.49
Aythya ferina	1	56.03
Aythya fuligula	1	61.10
Aythya nyroca	1	84.80
Bonasa bonasia	1	35.25
Botaurus stellaris	1	65.83
Bubo bubo	1	58.59
Bubulcus ibis	1	37.47

Burhinus oedicnemus	1	61.66
Buteo buteo	2	40.30
Calandrella brachydactyla	2	33.52
Calonectris diomedea	2	60.97
Caprimulgus europaeus	1	50.24
Carduelis cannabina	1	35.30
Carduelis carduelis	1	13.83
Carduelis chloris	1	21.11
Carduelis flammea	2	19.18
Carduelis spinus	1	41.00
Certhia brachydactyla	1	24.31
Certhia familiaris	2	23.15
Cettia cetti	2	18.15
Charadrius alexandrinus	1	46.53
Charadrius dubius	2	31.96
Charadrius morinellus	1	57.82
Chlidonias hybridus	2	60.29
Chlidonias leucopterus	1	52.45
Chlidonias niger	1	66.42
Ciconia ciconia	1	58.00
Ciconia nigra	1	51.81
Cinclus cinclus	1	37.10
Circaetus gallicus	1	64.72
Circus aeruginosus	2	53.16
Circus pygargus	1	50.55
Cisticola juncidis	1	17.68
Clamator glandarius	1	49.07
Coccothraustes coccothraustes	1	31.25
Columba livia	1	36.35
Columba oenas	1	55.70
Columba palumbus	1	16.21
Coracias garrulus	2	68.85
Corvus corax	2	27.49
Corvus corone	1	7.61
Corvus monedula	2	18.33
Coturnix coturnix	1	39.80
Crex crex	1	75.19
Cuculus canorus	1	16.55
Cygnus olor	0	--
Delichon urbica	2	27.92
Dryocopus martius	1	28.84
Egretta alba	1	29.39
Egretta garzetta	1	25.06
Emberiza cia	1	32.06
Emberiza cirlus	1	24.02
Emberiza citrinella	1	22.99
Emberiza hortulana	1	51.74
Emberiza melanocephala	1	50.28

Emberiza schoeniclus	1	22.28
Erithacus rubecula	1	21.58
Falco biarmicus	1	65.75
Falco eleonorae	2	68.88
Falco naumanni	1	73.25
Falco peregrinus	1	46.42
Falco subbuteo	2	41.90
Falco tinnunculus	1	37.71
Falco vespertinus	1	41.09
Ficedula albicollis	1	43.71
Fringilla coelebs	1	21.11
Fulica atra	1	19.18
Galerida cristata	1	34.15
Gallinago gallinago	2	37.60
Gallinula chloropus	2	13.49
Garrulus glandarius	1	11.65
Gelochelidon nilotica	1	61.04
Glareola pratincola	1	62.50
Glaucidium passerinum	1	45.81
Gyps fulvus	1	58.79
Haematopus ostralegus	1	53.80
Hieraaetus fasciatus	1	74.51
Himantopus himantopus	2	37.18
Hippolais polyglotta	2	24.02
Hirundo daurica	1	47.57
Hirundo rustica	1	28.02
Hydrobates pelagicus	2	50.26
Ixobrychus minutus	2	47.00
Jynx torquilla	1	31.03
Lagopus mutus	1	37.47
Lanius collurio	2	34.27
Lanius minor	1	65.03
Lanius senator	2	54.81
Larus audouinii	2	78.34
Larus cachinnans	2	24.40
Larus genei	1	66.88
Larus melanocephalus	1	54.69
Larus ridibundus	1	44.19
Limosa limosa	1	70.88
Locustella luscinioides	1	43.72
Loxia curvirostra	1	24.84
Lullula arborea	1	39.40
Luscinia megarhynchos	1	21.20
Melanocorypha calandra	1	46.62
Merops apiaster	1	34.09
Miliaria calandra	1	36.99
Milvus migrans	1	54.83
Milvus milvus	1	73.60

Monticola saxatilis	1	43.56
Monticola solitarius	2	33.00
Montifringilla nivalis	1	29.84
Motacilla alba	2	16.93
Motacilla cinerea	2	17.77
Motacilla flava	2	18.62
Muscicapa striata	1	31.08
Neophron percnopterus	1	73.19
Netta rufina	1	52.02
Nucifraga caryocatactes	1	20.41
Nycticorax nycticorax	1	36.15
Oenanthe hispanica	2	57.73
Oenanthe oenanthe	1	29.52
Oriolus oriolus	2	15.61
Otus scops	1	53.30
Panurus biarmicus	1	32.75
Parus ater	1	15.89
Parus caeruleus	1	24.31
Parus cristatus	1	37.28
Parus major	1	13.83
Parus montanus	1	24.47
Parus palustris	1	30.27
Passer domesticus	2	24.58
Passer hispaniolensis	1	17.87
Passer italiae	1	9.68
Passer montanus	1	25.76
Perdix perdix	1	45.91
Pernis apivorus	1	49.42
Petronia petronia	1	20.03
Phalacrocorax aristotelis	2	44.84
Phalacrocorax carbo	1	43.13
Phalacrocorax pygmeus	1	68.76
Phasianus colchicus	1	10.86
Phoenicopterus ruber	1	47.07
Phoenicurus ochruros	2	16.08
Phoenicurus phoenicurus	1	17.49
Phylloscopus bonelli	1	36.99
Phylloscopus collybita	1	16.18
Phylloscopus sibilatrix	2	36.24
Pica pica	1	9.75
Picoides leucotos	1	47.51
Picoides major	2	27.77
Picoides medius	1	47.16
Picoides minor	1	30.59
Picoides tridactylus	1	58.50
Picus canus	1	52.09
Picus viridis	1	48.40
Platalea leucorodia	1	53.13

Plegadis falcinellus	1	66.04
Podiceps cristatus	1	16.93
Porphyrio porphyrio	1	54.16
Porzana parva	1	59.98
Porzana porzana	1	53.94
Prunella collaris	1	18.71
Prunella modularis	1	23.84
Ptyonoprogne rupestris	2	18.62
Puffinus yelkouan	2	58.20
Pyrrhocorax graculus	2	29.18
Pyrrhocorax pyrrhocorax	1	57.41
Pyrrhula phyrrula	1	17.68
Rallus acquaticus	1	27.16
Recurvirostra avosetta	1	51.09
Regulus ignicapillus	1	24.49
Regulus regulus	1	23.27
Remiz pendulinus	2	18.81
Riparia riparia	1	33.75
Saxicola rubetra	2	26.47
Saxicola torquata	1	15.99
Scolopax rusticola	1	55.16
Serinus citrinella	2	37.78
Serinus serinus	1	21.39
Sitta europaea	1	16.74
Sterna albifrons	1	57.97
Sterna bengalensis	1	51.42
Sterna hirundo	1	36.75
Sterna sandvicensis	1	61.16
Streptopelia decaocto	1	12.04
Streptopelia turtur	1	26.09
Strix aluco	1	29.77
Strix uralensis	1	32.59
Sturnus unicolor	1	24.40
Sturnus vulgaris	1	22.04
Sylvia atricapilla	1	20.36
Sylvia borin	2	24.02
Sylvia cantillans	1	26.00
Sylvia communis	2	24.68
Sylvia conspicillata	2	21.91
Sylvia curruca	1	19.00
Sylvia hortensis	1	55.35
Sylvia melanocephala	1	23.84
Sylvia nisoria	1	39.43
Sylvia sarda	1	42.63
Sylvia undata	2	43.25
Tachybaptus ruficollis	2	18.53
Tadorna tadorna	1	49.95
Tetrao tetrix	1	34.22

Tetrao urogallus	1	42.60
Tetrax tetrax	1	77.63
Tichodroma muraria	2	30.31
Tringa totanus	1	66.26
Troglodytes troglodytes	1	15.99
Turdus merula	1	15.89
Turdus philomelos	1	20.31
Turdus pilaris	1	14.53
Turdus torquatus	1	25.90
Turdus viscivorus	1	25.81
Tyto alba	1	47.05
Upupa epops	1	30.65
Vanellus vanellus	1	38.97

CLASS MAMMALIA

Scientific name	Model/EOO	Conservation Interest
Erinaceus europaeus	1	23.00
Erinaceus concolor	1	13.83
Sorex minutus	1	23.00
Sorex araneus	2	23.00
Sorex samniticus	1	53.00
Sorex alpinus	1	13.00
Neomys fodiens	1	13.00
Neomys anomalus	1	13.00
Suncus etruscus	1	23.00
Crocidura leucodon	1	23.00
Crocidura suaveolens	1	23.00
Crocidura russula	1	57.00
Crocidura sicula	1	57.00
Talpa europaea	1	8.00
Talpa romana	1	48.00
Talpa caeca	1	18.00
Rhinolophus ferrumequinum	1	41.33
Rhinolophus hipposideros	1	48.67
Rhinolophus euryale	1	44.67
Rhinolophus mehelyi	2	44.67
Rhinolophus blasii	2	56.64
Myotis mystacinus	1	39.33
Myotis brandti	2	15.33
Myotis emarginatus	1	44.67
Myotis nattereri	1	43.33
Myotis bechsteini	1	48.67
Myotis myotis	1	41.33
Myotis blythi	1	38.67
Myotis daubentoni	1	39.33
Myotis capaccinii	1	48.67
Myotis dasycneme	2	15.33
Pipistrellus pipistrellus	1	19.33

Pipistrellus nathusii	1	26.67
Pipistrellus kuhli	1	19.33
Nyctalus leisleri	1	42.67
Nyctalus noctula	1	39.33
Nyctalus lasiopterus	1	46.67
Hypsugo savii	1	19.33
Amblyotus nilssonii	1	15.33
Eptesicus serotinus	1	19.33
Vespertilio murinus	1	12.00
Barbastella barbastellus	1	45.33
Plecotus auritus	1	19.33
Plecotus austriacus	1	19.33
Miniopterus schreibersi	1	21.33
Tadarida teniotis	1	19.33
Oryctolagus cuniculus	1	46.00
Lepus timidus	1	10.50
Lepus europaeus	1	30.00
Lepus capensis	1	70.50
Sylvilagus floridanus	0	--
Sciurus vulgaris	1	46.33
Sciurus carolinensis	0	--
Marmota marmota	1	15.50
Eliomys quercinus	1	65.00
Dryomys nitedula	1	33.50
Glis glis	1	27.00
Muscardinus avellanarius	1	61.50
Clethrionomys glareolus	1	18.00
Arvicola terrestris	1	18.00
Microtus subterraneus	1	8.00
Microtus multiplex	1	18.00
Microtus savii	1	48.00
Microtus arvalis	1	8.00
Microtus agrestis	1	8.00
Chionomys nivalis	1	18.00
Ondatra zibethicus	0	--
Apodemus agrarius	1	8.00
Apodemus flavicollis	1	18.00
Apodemus sylvaticus	1	18.00
Apodemus alpicola	1	4.00
Micromys minutus	1	31.33
Rattus rattus	1	8.00
Rattus norvegicus	1	8.00
Mus domesticus	1	8.00
Hystrix cristata	1	25.50
Myocastor coypus	0	--
Canis lupus	1	42.33
Canis aureus	1	25.90
Vulpes vulpes	1	18.00

Ursus arctos	1	68.10
Meles meles	1	13.00
Mustela erminea	1	13.00
Mustela nivalis	1	23.00
Mustela putorius	1	12.33
Mustela vison	0	--
Lutra lutra	1	51.83
Martes martes	1	36.33
Martes foina	1	13.00
Felis silvestris	1	43.00
Lynx lynx	1	43.83
Sus scrofa	1	20.50
Dama dama	1	10.50
Cervus elaphus	1	14.50
Capreolus capreolus	1	28.50
Ovis orientalis	1	46.90
Capra hircus	2	74.50
Capra ibex	1	31.17
Rupicapra rupicapra	1	13.00
Rupicapra pyrenaica	1	80.50
Lepus corsicanus	1	60.00
Cervus elaphus	1	82.00

CHAPTER 4: GAP ANALYSIS OF TERRESTRIAL VERTEBRATES IN ITALY: PRIORITIES FOR CONSERVATION PLANNING IN A HUMAN DOMINATED LANDSCAPE[1]

Introduction

One of the most efficient ways to protect endangered biodiversity is to maintain viable populations in natural ecosystems (Balmford et al. 1996; Redford and Richter 1999; Groves 2003; Rosenzweig 2003) through the creation of protected areas (PAs). The importance of PAs has been widely supported (Bruner et al. 2001; Sinclair et al. 2002; Sànchez-Azofeifa et al. 2003; Chape et al. 2005) and numerous national and international agreements and laws (the Convention on Biological Diversity, http://www.biodiv.org/; the US ESA, http://endangered.fws.gov/esa.html; the Bird and Habitat European Directives, http://europa.eu.int/comm/environment/) consider PAs as the core of any conservation strategy.

However, a number of studies have demonstrated that PAs often do not represent the biodiversity of a region (Pressey et al. 1993; Scott et al. 1993; Rodrigues et al. 1999; Margules and Pressey 2000). Worldwide, the PA network has yet to be completed: at least 12 of the species considered (terrestrial vertebrates) are not represented in any PA, and 75% do not achieve their representation targets (Rodrigues et al. 2003, 2004a). In fact, socio-economic, aesthetic and political criteria have often been used to choose PA location, resulting in unrepresentative sites of lesser conservation value (Pressey

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1994; Scott et al. 2001) and in the need to establish new PAs, especially in those regions that could contribute most to species conservation.

Building on already existing PA systems represent a pragmatic approach to biodiversity conservation (Pressey 1994; Groves 2003). The first step is to determine the degree to which biodiversity elements are represented in existing PAs – generally referred to as GAP analysis – followed by the identification of the elements that need further protection through systematic conservation planning (Scott et al. 1993; Margules and Pressey 2000). Gap analysis and systematic conservation planning began at the beginning of the 1980s (Kirkpatrick 1983; Scott et al. 1987; Burley 1988) and studies that adopted one or both of the two approaches have been carried out globally (Rodrigues et al. 2004a), on a continental level and in many countries worldwide (Hunter and Yonzon 1993; Scott et al. 1993; Fearnside and Ferraz 1995; Ramesh et al. 1997; Araùjo 1999; Rodrigues et al. 1999; Keith 2000; Powell et al. 2000; Scott et al. 2001; Sierra et al. 2002; De Klerk et al. 2004; Fjeldsa et al. 2004; Oldfield et al. 2004; Yip et al. 2004; Dietz and Czech 2005).

The target for conservation planning has always been towards areas of higher biodiversity and thus, especially in the North America and Australia, towards zones which are largely free of human disturbance.

However, such areas are usually too few, do not necessarily complement the existing PA system, and cannot be easily incorporated into a human dominated landscape (Grumbine 1990; Schwartz 1999; Miller and Hobbs 2002). A good example of this is the Mediterranean basin, one of the ‵‵hottest hotspots′′ in the world (Shi et al. 2005) and at the same time one of the most significantly altered (Myers et al. 2000). Since the glacial period, the area has always been used by humans (Blondel and Aronson 1999; Farina et al. 2003). As a result, only 4.7% of its primary vegetation remains, while a great part of the present day landscape and biodiversity are the result of anthropogenic disturbances and land-use legacies over several millennia (Cowling et al. 1996; Heywood 1999; Lobo et al. 2001; Foster et al. 2003).

The integration of natural ecosystems and traditional human activities is one of the reasons for the high environmental diversity that characterizes the region (Farina et al. 2003), and ignoring this historical legacy may lead to conservation plans that are unsuitable to regional contexts (Foster 2000; Foster et al. 2003).

In the last 40 years, the Mediterranean basin has seen important changes in the patterns of human occupation and land-use: human pressure is increasing in flat and coastal areas mainly because of increasing resident population and tourist presence (UNEP 1989), while internal mountainous areas are being abandoned and naturally reforested (Ales et al. 1992; Garcia-Ruiz et al. 1996; Debussche et al. 1999; Scozzafava and De Sanctis 2006; Falcucci et al. 2007). These changes have substantial impacts on both the landscape and the biodiversity of the region (Ales et al. 1992; Covas and Blondel 1998; Tellini-Florenzano 2004; Scozzafava and De Sanctis 2006), but no conservation strategy has ever explicitly considered them, and moreover, to our knowledge, no systematic assessment of the existing PAs has been carried out in the Mediterranean basin.

In this study, we provide an assessment of the effectiveness of the existing PAs for the conservation of terrestrial vertebrates (mammals, breeding birds, reptiles, amphibians and freshwater fish) in Italy, and we identify regions, species, and strategies that appear to be priorities for expanding and consolidating the existing network in a human dominated landscape.

Materials and Methods

We compiled a geographical database with 777 PAs (Fig. 1) out of a total of 1004 existing in Italy (Gambino and Negrini 2001). We omitted the terrestrial areas for which it was not possible to find a paper and/or digital map (usually very small) and all the marine PAs. The total area of the 777 PAs accounts for 96.6% of the Italian PAs system.

Figure 1. Protected areas in Italy.

We collected the extents of occurrence (EOO; sensu Gaston 1991) plus habitat preferences for all the 504 Italian vertebrate species (102mammals, 244 regularly breeding birds, 43 reptiles, 34 amphibians and 82 freshwater fish; the complete list of species is available at http://www.gisbau.uniroma1.it/ren.php). Species data were obtained from the published literature and experts' opinions in the context of the National Ecological Network project (Boitani et al. 2003a). The entire species database is freely available at http://www.gisbau.uniroma1.it/ren.php.

We excluded 8 fish species from the analysis, whose distribution is unknown, and 35 species that have been recently introduced in the Italian peninsula (5 mammals, 1 bird, 1 amphibian, and 28 fish); but we retained all the species introduced in historical times and now part of the naturally occurring Italian fauna (e.g. the fallow deer *Dama dama* introduced by Romans more than 2000 years ago). A final list of 462 species (91.7% of the total of 504) was used for the analyses (97 mammals, 243 birds, 43 reptiles, 32 amphibians, 47 fish).

The Italian landscape was characterized considering four environmental layers: land cover, elevation, distance to water, and distance to roads. The four layers were chosen on the basis of the data on species' habitat preferences and the availability of digital maps covering all Italy. In particular, we used the CORINE 2000 Land Cover map (http://terrestrial. eionet.eu.int/CLC2000), a Digital Elevation Model (pixel size 75 m), a digital map of water bodies and streams (originally at the scale 1:25,000), and a map of the road network (originally at the scale 1:200,000). All layers were provided by the Italian Ministry of the Environment – Directorate for Nature Protection. The layers were transformed from vector to raster with a pixel size of 100 m (the lowest possible resolution for the CORINE Land Cover map, which was the layer with the coarser spatial resolution); the DEM, already in raster format, was re-sampled to obtain a pixel size of 100 m.

We used the species–habitat relationships and the available environmental layers to build deductive species distribution models (DM; sensu Corsi et al. 2000) for 442 vertebrate species. We did not build a DM for seven amphibians which live mainly in subterranean habitats that are not mapped on a national level, for six birds that are primarily marine and/or breeds on small islands, for two birds linked to habitats poorly mapped on a national level, for four mammals whose ecology is poorly known, and for one reptile, endemic of a small island. These 20 species were included in the analysis using their full EOs. Full details on the model building procedures that have been used are available from http://www.gisbau.uniroma1.it/ren.php.

The DMs were clipped using the species EOs, allowing us to take into account historical constraints, complex disturbance regimes, and other non-ecological factors that influence species distribution (Morrison et al. 1998).

We tested the predictive power for the DMs of 146 bird species using points of presence independently collected during the project MITO2000 (Fornasari et al. 2001; http://www.ciso- coi.org/mito2000.htm). For each point and each species, we built a buffer of a 150-m radius and we measured the percentage of area classified as suitable by the DM: when more than 50% of the area was classified as suitable, the point and the model were considered to agree. A model was considered to be validated when it agreed with more than 50% of the points of presence. If a model was not positively tested, the corresponding EOO was considered in the analysis.

The DMs for 24 fish, 52 mammals, 32 reptiles and 25 amphibians had been previously evaluated using independent datasets on species presence (Boitani et al. 2002; http://www.gisbau.uniroma1.it/REN.php).

All 462 species were weighted for the analysis using their conservation value and their degree of vulnerability. We used 12 rules of international and national conventions, treaties and laws, and six published conservation-related indexes (Table 1) to assign species-specific weights. Conventions and laws were used to score each species on the number of times the species was listed and the total score was scaled between 0 and 100, 0 being assigned to species not considered in any convention or lawand 100 being assigned to species considered in all the conventions and laws relative to each taxonomic group. The same operation was performed with the conservationrelated indexes. The two scores were then summed and the total was rescaled from 0 to 100. The 118 species ranked with a final score greater or equal to 50 were considered to be of conservation interest. For irreplaceability analyses, the final listwas divided into five groups (e.g. the group of the most vulnerable species had scores 100–81; the last group, comprising the least vulnerable species, had scores 20–0).

	National Law 157/92 art. 1 (1)	National Law 157/92	EU Bird Directive 79/409/EEC Ann. 1	Berna Convention 1979 Ann. 2	Berna Convention 1979 Ann. 3	Washington Convention (CITES) App. A	Washington Convention (CITES) App. B	Bonn Convention 1979 Ann. 1	Bonn Convention 1979 Ann. 2	EU Habitat Directive 92/43/eec Ann. 2	EU Habitat Directive 92/43/EEC Ann. 4	Barcellona Convention App. 2	Endemic Taxon (2)	Intrinsic Value (3)	% of population/range (4)	Trend in Italy (4)	Threat Category (5)	SPEC (6)
Mammals	X	X		X	X	X	X		X	X	X		X		X	X	X	
Birds	X	X	X	X	X	X	X	X	X					X			X	X
Reptiles				X	X	X	X			X	X		X		X	X	X	
Amphibians				X	X	X	X			X	X		X		X	X	X	
Fish				X	X	X	X			X	X	X	X		X	X	X	

Table 1. List of variables considered for the classification into conservation categories of each taxa. (1) Special protected species; (2) Bulgarini et al. 1998, Amori et al. 1999, Societas Herpetologica Italiana 1996 , Gandolfi et al. 1991; (3) Brichetti and Gariboldi 1997; (4) Bulgarini et al. 1998; (5) Amori et al. 1999, Baillie et al. 2004, Bulgarini et al. 1998; (6) Tucker and Heath 1994.

We analyzed the dataset in three steps: analysis of species richness over the entire national territory and in the protected areas (hot spot analysis), identification of gaps in the current PA systems (gap analysis), identification of areas that need priority conservation attention (irreplaceability analysis).

We built two maps of species richness (one using all 462 species and one using only the 118 species of conservation interest) using the DMs (all the suitability classes) and, when these were not available, the EOs, and we compared the species richness outside and inside the PAs using a Kolmogorov– Smirnov test.

Both gap and irreplaceability analyses require the definition of a representation target. We used a species-specific representation target depending on the area occupied by each species. For this purpose, the area occupied was defined using the area inside the EOO classified as suitable by the DMs; for species where no DM was available, the area occupied was defined as the EOO.

We set the representation target to 100% of the area occupied for species with a narrow distribution (area occupied smaller than 500 km^2), to 10% for widespread species (area occupied greater than 25,000 km^2), and for species with ranges in between the target was interpolated between the two extremes using a linear regression on the log-transformed area occupied.

A species not represented at all in any PA was considered a total gap, a species whose representation target is only partially met was considered a partial gap, and a species whose representation target is met was considered covered.

The same representation targets were also used for irreplaceability analysis. We divided the study area into 78,207 non-overlapping square (2 by 2 km) spatial units (hereafter called sites). Each site was classified as protected if at least 10% of its area was covered by PAs. Moreover, for each site we measured the area occupied by each species.

The irreplaceability analysis was carried out first considering and then excluding the presence of protected sites. In the first case, we aimed at identifying the areas outside the existing PAs that best complement the system. In the second case, we aimed at obtaining a map of the most irreplaceable sites in Italy. In both cases we used all the 462 species weighted according to their conservation values. In particular, for species with conservation value 1 (the highest) the conservation target used in the irreplaceability was 100% of the original conservation target, for species with conservation value 2 (the second higher) the target used in the irreplaceability was 80%, for species with conservation value 3 the target used in the irreplaceability was

60%, for species with conservation value 4 the target used in the irreplaceability was 40%, for species with conservation value 5 (the lowest) the target used in the irreplaceability was 20% of the original conservation target.

The analyses were performed using ArcGIS 9.0 (ESRI ©), CPlan 3.2 (NSW NPW Service ©) and SAS 8 (SAS Institute ©).

Results

Altogether the 777 PAs cover more than 3.37 million hectares, corresponding to more than 11% of Italy. They cover a disproportionate percentage of mountain areas (PAs: median elevation = 941 m, interquartile range = 1159 m; Italy: median elevation = 337 m, interquartile range = 618 m), with a great variation in the different regions (e.g. more than 25% of the Abruzzi and less than 1% of Sardinia). The largest PA is the Pollino National Park (more than 190,000 ha), the smallest is the Sasso di Preguda Natural Monument (0.05 ha), the median PA is 265.4 ha; 9% of the PAs are smaller than 10 ha, while only five PAs have an area larger than 100,000 ha (Fig. 2).

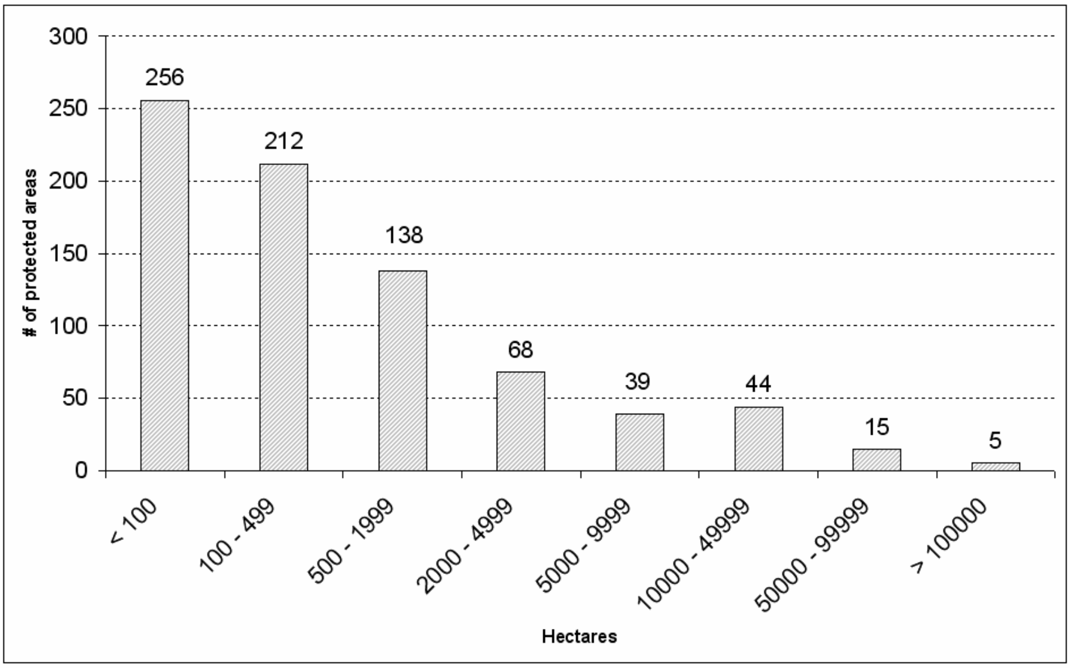

Figure 2. Number of protected areas per class area.

