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BIODIVERSITY OF SACRED NATURAL SITES IN ITALY

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Χαλεπά τά καλά

ABSTRACT

Here, I aimed to assess the value of Sacred Natural Sites (SNS) for biodiversity conservation, by asking: Which is the state of the art on SNS and biodiversity conservation and their overall effect on biodiversity? Do SNS in Italy complement the network of Protected Areas (PA)? Which is the effect of SNS and PA in Italy as drivers of forest cover changes? Do SNS in Italy conserve plant diversity? First, I carried out a systematic review on SNS and biodiversity conservation by means of descriptive statistics and vote counting. Then, I built a geo-referenced nation-wide survey of SNS and compared it with the national network of PA in terms of land uses and landscape characteristics, while investigating their spatial distributions. I also investigated the role of SNS and PA in respect with forest cover changes between 1936 and 2018. Finally, I evaluated the diversity of plant communities found at 30 SNS scattered across Italy by comparing them with related control areas. Most of research up-to-date has been dealing with plants, while largely focusing on asian and african SNS. Hence, the rest of this thesis aims to help close the aforementioned geographical gap. Among the main results, I found that SNS and PA hardly overlapped, being located in different landscape contexts and over different land uses, showing a potential complementarity. Moreover, SNS and PA were associated with a decrease in forest loss and an increase in forest gain, hinting both at a positive effect on forest conservation and an allarming abandonment of cultural landscapes. Finally, plant communities of SNS were highly diverse, but patterns varied across macrohabitats.

SNS provided positive conservation outcomes likely due to traditional management, social taboos and unique natural features. Legal recocgnition and innovative policies are required to protect SNS and their biological specificity.

EXTENDED ABSTRACT

Aims: in the present thesis, I aimed to assess the value of Sacred Natural Sites (SNS) for biodiversity conservation with a focus on Italy, by asking: 1) Which is the state of the art on SNS and biodiversity conservation and their overall effect on biodiversity? 2) Do SNS in Italy complement the network of Protected Areas (PA)? 3) Which is the effect of SNS and PA in Italy as drivers of forest cover changes across long term? 4) Do SNS in Italy conserve higher levels of plant diversity than comparable areas.

Study area: Global in the first chapter, Italy in the other chapters.

Methods: 1) I carried out a systematic review on SNS and biodiversity conservation, starting from 2750 unique papers, I performed a preliminary screening and after applying inclusion criteria I kept 27 papers as final pool. I carried out descriptive analyses and extracted comparisons between SNS and control areas, then I applied vote-counting, multinomial tests and binomial post-hoc tests. 2) I built a geo-referenced nation-wide survey of SNS consisting of 2,332 sites. Then, I characterized the national networks of SNS and PA according to land uses and landscape characteristics and investigated their distributions by means of quasi-poisson GLMs with autocovariates. 3) I produced a map of forest cover changes between 1936 and 2018 at 500 m resolution. Hence, I performed a descriptive analysis and modelled forest loss and forest gain by means of spatial binomial GLMs with residual autocovariates by using protection status, geographical-topographical and anthropogenic factors as explanatory variables. 4) I extracted 30 SNS across Italy and related control areas, where I sampled 243 vegetation plots across 4 different macrohabitats. I identified occurrences at species level and standardized the nomenclature according to *Flora d'Italia (2nd Edition)*. Then, I compared SNS and control areas for species richness, rarity, number of locally-exclusive species, beta and gamma diversity.

Results: 1) I found most of research has been focusing on Asia and Africa, with few papers in Europe and no papers from Americas and Oceania. The vast majority of papers dealt with plants, with a limited number of papers dealing with vertebrates, invertebrates, fungi and lichens. Moreover, SNS were seldom detrimental to biodiversity. 2) I found that SNS and PA hardly overlapped, being located in different landscape contexts and over different land uses. SNS were found at low-mid elevations mostly in cultural landscapes, whereas PA were usually in more remote and natural areas, at higher elevations. The distribution of the two networks was driven by different factors, hinting towards a complementarity between them. 3) Descriptive analysis showed that both SNS and PA had greater rates of forest gain and forest loss than Italy as a whole, but after accounting for the other variables employed in the models SNS and PA were negatively related with forest loss and positively related with forest gain. However, most of the explained variability was

due to the geographical-topographical variables and the residual autocovariates. 4) I collected 6510 plant occurrences, belonging to 1094 species. SNS had more rare assemblages and a larger proportion of locally-exclusive species than control, as well as greater beta and gamma diversity. In contrast, species richness showed no significant differences. When splitting by macrohabitat, only woodlands and annual crops found at SNS were clearly more diverse than control areas.

Conclusions: SNS provided positive conservation outcomes likely due to traditional management, social taboos and unique natural features. It has been highlighted the potential of SNS in enlarging and enhancing the national network of PA as well as contributing to its connectivity. Future research should address the geographical and taxonomical gaps, while innovative policies are required to protect SNS. Finally, incentives and dedicated policies are required to help maintain the continuous traditional management of SNS, to avoid losing their biological specificity.

CONTENTS

Preface

This thesis is the product of a three year PhD project at the Department of Biological, Geological and Environmental Sciences (BiGeA) of University of Bologna, based at the research group of Biodiversity and Macroecology. The thesis has been supervised by Prof. Alessandro Chiarucci, Prof. Juri Nascimbene collaborating as co-supervisor and Dr. Fabrizio Frascaroli acting as mentor despite not being an official supervisor. Moreover, this thesis is part of a larger research project named "BIOESSaNS", meaning Biodiversity and Ecosystem Services at Sacred Natural Sites. BIOESSaNS is a PRIN project, that is a research project funded by the Italian Ministry of Education, University and Research. Other than our research group, CNR-IRPI and CNR-IBBR, that are two research institutes of the National Research Council (CNR), and the University of Gastronomic Sciences of Pollenzo were involved in the project.

During my PhD I also spent a three months period at the Biological Applications & Technology Department, University of Ioannina, working on my PhD project under the supervision of Prof. John Maxwell Halley.

The thesis consists of four chapters preceded by a general introduction. The first part is a systematic review on Sacred Natural Sites and biodiversity conservation and is the only chapter with a global scope, whereas in the other three chapters I focused on Sacred Natural Sites in Italy. The second and third chapters deal with spatial analysis of biodiversity relevant phenomena and the last chapter is about plant diversity patterns at community scale.

The second chapter has already been published on an international journal and the first chapter is currently being reviewed. The remaining two chapters are written as scientific research papers and are close to being submitted.

Through my PhD, I had the unique opportunity for developing and honing a number of skills, which comprise project management, sampling design, field sampling, team leading, mentoring, scientific writing, spatial and statistical analysis. Moreover, I was able to deepen my knowledge in a number of research topics, such as conservation biology, island biogeography, vegetation ecology, biodiversity measurements, spatial ecology and macroecology.

Results from my PhD project were presented at the 1st Doctoral Workshop of UNA Europa - Cultural Heritage 2018, Madrid (oral communication), at the 2nd International Conference on Community Ecology 2019, Bologna (oral communication), at the 62nd IAVS Symposium 2019, Bremen (poster) and at the 114th Congress of Italian Botanical Society 2019, Padua (poster).

Finally, I was also able to support didactic activity by supervising five Bachelor Theses (Chiara Ferrara, Ilaria Zauli, Federico Tossani, Lorenzo Spina and Igor Da Rin Bettina) and four Master

Theses (Mirko Enea, Filippo De Vigili, Vito Sasso and Angela Persico), as well as teaching extra seminars for the courses of 1) Vegetation Ecology (Master's program, year 2018, University of Bologna), 2) Biogeography and Macroecology (Master's program, years 2019 and 2020, University of Bologna), 3) Environmental Data Analysis (Bachelor's program, year 2019, University of Ioannina).

Piero Zannini   Bologna, Italy, April 2021

INTRODUCTION

1 BIODIVERSITY

1.1 From the concept of biodiversity to its value (passing through its measure)

The term biodiversity derives from the contraction of "biological diversity" and it was coined in 1985 by Walter G. Rosen for "The National Forum on BioDiversity", which was held the following year in Washington DC. However, the term is often attributed to Edward O. Wilson, who indeed edited the proceedings of the aforementioned conference in 1988 by the title *Biodiversity*. Soon after, at the Convention on Biological Diversity, which was held in Rio de Janeiro in 1992, biodiversity was defined as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other acquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems".