DMs (101 out of 146 [69.2%]) for birds were positively validated, the other 45 were discarded and the EOs were used. Overall, we used DMs for 395 species out of 462 (85.3%), for the other 67 species we used their EOs.

Considering all 462 species of vertebrates, the Apennines and the Alps are the areas with the highest species richness (Fig. 3), whereas Sardinia, Sicily and the coastal areas and plains of continental Italy have low diversity values. At least three large areas are notable for their species richness: the central Apennines of Molise, Abruzzi and Lazio (A in Fig. 3), the Emilia-Romagna and Liguria Apennines (B in Fig. 3), and the eastern Alps (C in Fig. 3). Smaller areas of high species diversity can also be found in the southern Apennines, and in particular in southern Campania, along the boundaries with Calabria and Basilicata (D in Fig. 3). Other areas include the Gargano (E in Fig. 3) and in some of the wetlands.

Considering only the species of conservation interest (118 species), the areas hosting the higher number of species are located in the Po river basin, the areas usually being small and fragmented (F in Fig. 3), along the Apennines (especially central and southern) and in the two main islands of Sicily (G in Fig. 3) and, in particular, Sardinia (H in Fig. 3).

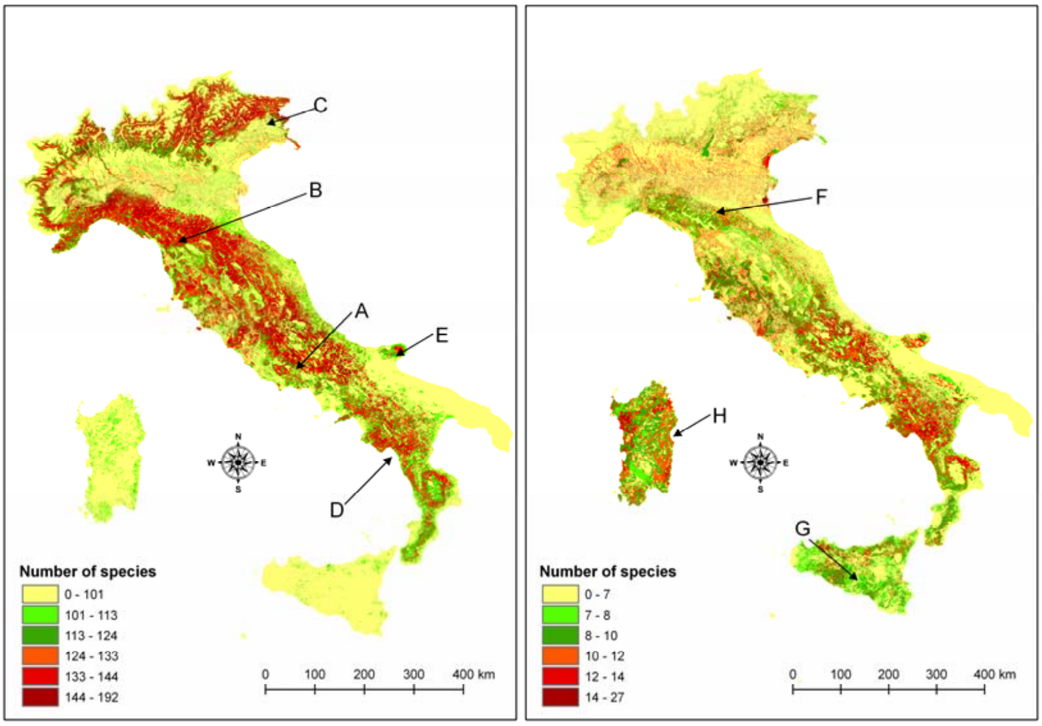

Figure 3. Vertebrate species richness in Italy (left: all the 462 species of vertebrates; right: 118 species of conservation interest). A = central Apennines (Molise, Abruzzi and Lazio); B = Emilia Romagna and Liguria Apennines; C = Eastern Alps; D = Southern Apennines (Campania, Calabria and Basilicata); E = Gargano; F = Po river basin; G = Sicily; H = Sardinia.

The pattern of species richness inside the PAs differs from that of Italy as a whole ($p < 0.0001$; Fig. 4), indicating a nonrandom selection of areas. In particular, areas with a high and low number of species are over-represented in PAs compared to Italy as a whole, while areas with a medium number of species are under-represented.

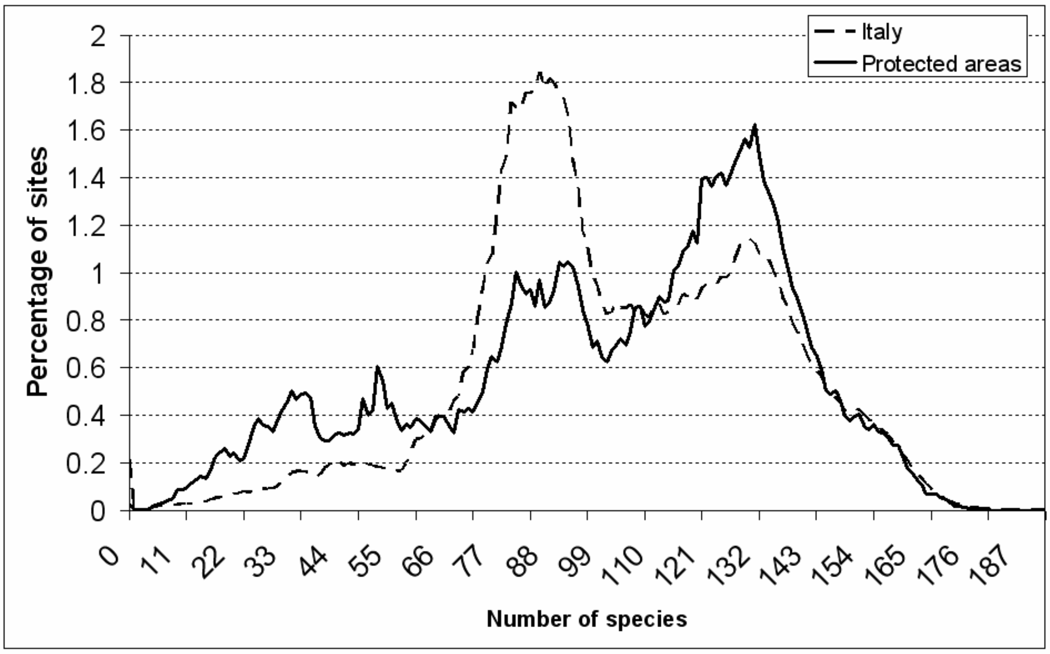

Figure 4. Frequency distribution of vertebrate species richness in Italy and in the protected areas. All the 462 species of vertebrates have been considered.

Considering only the species of conservation interest, the pattern of species richness inside PAs and in Italy as a whole is similar (Fig. 5), with a slight but significant preference of PAs for areas with higher numbers of endangered species ($p < 0.0001$).

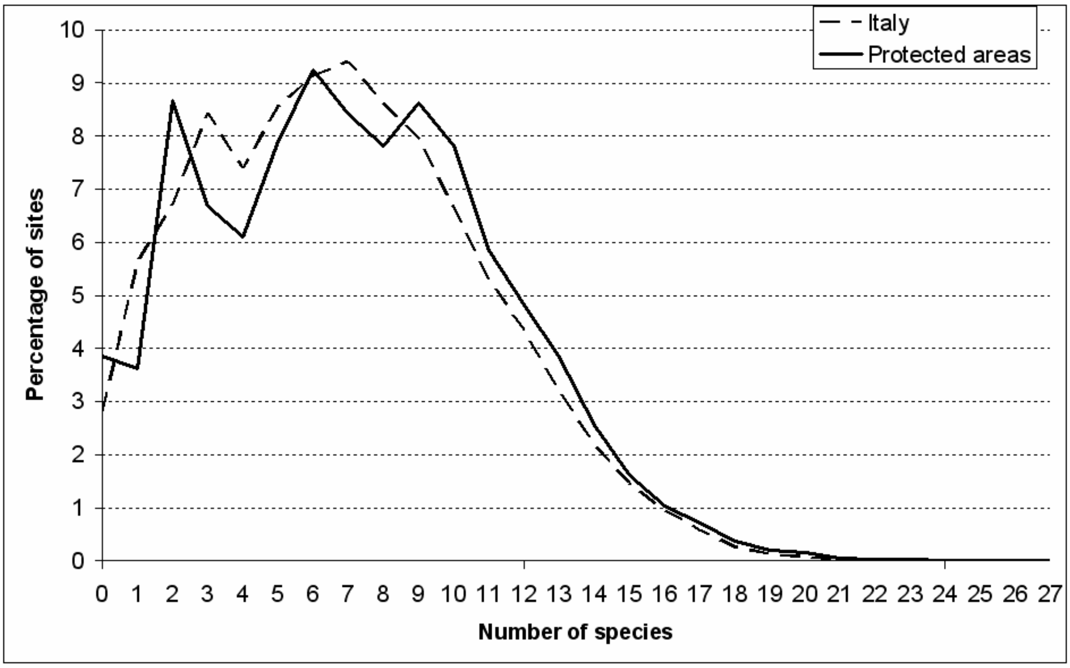

Figure 5. Frequency distribution of vertebrate species richness in Italy and in the protected areas. Only 118 species of conservation interest have been considered.

PAs cover less than 15% of the area occupied for 315 species (68%) while only three species are completely covered (Fig. 6). Following the pre-defined target, we identified 292 gap species (63.2% of those analyzed), of which 10 (2.2%) are total gaps and 282 (61.0%) are partial gaps (Fig. 7), with a general tendency towards species that meet smaller fractions of their representation targets. One hundred and seventy species were fully covered by the PA system. For the complete list of the gap species see the supplementary data.

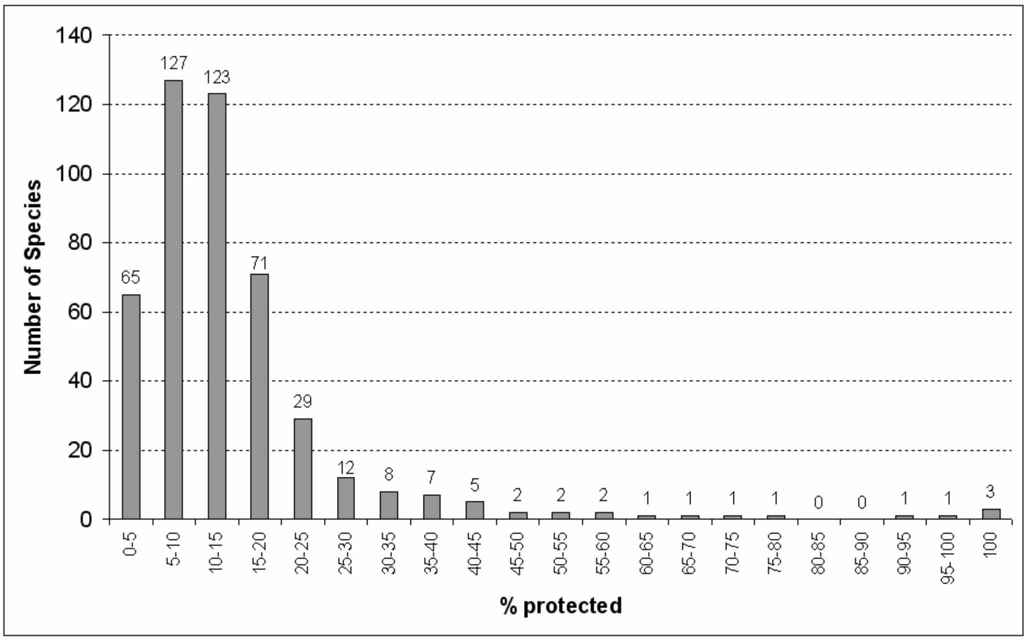

Figure 6. Percentage of the area occupied by each species that is covered by existing protected areas.

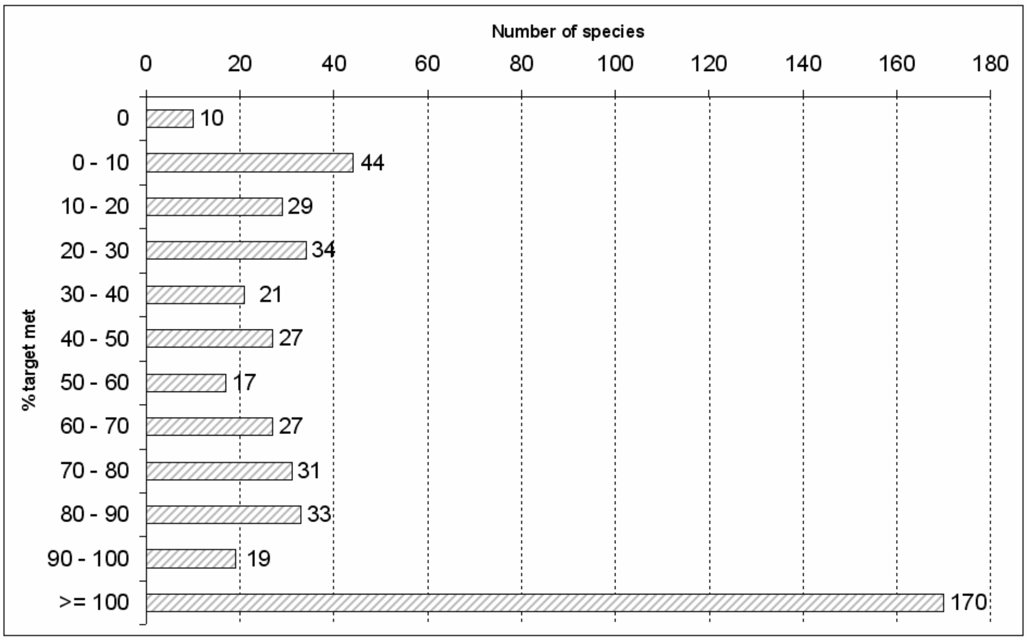

Figure 7. Number of total-gap species (0% of the target met), partial-gap species (from 0% of the target met to less than 100% of the target met) and covered species (≥100% of the target met).

The total-gap species are concentrated in Sardinia, with only three exceptions: a bird species in the eastern Alps, and two fish species, one in the Garda lake and the other in the central Apennines. Considering their conservation value, 8 out of 10 total-gap species are classified into the first three groups of conservation interest, while most of the partial-gap species and of the covered species are concentrated in the last three groups of conservation interest (Table 2).

Fish, amphibians, and reptiles are the less protected taxonomic groups (Table 2). Eighty-three percentage of fish species are gap (total or partial) and 49% of the gap species are classified in the first three groups of conservation interest; 79% of reptiles are gap and 47% of the gap species are classified in the first three groups of conservation interest. Amphibians (together with birds) have the highest absolute number of total-gap species, with more than 63% of gap species (total or partial) and 75% of the gap species being classified in the first three groups of conservation interest.

		Total Gap	Partial Gap	Covered species		Total Gap	Partial Gap	Covered species
Group 1	All taxa	1 (0.22%)	7 (1.52%)	0 (0.00%)	Mammals	0 (0.00%)	2 (2.06%)	0 (0.00%)
Group 2		5 (1.08%)	43 (9.31%)	9 (1.95%)		0 (0.00%)	2 (2.06%)	3 (3.09%)
Group 3		2 (0.43%)	95 (20.56%)	32 (6.93%)		0 (0.00%)	11 (11.34%)	15 (15.46%)
Group 4		2 (0.43%)	77 (16.67%)	62 (13.42%)		0 (0.00%)	7 (7.22%)	16 (16.49%)
Group 5		0 (0.00%)	60 (12.99%)	67 (14.50%)		0 (0.00%)	16 (16.49%)	25 (25.77%)
Group 1	Birds	0 (0.00%)	1 (0.41%)	0 (0.00%)	Reptiles	0 (0.00%)	2 (4.65%)	0 (0.00%)
Group 2		1 (0.41%)	27 (11.11%)	0 (0.00%)		0 (0.00%)	2 (4.65%)	0 (0.00%)
Group 3		2 (0.82%)	57 (23.46%)	14 (5.76%)		0 (0.00%)	12 (27.91%)	1 (2.33%)
Group 4		1 (0.41%)	53 (21.81%)	40 (16.46%)		0 (0.00%)	5 (11.63%)	2 (4.65%)
Group 5		0 (0.00%)	19 (7.82%)	28 (11.52%)		0 (0.00%)	13 (30.23%)	6 (13.95%)
Group 1	Amphibian	1 (3.13%)	1 (3.13%)	0 (0.00%)	Fish	0 (0.00%)	1 (2.13%)	0 (0.00%)
Group 2		3 (9.38%)	4 (12.50%)	4 (12.50%)		1 (2.13%)	8 (17.02%)	2 (4.26%)
Group 3		0 (0.00%)	6 (18.75%)	1 (3.13%)		0 (0.00%)	9 (19.15%)	1 (2.13%)
Group 4		0 (0.00%)	2 (6.25%)	4 (12.50%)		1 (2.13%)	10 (21.28%)	0 (0.00%)
Group 5		0 (0.00%)	3 (9.38%)	3 (9.38%)		0 (0.00%)	9 (19.15%)	5 (10.64%)

Table 2. Total-gap species, partial-gap species and covered species subdivided considering conservation value and taxonomic group.

The areas with high irreplaceability values (Fig. 8) are located throughout the entire peninsula, both in mountain areas (eastern and western Alps and central Apennines) and in coastal/plain areas (the Po river plain, western Liguria, the north-western coast of the peninsula, a few smaller areas in southern Italy, most of the smaller islands), but the most interesting result is Sardinia, which has almost been completely classified in the higher irreplaceability values.

When the existing PAs are included in the analysis, many of the highly irreplaceable areas in the mountains lose their importance, while Sardinia, the coastal areas, and the Po river basin retain their high irreplaceability values (Fig. 8).

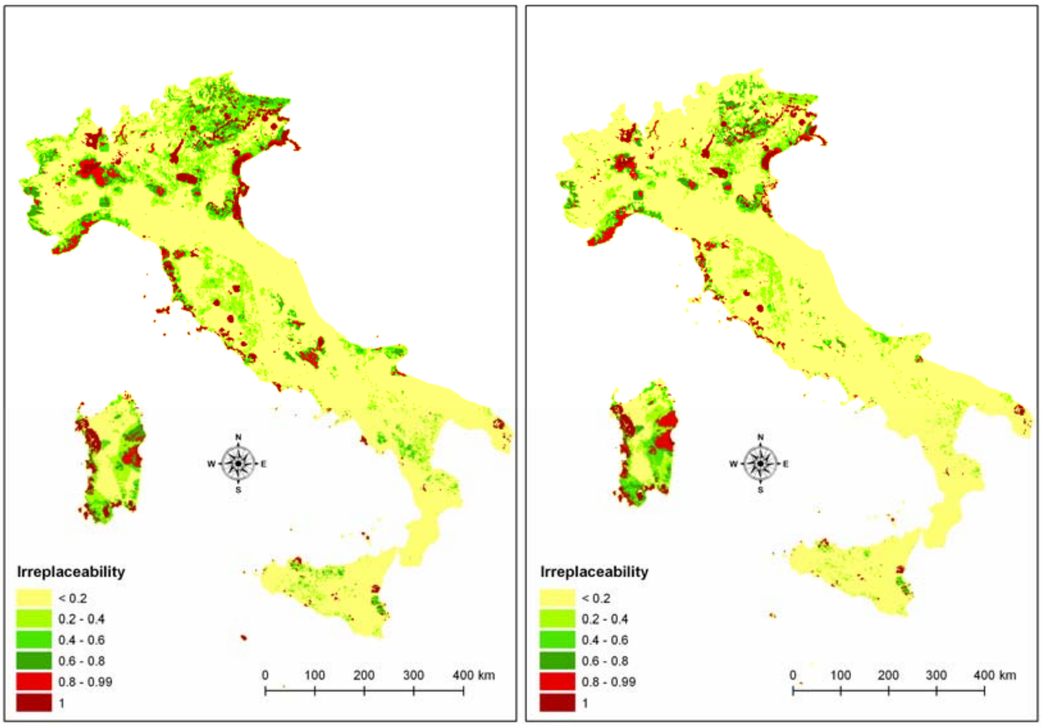

Figure 8. Distribution of irreplaceability values in the Italian peninsula: the map on the left does not consider existing PAs, the map on the right considers the existing PAs.

Discussion

Our study represents a first attempt to evaluate the efficacy of the Italian PA system for the conservation of terrestrial vertebrates. As in any gap analysis, our results depend heavily on the conservation target as well as on the quality of the dataset used. Changing the dataset or changing the targets the results we obtained would have been different; however, the methodology used and the availability of the dataset allow for a simple and quick re-analysis with better data or with different targets. Moreover, we provide full details on the conservation targets in the supplementary data.

We weighted each species with a conservation value obtained using biological and socio-political criteria. Future studies adopting the same approach should perform sensitivity analyses on the criteria included in the calculations to evaluate the results obtained, but the final ranking that we obtained seems to correctly reflect the relative conservation value of each species. Endemic species such as *Pelobates fuscus*, *Euproctus platycephalus*, *Podarcis wagleriana*, *Vipera ursinii*, *Lampetra zanandreai*, and *Rupicapra pyrenaica ornata* had the highest conservation values and species like *Anguilla anguilla*, *Cyprinus carpio*, and *Rattus norvegicus* had the lowest. However, some species appeared in the ranking with a relative value which was higher or lower than expected. The wildboar (*Sus scrofa*), usually considered a pest, was classified in the central part of the ranking because there is the possibility of an endemic sub-species (*Sus scrofa mediterraneus*) still surviving in Sardinia (Boitani et al. 2003b). Conversely species such as *Lepus corsicanus*, which was thought to be extinct and which has been rediscovered only very recently (Pierpaoli et al. 1999; Riga et al. 2001), are not considered in many conservation treaties or in the IUCN red lists, making this a low-ranked species.

In our analyses we used a combination of DMs and of EOs. To our knowledge, all studies on reserve selection and gap analysis that focus on species distribution as a surrogate for the biodiversity of a region have used raw distribution data in the form of points of presence and/or EOs (van Jaarsveld et al. 1998; Araùjo 1999; Rodrigues et al. 1999; Polasky et al. 2000; Dimitrakopoulos et al. 2004; Rodrigues et al. 2004a, 2004b; Yip et al. 2004; Solymos and Fehér 2005) or DMs (Scott et al. 1993; Clark and Slusher 2000; Allen et al. 2001; Loiselle et al. 2003; Carroll et al. 2003; Johnson et al. 2004; Rondinini et al. 2005; Wilson et al. 2005).

Both EOs and DMs have advantages and limitations. EOs are often not available, incomplete, spatially biased towards areas of easier accessibility, or taxonomically biased towards flagship species or taxa (Polasky et al. 2000; Margules et al. 2002; Williams et al. 2002; Wilson et al. 2005). Moreover, species do not completely occupy their EOs (Gaston 1991), making EOs particularly prone to commission errors (Fielding and Bell 1997; Loiselle et al. 2003; Rondinini et al. 2005; Wilson et al. 2005).

DMs can be used to reduce the level of commission errors (Corsi et al. 2000; Guisan and Zimmermann 2000; Scott et al. 2002), potentially the most ‘‘dangerous’’ type of error from a conservation point of view (Loiselle et al. 2003; Rondinini et al. 2005), but they cannot be developed for all species. Some species have small and well known distribution ranges that models cannot improve: for example, *Larus audouinii* in Italy breeds only on a few small islands and even the exact number of birds breeding on each island is known (Serra et al. 2001). Other species have a narrow distribution and are limited to particular habitat types that are not mapped on traditional land-use maps (e.g., *Proteus anguinus* is limited to caves in North-eastern Italy).

Even when DMs can be developed, there is no inherent assurance that model results portray reality (Guisan and Zimmermann 2000; Johnson and Gillingham 2004), and a model that poorly reflects the presence of a species may result in misleading and/or harmful conservation and management actions (Loiselle et al. 2003; Wilson et al. 2005). This is the case of species whose presence is constrained by micro-habitat characteristics that are not mapped on a small scale and whose distribution cannot be modeled over large areas (in Italy this is the case of, for example, *Sorex araneus*). Moreover, it is important to note that our DMs may potentially be prone to omission errors, thus our results should be further checked using larger and more detailed datasets.

Despite this, conservation planning cannot be delayed till complete biodiversity surveys or data to model distribution of a given species are available, or options for conservation would be dramatically reduced (Margules and Pressey 2000). Therefore, we argue that DMs and EOs can be integrated successfully: DMs should be thought of as a finer resolution version of EOs, practically representing a way of moving from EOs towards Areas of Occupancies (Gaston 1991). Our approach is a way of adding valuable information to the analysis, information that would be otherwise lost. In particular, considering only DMs, we would have lost 3 total-gap species (all classified in the second group of conservation interest) and 35 partial-gap species (20 of which were classified in the first three groups of conservation interest).

We were able to collect the largest and most detailed geographical database on the Italian PAs (the PAs that have not been considered in the analysis are all extremely small) and our results demonstrate that, even though the existing PAs cover more than 11% of the national territory, a figure comparable to the coverage of the worldwide global network of PAs (Rodrigues et al. 2004a) and much greater than the 4% coverage that characterize the Mediterranean basin (Médail and Myers 2005), the system cannot be considered complete. In particular, our analyses show apparently contrasting results for the ‘‘internal’’ mountainous part and the ‘‘Mediterranean’’ (mainly coastal areas, islands and flat areas) part of the peninsula.

Italian PAs tend to over-represent highland areas and other regions with low values. This has also been found in many studies elsewhere (Fearnside and Ferraz 1995; Scott et al. 2001; Oldfield et al. 2004). On the contrary, the coverage offered by PAs in the ‘‘Mediterranean’’ part of Italy is limited. Many of the existing PAs are small (the median size is 265 ha), with a tendency for smaller PAs in coastal areas, limiting the possibility of supporting viable populations of vertebrate species (Saunders et al. 1991; Rodrigues and Gaston 2001), even for species whose coverage is complete.

These results are important especially in light of the changes occurring in Italy over the last decades. The internal mountain areas have seen a trend of decreasing human population and activities (particularly important being the decrease in traditional agriculture and pastures), and increasing natural vegetation, forests in particular (Falcucci et al. 2007). Pastures and other traditional human activities have also decreased in coastal ‘‘Mediterranean’’ areas, where human pressure has increased (Falcucci et al. 2007). Vertebrate populations have been affected by these changes, with ‘‘forest’’ species increasing and ‘‘open space’’ species decreasing. This is illustrated by the fact that wolves increased from about 100 individuals in the 1970s to more than 500 individuals in 2000 (Boitani and Ciucci 2000); wild ungulate populations and temperate forest birds have increased both in numbers and distribution (Preiss et al. 1997; Boitani et al. 2003b; Laiolo et al. 2004; Tellini-Florenzano 2004).

Almost the opposite can be found in Sardinia, Sicily and in the coastal areas where Mediterranean bird and plant species are decreasing following a decrease in open areas and the intensification of agriculture (Preiss et al. 1997; Brotons et al. 2004; Scozzafava and De Sanctis 2006). The little bustard (*Tetrax tetrax*), an open space species, is disappearing from continental Italy and Sardinia (de Juana and Martinez 2001; Wolff et al. 2001); the griffon vulture (*Gyps fulvus*), dependent on traditional pasture, is extinct in the Italian peninsula (Dentesani et al. 1996), and is extremely rare in Sardinia; the Egyptian vulture (*Neophron percnopterus*) is decreasing drastically following the reduction of traditional cattle husbandry techniques (Liberatori and Penteriani 2001).

The number of total-gap species can be considered low (10 species, 2.2% of all species considered in the analysis compared to the 12% found by Rodrigues et al. [2004a] at a global level) but the number of partial-gap species is relatively high. The high number of gap species should be seen in relation to the conservation target that we defined: 10% was the lowest possible target and thus, even though PAs occupy more than 11% of Italy, they provide less than 10% of coverage to many vertebrate species.

This is a clear indication that the existing PAs cannot be considered to be fully representative of Italian vertebrate biodiversity.

Flagship and conspicuous species, such as large mammals and, at least in part, birds are the species that obtain the best coverage from the existing protected areas. Amphibians and freshwater fish, and in general the lesser known species, are those in greater need of further protection, together with species linked to particular habitats and with narrow distribution ranges, such as birds breeding in aquatic habitat types. This pattern is only partially linked to the greater conservation interest that flagship and conspicuous species can generate (Meffe and Carroll 1997). In fact, it is also connected to the distribution pattern of species such as amphibians and freshwater fish, which in Italy are usually limited to patchy habitats in areas of dense human settlements, mostly in the ''Mediterranean'' part of the peninsula.

It comes as no surprise that the total-gap species classified in the first three groups of conservation interest have a distribution which is limited to Sardinia, with the exception of one fish species that is only found in the Garda Lake (north Italy). For conservation planners, this represents an easy target to improve the Italian conservation network. At the same time, this is also an indication of the total lack of conservation interest that the island of Sardinia has received to date. This situation cannot be justified given the importance of Sardinia's biodiversity within the Mediterranean hotspot (Médail and Quezel 1999), but can be easily explained by the traditional political and social difficulties of establishing PAs on the island (Carrus and Bonnes 2002).

The pattern of species richness (Figs. 20–23) indicates that the Alps and the Apennines represent the strongholds for vertebrate biodiversity, with coastal areas and plains hosting a relatively low number of species. However, many species of conservation interest are concentrated in the Mediterranean part of the peninsula, and particularly in the two main islands as well as along the coasts, exactly where the number of protected areas is lower and human pressure is higher (UNEP 1989).

The irreplaceability analysis confirms the results discussed above, supporting the importance of areas such as Sardinia, where the number of endangered species with a narrow distribution range is high, and highlighting small areas in the Po river plain and along the coasts (particularly the Tyrrhenian one) that are especially important for freshwater fish and aquatic birds. The high irreplaceability of several areas in the Alps (particularly eastern) and in the Apennines is due to their high number of species.

Our results indicate that the existing network of PAs is inadequate in assuring the conservation of all vertebrate species. Completing the system to ensure coverage of all species would seem to be the most logical action to take: it is important to cover the total-gap species, implementing a system of protected areas in Sardinia, and to integrate the conservation provided to partial-gap species. In this context, an important role could be played by the Sites of Community Interest (SCIs) and the Special Protection Areas (SPAs) that Italy has proposed under the European Bird and Habitat Directives (http://europa.eu.int/comm/environment/) for the conservation of particular habitats and species. The proposed SCIs and SPAs, if approved by the European Union, would increase the percentage of protected Italian territory to more than 20% (Boitani et al. 2003b) providing the necessary coverage for gap species, particularly in Sardinia, and covering many of the areas indicated by the irreplaceability analysis as priorities outside the existing PAs.

However, Italy and the Mediterranean basin have seen thousands of years of intense human presence, with a complex integration of traditional human activities and natural ecosystems leading to high environmental diversity and also to high fragmentation. As a result a complex and ecologically rich cultural landscape has formed (Blondel and Aronson 1999).

In such a context PAs, when feasible, are necessarily small and fragmented and cannot protect viable populations of almost any species of vertebrate (Saunders et al. 1991; Tilman et al. 1994). Their role as components of a conservation network must be planned in conjunction with the conservation of areas actively managed to preserve the traditional cultural landscape, where species have coexisted for centuries with compatible human activities. In the Mediterranean more than anywhere else, the PAs must be planned and managed in conjunction with the matrix in which they are embedded and in the context of the environmental history of the region (Foster et al. 2003). Biodiversity and human presence are functionally linked through traditional agriculture, pasture, etc. and the only viable option for conservation is that of considering human presence and human activities as an integral part of the system. Ignoring the environmental history of a region can produce conservation and management schemes that simply cannot work (Foster 2002), because historical changes in the ecosystems caused by natural and humanrelated phenomena significantly limits available management options. Protecting extensive tracts of wild land has usually been considered the best way of preserving biodiversity (Redford and Richter 1999; Miller and Hobbs 2002) but in a human dominated landscape, a condition that is common to most of the biodiversity hotspots (Myers et al. 2000), the only viable approach to conservation is that of coupling the conservation of the few wild, semi-natural areas with the restoration and the maintenance of traditional cultural landscapes with all their species assemblages.

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CHAPTER 5: CONTRIBUTION OF THE NATURA2000 NETWORK TO BIODIVERSITY CONSERVATION IN ITALY[1]

Introduction

One of the main responses to the current biodiversity crisis has been to develop networks of conservation areas, often designed with the aim of maximizing returns from limited conservation investments while minimizing conflicts with human activities. Many different coarse-grained approaches have been developed in this context (Groves et al. 2002), including endemic bird areas (Statterfield et al. 1998), biodiversity hotspots (Myers et al. 2000), tropical wilderness areas (Mittermeier 1999), ecoregions (Groves et al. 2000), centers of plant diversity (WWF & IUCN 1994-1997). Many of these approaches have been criticized (Jepson & Canney 2001; Possingham & Wilson 2005) because of taxonomic issues or because their targets are ill-conceived (Kareiva & Marvier 2003). Moreover, cooperation among different conservation groups has been sporadic (Mace et al. 2000, but see Mittermeier et al. 1998), which has made it difficult to convince the world of the importance of these approaches (Redford et al. 2003).

The examples mentioned above identify large areas of global importance for biodiversity which, in nearly all cases, are too large to be protected entirely. To date, there are few examples of international initiatives aimed at the identification of specific local sites for conservation. Among these are international bird areas (Fishpool & Evans 2001), key biodiversity areas (Eken et al. 2004), alliance for zero extinction (Ricketts et al. 2005) and the Natura 2000 network developed by the European Union (EU; European Commission 2000a).

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Natura 2000 is the cornerstone of EU nature conservation policy, and it is regulated mainly by two directives: the 1979 Bird Directive and the 1992 Habitat Directive. The Bird Directive identified 193 endangered species and subspecies for which the member states are required to designate Special Protection Areas (SPAs). The Habitat Directive aims to protect animals (other than birds), plants and habitats for which each Member State is required to identify Sites of Community Importance (SCIs). The SCIs and SPAs make up the Natura 2000 network, whose aim is to conserve an extensive range of habitat types and wildlife species throughout Europe, maintaining listed habitat and species at “favorable conservation status” (European Commission 2000a; European Commission 2000b).

Natura 2000 is by far the most important conservation effort being implemented in Europe, it is one of the most important tools that the European nations have to improve their existing networks of conservation areas, and it has been proposed as the main strategy that the EU can use to meet the target of halting (or at least significantly reducing) biodiversity loss by 2010 (Balmford et al. 2005). As of June 2006, 20,789 SCIs (area > 559,000 km2) and 4,540 SPAs (area > 444,000 km2) have been submitted to the EU for approval. For SCIs the national territory covered ranges from 4.2% (Poland) to 22.6% (Spain) (mean [SD] = 10.5% [8.3%]). For SPAs the percentage of national territory covered ranges from 2.4% (Malta) to 23% (Slovenia) (mean [SD] = 13.2% [7.8%]). Overall, more than 20% of Europe (25 countries in its current configuration) has been proposed for incorporation in the Natura 2000 network (European Commission 2006) with an expected yearly cost of €6.1 billion to properly implement and manage the system (Torkler 2006).

Moreover, Natura 2000 represents the conservation scheme with the best “political” chances of success throughout the continent. In fact, specific policy and financial instruments have been developed to ensure the proper implementation and management of Natura 2000 sites (Miller & Kettunen 2005).

Whittaker et al. (2005) calls for systematic testing and evaluation of all conservation schemes in order to strengthen and improve their effectiveness for conservation. Many researchers provide a scientific evaluation of important bird areas (Pain et al. 2005; O'Dea et al. 2006; Tushabe et al. 2006) or other national fine-grained conservation initiatives (Oldfield et al. 2004; Czech 2005), but there has been no evaluation of the conservation effectiveness of Natura 2000, with the exception of plant species in Crete (Dimitrakopoulos et al. 2004). Dimitrakopoulos and co-authors used plant diversity hotspots and complementarity analysis to define priority areas for plant conservation and to calculate their spatial overlap with Natura 2000 in Crete. The authors showed that overlap among priority areas and Natura 2000 is low and they conclude that the Natura 2000 network in Crete is inadequate to fulfill its major goal of ensuring long term persistence of plant species.

We consider two questions in an evaluation of the Italian Natura 2000 network: (1) How well does Natura 2000 complement existing protected areas? (2) Is Natura 2000 suitable for maintaining listed species in a "favorable conservation status"? We limited our analysis to terrestrial vertebrates and freshwater fish, and we combined species distribution models (DMs) and extents of occurrence with data for the existing protected areas and the Natura 2000 network.

Methods

Officially, the European Union lists 2286 SCIs (total area=44,979 km2) and 566 SPAs (total area=34,683 km2; European Commission 2006) in Italy. We compiled a geographic database (Fig. 1) with 2255 SCIs (98.6% of the total) and 554 SPAs (97.9%). Overall, the database accounted for 98.1% (44,105 km2) of the SCIs' area and for 90.7% (31,442 km2) of the SPAs' area. We used the dataset for existing protected areas described by Maiorano et al. (2006).

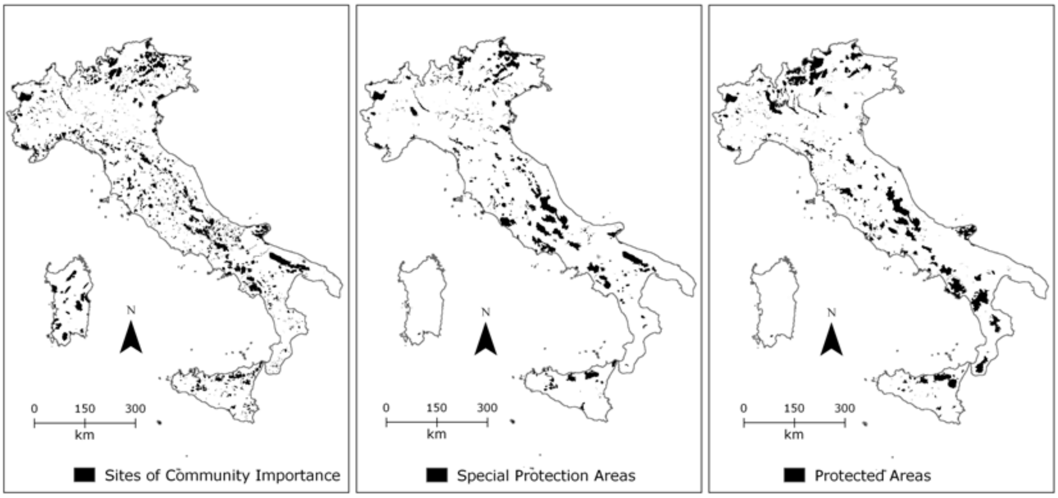

Figure 1. Sites of Community Importance, Special Protection Areas and Protected Areas in Italy.

Species distribution

We obtained species distribution data for terrestrial vertebrates (mammals, amphibians, reptiles, regularly breeding birds) and freshwater fish from Maiorano et al. (2006) in the form of DMs (pixel size 100X100m; 4 suitability classes: unsuitable, low suitability, medium suitability, high suitability) and extents of occurrence. Distribution models and extents of occurrence for amphibians and reptiles were updated following Sindaco et al. (2006). We updated DMs and extents of occurrence for freshwater fish following Smith and Darwall (2006).

We considered only species that are naturally present on the Italian peninsula or species introduced in historical times that have become naturalized. We considered 468 species in the analyses (98 mammals, 243 regularly breeding birds, 44 reptiles, 36 amphibians, 47 freshwater fish) that correspond to 62.6% of all the terrestrial *Chordata* present in Italy (Ministero dell'Ambiente e della Tutela del Territorio 2007). All 468 species were classified according to their conservation status as defined by Maiorano et al. (2006). Two hundreds and four out of the total 468 species (43.6%) are listed under the EU Bird directive (Annex I; 83 species hereafter referred to as BIRD species) or under the EU Habitat directive (Annex II and IV; 121 species hereafter referred to as HABITAT species) as species of European concern. In particular, we considered all species listed under the Bird directive and regularly breeding in Italy and 60.3% of the animal species listed under the Habitat directive and present in Italy.

Distribution models were available for 33 amphibians, 92 mammals, 207 breeding birds, 37 reptiles, and 45 freshwater fish, for a total of 414 species (88.5% of the total 468). For 54 species (6 mammals, 36 breeding birds, 7 reptiles, 3 amphibians, 2 freshwater fish) no distribution models were available because of limited ecological knowledge, ecological requirements that have no correspondence in the available environmental data, or extremely small extents of occurrence.

We tested the predictive power of 303 DMs (73.2% of the models) against a validation dataset (from 20 to 4392 points of presence per species) independently collected by various researcher: Fornasari et al. (2001), Ruffo and Stoch (2005), Abruzzo-Lazio-Molise National Park (unpublished data), M.Masi and P.Ciucci (unpublished data), A.Mortelliti (unpublished data), G.Puddu (unpublished data), W.Reggioni (unpublished data), and S.Sarrocco (unpublished data). For each DM we calculated the percentage of points in agreement with the predictions of the model. In particular, for each point of presence and for each species tested, we built a circular buffer with a radius corresponding to the location error associated with each point of presence (from 200 m to 3 km). A point was considered to agree with the respective DM if the circular buffer contained medium suitability or high suitability cells. To avoid using a subjectively predefined number of cells, we considered a range of possibilities, going from 1 cell inside the buffer classified as medium or high suitability to 20% of the area inside the buffer classified as medium or high suitability. Therefore we calculated a range of percentages of agreement for each species tested.

To test the significance of the agreement among points of presence and the DMs, we used a permutation test. For each species considered, we generated 1000 sets of random points sharing the same characteristics as the set of points of presence (same number of points, same distribution of buffer sizes). For each of the 1000 sets of random points we calculated the percentage of agreement with the DM as specified above for the points of presence. If the percentage of agreement calculated for the points of presence was in the top 5% of the agreements obtained from the 1000 random samples, the model was considered validated. We performed the same test for all the different percentages of high and medium suitability inside the buffer.

Vertebrate diversity, Natura 2000 and existing protected areas

To investigate how well the Natura 2000 network complements the existing network of protected areas, we analyzed the dataset in three steps: (1) analysis of species richness over all of Italy, in the existing protected areas and in protected areas plus Natura 2000 areas, (2) gap analysis of the existing protected areas and of protected areas plus Natura 2000 areas (Scott et al. 1993), and (3) irreplaceability analysis (Ferrier et al. 2000).

We built three maps of species richness, one with all 468 species (total richness), one with the 83 BIRD species, and one with the 121 HABITAT species. We measured the correlation existing among total species richness, BIRD richness and HABITAT richness with Pearson correlation (“correlation” function in the ArcInfo Grid module; ESRI, Redlands, California). To assess whether the Natura 2000 network will improve vertebrate conservation in Italy, we compared total species richness inside protected areas with that inside protected areas plus the Natura 2000 network, BIRD richness inside protected areas with that inside protected areas plus SPAs, and HABITAT richness inside protected areas with that inside protected areas plus SCIs.

To perform gap and irreplaceability analyses, we defined a species-specific representation target that depended on the area occupied by each species. We defined the area occupied as the sum of all the medium-suitability or high-suitability areas in each species DM. If no DM was available, the area occupied was defined as the area of the extent of occurrence. Following Maiorano et al. (2006), for narrowly distributed species (area occupied < 500 km2) the representation target was set to 100% of the area occupied, for widespread species (area occupied > 25,000 Km2) the representation target was set to 10% of the area occupied, and for species with ranges in between the target was interpolated between the two extremes using a linear regression on the log-transformed area occupied.

For each species we calculated the percentage of the representation target that was met by existing protected areas and by protected areas plus SCIs and SPAs. A species not represented in any conservation area was considered a total gap. A species whose representation target is only partially met was considered a partial gap. A species whose representation target is completely met was considered covered.

In the irreplaceability analysis (Ferrier et al. 2000) we weighted each species based on their conservation value as defined by Maiorano et al. (2006). We divided the study area into 78,207 nonoverlapping squares (2 km x 2 km) spatial units and for each square we measured the area occupied by each species. We performed 3 irreplaceability analyses without considering protected areas or Natura 2000 sites: one considering all 468 species (to identify the most irreplaceable sites in Italy); one considering the 83 BIRD species (to identify the most irreplaceable sites for species of concern under the Bird Directive) and one considering the 121 HABITAT species (to identify the most irreplaceable sites for species of concern under the Habitat Directive). We also performed the same analyses including protected areas and Natura 2000 sites. In particular, we included both protected areas and the entire Natura 2000 network while considering all 468 species (to identify where the representation targets are unmet if the entire system is implemented); we included both protected areas and SCIs while considering the 121 HABITAT species (to identify where the representation targets are unmet for species of concern under the Habitat directive); we included both protected areas and SPAs while considering the 83 BIRD species (to identify where representation targets are unmet for species of concern under the Bird directive).

All analyses were performed in ArcGis 9.1 (ESRI), C-Plan 3.2 (New South Wales Department of Environment & Conservation, Sydney, Australia) and SAS 8 (SAS Institute, Cary, North Carolina).

Measuring conservation status in the Natura 2000 network

To assess the contribution of the Natura 2000 system in maintaining listed species in a favorable conservation status, we estimated the number of individuals of species of European concern potentially occurring in SCIs and SPAs. Freshwater fish, amphibians, and reptiles were excluded from the analysis because of the scarcity of available information and because of the importance of microscale habitat factors for these taxa (all data collected are available from L.M.)

We obtained the mean number of breeding pairs in Italy for BIRD species from BirdLife (2004). For five mammal HABITAT species, we obtained the number of individuals present in Italy from Boitani et al. (2003) and from L.Carnevali and F.Riga (personal communications). For all the other mammal HABITAT species, no information was available on the number of individuals. For 13 mammal HABITAT species we collected home range size (minimum and maximum) and/or density (minimum and maximum) from Boitani et al. (2002; 2003) and Mitchell-Jones et al. (1999).

For all species for which density was available we estimated the minimum and maximum number of individuals living in Italy by dividing the area occupied by each species by the minimum and maximum density of each species. For all the species for which only home range size was available we estimated the minimum and maximum number of individuals living in Italy by dividing the area occupied by each species by the minimum and maximum home range sizes and multiplying the result by 2 to account for intersexual overlap among home ranges. Inter- and intrasexual home range overlap varies considerably among species, but we chose complete overlap among sexes to obtain conservative estimates.

We calculated the number of individual or breeding pairs present in protected areas, in protected areas plus SCIs (for HABITAT species) and in protected areas plus SPAs (for BIRD species) by multiplying the total (estimated through field counts or based on DMs) number of individuals by the percentage of the area occupied within protected areas and within protected areas plus Natura 2000.

We evaluated the conservation status of 101 species of European concern (100% of the BIRD species and 14.9% of the HABITAT species) based on the IUCN red list criterion D (Baillie et al. 2004). In particular, we classified each species as critically endangered (CE), endangered (En), or vulnerable (Vu) inside protected areas and inside protected areas plus Natura 2000 according to the following criteria: CE, < 50 mature individuals; En, < 250 mature individuals; and Vu, < 1000 mature individuals. A species classified as CE, En or Vu was not considered to be completely supported by the Natura 2000 network.

Results

It is proposed that Natura 2000 in Italy will cover almost 5 million ha (> 16% of Italy), which, combined with the existing protected areas, would amount to 20.5% of

Italy being committed to conservation areas. The SCIs’ median size was 520 ha. Only one area was larger than 100,000 ha, and 23% of the areas were smaller than 100 ha. The SPAs were bigger on average (median 1229 ha), but only two areas were bigger than 100,000 ha, and 12% were smaller than 100 ha. The SCIs and SPAs covered a disproportionate percentage of mountain areas, where the larger areas were located (Italy: median elevation 337m, interquartile range 318m; SCIs: median elevation 787m, interquartile range 1073m; SPAs: median elevation 915m, interquartile range 1162m), but many (and usually small) areas were located in lowlands, along the coasts and on the islands. Both SCIs and SPAs often overlapped with existing protected areas (55.4% of the SPA area and 41.6% of the SCI area is included in existing protected areas).

Model validation

For all taxa, concordance among points of presence and DMs was greater than the average concordance among random points and DMs for a minimum of 89.4% and a maximum of 92.7% of the DMs. For most of the DMs (from 68.9% to 78.9%) the difference was significant at the 0.05 level.

For each taxonomic group, the concordance among points of presence and DMs was greater than the average concordance among random points and DMs for most of the species, with a minimum of 83.3% for mammals and a maximum of 100% for amphibians (Table 1). However, the percentage of DMs for which the difference was significant varied from a minimum of 51.4% for mammals (the taxon with the lowest number of presence points available for validation) to a maximum of 85.9% for birds (the taxon with the highest number of presence points available for validation).

Taxon	Better than random[1]	Significant[2]	Number of points[3]
Mammals	83.3 - 88.9	51.4 - 69.5	76 (±168)
Birds	93.6 - 94.9	78.9 - 85.9	145 (±517)
Reptiles	92.0 - 96.0	64.0 - 72.0	105 (±236)
Amphibians	90.5 - 100	57.1 - 80.9	123 (±233)
Fish	86.2 - 89.7	68.9 - 79.3	127 (±299)

Table 1. Results of the validation of DMs for individual taxonomic groups. [1]Better than random reports the percentages (minimum – maximum) of distribution models that agreed with points of presence more than expected by chance alone. [2]Significant reports the percentage (minimum – maximum) of distribution models that agreed with points of presence significantly more than expected by chance. [3] median number of points of presence per taxonomic group (interquartile range).

Vertebrate diversity, Natura 2000 and existing protected areas

The areas of highest total richness (Fig. 2) were located mainly in the medium-elevation areas of the Alps (especially the eastern Alps) and the Apennines (the northern Apennines in particular). Plains, costal areas, and the islands had the lowest species richness, with the exception of some wetlands and a few small areas. HABITAT richness exhibited the same pattern as total richness (r=0.92), whereas BIRD richness exhibited a different pattern if compared to total richness (r=0.34). The eastern Alps and Sardinia were the main diversity centers, with many smaller but extremely rich areas located in flat areas and along the coasts (Fig. 2).

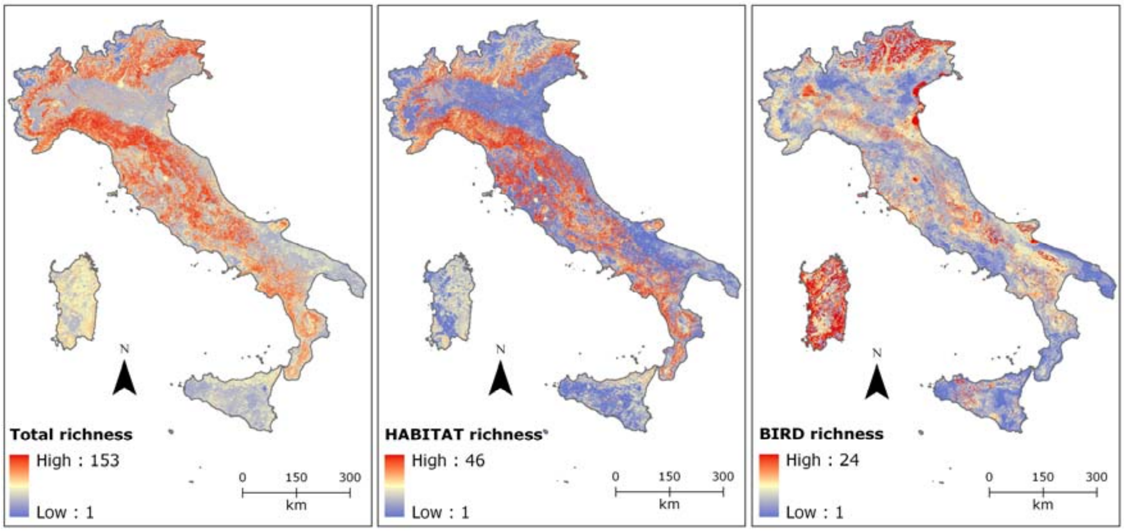

Figure 2. Species richness in Italy. Total richness includes all the 468 species considered in the analyses. HABITAT richness includes only the 121 species listed under the Habitat directive. BIRD richness includes only the 83 species listed under the Bird directive.

Considering that roughly 11% of Italy is committed to conservation as a protected area, it is clear that PAs covered a disproportionate percentage of areas with low diversity values (mainly corresponding to high mountains) and with medium to high diversity values, whereas areas with the highest diversity are underrepresented (Fig. 3). When we combined Natura 2000 with the existing protected areas, the general distribution of richness values did not change, even though a higher percentage of areas was covered.

Considering only BIRD species, protected areas covered a disproportionate percentage of areas with high diversity values. However, the areas with highest diversity values were underrepresented. Adding SPAs to existing protected areas, the percentage of areas with high and medium-high diversity that were protected increased and all the areas with the maximum possible species richness are covered, with the exception of the areas in Sardinia (Fig. 3). Considering only HABITAT species, protected areas and protected areas plus SCIs showed similar distributions (Fig. 3): high percentages of areas with low and medium to high diversity values were protected.