Since then, many other definitions – altough with different fortunes – have been proposed. However, from this early definition two key aspects of biodiversity can already be grasped: the first is that biodiversity manifests at different scales and the second is that the measurement of biodiversity is complex, as it can be carried out in many ways. These two features are clearly linked, with the latter being – at least in part – a consequence of the former. Indeed, biodiversity manifests from the sub-cellular scale, where we can observe for instance the variety of genes, proteins and transcripts, to the global scale, where we can observe different biomes and biogeographical realms, passing through a number of hierarchical levels. Considering the number of classes in which we can classify every element observable at each scale, it is easily understandable that despite the number of these classes is finite it is extremely demanding, if not impossible, to list every element.

Despite these intrinsic difficulties and limitations, ecologists and conservation biologists measured and compared biodiversity – or at least they tried to – hundreds and thousands times. The most straightforward way to measure biodiversity and probably the most intuitive is the count of the

different classes of elements in the given sampling unit, when the classes we are counting are species this measure goes by the name of species richness (Gotelli & Colwell 2001). Indeed, species richness is by far the most widely employed metric to measure biodiversity and it has been applied through a number of spatial scales, from microbial communities to biogeographical regions and the entire Earth. Also, derivatives of species richness have been widely used, for example genera and family richness are employed both in paleoecology and in ecology when species identification is not feasible. Similarly, the count of land uses, ecosystems or patches of habitats are very common in landscape ecology. By increasing complexity through adding the quantities of elements across different classes, such as the abundances of species (or haplotypes, land uses, etc.), other common diversity indices can be calculated like Shannon's H (Shannon 1948), 1 − Simpson's D (or Simpson's D, which, in contrast, is a dominance index; Simpson 1949), as well as equitability indices as Pielou's E or eveness (Pielou 1969). In order to briefly grasp different dimensions of diversity, such as the contribution of rare or common species in a given assemblage, more complex indices have been proposed like Rényi Entropy (Rényi 1961) or Hill Numbers (MacArthur 1965, Hill 1973). These indices provide different insights on the studied systems by varying their *order*, giving information related to the number of classes, their relative quantity, the number and variety of rare elements, etc. While this short excursus is not meant to provide a full overview on diversity indices, a mention to Rao's Q is likely due (Rao 1982). This index has been introduced in ecology in recent past and owes its fortune to measuring diversity not only based on the number of classes and their proportions but it also for accounting for differences - or better distances - among classes, adding further complexity and information (for applications of Rao's Q see e.g., Botta-Dukát 2005, Rocchini et al. 2017).

It should be remarked that when biodiversity indices are calculated based on the number of taxonomical classes (and their proportions), we are referring to taxonomical diversity. However, diversity can also account for the phylogenetic relationships among species or their functional roles,

in these cases we are dealing with phylogenetic and functional diversity indices. Full sets of the aforementioned (or analogous) indices exhist to measure the phylogenetic and functional diversities. Moreover, different types of diversity have been described for different scales. The concepts of α, β and γ diversity refer to the diversity *within* groups, *between* groups and the *overall* diversity of a given system. Also, diversity does not vary only in space but also in time and, indeed, can be measured in space and time.

However, the reason for which we should measure or even protect biodiversity has still to be addressed. One could say that we should protect biodiversity because of some self-proclaimed role of Mankind on Earth or due to the fact that the current biodiversity crisis is our responsibility. Moreover, it could be said that the rest of living organisms on Earth shared a common ancestor with us at a certain time in the past and hence we are somehow related. All this reasoning and positions towards the conservation of biodiversity are usually referred to the concept known as *biophilia* (Wilson 1984), which is defined as "the urge to affiliate with other forms of life". Therefore, these orientations towards biodiversity conservation are based on ethical or emotional reasons to protect biodiversity rather than on objective and pragmatic motivations. About that, the most straighforward reason for which we should protect biodiversity is that in this way we protect our own lives and the possiblity for future generations to live. Indeed, biodiversity is the fundamental phenomenon for which ecosystem services exhist and continue (Millenium Ecosystem Assessment 2005). Ecosystem services in turns are those benefits provided by natural systems that allow humanity to continue and thrive. Therefore, protecting biodiversity is critical to ensure the possiblity for current and future generations to live (Díaz et al. 2006).

1.2 The sixth mass extinction

Biodiversity is being eroded across taxa and biomes at unprecedented speed in the history of Mankind. Similar major periods of extinction have already happened in the past and we refer to

them as mass extinctions. In this sense, the current global extinction is the sixth mass extinction in Earth's history (Barnosky et al. 2011, Ceballos et al. 2015) and is likely to pose a major threat on humanity if no measures will be taken, ultimately destroying ecosystem services and a vast majority of life on Earth, Mankind included (Cardinale et al. 2012, Ceballos et al. 2017).

The actual biodiversity crisis has multiple causes, among which land use changes, habitat fragmentation, deforestation, biological invasions, expansion of urban and agricultural areas and climate change. Moreover, the complex interactions among these causes are likely to further increase extinction rates, through mutual exacerbation and positive feedbacks.

1.3 Area-based conservation measures

Protected areas (PA) are the cornerstone of biodiversity conservation, consisting of portions of land and sea with modest to high degree of naturality and hosting rare or threatened or *simply* precious species and ecosystems. The first PA in history dates back to 1872, with the establishment of Yellowstone National Park, the iconic and world-famous park in the Rocky Mountains, USA. As of October 2020, 15.0% of Earth's land surface and 7.6% of marine areas are under some regime of protection (UNEP-WCMC et al. 2020), being made up by 258,608 designated PA. Moreover, larger portions of land and sea are expected to become PA in the near future (see e.g., European Commission 2020) to face the current biodiversity crisis (Barnosky et al. 2011, Ceballos et al. 2015), following calls from the scientific community to preserve vast portions of Earth as the main opportunity to halt the sixth mass extinction (Locke 2013, Wilson 2016, Baillie & Zhang 2018). Recently, conservationists noticed the possibility given by other areas not primarily designed for biodiversity conservation to effectively deliver favourable conservation outcomes and named them other effective area-based conservation measures (OECMs). Consequently, interest on OECMs has been increasing fast, to the point they are currently accounted for within international conservation schemes as for instance Aichi Biodiversity Targets, while the IUCN has been delivering guidelines

to help practitioners recognising, managing and reporting OECMs (IUCN WCPA 2019). OECMs have the remarkable advantage of conserving biodiversity free-of-charge, while being usually valued by local communities, which in turns may oppose the establishment of official PA due to potential conflicts.

2 SACRED NATURAL SITES

Sacred Natural Sites (SNS) are defined as "areas of land or water having special spiritual significance for people and communities" (IUCN UNESCO 2008). Many different types of natural features have been regarded as SNS, from single trees as St. Francis Holm Oak or rocks, to mountains such as Ururu or Devil's to Tower and even entire mountain landscapes as those found in the Hymalaian Range, passing through groves, forests, springs, islands, valleys, rivers, islands, etc. SNS have been found in all major faiths and in many other, in all inhabited continents. It has been recognized that SNS often deliver positive conservation outcomes, usually as a by-product of traditional management, social taboos or unique natural features, and as such can be regarded as OECMs.

Interest on SNS started in India between the 70s and the 80s (Gadgil & Vartak 1975, 1981, Vartak & Gadgil, 1981, Guinko 1985), but in only started to catch on in the late 90s (see e.g., Balasubramanian & Induchoodan 1996, Barik et al. 1996, Ramakrishnan 1996, Deb et al. 1997, Decher 1997, Bagine 1998, Chandrashekara & Sankar 1998, Mwihomeke et al. 1998, Tiwari et al. 1998, Decher & Bahian 1999) and it further increased in the last 15 years (Figure 1).

Despite the discrete amount of research on SNS, many questions are still open and some approaches have yet to be employed, for instance no systematic review has ever been carried out on SNS and biodiversity conservation, although some literature reviews exhist (Bhagwat & Rutte 2006, Dudley et al. 2010). Moreover, it has been said that SNS make up a "shadow conservation network" (Dudley et al. 2009) but large-scale comparisons with PA are lacking. Also, SNS are typically