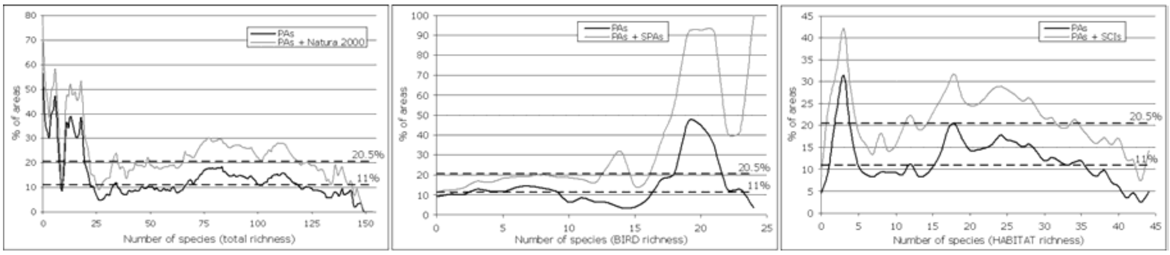

Figure 3. Richness values covered by protected areas and Natura 2000 areas. Total richness includes all the 468 species considered in the analyses. HABITAT richness includes only the 121 species listed under the Habitat directive. BIRD richness includes only the 83 species listed under the Bird directive. The two dashed bold lines in each graph indicate the level of coverage provided by existing protected areas (11%) and by protected areas plus Natura 2000 (20.5%).

Protected areas did not include any portion of the area occupied by two BIRD species (2.4% of the BIRD species), and for 74 BIRD species (89.2%) the representation target was met only partially. With the addition of SPAs to existing protected areas (Table 2) the number of total gap species was unchanged but the number of partial gap species fell to 62 (74.7%). On average protected areas covered < 16% (SD 14.2%) of the area occupied by BIRD species, and adding SPAs markedly increased the percentage of area covered (mean [SD] = 26.4% [18.1%]). Among HABITAT species, protected areas did not covered any portion of the area occupied by 7 species (5.8% of the HABITAT species) and for 85 species (70.2%) the representation target was met only partially. With the addition of SCIs, no species was left unprotected (Table 2) and the number of partial gap species dropped to 68 (56.2%). On average, protected areas covered 14.3% of the area occupied by each HABITAT species (SD 16.5%). Adding SCIs to protected areas, the mean area protected per species increased up to 25.8% (SD 18.5%).

Taxon	# of species	All Species (Pas)		ECB Species (PAs + SPA)		ECH Species (PAs + SCI)	
		Total Gap	Partial Gap	Total Gap	Partial Gap	Total Gap	Partial Gap
Mammals	98	1	42	\	\	0	12
Birds	243	2	178	2	62	\	\
Reptiles	44	1	33	\	\	0	22
Amphibians	36	6	21	\	\	0	20
Fish	47	1	43	\	\	0	14

Table 2. Number of gap species per taxonomic group.

Areas with high irreplaceability values (Fig. 4) occurred across the entire peninsula, but particularly in the eastern alpine range, along the coasts (north-east and north and central west), in the central Apennines, on the main plains (corresponding to the most important wet areas), in Sardinia, and on the smaller islands. For BIRD species the spatial distribution of irreplaceability values (Fig. 4) was different, with the most important areas occurring in the main plains south of the eastern alpine range (midmountain areas and the northeastern coast), in Sicily and Sardinia, and on the smaller islands. Irreplaceability values for HABITAT species followed the distribution of the irreplaceability calculated for all the species (Fig. 4) even though Sardinia was much more important.

For all the species, protected areas covered 18.3% of the areas with irreplaceability values of 1 and 20.5% of the areas with an irreplaceability value > 0.8. For HABITAT species, protected areas covered 19.7% of the totally irreplaceable areas and 25.6% of the areas with irreplaceability > 0.8. With the addition of SCIs, 53.8% of the totally irreplaceable areas and 49.8% of the areas with irreplaceability > 0.8 were protected. For BIRD species, protected areas covered 14.3% of the totally irreplaceable areas and 11.9% of the areas with irreplaceability > 0.8. With the addition of SPAs, 28.5% of the totally irreplaceable areas and 27.5% of the areas with irreplaceability > 0.8 were covered. The irreplaceability values calculated considering the existence of protected areas and Natura 2000 showed a spatial distribution that was different from the general pattern described above. In fact, most of the areas with high irreplaceability and no coverage were located along the coasts, in the lowlands and in Sardinia (Fig. 4).

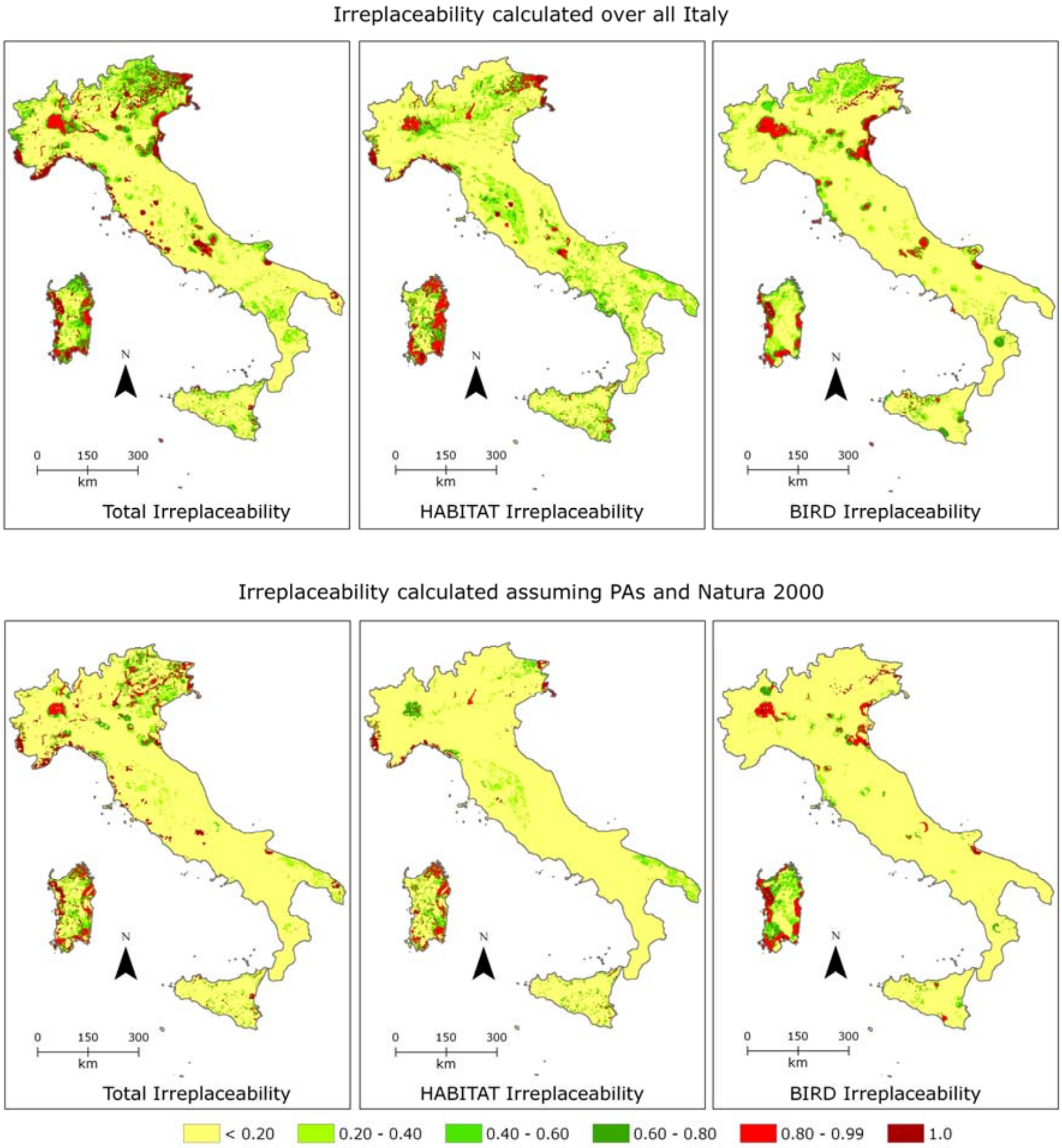

Figure 4. Maps of irreplaceability values in Italy. Total irreplaceability is calculated considering all the species; HABITAT irreplaceability is calculated considering only the 121 species of concern under the Habitat directive; BIRD irreplaceability is calculated considering only the 83 species of concern under the Bird directive. The three maps on the top represent irreplaceability values calculated without considering protected areas and Natura 2000 areas; the three maps on the bottom represent irreplaceability values calculated considering protected areas and Natura 2000 areas.

Measuring conservation status in the Natura2000 network

In protected areas alone, most of the BIRD species (from 69.9% to 80.7%) were classified as CE, En or Vu (Table 3). With the addition of SPAs to protected areas the percentages did not change markedly (from 63.9% to 69.9%) but the number of CE species decreased (Table 3). For HABITAT species, a minimum of 44.4% and a maximum of 55.6% were classified in protected areas as CE, En or Vu (Table 3); with the addition of SCIs, the percentage changed to a minimum of 33.3% and a maximum of 38.9%.

	PAs			**PAs + Natura2000[1]**		
	CE	**EN**	**VU**	**CE**	**EN**	**VU**
ECB species	24 - 27	12 - 18	22 - 22	12 - 16	19 - 20	22 - 22
ECH species	2 - 2	0 - 1	6 - 7	2 - 2	0 - 1	4 - 4

Table 3. Number of BIRD and HABITAT species classified as Critically Endangered (CE), Endangered (EN) or Vulnerable (VU) in protected areas and in protected areas plus Natura 2000 areas. [1] For BiRD species only SPAs have been considered; for HABITAT species only SCIs have been considered.

Discussion

To perform our analyses, we used a combination of DMs and extent of occurrences, an approach that has proved to be useful in conservation planning exercises (Maiorano et al. 2006). We collected a dataset with the most updated available information on species presence and ecology. Whenever possible (73% of the DMs), we validated the DMs with data on species presence independently collected in the field. The results of our validation process showed the reliability of our dataset, although better validation datasets could improve the process. In particular, we were unable to quantify the commission error associated with our DMs (i.e., how much our DMs overestimate species distribution) because no absence data were available.

Like any gap and irreplaceability analysis, our results depended heavily on the representation targets that were used. Our targets followed an approach that has been used previously (Rodrigues et al. 2004; Maiorano et al. 2006) and that is based on a strong assumption: species with restricted ranges require a more precautionary approach. In fact species with small ranges tend to be rare in terms of range size and in terms of local abundance (Lawton 1993; Gaston et al. 1997). The use of targets in conservation-planning exercises is always problematic (Soulé & Sanjayan 1998; Solomon et al. 2003; Svancara et al. 2005; but see Pressey et al. 2003; Desmet & Cowling 2004; Moilanen 2007), but in setting our targets we followed a clear logic: very localized species should have their entire range reserved, while widespread species (those for which we set a 10% target) should be on average neutral for the analysis (more than 10% of Italy is already covered by PAs). However, the 500 km2 and 25,000 km2 thresholds were admittedly somewhat arbitrary and our results should be interpreted accordingly.

Integration of Natura 2000 and protected areas

In the human-dominated Italian landscape, and more in general in Europe, it is not practical to ignore the existing reserve network when new conservation areas are identified (Pressey & Cowling 2001). Even though the Natura 2000 network does not explicitly provide a framework for integration of existing protected areas, one can assume that SCIs and SPAs are being used to fill the conservation gaps. However, generally speaking, the Italian Natura 2000 network does not seem to be particularly well integrated with the existing protected areas. In fact, like most protected area systems (Powell et al. 2000; Scott et al. 2001; Oldfield et al. 2004) both SCIs and SPAs tended to overrepresent highland areas and avoid lowlands with higher agricultural values. Moreover, the median size of the areas was small, with a tendency for smaller areas in lowlands. The same pattern has been documented in Italian protected areas (Maiorano et al. 2006) over which both SCIs and SPAs largely overlap.

Even though the Natura 2000 network increased the area devoted to conservation in Italy (from 11% of the protected areas to 20%) and many areas rich in biodiversity are included in the network (Fig. 3), some areas with a large number of species were still unprotected. Furthermore, protected areas had conservation gaps: 11 species were not covered by any protected area, 317 species were only partially covered, and many highly irreplaceable areas were not in protected areas. SCIs covered all the total gap species listed in the Habitat directive but SPAs did not cover the 2 total gap species listed in the Bird directive and the entire Natura 2000 network did not significantly improve the coverage of high irreplaceability areas.

Particularly important is the case of Sardinia, where many BIRD species are present (usually with endemic subspecies), but the number of protected areas and SPAs is extremely low and none of them are present in the areas of highest diversity. This is particularly striking if one considers that breeding birds probably represent the most well-known taxa among vertebrates and that Important Bird Areas (usually used as the basis for the identification of SPAs) in Sardinia represent an important conservation component and cover more than 20% of the region. However, all the areas with high irreplaceability values (Fig. 4) should be carefully considered for future establishment of conservation areas, and further field investigations should be considered.

Our results can easily be explained in light of how SCIs and SPAs have been selected in Italy. The selection process has been carried out by 21 different administrative units that provided a list of candidate areas to the central unit (the Ministry of the Environment). The process has been coordinated in the sense that each administrative unit has followed a common set of criteria to define SCIs and SPAs (criteria that are common to all EU member states; e.g. Annex III of the Habitat Directive) but in many cases other issues (mainly political and economic) have driven the selection process, which has resulted in a final network that is inefficient and incomplete. However, Natura 2000 can be improved easily by selecting a few more areas to cover the areas that, according to the irreplaceability analyses, do not meet their representation target.

Conservation status in Natura 2000

One of the main objectives of the Natura 2000 network is to maintain populations of species that are listed in a "favorable conservation status". Although there are no precise quantitative criteria that define "favorable conservation status" we suggest that the IUCN red list criteria (and in particular criterion D) could be used to define the conservation status of a species in the Natura 2000 network.

Red list criteria provide a quick means with which to assess a species' probability of extinction given its current population size and threats it faces. They have been used widely in conservation planning exercises (Ceballos et al. 1998; Allen et al. 2001; but see Possingham et al. 2002) and as a tool to evaluate achievements in conservation (Butchart et al. 2005). In fact, even though they provide a relatively coarse level of resolution, they are highly developed and widely tested, at least for the most well-known taxa (i.e., mammals and birds) (Rodrigues et al. 2006). Moreover, criterion D is a fairly conservative estimate of a minimum viable population for many vertebrate species (Reed et al. 2003; Brooks et al. 2006), which allows for safer extrapolations over large areas.

We have provided an estimate of the number of animals of BIRD and HABITAT species living in the Natura 2000 network in Italy. We assumed a uniform distribution of individuals over the entire DM. We recognize that habitat suitability models cannot be used as proxies for densities of individuals (van Horne 1983; Tyre et al. 2001; Jenkins et al. 2003) and, besides densities, many other factors that we have not been able to consider in our models could have biased our results, from small-scale habitat characteristics, to population processes (competition, predation, etc.), to historical factors that have shaped the distribution of the species.

However, we know our population estimates are not accurate or precise, and that criterion D alone can not provide an overall measure of the conservation status of a species. Moreover, we recognize that other factors (e.g., dispersal, fragmentation) can greatly influence the viability of a species (Carroll et al. 2004), especially in the part of the Italian Natura 2000 network embedded in a human-dominated landscape. Yet, our results are clear: even if SCIs and SPAs were to be effectively integrated with the existing protected areas, most of the species could not be considered supported, and this conclusion is particularly applicable to BIRD species with population sizes that in Italy are known to a high level of precision.

Conclusions

The Natura 2000 network is probably stronger than our analyses suggest. The system is based on a site-specific expert-based strategy (at least in Italy) and is driven by direct and detailed knowledge of local diversity. This approach can have many potential advantages from a practical, social and political point of view (Loiselle et al. 2003), but it also has many potential disadvantages, such as its tendency to produce misallocated and biased conservation schemes (Pressey et al. 1993; Cowling et al. 2003).

We did not seek to undermine the importance of Natura 2000 (that, by the way, may be doing a great job for biodiversity features that we have not considered); rather we wished to highlight the main limitations of the approach. Taken together, our results suggest that the Italian Natura 2000 network, although it does not cover the entire range of vertebrate biodiversity, does represent, together with existing protected areas, an important component on which to build future conservation efforts. Further studies are necessary to evaluate the importance of the Natura 2000 network for all those biodiversity features that we have not considered, namely plant species, vegetation communities, and invertebrates. Particularly important is the case of all the marine species for which SCIs represent the main conservation option.

To improve the system, four main points should be considered: (1) The Natura 2000 network cannot constitute the only conservation tool because in the highly fragmented, human-dominated European landscape, biodiversity conservation requires excessively large PAs (more than 20% of Italy is insufficient). (2) The matrix around Natura 2000 should be considered and managed as a functional part of the system, with limitations and control over human activities in areas outside the system to facilitate conservation activities inside. (3) Species viability should be explicitly considered in the management of Natura 2000, with functional connections being implemented between sites to obtain a real network of areas and not simply a “collection” of areas. (4) Biodiversity conservation in Europe should focus more on natural processes: emphasis on pattern rather than process is in many respects understandable, but if one wants to conserve biodiversity indefinitely, ecological and evolutionary mechanisms should also be considered (Smith et al. 1993; Moritz 2002; Rouget et al. 2005; Salomon et al. 2006).

The Natura 2000 system should be integrated into a more general conservation strategy, where it can represent the starting point from which conservation efforts can be developed. If it is thought of as representing the end point of all the EU conservation policies, it will inevitably fail.

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CHAPTER 6: SIZE-DEPENDENT RESISTANCE OF PROTECTED AREAS TO LAND-USE CHANGE

Introduction

Protected areas (PAs) are widely recognized as the most important tool available for "in situ" conservation (Bruner et al. 2001; Sinclair et al. 2002; Sànches-Azofeifa et al. 2003; Chape et al. 2005). However, in many cases it has been demonstrated that PAs do not represent adequately the biodiversity of a region (Pressey et al. 1993; Scott et al. 1993; Rodrigues et al. 1999; Maiorano et a. 2006). The recent world GAP analysis found that at least 12 of the species considered (terrestrial vertebrates) are not covered by any existing PA and that 75% of the species considered do not achieve their representation target (Rodrigues et al. 2003, 2004). Moreover, different studies have demonstrated that existing PA networks are too small to represent a viable solution for conservation of biodiversity, especially in human dominated landscapes (Tilman et al. 1994; Carroll et al. 2004; Maiorano et al. submitted).

One of the most important threats facing PAs is land-use change and related habitat loss (Sala et al. 2000; Brooks et al. 2002). In particular, Hoekstra et al. (2005) have demonstrated that habitat conversion exceeds habitat protection by a ratio of 8:1 in temperate grasslands and Mediterranean forests and 10:1 in more than 140 ecoregions.

Different studies have analyzed the effectiveness of PAs and the results are not so clear. Bruner et al. (2001) examined the effectiveness of PAs in the tropics, drawing on survey data to support the conclusion that parks have been effective at preventing land clearing within their boundaries. However, Vanclay (2001) re-examined the same dataset and obtained different results, concluding that the results obtained by Bruner et al. (2001) remain equivocal. Other studies have demonstrated that PAs have been effective in preventing deforestation and habitat loss (Nagendra et al. 2004; Nepstad et al. 2006) but there are many cases where existing PAs have not been able to stop habitat degradation (Schwartzman et al. 2000; Curran et al. 2004; Fuller et al. 2004; Sigel et al. 2006, Verburg et al. 2006; Gaveau et al. 2007).

Most of the studies have dealt with tropical PAs located in areas where high human population growth rates, land-use intensification and loss of natural habitat are the main features (Houghton 1994; Dobson et al. 1997; Matson et al. 1997; Lambin et al. 2003; Sodhi et al. 2004; Brown et al. 2005; Lepers et al. 2005).

It is not clear if the results of these studies can be applied to the completely different context found in the Mediterranean basin (Falcucci et al. 2007). We report here the first analysis of the efficacy of PAs at stopping (or at least reducing) habitat degradation in a human dominated landscape. Our study area is the Italian peninsula and our hypothesis is that there is a relationship among the size of PAs and the resistance that PAs can offer to land-use change, with smaller areas showing influences of surrounding environment and larger areas acting much more effectively.

Materials and methods

We measured land-cover/land-use change from 1990 to 2000 using two CORINE Land cover maps, one for 1990 (CLC1990) and one for 2000 (CLC2000). The two maps are part of the program started in 1985 by the European Community to generate digital land-use/land-cover maps covering the European continent (EC 1993) and were produced using satellite images (Landsat5 TM for CLC1990 and Landsat7 ETM+ for CLC2000) and other ancillary data (Digital Elevation Model, hydrology, and aerial photos). The maps have a spatial detail comparable to that of a paper map on a scale of 1:100,000 and a hierarchical legend with 5 classes at the first level. For the purposes of the analyses we divided the 5 classes into two main categories: artificial land-use/land-cover classes (Class 1: artificial surfaces; Class 2: agricultural areas) and natural land-use/land-cover classes (Class 3: forests and seminatural areas; Class 4: wetlands; Class 5: water bodies).

We obtained a detailed dataset on PAs from Maiorano et al. (2006), comprising 777 PAs covering 97% of the total area protected in Italy. The dataset considers 5 different types of areas: National Parks, National Reserves, Regional Parks, Regional Reserves and Other Protected Areas. In our analysis we considered each separated polygon as a single PA and we excluded all the areas for which no change in land-use/land-cover was measured inside the same PA or inside a 2.5km buffer built around the area. We obtained a final list of 716 areas that was used in all the analyses. All the protected areas considered have been officially instituted before 1990, or the same area was subject to some level of conservation before 1990.

For each PA considered we calculated the total rate of change in land-use/land-cover (number of cells that changed from artificial to natural and viceversa over total number of cells), the rate of change towards natural land-use/land-cover (number of cells that changed from artificial to natural over total number of cells that where classified as artificial in 1990), and the rate of change towards artificial land-use/land-cover (number of cells that changed from natural to artificial over total number of cells that where classified as natural in 1990).

Since the quantification of reserve performance in slowing or halting land-use/land-cover change is best measured against a baseline that describe the trajectory of the change (Nepstad et al. 2006), we have considered 3 different buffers around each area (1km; 2.5km; 5km) and we have calculated land-use/land-cover change (total change, change towards natural and change towards artificial) inside the 3 buffers.

Our analyses are sensitive to coregistration errors between land cover maps of different years and between land cover and park boundaries. We therefore performed all the analysis using three different cell sizes: 100m, 200m, and 300m.

Given the highly skewed distribution of the land-use/land-cover change dataset, we used non-parametric statistics to make the comparisons among different types of PAs and to measure the relationship among PAs size and inhibition of land-use/land-cover change. We used Mann-Whitney U-tests to compare land-use/land-cover changes (total change, change towards natural, change towards artificial) inside PAs and inside the 3 buffers. We used Kruskal-Wallis one-way analysis of variance for independent samples to compare inhibition of total land-use/land-cover change across the 5 reserve types followed by Fisher's protected least significant difference to perform comparisons among reserve types.

To explore the relationship existing between size of PAs and their efficacy in slowing down or stopping land-use/land-cover change (total change, change towards natural and change towards artificial) we divided the 716 PAs into categories according to their size and for each size-class we calculated the mean rates of change (total change, change towards natural, change towards artificial). Since the subdivision into classes is subjective and can potentially influence the outcome of the analyses we used 9 different possible legends, with a minimum of 7 classes and a maximum of 11 classes (Tab. 1).

Legend 1	<1 (250)	1-5 (205)	5-10 (68)	10-50 (105)	50-100 (35)	100-500 (36)	>=500 (17)				
Legend 2	<1 (250)	1-5 (205)	5-10 (68)	10-50 (105)	50-100 (35)	100-250 (22)	250-500 (14)	>=500 (17)			
Legend 3	<0.5 (186)	0.5 - 1 (64)	1 - 2.5 (117)	2.5 - 5 (88)	5-10 (68)	10-50 (105)	50-100 (35)	100-500 (36)	>=500 (17)		
Legend 4	<1 (250)	1-5 (205)	5-10 (68)	10-25 (63)	25-50 (42)	50-100 (35)	100-250 (22)	250-500 (14)	>=500 (17)		
Legend 5	<0.5 (186)	0.5 - 1 (64)	1 - 2.5 (117)	2.5 - 5 (88)	5-10 (68)	10-50 (105)	50-100 (35)	100-250 (22)	250-500 (14)	>=500 (17)	
Legend 6	<0.5 (186)	0.5 - 1 (64)	1 - 2.5 (117)	2.5 - 5 (88)	5-10 (68)	10-25 (63)	25-50 (42)	50-100 (35)	100-250 (22)	250-500 (14)	>=500 (17)
Legend 7	<0.5 (186)	0.5 - 1 (64)	1 - 2.5 (117)	2.5 - 5 (88)	5-10 (68)	10-50 (105)	50-100 (35)	100-250 (22)	>=250 (31)		
Legend 8	<1 (250)	1-5 (205)	5-10 (68)	10-25 (63)	25-50 (42)	50-100 (35)	100-250 (22)	>=250 (31)			
Legend 9	<0.5 (186)	0.5 - 1 (64)	1 - 2.5 (117)	2.5 - 5 (88)	5-10 (68)	10-25 (63)	25-50 (42)	50-100 (35)	100-250 (22)	>=250 (31)	

Table 1. Classification schemes adopted to classify PAs according to their area (measured as km 2). The number of PAs in each size-class is indicated in parentheses.

We used Spearman rank correlation analysis to test for a relationship between the mean land-use/land-cover change (total change, change towards natural and change towards artificial) and the size of PAs, performing a total of 27 (3 cell sizes by 9 different legends) correlation analyses. We performed the same analyses for the three types of buffers for a total of 81 possible combinations (3 buffer sizes by 3 cell sizes by 9 different legends).

The previous analyses treat the PAs as a group. We also compared individual PAs with their own surroundings to determine what percentage of individual parks are functioning. For each size-class we calculated the percentage of PAs that changed more than their buffers towards natural land-use/land-cover and the percentages of PAs that changed more than their buffers towards artificial land-use/land-cover and we measured the Spearman rank correlation existing between the two percentages and the size of PAs (3 cell sizes by 3 buffer sizes by 9 legends by 2 types of change for a total of 162 correlation analyses).

To investigate if the same pattern is common to the entire study area, we divided the Italian peninsula into 3 homogeneous ecological macro-regions (the Alps, the

Apennines, flat areas and coastal plains) modifying the scheme proposed by Falcucci et al. (2007). For each of the macro-regions we performed the same Spearman rank correlation analyses described above.

All significance tests were carried out at the $\alpha = 0.05$ level in SAS software.

Results

Differences in land-use/land-cover change among the 5 reserve types were significant (cell size 100m: K-W=19.065, p=0.0008; cell size 200m: K-W=15.219, p=0.043; cell size 300m: K-W=13.036, p=0.0111) with national PAs experiencing the lowest rates of land-use/land-cover change (National Reserves: median=1.1%, interquartile range=7.4%; National Parks: median=4.9%, interquartile range=13.66%) and with regional and local PAs experiencing the highest (Regional Parks: median=5.12%, interquartile range= 11.84%; Regional Reserves: median=7.76%, interquartile range=21.08%; Other Protected Areas: median=8.7%, interquartile range=23.81%). However, only National Reserves were significantly different from all the other types of reserves, while no significant difference was measured among National Parks, Regional Parks, Regional Reserves and Other Protected Areas due to the high variability of the land-use/land-cover change rates.

Overall, PAs changed significantly less than the surrounding buffers in 89% of the comparisons (8 out of 9 possible comparisons; Tab. 2). No clear pattern was found considering only changes towards natural land-use/land-cover classes: in one case we found a significant difference indicating that the rate of change towards natural land-use/land-cover classes in PAs was higher than that in the buffer, but we had also 5 comparisons where the difference was not statistically significant and 3 comparisons with a significant difference but in the opposite direction (Tab. 2). Considering only changes towards artificial classes we found that the rate of change was always significantly higher in the buffers than in the PAs (Tab. 2).

	100m	200m	300m
Total change			
PA vs B1km	-2.3 (0.0201)	-3.9 (<0.0001)	-4.5 (<0.0001)
PA vs B2.5km	-2.3 (0.0240)	-3.8 (0.0002)	-5.1 (<0.0001)
PA vs B5km	-0.9 (0.3697)	-3.2 (0.0014)	-4.5 (<0.0001)
Natural change			
PA vs B1km	1.8 (0.0769)	-0.2 (0.8435)	-2.6 (0.0096)
PA vs B2.5km	1.6 (0.1092)	-0.6 (0.5645)	-3.9 (<0.0001)
PA vs B5km	2.7 (0.0067)	-0.5 (0.6470)	-4.0 (<0.0001)
Artificial change			
PA vs B1km	-10.9 (<0.0001)	-12.2 (<0.0001)	-9.2 (<0.0001)
PA vs B2.5km	-13.4 (<0.0001)	-14.9 (<0.0001)	-13.4 (<0.0001)
PA vs B5km	-12.4 (<0.0001)	-14.6 (<0.0001)	-13.2 (<0.0001)

Table 2. Mann-Whitney U-test comparisons (z values followed by p values in parentheses) of land-use/land-cover changes (total change, change towards natural, change towards artificial) between PAs and the 1km buffer (B1km), PAs and the 2.5km buffer (B2.5km), and PAs and the 5km buffer (B5km). The three columns (100m, 200m, 300m) give the results obtained with 3 different cell sizes.

We found a clear relationship among PAs size and the total rate of land-use/land-cover change experienced: 96% of the available combinations (26 out of 27) gave a significant negative rank correlation, with the only exception of 1 combination with r = -0.714 and p=0.0713 (Fig. 1; Supporting material: Tab. 1). No clear relationship between buffer area and total rate of land-use/land-cover change was found for the three buffers, with only 6% of the combinations (5 out of 81) being significant (Supporting material: Tab. 1).

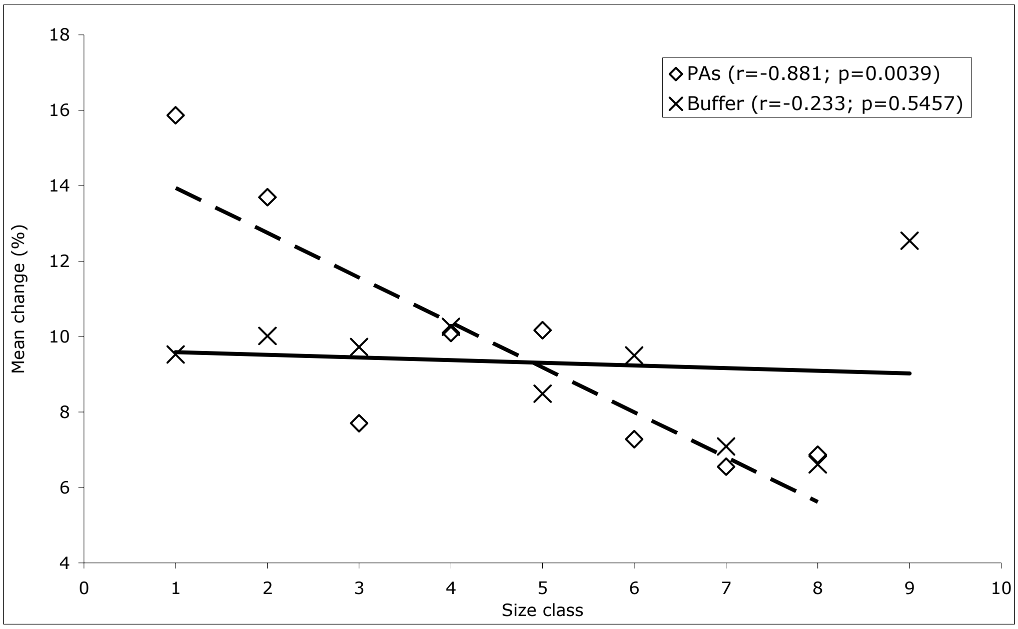

Figure 1. Spearman rank correlations measured between mean land-use/land-cover change and PAs size, and between mean land-use/land-cover change and the buffers size. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 1).

We measured a positive correlation between PA size and the rate of change towards natural land-use/land-cover classes, but the relationship was not strongly supported by the analyses: at the $\alpha = 0.05$ level only 11% of the combinations (3 out of 27) were significant, even though a number of combinations were nearly significant (48% of the combinations at the $\alpha = 0.1$ level; Fig. 2, Supporting material: Tab. 2). On the contrary we found a strong negative correlation among PA size and the rate of change towards artificial land-use/land-cover classes, with 59% of the combinations (16 out of 27) being significant at the $\alpha=0.05$ level and 85% of the combinations (23 out of 27) being significant at the $\alpha=0.1$ level (Supporting material: Tab. 2). No significant correlation (both at the $\alpha=0.05$ and at the $\alpha=0.1$ level) was found for the 3 buffers (Supporting material: Tab. 2).

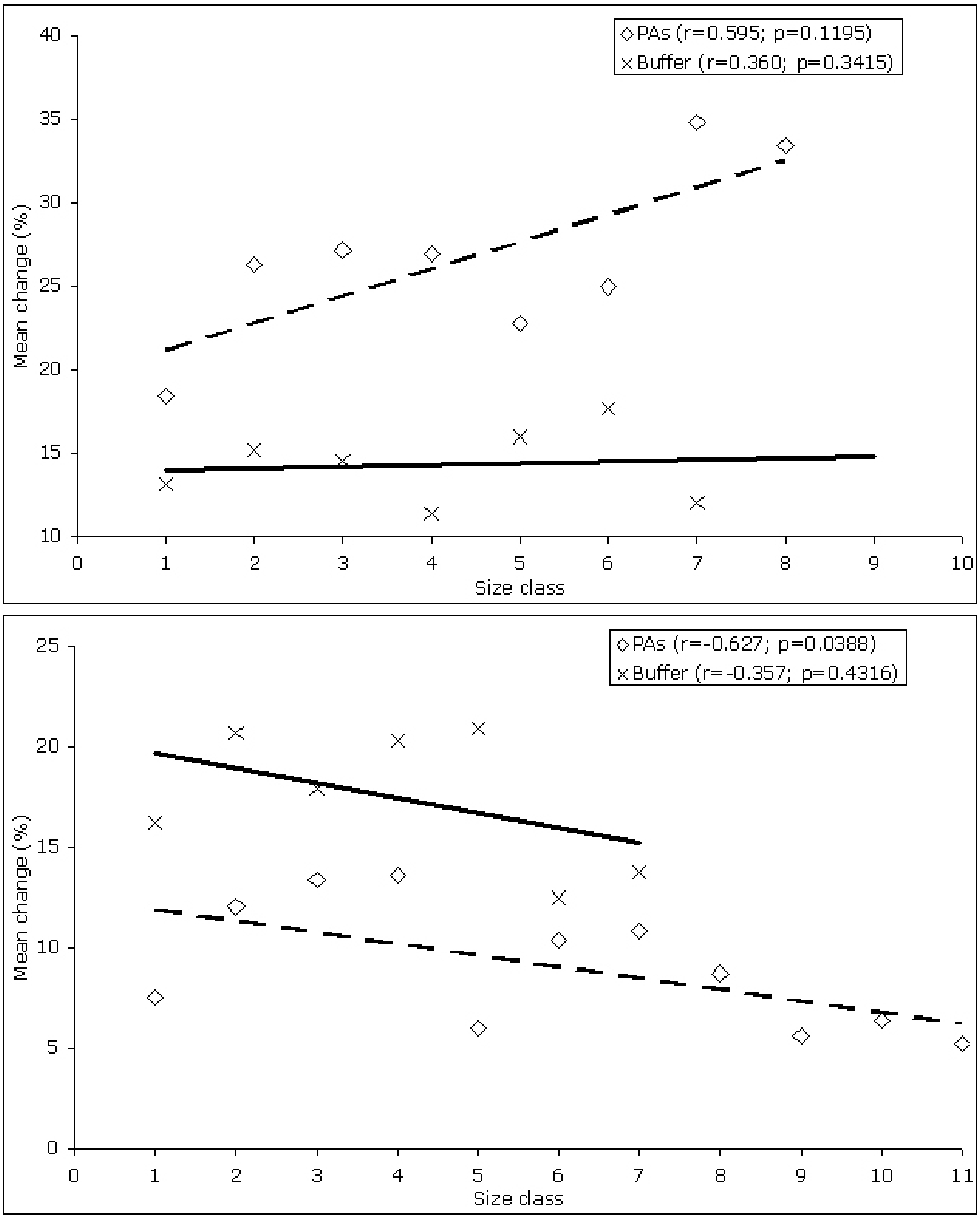

Figure 2. Spearman rank correlations measured between mean land-use/land-cover change (towards natural classes on the top; towards artificial classes on the bottom) and PAs size, and between mean land-use/land-cover change (towards natural classes on the top; towards artificial classes on the bottom) and the buffers size. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 2).

We measured a strong positive correlation among PA size and the percentage of PAs that changed more than their buffers towards natural land-use/land-cover, with 100% of the combinations being significant (Fig. 3; Supporting material: Tab. 3). We also measured a negative correlation among PA size and the percentage of PAs that changed more than their buffers towards artificial land-use/land-cover, with 31% of the combinations (25 out of 81) being significant at the α=0.05 level and 60% of the combinations (49 out of 81) being significant at the α=0.1 level.

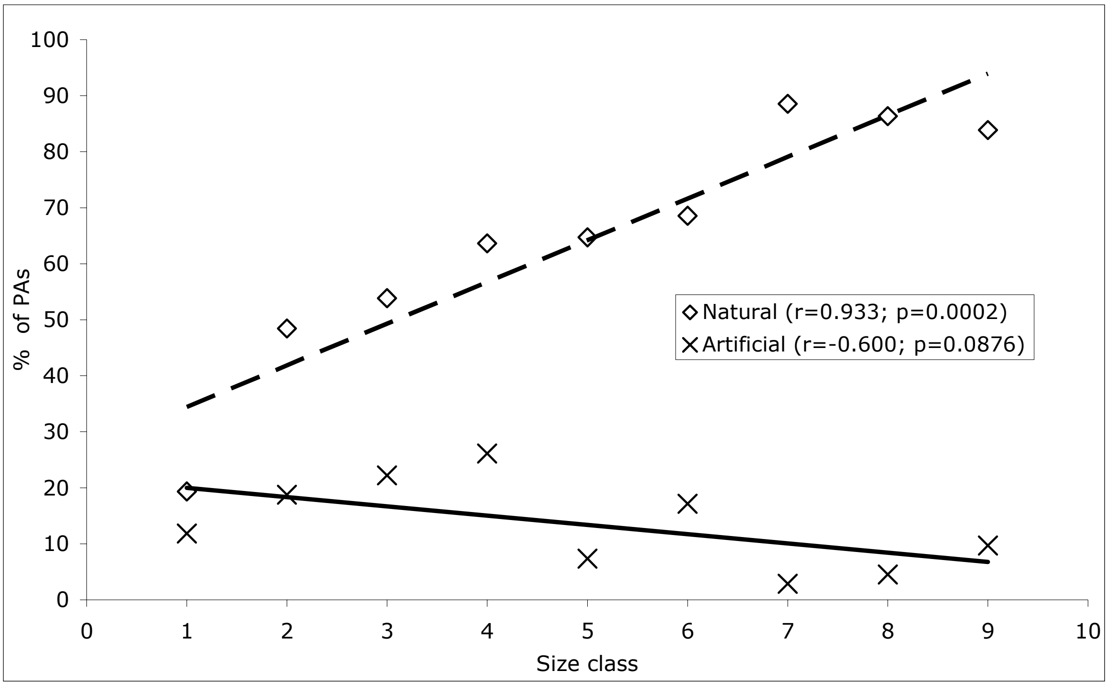

Figure 3. Spearman rank correlations measured between PA size and % of PAs with a mean change towards natural land-use/land-cover class that is greater than the respective buffer, and between PA size % of PAs with a mean change towards artificial land-use/land-cover class that is greater than the respective buffer. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 3).

The Alps

The results obtained over the Alpine region were comparable to those obtained at the national level, even though the number of PAs in each size-class was lower (total number of PAs in the Alps is 159), especially considering that only 40 PAs are bigger than 5 km2. Also in this case, we found a clear relationship among PAs size and the total rate of land-use/land-cover change experienced: 100% of the available combinations gave a significantly negative rank correlation (Fig. 4; Supporting material: Tab. 4). The relationship between buffer size and total rate of land-use/land-cover change was comparable to that obtained for the PAs but weaker, with 57% of the available combinations (46 out of 81) being significant (Supporting material: Tab. 4).

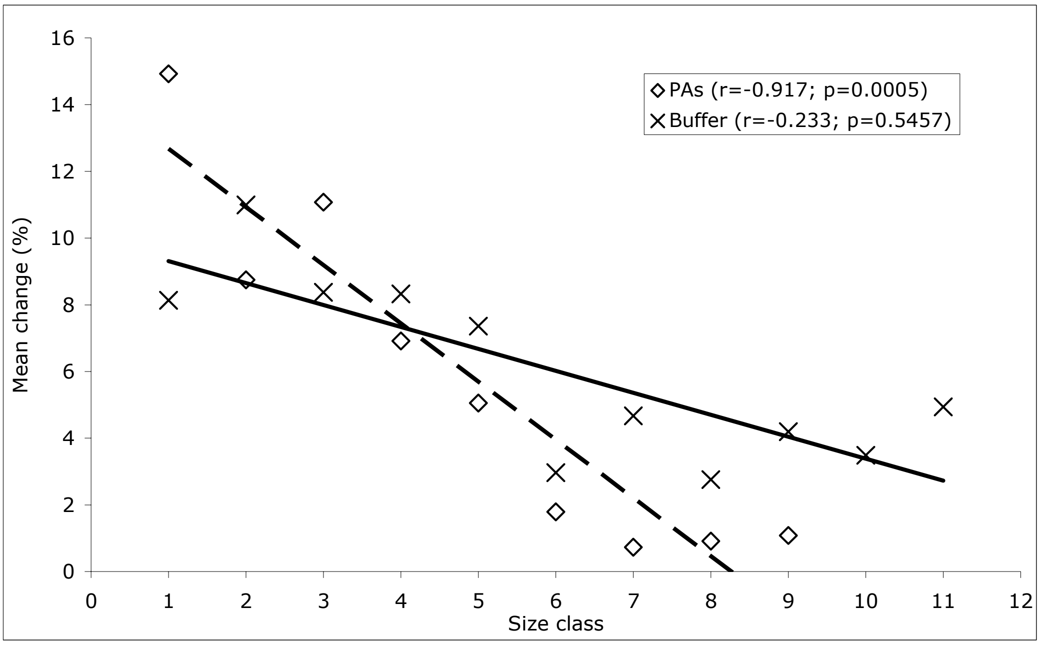

Figure 4. Alpine region: Spearman rank correlations measured between mean land-use/land-cover change and PAs size, and between mean land-use/land-cover change and the buffers size. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 4).

No clear relationship was measured in the Alpine region between PA size or buffer size and the rate of change towards natural land-use/land-cover classes (only 11% of the correlations were significant for PAs, and no significant result was found for the buffers; Fig. 5, Supporting material: Tab. 5). On the contrary, 100% of the combinations gave a significant negative correlation between PA size and the rate of change towards artificial land-use/land-cover classes (Supporting material: Tab. 5), while only 35% of the combinations (28 out of 81) gave comparable results for the buffer (Supporting material: Tab. 5).

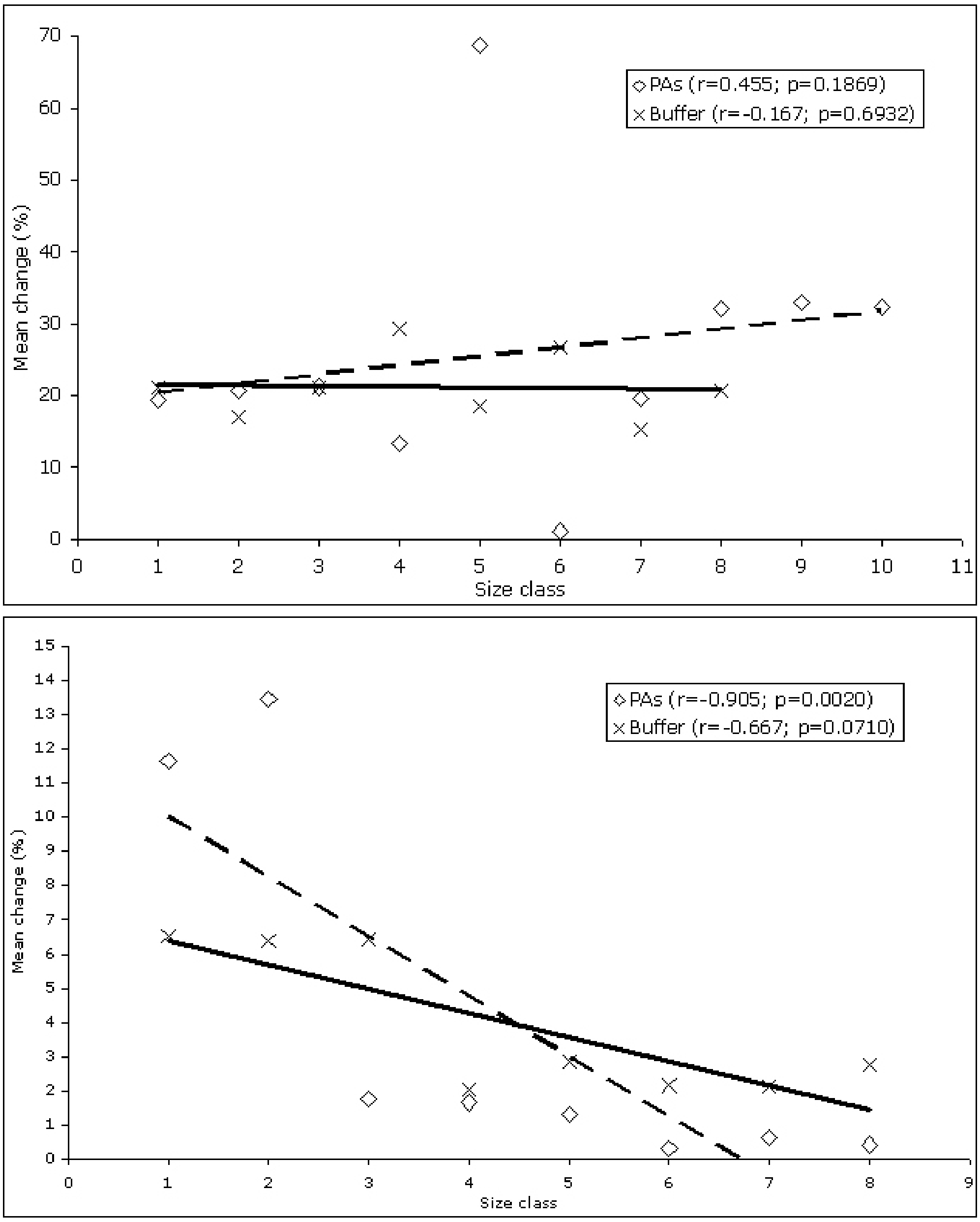

Figure 5. Alpine region: Spearman rank correlations measured between mean land-use/land-cover change (towards natural classes on the top; towards artificial classes on the bottom) and PAs size, and between mean land-use/land-cover change (towards natural classes on the top; towards artificial classes on the bottom) and the buffers size. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 5).

We measured a fairly strong positive correlation among Alpine PA size and the percentage of Alpine PAs that changed more than their buffers towards natural land-use/land-cover, with 63% of the combinations (51 out of 81) being significant at the α=0.05 level and 74% of the combinations (67 out of 81) being significant at the α=0.1 level (Fig. 6; Supporting material: Tab. 6). We also measured a negative correlation among Alpine PA size and the percentage of Alpine PAs that changed more than their buffers towards artificial land-use/land-cover, with 42% of the combinations (34 out of 81) being significant at the α=0.05 level and 67% of the combinations (54 out of 81) being significant at the α=0.1 level (Fig. 6; Supporting material: Tab. 6).

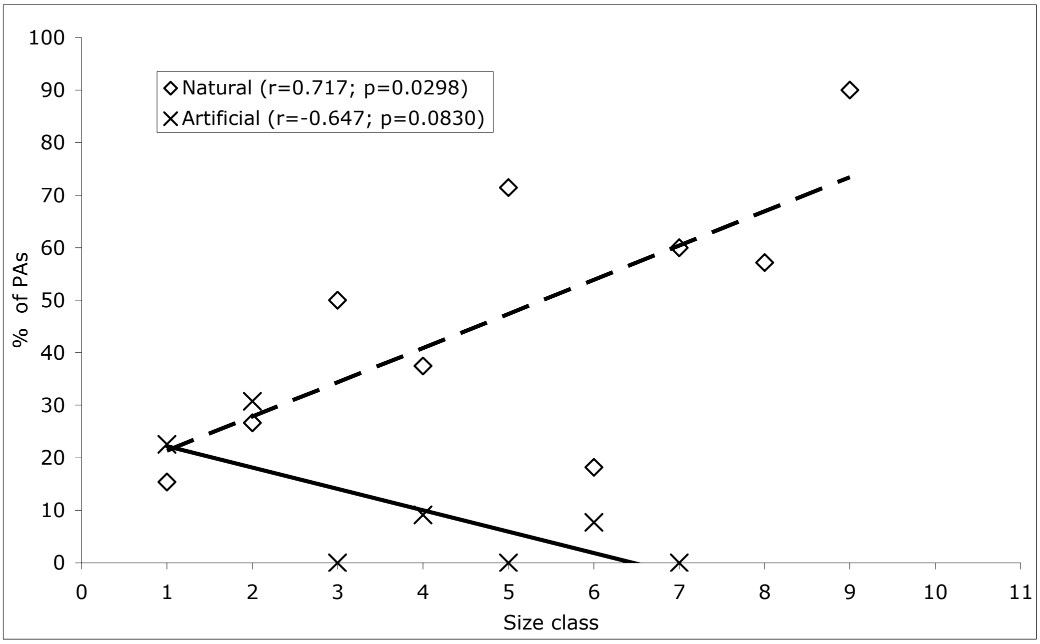

Figure 6. Alpine region: Spearman rank correlations measured between PA size and % of PAs with a mean change towards natural land-use/land-cover class that is greater than the respective buffer, and between PA size and % of PAs with a mean change towards artificial land-use/land-cover class that is greater than the respective buffer. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 6).

The Apennines

The Apennines hosts 167 PAs, but only 32 of them are bigger than 50 km2, making it difficult to obtain a reasonable number of PAs per size-class in higher classes. The results obtained for the Apennines were not comparable to those obtained at the national level. In fact, we did not find any clear relationship among PAs size and the total rate of land-use/land-cover change experienced, with only 37% of the available combinations giving a significantly negative rank correlation (Fig. 7; Supporting material: Tab. 7). The correlation between buffer size and total rate of land-use/land-cover change was never significant (Supporting material: Tab. 7).

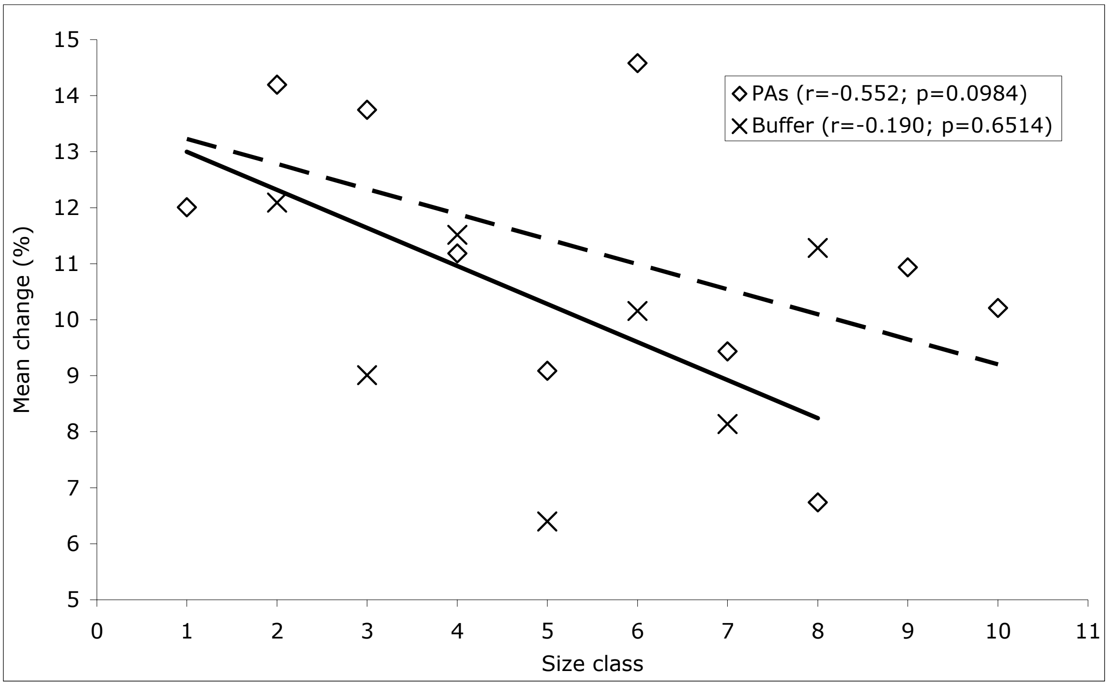

Figure 7. Apennines: Spearman rank correlations measured between mean land-use/land-cover change and PAs size, and between mean land-use/land-cover change and the buffers size. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 7).

No clear relationship was measured in the Apennines between PA size or buffer size and the rate of change towards natural land-use/land-cover classes (only 26% of the correlations was significant for PAs, and only 4% of the correlations was significant for the buffers; Fig. 8, Supporting material: Tab. 8). The same was true for changes towards artificial land-use/land-cover classes (Fig. 8, Supporting material: Tab. 8).