regarded as the oldest form of habitat protection (Dudley et al. 2010) but analyses on SNS and land use and land cover changes have seldom been addressed. Finally, while a number of papers tried to evaluate the conservation value of SNS in given areas by studying biological communities (see e.g., Salick et al. 2007, Ambinakudige & Sathish 2009, Frascaroli et al. 2016, Avtzis et al. 2018, Shepheard-Walwyn & Bhagwat 2018, Kühnert et al. 2019, Marini Govigli et al. 2020) none has ever tried to assess it across a whole country.
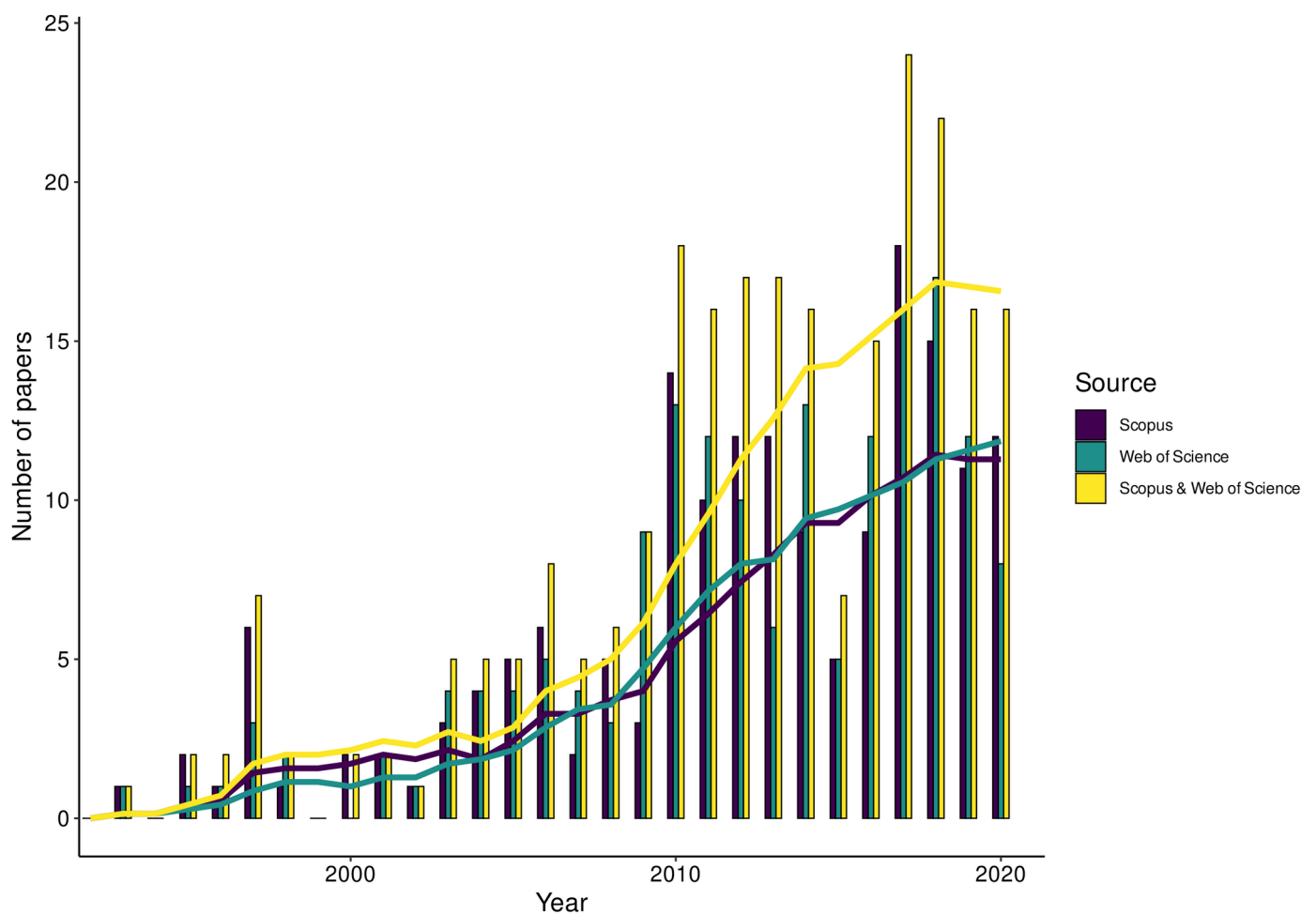

Figure 1. Number of papers on Sacred Natural Sites and biodiversity from 1993 to 2020, coloured according to data source (in yellow are shown all unique papers from Scopus and Web of Science). Each triplet of bars represents a year, except for the year 2000 where a pair is shown as no papers were retrieved from Web of Science. Lines represent 7-year moving averages. Data were retrieved in December 2020 with the following research string applied to abstract, title and keywords: ("sacred grove*" OR "sacred natural site*") AND biodiversity.

3 OUTLINE OF THE THESIS

In this thesis I investigate the role of SNS for biodiversity conservation in Italy, trying to fill the aforementioned knowledge gaps. To do so, I first start with a chapter with global scope, in order to review literature up to now. Then, three chapters cover SNS and biodiversity relevant phenomena in Italy, that are landscape and landuse characteristics of SNS and a comparison with PA, SNS and PA as drivers of forest cover changes across an 82 years period and classical community ecology of SNS. Below I provide a short summary of the 4 chapters composing this thesis.

3.1 Chapter 1

In Chapter 1, I present the first-ever systematic review on SNS and biodiversity conservation. I formulated a research string and extracted 2750 unique papers from Scopus, 27 of which where retained for the review after applying inclusion criteria. I permormed descriptive analyses to summarise where the research has been focusing and on which organisms, then I applied vote-counting to evaluate the effect of SNS on biodiversity conservation. Most of research has been focusing on Asia and Africa and the most studied organisms were plants, while few papers focused on SNS in Europe and on fungi, vertebrates and invertebrates. Future research should investigate the role of SNS in the less studied areas, while applying multitaxa approach.

3.2 Chapter 2

In Chapter 2, I present one of the first-ever nation-wide surveys of SNS and take advantage from it to test if the national networks of SNS and PA may complement each other. I georeferenced 2,332 SNS across Italy and characterized them by landscape and land use characteristics. Then, I characterized PA in the same way and compared them with SNS. I also compared elevational distributions and percentages of cover within administrative regions of the two networks. Then, in order to disentangle the drivers of their spatial distribution, I modelled them by means of quasi-

poisson GLMs with autocovariates. I found scarce overlap between SNS and PA, hinting at independent distributions of the two networks. SNS were typically distributed at mid-low elevations within cultural landscapes and PA were found at mid-high elevations in more natural contexts, therefore different factors shaped their distributions. Results confirmed the complementarity of the networks, pointing at the legal recognition of SNS through innovative policies.

3.3 Chapter 3

In Chapter 3, I explore forest cover changes, notably forest gain and forest loss, in relation with SNS and PA. The chapter focuses on changes that happened in Italy between 1936 and 2018 by taking advantage of a national forest inventory (the Italian Kingdom Forest Map 1936) and the European Union CORINE program products (Corine Land Cover 2018). After making a nation-wide map of forest cover changes for the period 1936-2018, I applied spatial binomial GLMs with residual autocovariates to explore the factors driving forest gain and forest loss. Despite geographical factors such as the latitude, longitude and elevation were the most important predictors in both cases, SNS and PA were significantly associated with forest loss and forest gain. In particular, both types of protection were positively associated with forest gain and negatively associated with forest loss. While SNS confirmed their role in protecting forest patches, it is desirable that policies and public recognition help halting the abandonment of SNS and related practices, as it became clear from forest gain patterns.

3.4 Chapter 4

In Chapter 4, I analyse plant species assemblages found at SNS across Italy and compared them with control areas. I selected 30 sites scattered across the country and developed a sampling design allowing me to sample different habitats at SNS and control areas. I sampled 243 plots, consisting of 6510 occurrences and 1094 species, across 4 macrohabitat types. I compared species richness,

rarity, locally-exclusive species, beta and gamma diversity between SNS and control areas. While I

found weak signal in terms of species richness, SNS showed larger values of rarity, beta and gamma

diversity, as well as a larger number of locally-exclusive species than control areas. Among the

habitats considered, woodlands and annual crops showed positive effects on biodiversity when

found at SNS. Here I call again for a legal recognition of SNS, while highlighting the need for

habitat-based conservation measures.

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CHAPTER I

Sacred Natural Sites and Biodiversity Conservation:

a systematic review

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ABSTRACT

Sacred Natural Sites (SNS) have gained recognition from conservationists, and are regarded as the oldest form of habitat protection in human history. Many case studies and literature reviews have been published on the subject. However, an updated and global-level synthesis on the effect of SNS on biodiversity conservation is still lacking. Here, we provide the first-ever systematic review on SNS and biodiversity conservation, aiming to evaluate the effect of SNS across different: i) continents; ii) taxa; iii) metrics. Starting from 2 750 papers and by applying inclusion criteria, we retained 27 relevant papers. From these, we extracted descriptive data and 131 comparisons between SNS and Reference Sites (RS). We applied vote-counting, multinomial and binomial post-hoc tests to the 131 comparisons. We found strong evidence that SNS have a positive effect on biodiversity, but also strong geographical and taxonomical biases, with most research focusing on Asia and Africa and on plants. We found that SNS have mainly positive effects on taxonomical diversity, vegetation structure and cultural uses of biodiversity. Our results strongly support the view that SNS have positive effects on biodiversity across continents and geographical settings, as found in a number of local studies and earlier overviews. These effects should be given official recognition in appropriate conservation frameworks, together with the specific forms of governance and management that characterise SNS. At the same time, further efforts are also required to fill the geographical and taxonomical gaps here highlighted, and to advancing our knowledge of SNS through further systematic research.