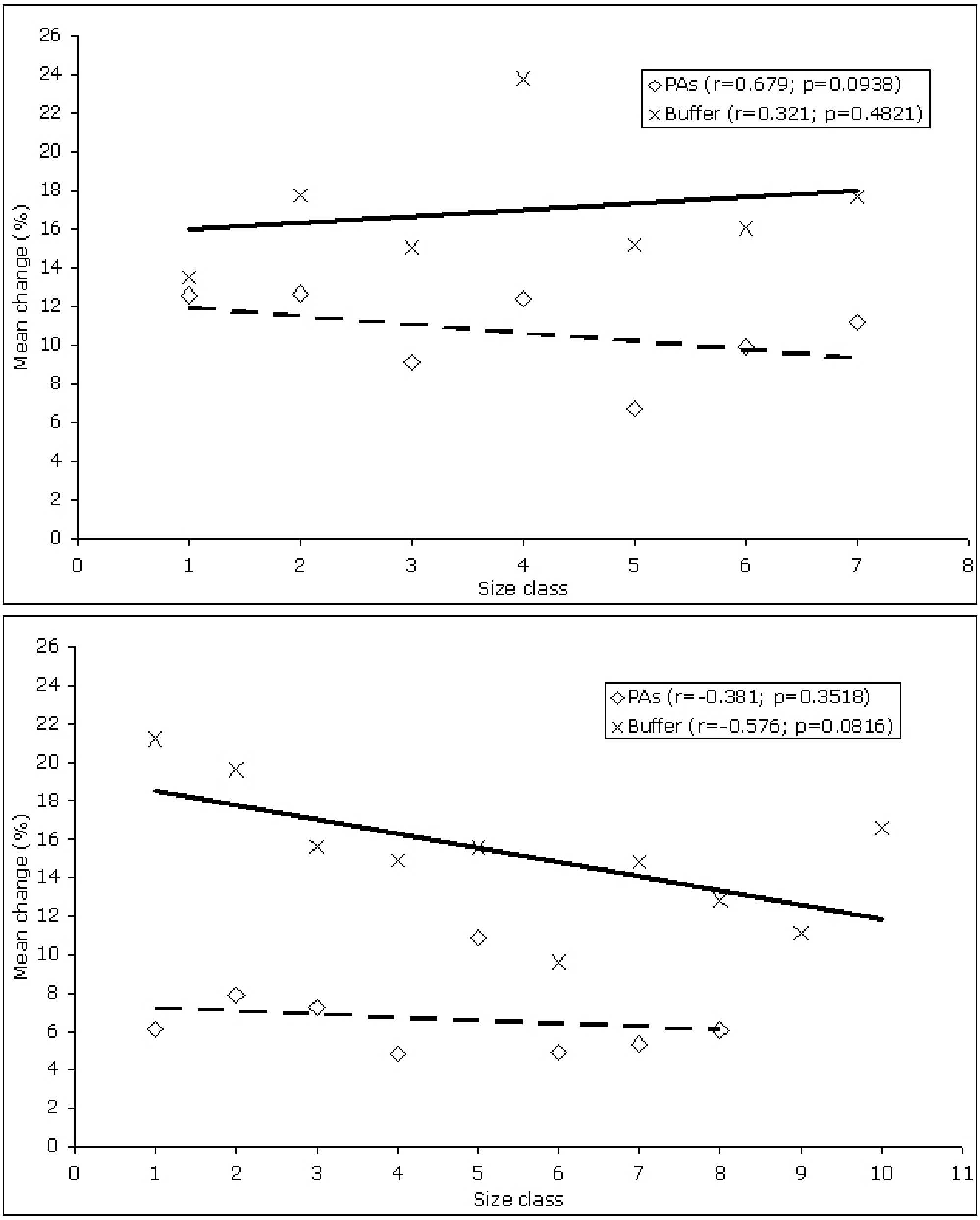

Figure 8. Apennines: Spearman rank correlations measured between mean land-use/land-cover change (towards natural classes on the top; towards artificial classes on the bottom) and PAs size, and between mean land-use/land-cover change (towards natural classes on the top; towards artificial classes on the bottom) and the buffers size. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 8).

The positive correlation between PA size and the percentage of PAs that changed more than their buffers towards natural land-use/land-cover was, in the Apennines, extremely strong, with 97% of the combinations (79 out of 81) being significant at the α=0.05 level and 100% of the combinations being significant at the α=0.1 level (Fig. 9; Supporting material: Tab. 9). On the contrary, no significant correlation was found between PA size and the percentage of PAs that changed more than their buffers towards artificial land-use/land-cover (Fig. 9; Supporting material: Tab. 9).

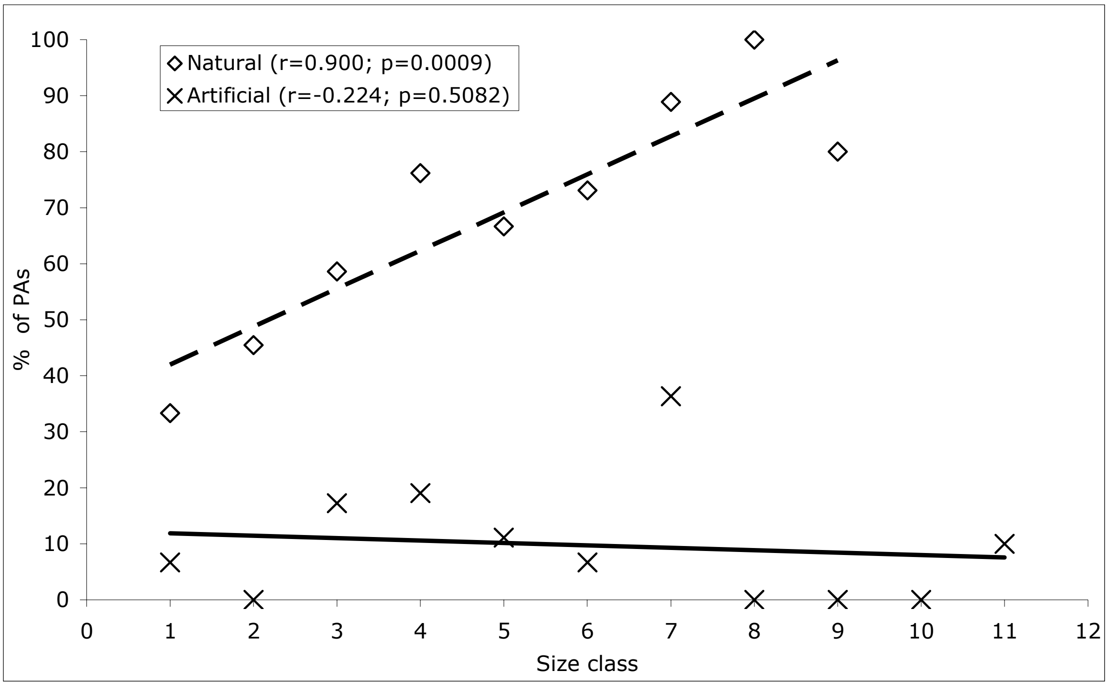

Figure 9. Apennines: Spearman rank correlations measured between PA size and % of PAs with a mean change towards natural land-use/land-cover class that is greater than the respective buffer, and between PA size and % of PAs with a mean change towards artificial land-use/land-cover class that is greater than the respective buffer. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 9).

Flat areas and coastal plains

Flat areas and coastal plains host 390 PAs, but also in this case only 34 of them are bigger than 50 km^2. In this macro-region, the negative correlation between PAs size and the total rate of land-use/land-cover change is significant at the α=0.05 level for 70% (19 out of 27) of the combinations, and for 100% of the combinations at the α=0.1 level (Fig. 10; Supporting material: Tab. 10). The correlation between buffer size and total rate of land-use/land-cover change was never statistically significant (Supporting material: Tab. 10).

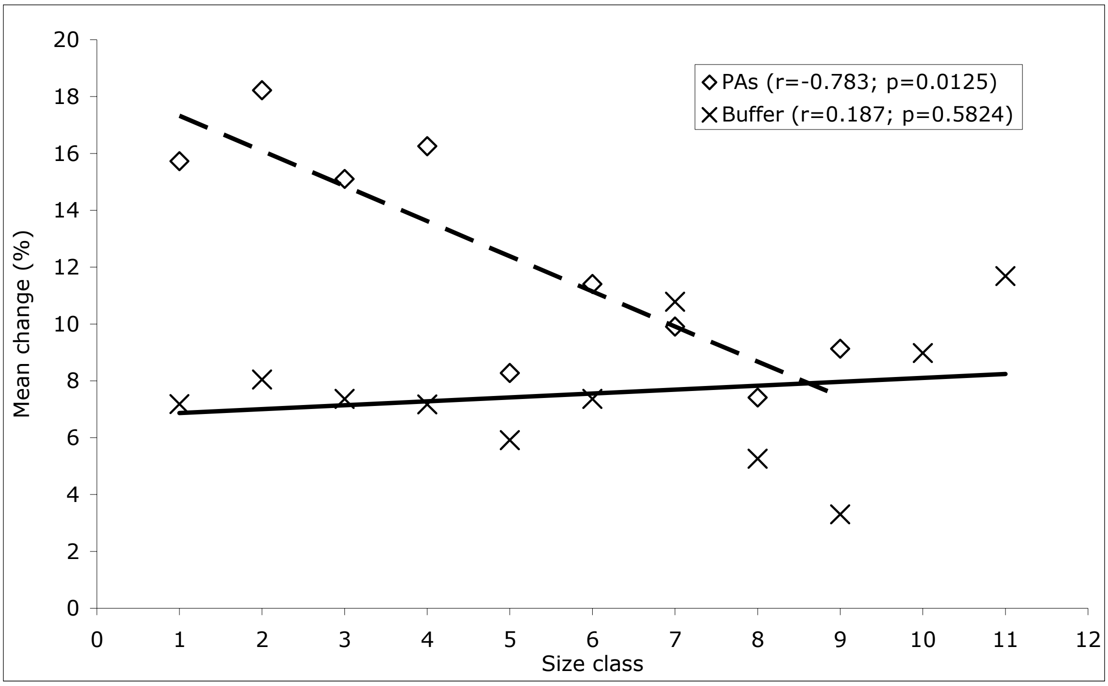

Figure 10. Flat areas and coastal plains: Spearman rank correlations measured between mean land-use/land-cover change and PAs size, and between mean land-use/land-cover change and the buffers size. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 10).

No clear relationship was measured between PA size or buffer size and the rate of change towards natural land-use/land-cover classes (Fig. 11; Supporting material: Tab. 11). The same was true for changes towards artificial land-use/land-cover classes (Supporting material: Tab. 11).

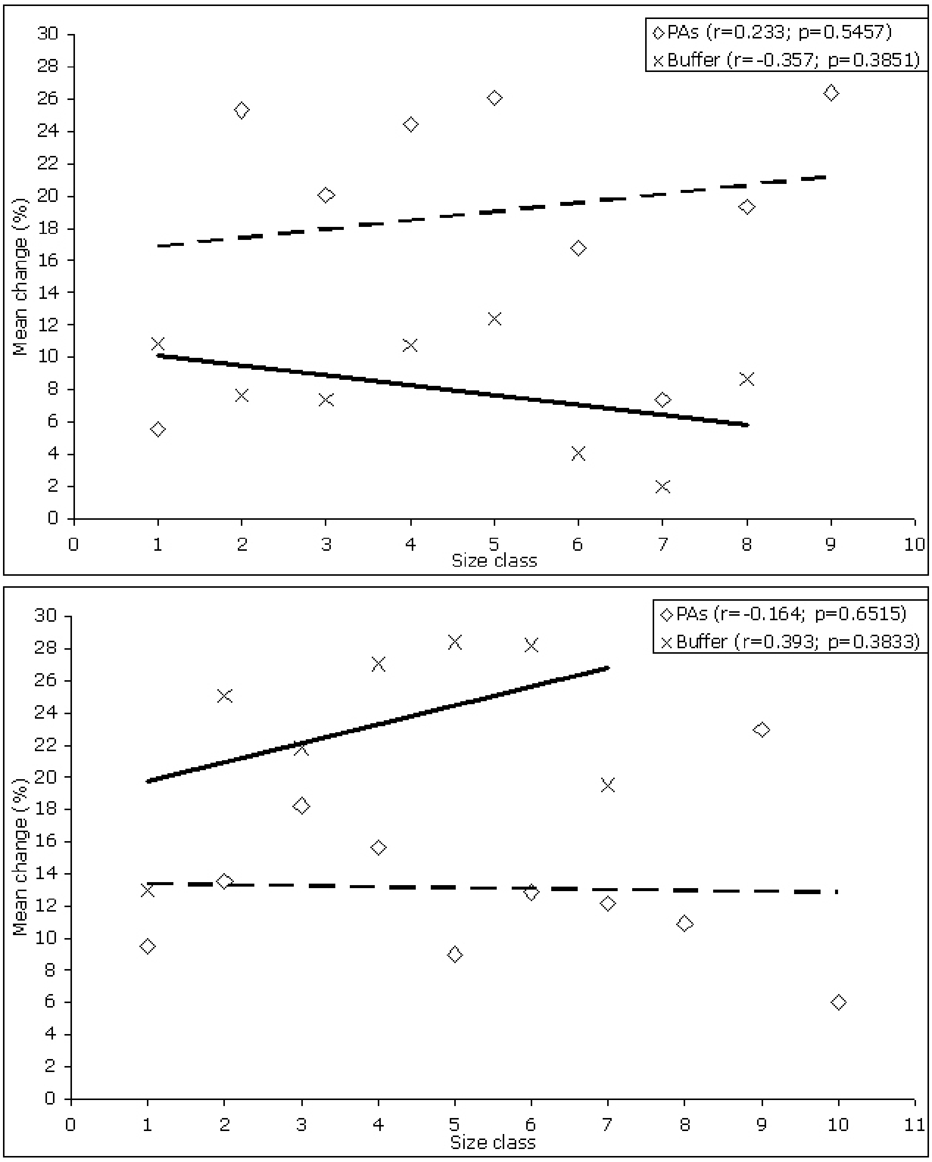

Figure 11. Flat areas and coastal plains: Spearman rank correlations measured between mean land-use/land-cover change (towards natural classes on the top; towards artificial classes on the bottom) and PAs size, and between mean land-use/land-cover change (towards natural classes on the top; towards artificial classes on the bottom) and the buffers size. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 11).

The positive correlation between PA size and the percentage of PAs that changed more than their buffers towards natural land-use/land-cover was extremely strong, with 100% of the combinations being significant at the α=0.05 level (Fig. 12; Supporting material: Tab. 12). On the contrary, no significant correlation was found between PA size and the percentage of PAs that changed more than their buffers towards artificial land-use/land-cover (Supporting material: Tab. 12).

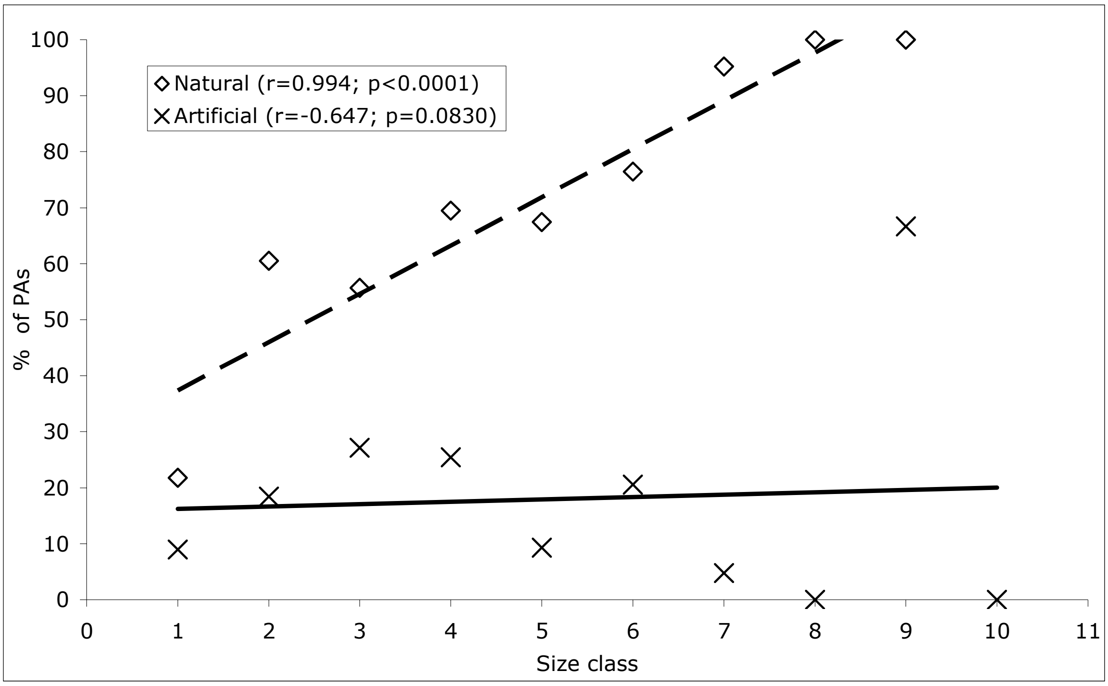

Figure12. Flat areas and coastal plains: Spearman rank correlations measured between PA size and % of PAs with a mean change towards natural land-use/land-cover class that is greater than the respective buffer, and between PA size and % of PAs with a mean change towards artificial land-use/land-cover class that is greater than the respective buffer. Only the r value with the median p-value is shown. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends (Supporting material: Tab. 12).

Discussion

Even though any analysis of land-use/land-cover change is subject to technical problems (Coppin and Bauer 1996; Petit and Lambin 2001, 2002), our results can be considered fairly robust. In fact, while the two Corine Land Cover maps have been created from two basically different datasets (the Corine Land Cover 1990 has been realized using Landsat5 images and other ancillary maps; the Corine Land Cover 2000 has been realized using Landsat7 ETM+ images and a different set of ancillary maps), both have been created using the same methodology and the same legend (for more information visit the official Corine Land Cover web site at the European Topic Center on

Land Use and Spatial Information: http://terrestrial.eionet.europa.eu/CLC2000). Moreover, EEA (2006) found that Corine Land Cover 2000, considering the 3rd level of its hierarchical legend, classified 87% of 8115 field samples correctly. We have no validation for Corine Land Cover 1990 but we can assume that the error rate was not much different. Moreover, it is important to consider that we have used only the 1st level classes of the Corine legend, thus further minimizing the errors.

The PAs coverage has already been extensively checked by Maiorano et al. (2006) and most of the errors have been corrected. We cannot be sure that co-registration errors (between the two land-use/land-cover maps and between the land-cover maps and the PA map) do not remain, thus we performed our analyses using three different cell sizes.

Our main finding is that PAs (both considered singularly and as a system) have been effective at protecting the ecosystems within their borders, even in areas with significant land use pressures (see our results for flat areas and coastal plains). In fact, comparing PAs with neighboring areas, we clearly demonstrated that PAs are effective at slowing down land-use/land-cover change. Bruner et al. (2001) obtained similar results but used a dataset coming from questionnaires, and their study was harshly criticized (Vanclay 2001), mainly because their dataset was considered anecdotal rather than substantive. Our results, on the contrary, are based on a dataset that can be easily verified (both Corine Land Cover maps can be freely downloaded from the European Environmental Agency web-site, and the PAs coverage can be obtained from the Italian Ministry of the Environment – Directorate for Nature Conservation), and that have been extensively validated in the field. Moreover, our results go further: if we consider land-use/land-cover change without distinguishing the direction of change, there is a clear and statistically significant negative correlation among mean change and PA size. In particular we were not able to find any relationship among PAs considered all together and change towards natural land-use/land-cover classes, but we found that PAs change towards artificial land-use/land-cover classes significantly less than neighboring control areas.

We have also been able to confirm our initial hypothesis. In particular, we found that the capacity of PAs to slow down habitat degradation and to favor habitat restoration is clearly related to their size, with smaller areas that on the average follow the dominant land-use/land-cover change pattern into which they are embedded. Furthermore, at the national level, increasing the size of PAs it is possible to favor the change towards more natural habitats and to slow down the change towards artificial habitats (Fig. 3).

The size of PAs has already been analyzed considering species survival during the SLOSS (Single Large or Several Small) debate (Margules et al. 1982; Soulé and Simberloff 1986; Ovaskainen 2002). A number of papers (reviewed in Ovaskainen 2002) have demonstrated several small PAs are better if the objective is that of maximizing the number of species occurring in a system of conservation areas. However, if the objective is that of maximizing the number of species that will eventually survive, the advantages of large PAs over small PAs are not always clear (Simberloff and Abele 1976), if the objective is that of maximizing the time to extinction large PAs should be the preferred solution (Burkey 1989, 1995, 1997; Ovaskainen 2002), and if the object is that of maximizing the metapopulation capacity of a PA system (Hanski and Ovaskainen 2000) an intermediate solution is the best. Clearly, no single solution is always preferable to the others, because there is no possibility of generalizing number, size, and location of habitat patches needed to preserve biodiversity (Soulé and Simberloff 1986).

However, the problem has never been analyzed considering the efficacy of PAs in slowing and/or halting habitat degradation and in favoring habitat restoration. Our results provide, from this point of view, very clear indications towards the importance of large PAs, not only in pristine environments but also in areas where the main habitat characteristics are and have been shaped by traditional human activities for thousands of year.

This is particularly clear if one analyzes in depth the results that we obtained for the Alps, the Apennines and the flat areas and coastal plains. In fact, even with the obvious interpretation problems (splitting our sample of PAs, we obtained 3 sub-samples with an extremely low number of big PAs, especially for the Alps and the Apennines; this implies that the results obtained for the single macro-regions should be considered with more caution), we obtained a confirmation of our general results both for areas dominated by land-cover changes towards natural habitats, and for areas with a really strong human influence. In particular, the Alps showed a pattern of land-use/land-cover change in PAs similar to that obtained for the entire peninsula. Falcucci et al. (2007) showed that the Italian alpine range changed from 1990 to 2000 towards a more natural condition, and we have demonstrated that PAs along the alpine range changed towards natural land-use/land-cover classes more than the rest of the macro-region (with bigger PAs changing the most) while the change towards artificial land-use/land-cover classes was lower (with bigger PAs changing the least).

From 1990 to 2000, the Apennines also showed a marked change towards natural land-use/land-cover classes (Falcucci et al. 2007). We were not able to demonstrate a correlation among PA size and the change towards artificial land-use/land-cover classes (Fig. 9). This is probably linked to the fact that most of the mid mountain areas along the Apennines (i.e. most of the areas along the borders of PAs) have been abandoned and naturally reforested (Falcucci et al. 2007), favoring a land-use/land-cover change towards natural classes. However, we demonstrated that PAs almost always change towards natural land-use/land-cover classes more than their buffers, and larger PAs change the most (Fig. 9).

The flat areas and the coastal plains are the areas where the contrast among PAs and their buffers is greatest. In these areas, most of the changes in the 1990-2000 time frame have been oriented towards artificial land cover classes (Falcucci et al. 2007) and, even though all PAs were efficient in slowing down changes towards artificial land-use/land-cover classes, we found a particularly clear relationship among PA size and their efficacy.

Generally, our results suggest that small protected areas are not going to be viable in the long term if they are considered islands surrounded by a "human dominated ocean". In fact, it is highly probable that "negative" land-use/land-cover changes will continue in the foreseeable future, even exacerbated by climate change (Chapin III et al. 2000). This implies that, in a human dominated landscape, small PAs will sooner or later (probably later than the surrounding areas) lose all the characteristics for which they have been established. However, small PAs are, in most of the western European countries, the only option available for in situ conservation, and actually they are important for conservation of small habitat features and of species with limited habitat requirements.

In fact, we are not suggesting that we should dismiss PAs. Conservation areas are still the most important tool available for conservation (Chape et al. 2005) but we need to make an important shift in our strategies. We cannot rely solely PAs alone but we need to change our management strategy and devote much more attention to the non-protected matrix in which PAs must survive.

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Supporting materials

		PAs		Buffer 1km		Buffer 2.5km		Buffer 5km	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value
Legend 1	100	-0.714	0.0713	-0.179	0.7017	-0.071	0.8790	0.036	0.0939
	200	-0.857	0.0137	-0.214	0.6445	-0.143	0.7599	-0.179	0.7017
	300	-0.821	0.0234	-0.214	0.6445	-0.179	0.7017	0.000	1.0000
Legend 2	100	-0.714	0.0465	-0.214	0.6103	-0.143	0.7358	0.262	0.5309
	200	-0.833	0.0102	-0.238	0.5702	-0.190	0.6514	-0.238	0.5702
	300	-0.810	0.0149	-0.238	0.5702	-0.214	0.6103	-0.119	0.7789
Legend 3	100	-0.833	0.0053	-0.267	0.4879	-0.133	0.7324	0.017	0.9661
	200	-0.900	0.0009	-0.333	0.3807	-0.250	0.5165	0.283	0.4600
	300	-0.817	0.0072	-0.367	0.3317	-0.233	0.5457	0.017	0.9661
Legend 4	100	-0.733	0.0246	-0.267	0.4879	-0.150	0.7001	0.285	0.4581
	200	-0.833	0.0053	-0.333	0.3807	-0.200	0.6059	-0.167	0.6682
	300	-0.833	0.0053	-0.300	0.4328	-0.200	0.6059	-0.183	0.6368
Legend 5	100	-0.830	0.0029	-0.321	0.3655	-0.244	0.5334	0.067	0.8548
	200	-0.891	0.0005	-0.370	0.2931	-0.309	0.3848	-0.345	0.3282
	300	-0.830	0.0029	-0.394	0.2600	-0.297	0.4047	-0.152	0.6761
Legend 6	100	-0.836	0.0013	-0.373	0.2589	-0.236	0.4841	0.091	0.7904
	200	-0.891	0.0002	-0.436	0.1797	-0.327	0.3259	-0.291	0.3855
	300	-0.855	0.0008	-0.436	0.1797	-0.264	0.4334	-0.164	0.6307
Legend 7	100	-0.933	0.0002	-0.583	0.0992	-0.133	0.7324	-0.033	0.9322
	200	-0.933	0.0002	-0.767	0.0159	-0.250	0.5165	-0.417	0.2646
	300	-0.850	0.0037	-0.717	0.0298	-0.233	0.5457	-0.033	0.9322
Legend 8	100	-0.881	0.0039	-0.524	0.1827	-0.048	0.9108	0.048	0.9103
	200	-0.881	0.0039	-0.714	0.0465	-0.119	0.7789	0.071	0.8665
	300	-0.881	0.0039	-0.571	0.1390	-0.119	0.7789	0.095	0.8225
Legend 9	100	-0.915	0.0002	-0.588	0.0739	-0.139	0.0709	-0.006	0.9867
	200	-0.915	0.0002	-0.818	0.0038	-0.261	0.4671	-0.309	0.3848
	300	-0.867	0.0012	-0.697	0.0251	-0.176	0.6272	-0.042	0.9074

Supporting materials: table 1. Spearman rank correlations between mean land-use/land-cover change and PAs size, between mean change and the 1km-buffer size, between mean change and the 2.5km-buffer size, between mean change and 5km-buffer size. The correlation was measured across 3 different cell sizes and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		Protected areas				Buffer 1km				Buffer 2.5km				Buffer 5km			
		Natural		Artificial		Natural		Artificial		Natural		Artificial		Natural		Artificial	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value
Legend 1	100	0.393	0.3833	-0.857	0.0137	0.321	0.4821	-0.500	0.2532	0.464	0.2939	-0.357	0.4316	0.429	0.3374	-0.286	0.5345
	200	0.750	0.0522	-0.750	0.0522	0.357	0.4316	-0.393	0.3833	0.429	0.3374	-0.429	0.3374	0.429	0.3374	-0.286	0.5345
	300	0.679	0.0938	-0.607	0.1482	0.429	0.3374	-0.321	0.4821	0.429	0.3374	-0.286	0.5345	0.536	0.2152	-0.143	0.7599
Legend 2	100	0.548	0.1600	-0.881	0.0039	0.286	0.4927	-0.452	0.2604	0.524	0.1827	-0.571	0.1390	0.595	0.1195	-0.524	0.1827
	200	0.595	0.1195	-0.667	0.0710	0.310	0.4556	-0.381	0.3518	0.500	0.2070	-0.619	0.1017	0.476	0.2329	-0.476	0.2329
	300	0.690	0.0580	-0.619	0.1017	0.357	0.3851	-0.286	0.4927	0.571	0.1390	-0.524	0.1827	0.475	0.2329	-0.286	0.4927
Legend 3	100	0.450	0.2242	-0.800	0.0096	0.317	0.4064	-0.433	0.2440	0.450	0.2242	-0.483	0.1875	0.433	0.2440	-0.033	0.9322
	200	0.633	0.0671	-0.717	0.0298	0.317	0.4064	-0.367	0.3317	0.467	0.2054	-0.450	0.2242	0.300	0.4328	0.000	1.0000
	300	0.667	0.0499	-0.600	0.0876	0.360	0.3415	0.000	1.0000	0.433	0.2449	-0.267	0.4879	0.483	0.1875	0.083	0.8312
Legend 4	100	0.583	0.0992	-0.850	0.0037	0.267	0.4879	-0.383	0.3085	0.467	0.2054	-0.517	0.1544	0.467	0.2054	-0.550	0.1250
	200	0.533	0.1392	-0.667	0.0499	0.250	0.5165	-0.350	0.3558	0.450	0.2242	-0.550	0.1250	0.317	0.4064	-0.483	0.1875
	300	0.617	0.0769	-0.567	0.1116	0.233	0.5457	-0.183	0.6368	0.517	0.1544	-0.483	0.1188	0.333	0.3807	-0.283	0.4600
Legend 5	100	0.442	0.2004	-0.842	0.0022	0.273	0.4458	-0.455	0.1869	0.479	0.1615	-0.539	0.1076	0.491	0.1497	-0.200	0.5796
	200	0.600	0.0667	-0.709	0.0217	0.273	0.4458	-0.406	0.2443	0.467	0.1739	-0.600	0.0667	0.321	0.3655	-0.152	0.6761
	300	0.709	0.0217	-0.648	0.0425	0.394	0.2600	0.006	0.9867	0.564	0.0897	-0.382	0.2763	0.418	0.2291	-0.030	0.9338
Legend 6	100	0.464	0.1509	-0.845	0.0010	0.291	0.3855	-0.427	0.2899	0.473	0.1420	-0.545	0.0827	0.409	0.2115	-0.282	0.4011
	200	0.600	0.0510	-0.636	0.0353	0.273	0.4171	-0.382	0.2466	0.464	0.1509	-0.536	0.0890	0.236	0.4841	-0.209	0.5372
	300	0.673	0.0233	-0.627	0.0388	0.355	0.2847	0.018	0.9577	0.555	0.0767	-0.355	0.2847	0.336	0.3118	-0.045	0.8944
Legend 7	100	0.233	0.5457	-0.783	0.0125	0.083	0.8312	-0.267	0.4879	0.367	0.3317	-0.383	0.3085	0.317	0.4064	-0.067	0.8647
	200	0.467	0.2054	-0.700	0.0358	0.083	0.8312	-0.200	0.6059	0.300	0.4328	-0.467	0.2054	0.083	0.8312	-0.033	0.9322
	300	0.600	0.0876	-0.583	0.0992	0.183	0.6368	0.217	0.5755	0.483	0.1875	-0.167	0.6682	0.217	0.5755	0.033	0.9322
Legend 8	100	0.405	0.3199	-0.786	0.0208	0.071	0.8665	-0.143	0.7358	0.357	0.3851	-0.333	0.4198	0.357	0.3851	-0.381	0.3518
	200	0.357	0.3851	-0.690	0.0508	0.095	0.8225	-0.095	0.8225	0.333	0.4198	-0.381	0.3518	0.143	0.7358	-0.333	0.4198
	300	0.524	0.1827	-0.500	0.2070	0.024	0.9554	0.143	0.7358	0.429	0.2894	-0.286	0.4927	0.167	0.6932	-0.190	0.6514
Legend 9	100	0.285	0.4250	-0.794	0.0061	0.115	0.7514	-0.248	0.4888	0.358	0.3104	-0.406	0.2443	0.273	0.4458	-0.176	0.6272
	200	0.479	0.1615	-0.600	0.0607	0.115	0.7514	-0.188	0.6032	0.355	0.3282	-0.394	0.2600	0.042	0.9074	-0.103	0.7770
	300	0.600	0.0667	-0.564	0.0897	0.200	0.5796	0.224	0.5334	0.467	0.1739	-0.152	0.6761	0.176	0.6272	0.030	0.9338

Supporting material: table 2. Spearman rank correlations between mean change towards natural land-use/land-cover classes and PAs size, between mean change towards artificial land-use/land-cover classes and PAs size, between mean change towards natural land-use/land-cover classes and the 1km buffer area, between mean change towards artificial land-use/land-cover classes and the 1km buffer area, between mean change towards natural land-use/land-cover classes and the 2.5km buffer area, between mean change towards artificial land-use/land-cover classes and the 2.5km buffer area, between mean change towards natural land-use/land-cover classes and 5km buffer area, between mean change towards artificial land-use/land-cover classes and 5km buffer area. The correlation was measured across 3 different cell sizes and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		Natural						Artificial					
		Pas vs B1km		PAs vs B2.5km		Pas vs B5km		Pas vs B1km		PAs vs B2.5km		Pas vs B5km	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value
Legend 1	100	0.893	0.0068	0.857	0.0137	0.893	0.0068	-0.714	0.0713	-0.714	0.0713	-0.821	0.0234
	200	0.857	0.0137	0.893	0.0068	1.000	<0.0001	-0.679	0.0938	-0.680	0.0938	-0.714	0.0713
	300	0.821	0.0234	0.857	0.0137	1.000	<0.0001	-0.536	0.2152	-0.679	0.0938	-0.786	0.0362
Legend 2	100	0.833	0.0102	0.762	0.0280	0.833	0.0102	-0.619	0.1017	-0.429	0.2894	-0.761	0.0280
	200	0.762	0.0280	0.857	0.0065	1.000	<0.0001	-0.667	0.0710	-0.476	0.2329	-0.571	0.1390
	300	0.838	0.0093	0.790	0.0195	0.958	0.0002	-0.452	0.2604	-0.571	0.1390	-0.548	0.1600
Legend 3	100	0.900	0.0009	0.917	0.0005	0.950	<0.0001	-0.683	0.0424	-0.717	0.0298	-0.800	0.0096
	200	0.933	0.0002	0.950	<0.0001	1.000	<0.0001	-0.667	0.0499	-0.667	0.0499	-0.633	0.0671
	300	0.917	0.0005	0.933	0.0002	0.983	<0.0001	-0.400	0.2861	-0.667	0.0499	-0.533	0.1392
Legend 4	100	0.883	0.0016	0.833	0.0053	0.867	0.0025	-0.600	0.0876	-0.383	0.3085	-0.812	0.0079
	200	0.817	0.0072	0.900	0.0009	1.000	<0.0001	-0.700	0.0358	-0.433	0.2440	-0.567	0.1116
	300	0.887	0.0014	0.853	0.0034	0.971	<0.0001	-0.427	0.2520	-0.567	0.1116	-0.550	0.1250
Legend 5	100	0.879	0.0008	0.867	0.0012	0.915	0.0002	-0.673	0.0330	-0.491	0.1497	-0.794	0.0061
	200	0.879	0.0008	0.927	0.0001	1.000	<0.0001	-0.697	0.0251	-0.491	0.1497	-0.515	0.1276
	300	0.918	0.0002	0.894	0.0005	0.967	<0.0001	-0.285	0.4250	-0.624	0.0537	-0.564	0.0897
Legend 6	100	0.909	0.0001	0.900	0.0002	0.927	<0.0001	-0.582	0.0604	-0.427	0.1899	-0.834	0.0014
	200	0.882	0.0003	0.945	<0.0001	1.000	<0.0001	-0.645	0.0320	-0.482	0.1334	-0.491	0.1252
	300	0.939	0.0001	0.920	<0.0001	0.975	<0.0001	-0.232	0.4918	-0.627	0.0388	-0.545	0.0827
Legend 7	100	0.900	0.0009	0.917	0.0005	0.933	0.0002	-0.667	0.0499	-0.683	0.0424	-0.800	0.0096
	200	0.933	0.0002	0.983	<0.0001	1.000	<0.0001	-0.667	0.0499	-0.600	0.0876	-0.617	0.0769
	300	0.933	0.0002	0.933	0.0002	0.983	<0.0001	-0.350	0.3558	-0.683	0.0424	-0.600	0.0876
Legend 8	100	0.929	0.0009	0.905	0.0020	0.881	0.0039	-0.619	0.1017	-0.571	0.1390	-0.881	0.0039
	200	0.881	0.0039	0.976	<0.0001	1.000	<0.0001	-0.690	0.0580	-0.524	0.1827	-0.619	0.1017
	300	0.905	0.0020	0.905	0.0020	1.000	<0.0001	-0.429	0.2894	-0.667	0.0710	-0.643	0.0856
Legend 9	100	0.927	0.0001	0.939	<0.0001	0.939	<0.0001	-0.539	0.1076	-0.588	0.0739	-0.855	0.0016
	200	0.915	0.0002	0.988	<0.0001	1.000	<0.0001	-0.588	0.0739	-0.600	0.0667	-0.552	0.0984
	300	0.952	<0.0001	0.952	<0.0001	0.988	<0.0001	-0.285	0.4250	-0.673	0.0330	-0.564	0.0897

Supporting material: table 3. Spearman rank correlations between PA size and % of PAs with a mean change towards natural land-use/land-cover class that is greater than the respective buffer, and between PA size and % of PAs with a mean change towards artificial land-use/land-cover class that is greater than the respective buffer. The correlation was measured across 3 different cell sizes, 3 different buffer sizes, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		PAs		Buffer 1km		Buffer 2.5km		Buffer 5km	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value
Legend 1	100	-0.857	0.0137	-0.714	0.0713	-0.714	0.0713	-0.643	0.1194
	200	-0.857	0.0137	-0.714	0.0713	-0.714	0.0713	-0.679	0.0938
	300	-0.893	0.0068	-0.679	0.0938	-0.714	0.0713	-0.750	0.0522
Legend 2	100	-0.881	0.0039	-0.571	0.1390	-0.667	0.0710	-0.571	0.1390
	200	-0.881	0.0039	-0.571	0.1390	-0.667	0.0710	-0.643	0.0856
	300	-0.862	0.0059	-0.571	0.1390	-0.667	0.0710	-0.643	0.0856
Legend 3	100	-0.917	0.0005	-0.850	0.0037	-0.783	0.0125	-0.667	0.0499
	200	-0.917	0.0005	-0.850	0.0037	-0.783	0.0125	-0.800	0.0096
	300	-0.917	0.0005	-0.817	0.0072	-0.783	0.0125	-0.783	0.0125
Legend 4	100	-0.850	0.0037	-0.583	0.0992	-0.683	0.0424	-0.517	0.1544
	200	-0.900	0.0009	-0.533	0.1392	-0.617	0.0769	-0.533	0.1392
	300	-0.753	0.0191	-0.533	0.1392	-0.653	0.0567	-0.567	0.1116
Legend 5	100	-0.927	0.0001	-0.770	0.0092	-0.770	0.0092	-0.661	0.0376
	200	-0.927	0.0001	-0.770	0.0092	-0.770	0.0092	-0.782	0.0075
	300	-0.906	0.0003	-0.758	0.0111	-0.770	0.0092	-0.745	0.0133
Legend 6	100	-0.909	0.0001	-0.764	0.0062	-0.782	0.0045	-0.645	0.0320
	200	-0.936	<0.0001	-0.736	0.0098	-0.745	0.0085	-0.718	0.0128
	300	-0.847	0.0010	-0.727	0.0112	-0.765	0.0060	-0.709	0.0146
Legend 7	100	-0.954	<0.0001	-0.850	0.0037	-0.833	0.0053	-0.650	0.0581
	200	-0.933	0.0002	-0.850	0.0037	-0.833	0.0053	-0.817	0.0072
	300	-0.950	<0.0001	-0.817	0.0072	-0.833	0.0053	-0.767	0.0159
Legend 8	100	-0.862	0.0059	-0.690	0.0580	-0.690	0.0580	-0.667	0.0710
	200	-0.905	0.0020	-0.690	0.0580	-0.690	0.0580	-0.690	0.0580
	300	-0.881	0.0039	-0.595	0.1195	-0.690	0.0580	-0.738	0.0366
Legend 9	100	-0.918	0.0002	-0.830	0.0029	-0.782	0.0075	-0.709	0.0217
	200	-0.939	<0.0001	-0.830	0.0029	-0.782	0.0075	-0.806	0.0049
	300	-0.915	0.0002	-0.770	0.0092	-0.782	0.0075	-0.794	0.0061

Supporting material: table 4. Alpine region: Spearman rank correlations between mean land-use/land-cover change and PAs size, between mean change and the 1km-buffer size, between mean change and the 2.5km-buffer size, between mean change and 5km-buffer size. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		Protected areas				Buffer 1km				Buffer 2.5km				Buffer 5km			
		Natural		Artificial		Natural		Artificial		Natural		Artificial		Natural		Artificial	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value
Legend 1	100	0.321	0.4821	-0.821	0.0234	-0.072	0.8780	-0.750	0.0522	0.143	0.7599	-0.714	0.0713	0.250	0.5887	-0.607	0.1482
	200	0.429	0.3374	-0.821	0.0234	-0.036	0.9394	-0.750	0.0522	0.143	0.7599	-0.714	0.0713	0.143	0.7559	-0.536	0.2152
	300	0.500	0.2532	-0.821	0.0234	0.286	0.5345	-0.679	0.0938	0.143	0.7599	-0.607	0.1482	0.357	0.4316	-0.607	0.1482
Legend 2	100	0.333	0.4198	-0.810	0.0149	-0.156	0.7128	-0.595	0.1195	0.024	0.9554	-0.667	0.0710	0.000	1.0000	-0.595	0.1195
	200	0.500	0.2070	-0.810	0.0149	-0.095	0.8225	-0.643	0.0856	0.024	0.9554	-0.667	0.0710	-0.167	0.6932	-0.548	0.1600
	300	0.524	0.1827	-0.810	0.0149	0.190	0.6514	-0.571	0.1390	0.000	1.0000	-0.595	0.1195	0.000	1.0000	-0.595	0.1195
Legend 3	100	0.450	0.2242	-0.883	0.0016	0.251	0.5147	-0.867	0.0025	0.283	0.4600	-0.700	0.0358	0.383	0.3085	-0.550	0.1250
	200	0.550	0.1250	-0.867	0.0025	0.250	0.5165	-0.850	0.0037	0.467	0.2054	-0.683	0.0424	0.183	0.6368	-0.517	0.1544
	300	0.683	0.0424	-0.867	0.0025	0.267	0.4879	-0.783	0.0125	0.167	0.6682	-0.617	0.0769	0.450	0.2242	-0.550	0.1250
Legend 4	100	0.433	0.2440	-0.850	0.0037	-0.209	0.5890	-0.550	0.1250	-0.050	0.8984	-0.550	0.1250	-0.033	0.9322	-0.500	0.1705
	200	0.433	0.2440	-0.867	0.0026	-0.167	0.6282	-0.567	0.1116	0.000	1.0000	-0.517	0.1544	-0.117	0.7650	-0.467	0.2054
	300	0.383	0.3085	-0.850	0.0037	0.100	0.7980	-0.483	0.1875	-0.050	0.8984	-0.500	0.1705	-0.033	0.9322	-0.500	0.1705
Legend 5	100	0.491	0.1497	-0.879	0.0008	0.146	0.6876	-0.782	0.0075	0.236	0.5109	-0.709	0.0217	0.103	0.7770	-0.600	0.0667
	200	0.612	0.0600	-0.867	0.0012	0.164	0.6515	-0.794	0.0061	0.382	0.2763	-0.685	0.0289	-0.006	0.9067	-0.576	0.0816
	300	0.697	0.0251	-0.867	0.0012	0.176	0.6272	-0.733	0.0158	0.042	0.9074	-0.648	0.0425	0.127	0.7261	-0.600	0.0667
Legend 6	100	0.482	0.1334	-0.900	0.0002	0.105	0.7591	-0.745	0.0085	0.191	0.5739	-0.664	0.0260	0.118	0.7293	-0.582	0.0604
	200	0.536	0.0890	-0.900	0.0002	0.118	0.7293	-0.745	0.0085	0.327	0.3259	-0.627	0.0388	0.055	0.8734	-0.564	0.0710
	300	0.564	0.0710	-0.891	0.0002	0.164	0.6307	-0.682	0.0208	0.036	0.9155	-0.618	0.0426	0.136	0.6893	-0.582	0.0604
Legend 7	100	0.483	0.1875	-0.900	0.0009	0.267	0.4879	-0.867	0.0025	0.350	0.3558	-0.700	0.0358	0.300	0.1250	-0.550	0.1250
	200	0.567	0.1116	-0.883	0.0016	0.267	0.4879	-0.850	0.0037	0.533	0.1392	-0.683	0.0424	0.117	0.7650	-0.517	0.1544
	300	0.683	0.0424	-0.883	0.0016	0.350	0.3558	-0.783	0.0125	0.233	0.5457	-0.617	0.0769	0.067	0.8647	-0.550	0.1250
Legend 8	100	0.476	0.2329	-0.881	0.0039	-0.095	0.8225	-0.714	0.0465	-0.071	0.8665	-0.690	0.0580	-0.071	0.8667	-0.548	0.1600
	200	0.357	0.3851	-0.905	0.0020	-0.071	0.8665	-0.690	0.0580	0.143	0.7358	-0.667	0.0710	0.143	0.7358	-0.500	0.2070
	300	0.286	0.4927	-0.881	0.0039	0.190	0.6514	-0.595	0.1195	0.095	0.8225	-0.548	0.1600	-0.071	0.8667	-0.619	0.1017
Legend 9	100	0.455	0.1869	-0.915	0.0002	0.224	0.5334	-0.842	0.0022	-0.309	0.3848	-0.721	0.0186	0.321	0.3655	-0.576	0.0816
	200	0.467	0.1739	-0.915	0.0002	0.224	0.5334	-0.818	0.0038	0.479	0.1615	-0.685	0.0289	0.212	0.5563	-0.552	0.0984
	300	0.503	0.1383	-0.903	0.0003	0.333	0.3466	-0.745	0.0133	0.224	0.5334	-0.624	0.0537	0.127	0.7261	-0.612	0.0600

Supporting material: table 5. Alpine region: Spearman rank correlations between mean change towards natural land-use/land-cover classes and PAs size, between mean change towards artificial land-use/land-cover classes and PAs size, between mean change towards natural land-use/land-cover classes and the 1km buffer area, between mean change towards artificial land-use/land-cover classes and the 1km buffer area, between mean change towards natural land-use/land-cover classes and the 2.5km buffer area, between mean change towards artificial land-use/land-cover classes and the 2.5km buffer area, between mean change towards natural land-use/land-cover classes and 5km buffer area, between mean change towards artificial land-use/land-cover classes and 5km buffer area. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		Natural						Artificial					
		Pas vs B1km		PAs vs B2.5km		Pas vs B5km		Pas vs B1km		PAs vs B2.5km		Pas vs B5km	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value
Legend 1	100	0.643	0.1194	0.643	0.1994	0.571	0.1802	-0.667	0.1016	-0.667	0.1016	-0.667	0.1016
	200	0.714	0.0713	0.714	0.0713	0.714	0.0713	-0.667	0.1016	-0.667	0.1016	-0.667	0.1016
	300	0.821	0.0234	0.821	0.0234	0.714	0.0713	-0.541	0.2103	-0.667	0.1016	-0.667	0.1016
Legend 2	100	0.762	0.0280	0.762	0.0280	0.619	0.1017	-0.647	0.0830	-0.647	0.0830	-0.698	0.0544
	200	0.762	0.0280	0.778	0.0229	0.714	0.0465	-0.647	0.0830	-0.647	0.0830	-0.647	0.0830
	300	0.810	0.0149	0.850	0.0075	0.714	0.0465	-0.515	0.1915	-0.647	0.0830	-0.647	0.0830
Legend 3	100	0.667	0.0499	0.667	0.0499	0.600	0.0876	-0.797	0.0102	-0.712	0.0314	-0.746	0.0210
	200	0.733	0.0246	0.783	0.0125	0.733	0.0246	-0.746	0.0210	-0.746	0.0210	-0.746	0.0210
	300	0.883	0.0016	0.883	0.0016	0.833	0.0053	-0.485	0.1854	-0.809	0.0083	-0.797	0.0102
Legend 4	100	0.733	0.0246	0.733	0.0246	0.600	0.0876	-0.584	0.0985	-0.584	0.0985	-0.714	0.0308
	200	0.683	0.0424	0.736	0.0237	0.617	0.0769	-0.584	0.0985	-0.602	0.0860	-0.657	0.0544
	300	0.700	0.0358	0.770	0.0152	0.650	0.0581	-0.451	0.2229	-0.584	0.0985	-0.511	0.1596
Legend 5	100	0.758	0.0111	0.758	0.0111	0.661	0.0376	-0.738	0.0148	-0.725	0.0176	-0.775	0.0084
	200	0.794	0.0061	0.827	0.0032	0.758	0.0111	-0.750	0.0124	-0.750	0.0124	-0.750	0.0124
	300	0.879	0.0008	0.899	0.0004	0.830	0.0029	-0.389	0.2665	-0.659	0.0383	-0.650	0.0418
Legend 6	100	0.702	0.0152	0.706	0.0152	0.591	0.0556	-0.677	0.0221	-0.706	0.0153	-0.791	0.0037
	200	0.800	0.0165	0.729	0.0109	0.664	0.0260	-0.725	0.0117	-0.715	0.0134	-0.715	0.0134
	300	0.745	0.0085	0.784	0.0043	0.718	0.0128	-0.354	0.2854	-0.550	0.0799	-0.543	0.0840
Legend 7	100	0.667	0.0499	0.667	0.0499	0.600	0.0876	-0.695	0.0377	-0.678	0.0447	-0.746	0.0210
	200	0.717	0.0298	0.762	0.0171	0.733	0.0246	-0.712	0.0314	-0.712	0.0314	-0.712	0.0314
	300	0.833	0.0053	0.862	0.0028	0.767	0.0159	-0.417	0.2646	-0.587	0.0963	-0.576	0.1043
Legend 8	100	0.619	0.1017	0.619	0.1017	0.548	0.1600	-0.545	0.1621	-0.545	0.1621	-0.683	0.0618
	200	0.548	0.1600	0.623	0.0991	0.571	0.1390	-0.545	0.1621	-0.558	0.1506	-0.635	0.0913
	300	0.571	0.1390	0.671	0.0687	0.500	0.2070	0.527	0.1796	-0.545	0.1621	-0.444	0.2750
Legend 9	100	0.608	0.0623	0.309	0.3838	0.321	0.3644	-0.842	0.0022	-0.721	0.0186	-0.576	0.0816
	200	0.600	0.0667	0.638	0.0470	0.612	0.0600	-0.700	0.0241	-0.682	0.0300	-0.682	0.0300
	300	0.661	0.3760	0.711	0.0211	0.624	0.0537	-0.395	0.2584	-0.471	0.1700	-0.463	0.1781

Supporting material: table 6. Alpine region: Spearman rank correlations between PA size and % of PAs with a mean change towards natural land-use/land-cover class that is greater than the respective buffer, and between PA size and % of PAs with a mean change towards artificial land-use/land-cover class that is greater than the respective buffer. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		PAs		Buffer 1km		Buffer 2.5km		Buffer 5km	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value
Legend 1	100	-0.571	0.1802	-0.143	0.7599	0.000	1.0000	0.071	0.8790
	200	-0.571	0.1802	-0.107	0.8192	-0.143	0.7599	-0.214	0.6445
	300	-0.536	0.2152	-0.143	0.7599	0.071	0.8790	-0.179	0.7017
Legend 2	100	-0.571	0.1390	-0.262	0.5309	0.000	1.0000	0.262	0.5309
	200	-0.571	0.1390	-0.238	0.5702	-0.190	0.6514	0.190	0.6514
	300	-0.548	0.1600	-0.262	0.5309	-0.048	0.9108	0.190	0.6514
Legend 3	100	-0.667	0.0499	-0.183	0.6368	-0.033	0.9322	0.217	0.5755
	200	-0.750	0.0199	-0.250	0.5165	-0.067	0.8647	-0.183	0.6368
	300	-0.550	0.1250	-0.200	0.6059	0.067	0.8647	-0.017	0.9661
Legend 4	100	-0.500	0.1705	-0.300	0.4328	-0.083	0.8312	0.200	0.6059
	200	-0.500	0.1705	-0.317	0.4064	-0.183	0.6368	0.183	0.6368
	300	-0.433	0.2440	-0.250	0.5165	-0.133	0.7324	0.133	0.7324
Legend 5	100	-0.673	0.0330	-0.297	0.4047	-0.030	0.9338	0.273	0.4458
	200	-0.745	0.0133	-0.345	0.3282	-0.103	0.7770	0.042	0.9074
	300	-0.588	0.0739	-0.309	0.3848	-0.091	0.8029	0.030	0.9338
Legend 6	100	-0.627	0.0388	-0.364	0.2716	-0.109	0.7495	0.100	0.5554
	200	-0.691	0.0186	-0.355	0.2847	-0.127	0.7092	0.055	0.8734
	300	-0.527	0.0956	-0.264	0.4334	0.027	0.9366	-0.064	0.8525
Legend 7	100	-0.767	0.0159	-0.317	0.4064	-0.033	0.9322	0.000	1.0000
	200	-0.750	0.0199	-0.383	0.3085	-0.067	0.8647	-0.183	0.6368
	300	-0.650	0.0581	-0.283	0.4600	0.067	0.8647	-0.200	0.6059
Legend 8	100	-0.476	0.2329	-0.452	0.2604	-0.143	0.7358	-0.143	0.7358
	200	-0.452	0.2604	-0.431	0.2862	-0.167	0.6932	-0.167	0.6932
	300	-0.381	0.3518	-0.214	0.6103	-0.095	0.8225	-0.238	0.5702
Legend 9	100	-0.685	0.0289	-0.430	0.2145	-0.152	0.6761	-0.067	0.8548
	200	-0.673	0.0330	-0.395	0.2584	-0.115	0.7514	-0.297	0.4047
	300	-0.552	0.0984	-0.261	0.4671	-0.018	0.9602	-0.309	0.3848