1 INTRODUCTION

The accelerated rate at which biodiversity is being eroded (IPBES 2019) is moving scientists to call

for the conservation of large portions of Earth (e.g., Locke 2013; Wilson 2016; Baillie & Zhang

2018; Dinerstein et al. 2020). While protected areas (PA) are regarded as the cornerstone of nature

conservation, concerns are being raised about their capacity to effectively halt the current

biodiversity crisis (Laurance et al. 2012; Watson et al. 2014). Interest has thus been increasing in

complementary conservation measures, which may integrate and possibly overcome some of the

limitations of PA. One such type of measures has been defined and legally ratified by IUCN and the

Convention on Biological Diversity as other effective area-based conservation measures (OECMs).

These include a broad range of cases, from private estates to ancestral lands, in which conservation

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is not the primary purpose of land management, as in the case of PA, but consistently delivered as a by-product of other management goals (IUCN WCPA 2019).

Sacred Natural Sites (SNS) are another example of nature protection independent of PA, which has received considerable recognition from conservationists over the last two decades. SNS have been defined as "areas of land or water having special spiritual significance for peoples and communities" (IUCN-UNESCO 2008). SNS are associated to a wide range of natural features, such as single trees or rock outcrops, rivers, mountains, islands and even entire landscapes (Dudley et al. 2006), although it seems that the majority are sacred groves and forest patches (Dudley et al. 2010). Sometimes, SNS can also consist of built features, such as temples, shrines or monasteries, surrounded by natural or semi-natural areas (Dudley et al. 2009; Frascaroli et al. 2016).

From a conservation perspective, SNS are often regarded as the oldest form of conservation of habitats and other types of natural resources in human history, in some cases having existed even for millennia. Like OECMs at large, it has been suggested that SNS may constitute a large network of "informal" PA (Dudley et al. 2009), potentially reinforcing the official PA networks (Frascaroli et al. 2019) and contributing to biodiversity conservation at global scale. Indeed, it has been reported that SNS have the same, or even higher, levels of species richness as comparable areas and even official reserves (see e.g. Bhagwat et al. 2005; Brown et al. 2006; Boadi et al. 2017). Moreover, SNS sometimes act as refugia for endemic, threatened, rare or specialist species (see e.g. Gunaga et al. 2013; Künhert et al. 2019) and preserve old-growth trees and forest patches (see e.g. Salick et al. 2007; Tiwari et al. 2010; Frascaroli et al. 2016; Stara et al. 2015). In some cases, SNS can also represent the only natural or semi-natural patches of habitat within highly modified landscapes (Dudley at al. 2010). SNS are found in association with both indigenous and mainstream faiths and in all continents, except for Antarctica (Dudley at al. 2006). However, most of the research concerning SNS has so far focused on Africa and Asia (Dudley at al. 2010).

Despite the growing awareness of the importance of SNS for both biodiversity conservation and local livelihoods, no systematic review based on quantitative or semi-quantitative methods has been published on the topic to date. Existing syntheses have introduced SNS to the scientific and conservation communities and demonstrated their significance in a wide range of contexts (Bhagwat & Rutte 2006; Dudley at al. 2006, 2010). While classic literature reviews have provided major insights, enabling syntheses and highlighting various gaps in our knowledge, the growing number of primary studies offers in addition an opportunity to deploy techniques of systematic review and meta-analysis. Systematic reviews can increase transparency, reproducibility and objectivity by applying strict guidelines throughout the whole review process (Pullin & Stewart 2006; Haddaway et al. 2015). Meta-analyses further develop such results by investigating statistically the size, direction and significance of various treatments across multiple studies (Vetter et al. 2013).

In the present work, we carry out the first-ever review that applies a systematic approach and semi-quantitative methods to address the following question: do SNS worldwide have significant benefits for biodiversity conservation, compared to surrounding areas. We take the comparison of SNS with nearby non-sacred areas as fundamental to objectively evaluate their effectiveness in delivering area-based conservation of nature (see Coetzee et al. 2014 for a similar case). In particular, we asked if the reported effects of SNS on biodiversity varies across: 1) continents; 2) taxa; 3) biodiversity measures. We investigated these three facets through systematic review, while collecting evidence on the state-of-art literature on SNS and paving the way to future evidence-based syntheses.

2 METHODS

2.1 Data collection

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To carry out the review, we followed the guidelines proposed by Pullin & Stewart (2006) for reviews in conservation and environmental management. A literature search was performed using Scopus Database (last access on May 27, 2019). The search string was composed of 'OR' and 'AND' statements combining key terms like "biological conservation" and "biodiversity" with SNS-related synonyms, such as "shrine", "temple" and "church" (Appendix A). We limited our search to peer-reviewed articles and book chapters (henceforth "papers"). We excluded conference papers, as they often consist of still partial, temporary or non-peer reviewed data (for the same reasons we removed records classified in Scopus as "note", "erratum", "letter" and "short survey"). No limitations were placed on the year or country of publication. The literature search provided 2 750 unique results that were screened by title and abstract to remove entries clearly not relevant to the review (Appendix B). This left a set of 473 papers, 95 of which, however, could not be retrieved. We thus read the full-text of the 378 resulting papers and chose only papers in English offering systematic comparisons between SNS and Reference Sites (RS) either of taxonomical diversity indices or of vegetation structure measurements or both. Based on these criteria, a final pool of 27 studies were identified as eligible for the review (see section "Data sources" for the full list of references). For each study we extracted publication metadata, geographical location (or its approximate centroid) and the taxonomic group investigated. Taxa were aggregated at high taxonomical level, distinguishing among vertebrates, invertebrates, plants and fungi. As lichens appeared in only one study, we decided to group them with fungi. We finally determined the biome in which each study was located, based on the WWF Ecoregion Map (Olson et al. 2001). Also, it has to be noted that the number of SNS studied within each paper was rather varied (up to 32, with a median of 6).

The selected studies provided very heterogeneous information. To keep the greatest portion of available knowledge while at the same time enabling meaningful comparisons between such different studies, we opted for carrying out a semi-quantitative evaluation through the vote counting method (Koricheva et al. 2013). Based on this method, different outcomes of the tested comparisons

are categorised as having an effect that is significantly positive (+), significantly negative (-) or non-significant (0, henceforth "neutral effect"). Hence, each comparison between SNS and RS casts a "vote" and the numbers of votes are counted. In order to minimise subjectivity and enhance reproducibility, we chose to include in the vote counting only comparisons whose statistical significance was reported in the studies or could be determined based on the data. Comparisons stated as marginally significant ($.05 < p < .10$) were still retained as significant. When p-values were not reported, we tried to use relevant data, such as raw or mean values, sample sizes, and confidence intervals, to calculate the significance of the effect. When p-values were neither reported nor retrievable on the basis of other available information, we excluded the comparisons from the vote counting. Similarly, we did not consider multiple comparisons on the same data that were not followed by appropriate post-hoc testing. Finally, given the large heterogeneity of metrics in the pool of target studies, we reclassified them into a set of 6 classes (Appendix C) and we excluded metrics that could not be reclassified, resulting in 131 comparisons. Descriptive data about the 27 selected papers as well as the 131 comparisons can be found in Appendix D.

We divided the retrieved comparisons into classes according to different aspects of biodiversity *sensu lato*. The first three classes ("taxonomic diversity", "evenness" and "beta diversity") are related to typical biodiversity measures. In particular, "taxonomic diversity" is related to local diversity irrespective of taxonomical resolution and aims at measuring that component of diversity that is the number of different classes without accounting for their relative sizes, which in turn is measured by the "evenness". As in our dataset Shannon's H was always employed along with species richness and in order not to use redundant comparisons, we chose to use Shannon's H as a proxy of "evenness" when a direct measure (like Pielou's E o Simpson's 1-D) was absent, otherwise we discarded it. "Policy species" gathers all metrics related to threatened, rare or endemic taxa. "Vegetation structure" is made up by metrics calculated on stem characteristics, such as mean basal area, mean diameter at breast height and number of stems per unit area, and is meant to be a proxy

of mature vegetation. Finally, within "cultural use" we included all the aspects related to the use of

biodiversity resources, such as medicinal or culturally important species. This is also referred as or

considered part of the Traditional Environmental Knowledge (TEK) of local communities. Although

these two last classes (i.e. "vegetation structure" and "cultural use") are not biodiversity measures

strictly speaking, we considered them as complementary measures of biodiversity and conservation

potential. Indeed, both mature vegetation and TEK are often related to habitat conservation or

traditional sustainable management.