Supporting material: table 7. Apennines: Spearman rank correlations between mean land-use/land-cover change and PAs size, between mean change and the 1km-buffer size, between mean change and the 2.5km-buffer size, between mean change and 5km-buffer size. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		Protected areas				Buffer 1km				Buffer 2.5km				Buffer 5km			
		Natural		Artificial		Natural		Artificial		Natural		Artificial		Natural		Artificial	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value
Legend 1	100	0.179	0.7017	-0.393	0.3833	0.576	0.2152	-0.679	0.0938	0.107	0.8192	-0.464	0.2939	0.250	0.5887	-0.321	0.4821
	200	0.750	0.0522	-0.500	0.2532	0.286	0.5345	-0.679	0.0938	0.286	0.5345	-0.464	0.2939	0.143	0.7599	-0.321	0.4821
	300	0.679	0.0938	-0.393	0.3833	0.393	0.3833	-0.536	0.2152	0.429	0.3374	-0.464	0.2939	0.321	0.4821	-0.321	0.4821
Legend 2	100	0.310	0.4556	-0.476	0.2329	0.429	0.2894	-0.667	0.0710	0.214	0.6103	-0.524	0.1827	0.214	0.6103	-0.333	0.4198
	200	0.595	0.1195	-0.571	0.1390	0.310	0.4556	-0.714	0.0465	0.357	0.3851	-0.524	0.1827	0.214	0.6013	-0.333	0.4198
	300	0.643	0.0856	-0.500	0.2070	0.333	0.4198	-0.548	0.1600	0.453	0.2604	-0.524	0.1827	0.262	0.5309	-0.333	0.4198
Legend 3	100	0.400	0.2861	-0.650	0.0581	0.717	0.0298	-0.817	0.0072	0.267	0.4879	-0.733	0.0246	0.550	0.1250	-0.583	0.0922
	200	0.600	0.0876	-0.267	0.4879	0.467	0.2054	-0.833	0.0053	0.500	0.1705	-0.728	0.0262	-0.067	0.8647	-0.500	0.1705
	300	0.833	0.0053	-0.067	0.8647	0.183	0.6368	-0.700	0.0358	0.183	0.6368	-0.733	0.0246	0.283	0.4600	-0.267	0.4879
Legend 4	100	0.300	0.4328	-0.317	0.4064	0.483	0.1875	-0.600	0.0876	0.133	0.7324	-0.467	0.2054	0.133	0.7324	-0.333	0.3807
	200	0.533	0.1392	-0.317	0.4064	0.300	0.4328	-0.633	0.0671	0.283	0.4600	-0.467	0.2054	0.100	0.7980	-0.333	0.3807
	300	0.617	0.0769	-0.367	0.3317	0.333	0.3807	-0.483	0.1875	0.417	0.2646	-0.467	0.2054	0.183	0.6368	-0.267	0.4879
Legend 5	100	0.418	0.2291	-0.685	0.0289	0.515	0.1276	-0.806	0.0049	0.263	0.4671	-0.745	0.0133	0.455	0.1869	-0.576	0.0816
	200	0.624	0.0537	-0.321	0.3655	0.479	0.1615	-0.842	0.0022	0.455	0.1869	-0.742	0.0141	0.006	0.9867	-0.515	0.1276
	300	0.806	0.0049	-0.176	0.6272	0.236	0.5109	-0.709	0.0217	0.212	0.5563	-0.745	0.0133	0.321	0.3655	-0.333	0.3455
Legend 6	100	0.400	0.2229	-0.518	0.1025	0.527	0.0956	-0.764	0.0062	0.209	0.5372	-0.700	0.0165	0.400	0.2229	-0.564	0.0710
	200	0.600	0.0510	-0.155	0.6500	0.436	0.1797	-0.791	0.0037	0.418	0.2006	-0.697	0.0171	0.018	0.9577	-0.509	0.1097
	300	0.782	0.0045	-0.109	0.7495	0.273	0.4171	-0.673	0.0233	0.191	0.5739	-0.700	0.0165	0.245	0.4669	-0.327	0.3259
Legend 7	100	0.517	0.1544	-0.717	0.0298	0.467	0.2054	-0.850	0.0037	0.133	0.7324	-0.767	0.0159	0.300	0.4328	-0.867	0.0025
	200	0.683	0.0424	-0.367	0.3317	0.467	0.2054	-0.867	0.0025	0.283	0.4600	-0.798	0.0099	0.017	0.9661	-0.783	0.0125
	300	0.833	0.0053	-0.200	0.6059	0.233	0.5457	-0.800	0.0096	0.183	0.6368	-0.833	0.0053	0.133	0.7324	-0.667	0.0499
Legend 8	100	0.190	0.6514	-0.524	0.1827	0.381	0.3518	-0.619	0.1017	0.000	1.0000	-0.429	0.2894	0.071	0.8665	-0.595	0.1195
	200	0.476	0.2329	-0.381	0.2518	0.214	0.6103	-0.619	0.1017	0.095	0.8225	-0.491	0.2166	0.119	0.7789	-0.595	0.1195
	300	0.619	0.1017	-0.381	0.3518	0.286	0.4927	-0.595	0.1195	0.286	0.4927	-0.548	0.1600	0.143	0.7358	-0.595	0.1195
Legend 9	100	0.467	0.1739	-0.539	0.1076	0.430	0.2145	-0.782	0.0075	0.115	0.7514	-0.697	0.0025	0.285	0.4250	-0.784	0.0061
	200	0.636	0.0479	-0.176	0.6272	0.358	0.3104	-0.794	0.0061	0.285	0.4250	-0.726	0.0175	0.055	0.8810	-0.721	0.0186
	300	0.794	0.0061	-0.091	0.8028	0.273	0.4458	-0.733	0.0158	0.176	0.6272	-0.758	0.0111	0.091	0.8028	-0.636	0.0479

Supporting material: table 8. Apennines: Spearman rank correlations between mean change towards natural land-use/land-cover classes and PAs size, between mean change towards artificial land-use/land-cover classes and PAs size, between mean change towards natural land-use/land-cover classes and the 1km buffer area, between mean change towards artificial land-use/land-cover classes and the 1km buffer area, between mean change towards natural land-use/land-cover classes and the 2.5km buffer area, between mean change towards artificial land-use/land-cover classes and the 2.5km buffer area, between mean change towards natural land-use/land-cover classes and 5km buffer area, between mean change towards artificial land-use/land-cover classes and 5km buffer area. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		Natural						Artificial					
		Pas vs B1km		PAs vs B2.5km		Pas vs B5km		Pas vs B1km		PAs vs B2.5km		Pas vs B5km	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value
Legend 1	100	0.893	0.0068	0.750	0.0522	0.964	0.0005	-0.214	0.6445	-0.607	0.1482	-0.559	0.1925
	200	0.893	0.0068	0.857	0.0137	0.964	0.0005	-0.342	0.4523	-0.432	0.3325	-0.414	0.3553
	300	0.786	0.0362	0.786	0.0362	0.964	0.0005	-0.179	0.7017	-0.685	0.0897	-0.214	0.6445
Legend 2	100	0.802	0.0165	0.667	0.0710	0.881	0.0039	0.000	1.0000	-0.275	0.5091	-0.610	0.1084
	200	0.786	0.0208	0.826	0.0114	0.922	0.0011	-0.439	0.2763	-0.293	0.4816	-0.488	0.2199
	300	0.738	0.0366	0.738	0.0366	0.922	0.0011	0.095	0.8225	-0.430	0.2881	-0.407	0.3167
Legend 3	100	0.900	0.0009	0.850	0.0037	0.967	<0.0001	-0.050	0.8984	-0.400	0.2861	-0.356	0.3471
	200	0.933	0.0002	0.883	0.0016	0.967	<0.0001	-0.085	0.8284	-0.136	0.7279	-0.136	0.7279
	300	0.900	0.0009	0.900	0.0009	0.950	<0.0001	0.050	0.8979	-0.519	0.1524	0.025	0.9489
Legend 4	100	0.740	0.0228	0.667	0.0499	0.862	0.0028	-0.017	0.9661	-0.276	0.4720	-0.627	0.0706
	200	0.733	0.0246	0.795	0.0104	0.929	0.0003	-0.390	0.2996	-0.271	0.4802	-0.407	0.2772
	300	0.750	0.0199	0.750	0.0199	0.862	0.0028	-0.017	0.9661	-0.426	0.2534	-0.469	0.2032
Legend 5	100	0.863	0.0013	0.806	0.0049	0.927	0.0001	0.030	0.9336	-0.183	0.6130	-0.431	0.2131
	200	0.879	0.0008	0.875	0.0009	0.948	<0.0001	-0.219	0.5435	-0.075	0.8368	-0.256	0.4746
	300	0.867	0.0010	0.867	0.0012	0.936	<0.0001	0.109	0.7635	-0.313	0.3787	-0.166	0.6474
Legend 6	100	0.831	0.0015	0.800	0.0031	0.916	<0.0001	0.023	0.9470	-0.183	0.5909	-0.423	0.1945
	200	0.845	0.0001	0.861	0.0007	0.952	<0.0001	-0.224	0.5082	-0.074	0.8278	-0.215	0.5265
	300	0.864	0.0006	0.864	0.0006	0.907	0.0001	0.064	0.8522	-0.275	0.4127	-0.220	0.5153
Legend 7	100	0.900	0.0009	0.900	0.0009	0.933	0.0002	-0.033	0.9322	-0.310	0.4175	-0.356	0.3471
	200	0.933	0.0002	0.917	0.0005	0.967	<0.0001	-0.153	0.6939	-0.153	0.6939	-0.204	0.5980
	300	0.950	<0.0001	0.950	<0.0001	0.950	<0.0001	0.050	0.8984	-0.494	0.1768	-0.059	0.8805
Legend 8	100	0.850	0.0075	0.838	0.0093	0.922	0.0011	-0.048	0.9103	-0.422	0.2980	-0.563	0.1463
	200	0.857	0.0065	0.857	0.0065	0.952	0.0003	-0.325	0.4317	-0.371	0.3652	-0.349	0.3962
	300	0.833	0.0102	0.833	0.0102	0.952	0.0003	-0.168	0.6915	-0.635	0.0909	-0.333	0.4198
Legend 9	100	0.888	0.0006	0.894	0.0005	0.948	<0.0001	-0.030	0.9336	-0.293	0.4118	-0.337	0.3403
	200	0.915	0.0002	0.891	0.0005	0.964	<0.0001	-0.180	0.6184	-0.172	0.6340	-0.168	0.6432
	300	0.915	0.0002	0.915	0.0002	0.952	<0.0001	0.043	0.9071	-0.438	0.2058	-0.122	0.7372

Supporting material: table 9. Apennines: Spearman rank correlations between PA size and % of PAs with a mean change towards natural land-use/land-cover class that is greater than the respective buffer, and between PA size and % of PAs with a mean change towards artificial land-use/land-cover class that is greater than the respective buffer. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		PAs		Buffer 1km		Buffer 2.5km		Buffer 5km	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value
Legend 1	100	-0.786	0.0362	-0.214	0.6445	-0.143	0.7599	-0.036	0.9394
	200	-0.786	0.0362	-0.179	0.7017	-0.252	0.5852	-0.036	0.9394
	300	-0.750	0.0522	-0.429	0.3374	-0.429	0.3374	0.071	0.8790
Legend 2	100	-0.690	0.0580	-0.071	0.8665	-0.238	0.5702	0.310	0.4556
	200	-0.690	0.0580	-0.119	0.7789	0.012	0.9775	0.310	0.4556
	300	-0.667	0.0710	-0.119	0.7789	-0.143	0.7358	0.381	0.3518
Legend 3	100	-0.867	0.0025	-0.317	0.4064	-0.317	0.4064	-0.100	0.7980
	200	-0.817	0.0072	-0.167	0.6682	-0.350	0.3558	-0.267	0.4879
	300	-0.783	0.0125	-0.483	0.1875	-0.467	0.2054	0.075	0.8473
Legend 4	100	-0.683	0.0424	-0.133	0.7324	0.133	0.7324	0.233	0.5457
	200	-0.683	0.0424	-0.117	0.7650	-0.017	0.9659	0.200	0.6059
	300	-0.617	0.0769	-0.117	0.7650	0.133	0.7324	0.267	0.4879
Legend 5	100	-0.818	0.0038	-0.188	0.6032	-0.067	0.8548	0.200	0.5796
	200	-0.782	0.0075	-0.200	0.5796	-0.091	0.8028	-0.018	0.9602
	300	-0.758	0.0111	-0.345	0.3282	-0.188	0.6032	0.328	0.3544
Legend 6	100	-0.809	0.0026	-0.200	0.5554	-0.100	0.7699	0.187	0.5824
	200	-0.782	0.0045	-0.264	0.4334	-0.145	0.6696	0.027	0.9366
	300	-0.736	0.0098	-0.327	0.3259	-0.164	0.6307	0.214	0.5272
Legend 7	100	-0.833	0.0053	-0.450	0.2242	0.317	0.4064	-0.100	0.7890
	200	-0.783	0.0125	-0.483	0.1875	-0.350	0.3558	-0.267	0.4879
	300	-0.750	0.0199	-0.600	0.0876	-0.467	0.2054	0.075	0.8473
Legend 8	100	-0.690	0.0580	-0.405	0.3199	-0.143	0.7358	-0.095	0.8225
	200	-0.690	0.0580	-0.333	0.4198	-0.262	0.5309	-0.143	0.7358
	300	-0.667	0.0710	-0.333	0.4198	-0.310	0.4556	-0.048	0.9108
Legend 9	100	-0.818	0.0038	-0.406	0.2443	-0.297	0.4047	-0.085	0.8152
	200	-0.782	0.0075	-0.539	0.1076	-0.358	0.3104	-0.188	0.6032
	300	-0.758	0.0111	-0.527	0.1173	-0.370	0.2931	-0.049	0.8939

Supporting material: table 10. Flat areas and coastal plains: Spearman rank correlations between mean land-use/land-cover change and PAs size, between mean change and the 1km-buffer size, between mean change and the 2.5km-buffer size, between mean change and 5km-buffer size. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		Protected areas				Buffer 1km				Buffer 2.5km				Buffer 5km			
		Natural		Artificial		Natural		Artificial		Natural		Artificial		Natural		Artificial	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value
Legend 1	100	0.143	0.7599	-0.143	0.7599	-0.536	0.2152	0.500	0.2532	-0.500	0.2532	0.393	0.3833	-0.286	0.5345	0.464	0.2939
	200	0.321	0.4821	0.214	0.6445	-0.357	0.4316	0.500	0.2532	-0.571	0.1802	0.536	0.2152	-0.286	0.5345	0.464	0.2939
	300	0.393	0.3833	0.071	0.8790	-0.286	0.5345	0.750	0.0522	-0.643	0.1194	0.393	0.3833	-0.464	0.2939	0.571	0.1802
Legend 2	100	0.119	0.7789	0.000	1.0000	-0.452	0.2604	0.167	0.6932	-0.333	0.4198	0.143	0.7358	-0.048	0.9108	-0.262	0.5309
	200	0.262	0.5309	0.286	0.4927	-0.286	0.4927	0.310	0.4556	-0.238	0.5702	0.262	0.5309	-0.119	0.7789	0.238	0.5702
	300	0.405	0.3199	0.167	0.6932	-0.143	0.7358	0.667	0.0710	-0.310	0.4556	0.143	0.7358	-0.357	0.3851	-0.262	0.5309
Legend 3	100	-0.317	0.4064	-0.300	0.4328	-0.733	0.0246	0.483	0.1875	-0.533	0.1392	0.183	0.6368	-0.083	0.8312	0.400	0.2861
	200	-0.233	0.5457	-0.233	0.5457	-0.678	0.0448	0.483	0.1875	-0.633	0.0671	0.067	0.8647	-0.200	0.6059	0.350	0.3558
	300	-0.133	0.7324	-0.400	0.2861	-0.583	0.0992	0.717	0.0298	-0.683	0.0424	0.100	0.7980	-0.050	0.8984	0.517	0.1544
Legend 4	100	0.067	0.8647	0.000	1.0000	-0.433	0.2440	0.217	0.5755	-0.667	0.3317	0.167	0.6682	-0.133	0.7324	0.267	0.4879
	200	0.233	0.5457	0.267	0.4879	-0.317	0.4064	0.333	0.3807	-0.267	0.4879	0.250	0.5165	-0.133	0.7324	0.250	0.5165
	300	0.367	0.3317	0.117	0.7650	-0.183	0.6368	0.700	0.0358	-0.300	0.4328	0.167	0.6682	-0.367	0.3317	0.267	0.4879
Legend 5	100	-0.285	0.4250	-0.164	0.6515	-0.697	0.0251	0.273	0.4458	-0.321	0.3655	-0.115	0.7514	-0.200	0.5796	0.091	0.8028
	200	-0.212	0.5563	-0.103	0.7770	-0.705	0.0227	0.333	0.3466	-0.491	0.1497	-0.127	0.7261	-0.018	0.9602	0.103	0.7770
	300	0.115	0.7514	-0.164	0.6515	-0.576	0.0816	0.697	0.0251	-0.455	0.1869	-0.176	0.6272	0.127	0.7261	0.152	0.6761
Legend 6	100	-0.300	0.3701	-0.173	0.6115	-0.636	0.3855	0.327	0.3259	-0.245	0.4669	-0.055	0.8734	0.118	0.7293	0.155	0.6500
	200	-0.173	0.6115	-0.055	0.8734	-0.597	0.0526	0.382	0.2466	-0.409	0.2115	-0.045	0.8944	0.018	0.9577	0.127	0.7092
	300	0.127	0.7092	-0.182	0.5926	-0.555	0.0767	0.700	0.0165	-0.409	0.2115	-0.064	0.8525	0.092	0.7904	0.209	0.5372
Legend 7	100	-0.417	0.2646	-0.033	0.9322	-0.750	0.0199	0.400	0.2861	-0.450	0.2242	0.200	0.6059	-0.100	0.7980	0.250	0.5165
	200	-0.467	0.2054	-0.033	0.9322	-0.678	0.0488	0.483	0.1875	-0.450	0.2242	0.167	0.6682	-0.350	0.3558	0.267	0.4879
	300	-0.050	0.8984	-0.150	0.7001	-0.600	0.0876	0.817	0.0072	-0.450	0.2242	0.117	0.7650	-0.067	0.8647	0.383	0.3085
Legend 8	100	0.048	0.9108	0.286	0.4927	-0.467	0.2329	0.381	0.3518	-0.357	0.3851	0.452	0.2604	-0.333	0.4198	0.452	0.2604
	200	0.024	0.9554	0.333	0.4198	-0.333	0.4198	0.548	0.1600	-0.357	0.3851	0.548	0.1600	-0.333	0.4198	0.429	0.2894
	300	0.190	0.6514	0.262	0.5309	-0.286	0.4927	0.810	0.0149	-0.286	0.4927	0.452	0.2604	-0.524	0.1827	0.571	0.1390
Legend 9	100	-0.430	0.2145	-0.055	0.8810	-0.648	0.0425	0.479	0.1615	-0.309	0.3848	0.248	0.4888	-0.127	0.7261	0.333	0.3466
	200	-0.394	0.2600	-0.030	0.9338	-0.523	0.1210	0.552	0.0984	-0.333	0.3466	0.248	0.4888	-0.261	0.4671	0.297	0.4047
	300	-0.018	0.9602	0.164	0.6515	-0.552	0.0984	0.818	0.0038	-0.370	0.2931	0.236	0.5109	-0.091	0.8028	0.455	0.1869

Supporting material: table 11. Flat areas and coastal plains: Spearman rank correlations between mean change towards natural land-use/land-cover classes and PAs size, between mean change towards artificial land-use/land-cover classes and PAs size, between mean change towards natural land-use/land-cover classes and the 1km buffer area, between mean change towards artificial land-use/land-cover classes and the 1km buffer area, between mean change towards natural land-use/land-cover classes and the 2.5km buffer area, between mean change towards artificial land-use/land-cover classes and the 2.5km buffer area, between mean change towards natural land-use/land-cover classes and 5km buffer area, between mean change towards artificial land-use/land-cover classes and 5km buffer area. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

		Natural						Artificial					
		Pas vs B1km		PAs vs B2.5km		Pas vs B5km		Pas vs B1km		PAs vs B2.5km		Pas vs B5km	
	Cell size (m)	r	p value	r	p value	r	p value	r	p value	r	p value	r	p value
Legend 1	100	0.927	0.0027	0.964	0.0005	0.927	0.0027	-0.714	0.0713	-0.429	0.3374	-0.429	0.3374
	200	0.991	<0.0001	0.991	<0.0001	0.964	0.0005	-0.714	0.0713	-0.357	0.4316	-0.500	0.2532
	300	0.927	0.0027	0.927	0.0027	0.927	0.0027	-0.500	0.2532	-0.607	0.1482	-0.286	0.5345
Legend 2	100	0.913	0.0015	0.939	0.0006	0.913	0.0015	-0.311	0.4528	-0.359	0.3821	-0.359	0.3821
	200	0.980	<0.0001	0.976	<0.0001	0.939	0.0006	-0.311	0.4528	-0.311	0.4528	-0.359	0.3821
	300	0.913	0.0015	0.913	0.0015	0.913	0.0015	-0.216	0.6081	-0.311	0.4528	-0.216	0.6081
Legend 3	100	0.966	<0.0001	0.966	<0.0001	0.966	<0.0001	-0.533	0.1392	-0.417	0.2646	-0.433	0.2440
	200	0.962	<0.0001	0.962	<0.0001	0.898	0.0010	-0.533	0.1392	-0.460	0.2125	-0.333	0.3807
	300	0.932	0.0002	0.966	<0.0001	0.966	<0.0001	-0.100	0.7980	-0.350	0.3558	-0.417	0.2646
Legend 4	100	0.940	0.0002	0.957	<0.0001	0.940	0.0002	-0.402	0.2839	-0.402	0.2839	-0.435	0.2418
	200	0.983	<0.0001	0.983	<0.0001	0.957	<0.0001	-0.301	0.4308	-0.351	0.3537	-0.318	0.4043
	300	0.940	0.0002	0.940	0.0002	0.940	0.0002	-0.201	0.6044	-0.285	0.4581	-0.201	0.6044
Legend 5	100	0.957	<0.0001	0.957	<0.0001	0.957	<0.0001	-0.310	0.3833	-0.359	0.3088	-0.371	0.2915
	200	0.963	<0.0001	0.963	<0.0001	0.907	0.0003	-0.310	0.3833	-0.347	0.3267	-0.274	0.4444
	300	0.932	<0.0001	0.957	<0.0001	0.957	<0.0001	-0.079	0.8282	-0.213	0.5551	-0.261	0.4657
Legend 6	100	0.968	<0.0001	0.968	<0.0001	0.968	<0.0001	-0.282	0.4000	-0.419	0.1994	-0.410	0.2104
	200	0.973	<0.0001	0.973	<0.0001	0.930	<0.0001	-0.328	0.3247	-0.346	0.2969	-0.246	0.4659
	300	0.930	<0.0001	0.968	<0.0001	0.949	<0.0001	-0.046	0.8942	-0.164	0.6299	-0.210	0.5363
Legend 7	100	0.966	<0.0001	0.966	<0.0001	0.966	<0.0001	-0.417	0.2646	-0.133	0.7324	-0.150	0.7001
	200	0.962	<0.0001	0.962	<0.0001	0.898	0.0010	-0.367	0.3317	-0.117	0.7650	-0.017	0.9661
	300	0.932	0.0002	0.966	<0.0001	0.966	<0.0001	-0.050	0.8984	-0.167	0.6682	0.000	1.0000
Legend 8	100	0.952	0.0003	0.976	<0.0001	0.952	0.0003	-0.448	0.3199	-0.167	0.6932	-0.214	0.6103
	200	0.994	<0.0001	0.994	<0.0001	0.975	<0.0001	-0.167	0.6932	-0.095	0.8225	-0.048	0.9108
	300	0.952	0.0003	0.952	0.0003	0.952	0.0003	-0.238	0.5702	-0.238	0.5702	0.119	0.7789
Legend 9	100	0.976	<0.0001	0.976	<0.0001	0.976	<0.0001	-0.358	0.3104	-0.236	0.5109	-0.224	0.5334
	200	0.973	<0.0001	0.973	<0.0001	0.926	0.0001	-0.370	0.2931	-0.139	0.7009	-0.006	0.9867
	300	0.926	0.0001	0.976	<0.0001	0.951	<0.0001	-0.030	0.9334	-0.103	0.7770	0.042	0.9074

Supporting material: table 12. Flat areas and coastal plains: Spearman rank correlations between PA size and % of PAs with a mean change towards natural land-use/land-cover class that is greater than the respective buffer, and between PA size and % of PAs with a mean change towards artificial land-use/land-cover class that is greater than the respective buffer. The correlation was measured across 3 different cell sizes, 3 different buffer size, and 9 different size-classes legends. See Tab. 1 in the text for the details on the different legends.

CONCLUSION

This dissertation was started as a response to a specific question from the Italian Ministry of the Environment and Territorial Protection (Directorate of Nature Conservation) that requested a series of studies on the conservation status of the Italian protected areas. As a response, I developed and validated deductive habitat suitability models for almost 500 species and I used the dataset to perform GAP and irreplaceability analyses with the aim of evaluating if (and how well) the Italian protected areas represent vertebrate species diversity in Italy. Moreover, using Red-List criteria, I evaluated the conservation status of endangered species in the Natura2000 network, a network of conservation areas that Italy (together with all the other European Community countries) has proposed according to a series of European Community Directives. Finally, I have also verified the efficacy of the Italian protected areas in face of the important land-use/land-cover scenarios that characterize the Italian landscape over the last 40 years.

These results are not really encouraging. Even if more than 11% of the national territory is legally protected, Italian protected areas are on the average really small (many of the areas are smaller than 10 hectares). Moreover, even considering that the number of GAP species in Italy is relatively low, it must be underlined that most of the species presented a conservation deficit and were not represented enough by the existing protected areas.

The Natura2000 network is an extremely important conservation effort that will raise the percentage of national territory to be protected to almost 20%. However, even with this really high percentage, the system of conservation areas is not able to preserve in a favorable conservation status the species for which it was instituted with important gaps especially in the Mediterranean islands, exactly where the concentration of species important for conservation is highest.

Finally, given their size, protected areas are not able (with few exceptions) to arrest or even to slow down the land-use/land-cover change that is ongoing in Italy. This is particularly important in flat areas and coastal plains, where the areas are smaller and most of the changes go towards artificial land-use/land-cover classes.

From these results it is clear that the Italian conservation strategy cannot be based on proteced areas alone, especially if we consider that Italian protected areas are embedded in a human dominated landscape. Their role as components of a conservation network must be planned in conjunction with the conservation of areas actively managed to preserve the traditional cultural landscape, where species have coexisted for centuries with compatible human activities. In the Mediterranean more than anywhere else, the PAs must be planned and managed in conjunction with the matrix in which they are embedded and in the context of the environmental history of the region. Biodiversity and human presence are functionally linked through traditional agriculture, pasture, etc. and the only viable option for conservation is that of considering human presence and human activities as an integral part of the system. Ignoring the environmental history of a region will produce conservation and management schemes that simply cannot work, because historical changes in the ecosystems caused by natural and human-related phenomena significantly limits available management options. Protecting extensive tracts of wild land has usually been considered the best way of preserving biodiversity but in a human dominated landscape the only viable approach to conservation is that of coupling the conservation of the few wild, semi-natural areas with restoration and maintenance of traditional cultural landscapes with all their species assemblages.