2.2 Statistical analyses

To assess whether the positive, negative and neutral effects of SNS occurred in statistically

significant proportions, we used a multinomial exact test. This kind of test implies that the response

variable is a categorical variable with more than two values. In our case, the possible values were:

positive, *negative* and *neutral* (no effect). The observed frequencies of each value are then

compared against a set of expected frequencies. We tested our observations against the assumption

that positive, negative and neutral effects of SNS would have equal odds to occur. We tested both

for all the recorded effects together and for separate subsets, defined on the basis of relevant

variables: the continent where the effect of SNS was investigated, the taxonomic group under

scrutiny and the class of metrics used in the comparison between SNS and RS. For each test, we

calculated the log-likelihood ratio statistic (LLR) and its relative *p*-value. We chose this metric

given its demonstrated advantages over more popular alternatives, such as Pearson's Chi-square

(Kotze & Gokhale 1980).

To test further which effects significantly deviate from their expected frequencies in instances

where the multinomial exact test was significant, we applied exact binomial test (McDonald 2014).

This test is based on the same logic as the exact multinomial but is applied to categorical variables

with two possible values. We thus tested the observed frequency of each effect against the pooled

frequencies of the other two, and an expected odd ratio 1:2. Since this procedure entails multiple comparisons from the same sample data, we applied Holm correction (Holm 1979) to the resulting p-values. This is similar to a Bonferroni correction but retains slightly higher statistical power, while still controlling the overall Type I error rate (Olejnik et al. 1997). Tests were performed in R version 3.6.3 (R Core Team 2020) with the package XNomial v 1.0.4 (Engels 2015), while graphical outputs were produced with ggplot2 (Wickham 2016), patchwork (Pedersen 2020), ggthemes (Arnold 2019), ggnewscale (Campitelli 2020), sf (Pebesma 2018) and rnaturalearth (South 2017).

3. RESULTS

3.1 Distribution across continents and taxa

The geographical distribution of the study areas investigated in the 27 papers eligible for systematic review was highly uneven (Figure 1.1a), as 8 studies focused on the Kodagu District (India) and another 8 on West Africa. Overall, we found 13 papers related to SNS located in Asia, 11 in Africa and only 3 papers related to SNS in Europe (Figure 1.1b). The final pool did not yield any paper concerning SNS from Oceania or Americas. India, Ghana and China were the only countries for which more than 1 paper was found, with 11, 3 and 2 papers, respectively (Figure 1.1c). The distribution of taxa studied within papers was dominated by plants (n = 21), followed by vertebrates, fungi and invertebrates (n = 6, 5 and 4, respectively; Figure 1.1d). In 5 papers a multitaxa approach was employed. The tropical and subtropical moist broadleaf forests biome, finally, was by far the most common biome (n = 16), whereas other biomes were more scarcely represented or not represented at all in the pool of papers (Figure 1.1e).
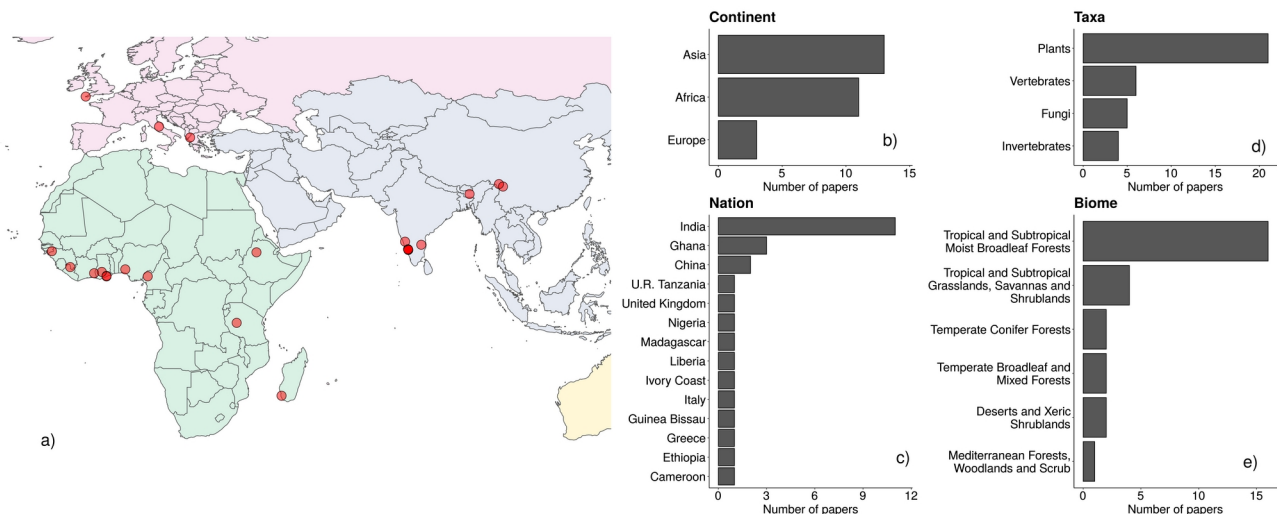
Figure 1.1. Descriptive statistics of selected papers (n = 27): a) geographical distribution of study areas; b) papers by continent; c) papers by country; d) papers by taxa; e) papers by biome.

3.2 Effects of SNS

Overall comparisons extracted from our pool of papers showed that SNS had positive or neutral effects in the vast majority of cases, whereas negative effects accounted for only ca. 10% of the total (Figure 1.2). Dividing the comparisons by continent, Europe had the largest proportion of positive effects and the smallest proportion of negative effects (about 55% and 5%, respectively), with Africa and Asia also showing similar patterns. While vertebrates, plants and fungi had similar patterns to all comparisons grouped together, invertebrates stood out as they had no positive effects. When considering different types of indicator, vegetation structure and cultural uses had no recorded negative effects and the former even had positive effects in ca. 70% of cases.
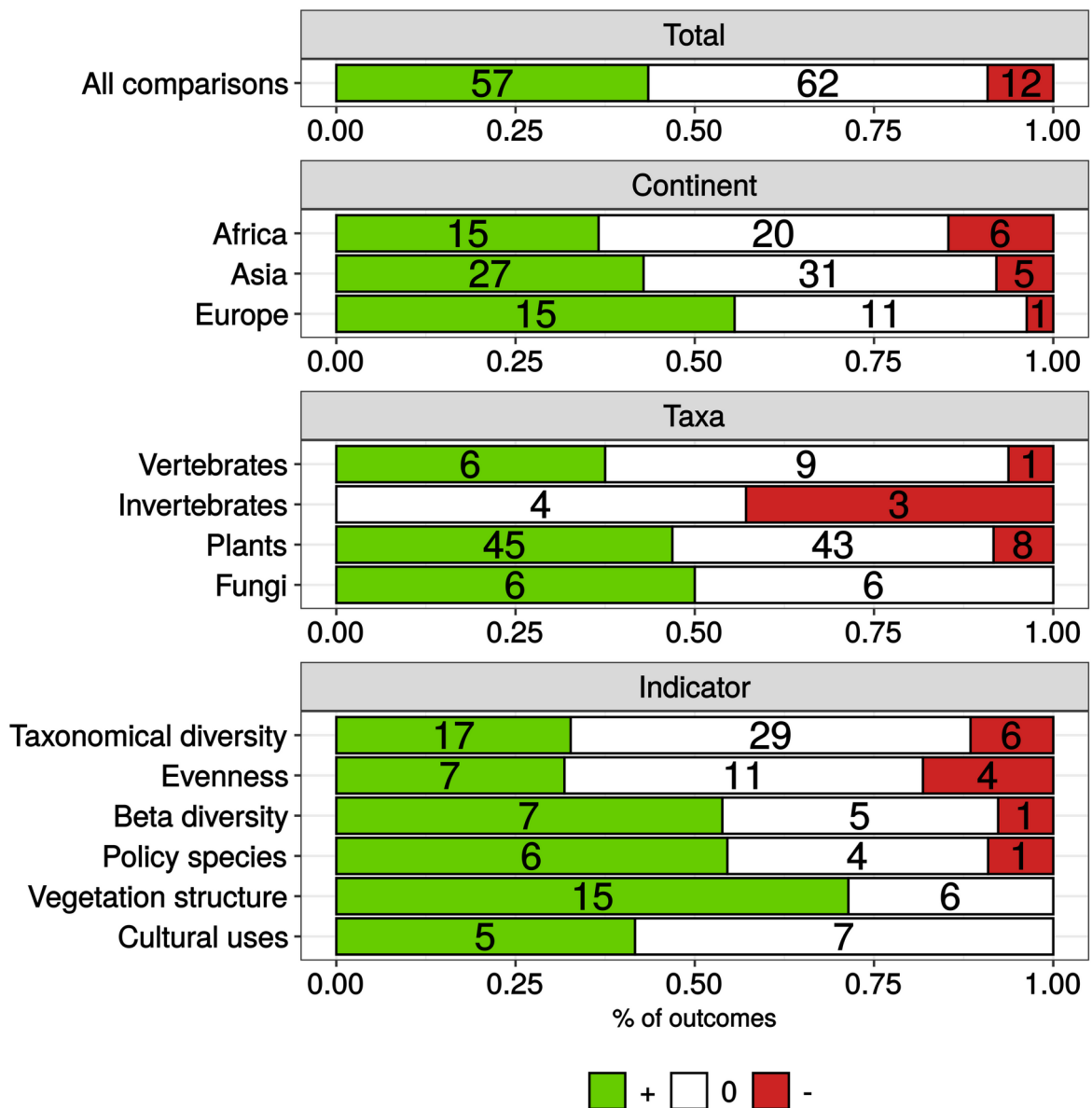
Figure 1.2. Summary of the effect of SNS on biodiversity conservation. Bar-plots show the distribution of positive (+), negative (-) and neutral (0) effects according to different classifications: *Total* comprises all the comparisons, while *Continent, Taxa* and *Indicator* show comparisons grouped by geography, taxonomy and types of metrics, respectively.

Multinomial tests for all groups of comparisons were statistically significant (Table 1), whereas post-hoc binomial tests were significant only in a limited number of cases. In particular, for all comparisons the pooled positive and neutral effects were significantly more than expected under the null hypothesis and negative effects were significantly less. When looking at comparisons split by continent, negative effects were always less than expected, while in the case of Europe we also found more positive effects and in Asia we found more neutral effects. Plants and fungi showed

significantly fewer negative effects, with plants also displaying more positive and neutral effects than expected under null hypothesis. Finally, taxonomical diversity, vegetation structure and cultural uses had fewer negative outcomes than expected under the null hypothesis, with first also showing more cases with neutral effects and the second more positive effects.

Table 1.1. Summary of the effect of SNS on biodiversity conservation displayed as results of Multinomial and Binomial *Post-Hoc* tests. Each row represents a different group (see "Group" column) arising in a class of comparisons, classes are grouped according to different classifications (see "Class" column). The first row shows all comparisons grouped together indiscriminately. *N* is the number of comparisons. Bold numbers displays significant results (alpha = .05). Arrows show the direction of significant binomial *post -hoc* tests, up-arrows mean "more than expected", down-arrows mean "less than expected".

Class	Group	*N*	*p*-value Multinomial test	*p*-value Positive effects	*p*-value Neutral effects	*p*-value Negative effects
Total	All comparisons	131	**< .001**	**.012 ↑**	**.002 ↑**	**< .001 ↓**
Continent	Africa	41	**< .001**	.621	.089	**.036 ↓**
	Asia	63	**< .001**	.108	**.02 ↑**	**< .001 ↓**
	Europe	27	**< .001**	**.044 ↑**	.416	**.001 ↓**
Taxa	Vertebrates	16	**.002**	.791	.123	.088
	Invertebrates	7	**.016**	.312	.458	.691
	Plants	96	**< .001**	**.009 ↑**	**.017 ↑**	**< .001 ↓**
	Fungi	12	**.002**	.455	.455	**.035 ↓**
Indicator	Taxonomical diversity	52	**< .001**	1	**.002 ↑**	**.002 ↓**
	Evenness	22	**.007**	1	.334	.351
	Beta diversity	13	**.006**	.279	.77	.22
	Policy species	11	**.013**	.391	.759	.345
	Vegetation structure	21	**< .001**	**.001 ↑**	.818	**< .001 ↓**
	Cultural uses	12	**.001**	.546	.142	**.035 ↓**

4 DISCUSSION

Interest in SNS from a conservation perspective has been growing over the years. However, there have been only scant efforts at systematising available knowledge and drawing key messages regarding their conservation effectiveness at a global scale. Our results strongly support the view, already advanced in a number of local studies (see e.g. Bossart et al. 2006; Ambinakudige & Sathis 2009; Brandt et al. 2013; Frascaroli et al. 2016; Nopper et al. 2017; Avtzis et al. 2018; Shepheard-Walwyn & Bhagwat 2018) and earlier overviews (Bhagwat & Rutte 2006; Dudley et al. 2006, 2010), that SNS have positive effects on biodiversity across continents and geographical settings. At

the same time, our review also underscores that the quantitative research to date has been limited to relatively few areas and taxa, resulting in evident knowledge gaps.

In the first place, the final pool of papers showed strongly uneven distributions, both spatially and taxonomically. As Dudley et al. (2010), we also found that research has largely focused on two main regions, namely the West-Central portions of India and Africa. These two areas approximate the Western African Forests and the Western Ghats of India, two global hotspots of biodiversity (Myers et al. 2000). At the same time, no papers from Americas, Oceania and large portions of Asia were retrieved. This limited geographical range is likely to hide part of the story, as there are indications that important SNS also occur in those macro-areas, although they have not been so far studied from a quantitative-ecological perspective.

Similarly, we detected a strong bias towards plants, partly reflecting a common taxonomical bias found in ecological and conservation research. Notably, plants are easier - and probably cheaper (Geijzendorffer et al. 2016) - to study than many other taxa, as they can be surveyed with relatively little time and resources and also as they already are much better known than other taxa. Several high-rank taxa which are widely studied in ecology and conservation were totally absent (e.g. fishes) or present in only one study (e.g. lichens), while the microbiota were not studied at all. These spatial and taxonomical biases that we recorded are likely due to the fact that the literature on the topic is small and largely contributed by a restricted group of scientists, specialised in the geographical areas and taxa in question. At the same time, we cannot rule out the possibility that the review protocol we adopted may have contributed to partly exacerbating those biases. For example, some papers focusing on additional geographical areas and taxa were excluded from our final pool because they did not satisfy some inclusion criteria (e.g. Castro & Aldunate 2003; Dafni 2006; García-Frapolli et al. 2007) or they were not present in Scopus at all (e.g. Deb at al. 1997).

In the second place, we noted some methodological limitations in the literature, which could be overcome in relatively easy ways. For example, some contributions have surveyed elements of

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biodiversity at SNS without comparing them with appropriate RS. While studies of this kind can have very high value from the point of view of community ecology, they do not offer a possibility of testing specific hypotheses to benchmark the conservation effect of SNS compared with neighboring areas. In other cases, the possibility to quantitatively synthesise available knowledge can be inhibited by missing information about the statistics employed, such as means and standard errors, or, more generally, by the unavailability of primary data.

In the third place, details about belief systems, management and governance would be key to understand the mechanisms whereby SNS deliver successful conservation, and evaluate their resilience or possible threats in the future (Dudley et al. 2006). Previous research has highlighted how deeply belief systems can affect the relationship between local communities and their environment. Also, different religions have different potential to influence *in situ* conservation (Mikusiński et al. 2013). Systematic data would enable the testing of possible relations between different faiths (e.g. monotheistic religions, animistic systems) and specific conservation outcomes. However, even basic details about belief systems are frequently omitted in the ecological literature on SNS. Similarly, governance regimes regulate the way communities manage their environmental resources. It has been suggested that one of the key characteristics of SNS is to include a set of mechanisms that contribute to enforcing effective and environmentally beneficial governance (Rutte 2011). However, no studies to date have systematically looked into the governance and management created under SNS. This is a gap that should be addressed in future interdisciplinary research.

Besides these limitations, our results provided overall confirmation of the biological importance of SNS worldwide. Indeed, all multinomial tests were statistically significant as well as a part of the post-hoc tests, indicating that SNS positively affect local biodiversity. These effects may be the outcomes of both unique environmental characteristics of SNS, as they often overlap with outstanding natural features (see e.g. Anderson et al. 2005; Frascaroli et al. 2016), and particular cultural factors and governance regimes, which regulate SNS use and prevent over-exploitation

(e.g., Rutte 2011; Marini Govigli et al. 2020). Moreover, while our analysis yielded no definitive proof that SNS are always beneficial to biodiversity, it produced very clear evidence that only seldom they are detrimental, despite being areas primarily dedicated to cultural uses. Indeed, in the overall pool of comparisons, positive and neutral effects were significantly more frequent and negative effects significantly less than expected. Similarly, when post-hoc tests resulted in significant outcomes, which occurred in less than half of all cases, positive and neutral effects were always observed more frequently than expected under the null hypothesis, while negative effects were always observed less frequently. The fact that the majority of the *post hoc* tests was not significant, in contrast, may indicate the lack of a clear trend in the results or suggest that the small sample size was enough to catch some signal at first, but insufficient to confirm the results when checking for the single effects.

Comparisons grouped by continent showed that Africa had the largest proportion of negative effects, although also in this case they remained significantly less frequent than expected, while Asia and Europe showed higher proportions of neutral effects and positive effects, respectively. This is an interesting finding, given that SNS are commonly thought of in connection with indigenous or non-Western contexts. The evidence we reviewed, in contrast, indicates that they have a significant positive effect on biodiversity also in relatively modernised areas of Europe. A possible explanation may be that the studies on SNS in Europe are often located in relatively less populated and accessible areas, e.g. Central Apennines (Frascaroli et al. 2016) and Epirus (Avtzis et al. 2018), while papers from Asia are mostly from Kodagu District in India, which has a moderately high human density. Therefore, we could hypothesise that geographical isolation has favored a continuity of traditional management and governance of those European SNS, although in a context of broader secularization, with positive repercussions on their biodiversity.

Results by taxa showed that SNS have a significantly positive effect on plants. While the reason for this pattern is not clear, the discrepancies among significant results for different taxa is likely due to

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the different sizes of observations. Moreover, no positive effects were found for invertebrates, although this result was not statistically significant due to the small number of observations. However, there could also be ecological reasons for the lack of positive effects on invertebrates, as groups like butterflies or nematodes, for instance, are known to produce very different assemblages as a response to the type of management and disturbance (see e.g. Hamer et al. 1997; Fu et al. 2000).

Results by type of metrics demonstrated that SNS affect mostly the number of taxa, vegetation structure and cultural uses. Available evidence suggests that all these patterns are likely to be dependent on the activities typically allowed or forbidden at SNS, often codified in the forms of taboos (Colding & Folke 2001). Stara et al. (2016), for example, reported that grazing and hunting as well as deadwood and non-timber products collection are often permitted at SNS in Epirus (Greece), whereas bans and restrictions are mostly connected to trees. A similar situation has been described with regards to the SNS of Central Italy (Frascaroli et al. 2016). In both cases, the traditional and extensive use of these sites is likely to have produced a moderate degree of disturbance, which is known to increase the local number of taxa, that is, the so called "intermediate disturbance hypothesis" (Connell 1979). Similarly, taboos related to trees are likely to have promoted the conservation of mature vegetation stands, explaining the significant effect found for vegetation structure (Marini Govigli et al. 2020). Finally, the possibility to collect forest and understory products in a culturally prominent area can explain the persistence of TEK and cultural uses of biodiversity at SNS. While these examples are both from SNS in Europe, the importance of resource-related taboos and collection of useful plants is far from limited to European SNS, having been similarly recorded in other different contexts such as, for example, India (Ormsby 2013) and the Himalaya (Salick et al. 1999).

Overall, our results newly underline the importance of recognizing and possibly ratifying the effectiveness of SNS for area-based conservation, while at the same time considering that they are

culturally sensitive areas whose primary purpose is not conservation. This would foster the application of appropriate policies and measures (IUCN-WCPA, 2019), while addressing such aims as increasing the coverage of rare or important ecosystems, supporting the conservation of threatened species and enhancing the connectivity of PA networks, among the others. Beliefs and practices associated with SNS should be part of any form of official recognition or institutionalization, as research shows that they are often key to driving the biodiversity patterns found in these areas. In this perspective, the broader framework currently developed with regards to OECMs can serve as a valuable example on which to build, to ratify the role of SNS without assimilating them to PA in the classical sense. At the same time, our review also shows that our knowledge of SNS needs to be systematised, and a number of gaps filled, before the conservation potential of SNS can be fully understood and recognised. Firstly, it is pivotal to undertake large scale surveys of SNS networks at a national or regional scale, as done for instance by Reynolds et al. (2016) or Frascaroli et al. (2019), to acquire a better sense for the extent of those networks and their distribution across biomes and elevational gradients. Secondly, it would be desirable to elaborate standardised protocols for future studies of SNS (see Borges et al. 2018 for an analogous case). Such protocols would concern the sampling of biological communities, the characterization and classification of SNS (e.g., based on their geographical extent, ecological characteristics, belief systems), as well as the analysis of governance and management mechanisms currently in place. Finally, a number of local studies should address the taxonomic and geographical gaps evidenced by our reviews, to understand whether SNS contribute to conservation of other taxa than just plants and on a truly global scale. These efforts would advance our understandings of SNS and help maximise their contribution to area-based conservation schemes.

5 CONCLUSIONS AND OUTLOOK FOR FUTURE RESEARCH

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In light of the growing recognition of SNS and OECMs in conservation policy and approaches globally, this study aimed at providing the first-ever systematic review about the effect of SNS on biodiversity conservation. Out of 2 750 papers extracted, we found 27 that met our criteria for quantitative information and analysis of biodiversity or vegetation structure. These came from studies scattered across Africa, Asia and Europe and investigated plants, animals and fungi. According to our analyses and based on a vote-counting approach, the evidence that SNS improve biodiversity conservation is quite strong, although this is not a conclusive proof. Also, the research on the topic has left large geographical areas unexplored, while studying predominantly plants. New efforts towards filling these knowledge gaps and towards improving our broader knowledge of SNS are urgent, to maximise their synergy with larger conservation schemes.

6 SUPPORTING MATERIAL

All Appendices can be found and freely downloaded at http://doi.org/10.5281/zenodo.4399641.

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CHAPTER II

Sacred Natural Sites in Italy have landscape characteristics

complementary to protected areas:

Implications for policy and planning

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ABSTRACT

There is growing awareness that protected areas (PA) may not suffice to deliver all the targets set by international conventions and guarantee the conservation of biodiversity and ecosystem services in anthropogenic landscapes. However, landscapes such as sacred natural sites (SNS), which are managed with clear benefits for people and nature although not having conservation as their primary purpose, can help deliver those goals. While a number of studies have demonstrated SNS' values for biodiversity and ecosystem services, the fundamental question of whether and how SNS may complement PA at a national scale has never been addressed. Here, we assembled a nationwide inventory of 2,332 SNS in Italy and compared their spatial distribution and landscape features with those of PA. We showed that there is scarce overlap between SNS and PA and that different factors drive the density of the two networks. SNS are more frequently associated with cultural landscapes at low and medium elevations and in extensively agricultural and peri-urban settings. PA, in contrast, are mainly found in more natural environments, at higher elevations, and farther from human settlements. These results indicate that the two networks largely complement each other and have different benefits for people and biodiversity. Land planning approaches should aim to valorize this complementarity. Instead of simply including SNS into PA, SNS could obtain a legal status through other emerging policy frameworks, such as the recognition of "other effective area-based conservation measures".

1 INTRODUCTION

Intensive agriculture and urban expansion are among the main drivers of habitat losses and land use

changes (Emmerson et al., 2016; McDonald et al., 2008; Plieninger et al., 2016) that can have

severe impacts not only on biodiversity but also social resilience and human wellbeing (Díaz et al.,

2006; Jongman, 2002; Oliver et al., 2015; Rescia et al., 2010). In the modern world, protected areas

(PA), such as parks and reserves, have been the primary instrument for conserving landscapes and

preventing negative land transformations (Dudley and Stolton, 2008; Joppa et al., 2016). Although

PA will continue to represent a pillar of environmental policies, questions are mounting about their

capacity to realistically fulfill the targets set by international conventions (SCBD, 2010) and

guarantee human and environmental wellbeing in increasingly anthropogenic landscape matrices

(Trzyna, 2007).

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A first limitation is that PA are more commonly established in mountainous and sparsely populated areas (Joppa and Pfaff, 2009). As such, they often fail to protect biodiversity and ecosystem functions in lowland landscapes, or bring benefits to urban or peri-urban populations. Other questions are strictly socio-economic and related to the scarce support from which PA suffer at times. Scarce support can range from direct conflict between PA and human communities to insufficient or constrained budgets, especially in times of economic austerity (Andrade and Rhodes, 2012; Bruner et al., 2004).

Similar considerations have contributed to sparking a growing interest in landscapes that are successfully protected or sustainably managed even outside of official conservation schemes (Kothari 2008; Naniwadekar et al., 2015; Willis et al., 2012). These can include private estates, indigenous lands, common property areas, and sites of religious importance. These categories are not necessarily separate and can overlap. What they share is that, despite not being PA proper, they often deliver positive conservation outcomes as by-products of other primary management purposes (Dudley et al., 2018). International conservation policy is giving instances of this kind increasing recognition (Borrini-Feyerabend et al. 2013; Wild and McLeod, 2008). The expression "other effective area-based conservation measures" (OECMs) has been coined as an umbrella label and the Convention on Biological Diversity has recently recognized the legal status of OECMs (Dudley et al., 2018).

These alternative forms of land protection can offer benefits in terms of both biodiversity conservation and human wellbeing (e.g., Garnett et al., 2018). However, there are challenges in estimating those benefits and maximizing the synergy with official conservation frameworks and land planning approaches. Part of these challenges derive from incomplete knowledge of the distribution and characteristics of these areas of informal protection (Dudley et al., 2018), leading to uncertainties on their possible role in planning and conservation, and the legal status that they should be given.

This situation fully apply to Sacred Natural Sites (SNS). On the one hand, SNS have probably been better studied than most other forms of OECMs (Verschuuren et al., 2010). SNS are often regarded as forming "shadow" conservation networks in many countries (Dudley et al., 2009), and a significant body of literature documents their benefits for biodiversity and human wellbeing in both natural and urban contexts (last reviewed in Dudley et al., 2010). These findings are commonly based on field investigations of restricted regions and a limited number of sample sites. On the other hand, there are very few examples of systematic mapping and large-scale analyses that examine how the SNS network is arranged and characterized at a national scale, and draw the implications that this has on policy and planning. Similarly, there are contrasting views as to how SNS should be treated in relation to legal status and official conservation schemes. Some argue that SNS should be included in official PA networks, as they are exposed to various threats due to a lack of legal recognition (Avtzis et al., 2018). This may be especially true for the SNS of indigenous people in ancestral lands, which face severe pressures from mining and resource extraction (Verchuuren et al., 2010). Others, in contrast, underline that SNS are not tantamount to PA, as their primary function is not to conserve biodiversity, and this basic difference should be considered in policy and planning (Rutte, 2011). Also, there are some indications that including SNS in PA may be detrimental, if it leads to standardized management and, consequently, loss of biological specificity (Frascaroli et al., 2016).

Here, we provide one of the first quantitative assessments of whether SNS and PA networks may complement each other, as commonly assumed, using a nationwide geo-referenced database of SNS in Italy. We focus on SNS in Italy, as they constitute a rich and relatively well documented model system. Indeed earlier studies have underlined the abundance and heterogeneity of SNS in the central part of Italy (Frascaroli, 2013), their importance for both biodiversity and humans (Frascaroli et al., 2014, 2016), and their role in the governance of natural resources (Frascaroli, 2018; Frascaroli and Fjeldsted, 2017). Despite the difference in study settings, these findings are

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quite close to those from apparently distant cultural and geographical contexts (e.g., Rutte, 2011; Dudley et al., 2010; Bhagwat and Rutte, 2006). .

In the present study, we advanced our understanding of Italian SNS and their role in conservation, by mapping SNS over the entire Country and analyzing the spatial distribution and key landscape-scale attributes of this network, as compared to the national network of PA. In particular, we tested the hypotheses that: (1) There is only limited spatial overlap between SNS and PA and they represent largely distinct networks; (2) the density of the SNS and PA networks are driven by different factors and landscape characteristics; and (3) despite their inherent differences, both SNS and PA networks may retain important values for conservation and land planning. While we focused on SNS in Italy, our approach and results can be applicable to a broader range of geographical contexts and cultural landscapes.

2 METHODS

2.1 Study area

Italy is characterized by wide altitudinal and latitudinal gradients (0 to 4800 m ASL, and 35° N 29′ 24″ to 47° N 04′ 22″, respectively), which create a diverse range of biophysical conditions across an overall surface area of ca. 301,000 km^2. The elevation profile is rugged, with a prevalence of mountainous 35.2% and hillside 41.7% areas (ISTAT, 2018).

Italy has a population density of ca. 201 inhabitants/km^2, which is above the average of Europe and especially Southern Europe. However, the population is unevenly distributed, with the Po Valley alone accounting for nearly half of the national population, and vast mountainous areas of the Alps and Apennines being only sparsely inhabited. There are only ten large cities with more than 300,000 inhabitants, and 70% of the Italian population is categorized as 'urban' (CIA, 2018), which is one of the lowest figures in Western Europe. Conversely, there are above 5,500 settlements with less than 5,000 inhabitants, often located in mountainous or peripheral areas. The Country is administratively

divided into 20 regions.

Human presence in the Italian islands and peninsula has very remote origins, as in the rest of the Mediterranean, and dates back to the Paleolithic (Blondel et al., 2010). This millennial history has produced layered interactions between people and environment, resulting into a number of cultural landscapes (Antrop, 2005), some of which retain high ecological and aesthetic value (Agnoletti, 2014). Italian SNS can be viewed as particular instances of cultural landscape (Frascaroli and Verschuuren, 2016). They are invariably associated with Roman Catholicism, although they are only in part a direct expression of Church-driven religion. Indeed a significant proportion of Italian SNS (about 33% in Central Italy, according to Frascaroli (2013)) rather includes worship sites that are directly managed by local people, possibly in collaboration with the priests. These SNS in particular resemble the sacred groves and SNS found in other cultural contexts, such as parts of Africa and India (Frascaroli et al., 2016; Bhagwat and Rutte, 2006), as they play a direct role in the social dynamics of rural communities and governance of natural and common-pool resources (Frascaroli, 2018; *cf.* Rutte, 2011). While this role has clearly declined following processes of secularization, modernization and erosion of traditional institutions, it remains recognizable and vital in a number of instances (Frascaroli and Fjelsted, 2017). Further, these local shrines and worship sites appear to be quite frequently established on earlier pre-Christian sacred sites. In Central Italy, this is the case for about 12% of all shrines based on strict archaeological evidence alone (Frascaroli, 2013), although this proportion may be higher if vernacular and oral tradition is also considered (De Waal, 2012). In contrast, only 2% ca. of SNS related to more official forms of Catholicism (e.g., monasteries) have a comparable pre-Christian origin.

While there have not been systematic mapping efforts of SNS in Italy to date, available evidence suggests that their spatial distribution tends to follow a broader pattern recorded at the European level (Nolan and Nolan, 1989), with SNS being generally found either in very remote settings or in natural and semi-natural areas in close proximity of human settlements (Frascaroli, 2018). This is

reflected in the land cover types observed around SNS in Central Italy, which are natural (i.e., forests and mountain grasslands) in nearly half of cases (48%), while consisting of matrices of semi-urban and semi-natural patches for the rest (Frascaroli, 2013). Field investigations of Italian SNS indicate that they tend to be located in a 200-1800 m ASL elevation range, with a large majority found below 1000 m (Frascaroli et al., 2016). Field studies have also demonstrated the conservation value of these sites, as expressed by plant richness and the occurrence of old-growth trees and plant species of conservation importance in comparison to surrounding areas (Frascaroli et al., 2016), similarly to what found in non-European contexts (e.g., Salick et al., 2007). Finally, the occurrence of useful plant species in SNS, despite significant, appears to be related more, although not exclusively, to past uses than current livelihoods and practices (Frascaroli et al., 2014).

2.2 Sacred natural sites identification

To compile a comprehensive database of SNS in Italy, we built on a similar approach as previously used to identify SNS in Central Italy (Frascaroli, 2013). We searched for possible published sources containing indications about sacred sites in Italy. To minimize the geographic bias, we only focused on sources with a national coverage and excluded local publications. We identified seven sources, that is, five books (Romanò, 1990; Bosi, 1992; Farnedi, 1996; Grasselli & Tarallo, 1994; Gottardo, 1994) and two online repositories (CSC, 2003; CNS, 2016).

We geo-located every site listed in these sources through a remote sensing search engine (Google Earth®). To determine which of all those sites qualify as 'SNS', we used a Corine Land Cover map (henceforth CLC) from 2012, which classifies Italy's land surface in 95 categories organized along 5 hierarchical levels (MATTM, 2012). To work with a more manageable number of categories, we reclassified the map in 7 macro-categories: 'Urban / built', 'Urban green', 'Intensive agriculture', 'Extensive agriculture', 'Natural', 'Alien' and 'Freshwater and wetlands'. With Geographic Information Systems (GIS) software, we then examined land cover composition within a radius of

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250 m around each mapped sacred site. We considered as SNS all sites that had at least 1 ha of land cover in the categories: 'Natural', 'Extensive agriculture', 'Freshwater and wetlands'. We included 'Extensively agricultural' land cover, as these land covers are often associated with traditional management and cultural landscapes with high aesthetic and ecological values (Barthel et al., 2013; Fischer et al., 2012). A 250 m buffer around each site was chosen because this size should be sufficiently small to capture landscape composition at a local scale, but sufficiently large to account for SNS that include extensive built structures. The threshold of 1 ha was selected on the basis of existing literature indicating that many SNS around the world have a very limited area but can still retain high relevance for conservation and ecosystem services (Bhagwat and Rutte, 2006).

2.3 Characterization of sacred natural sites and protected areas

We used GIS to characterize SNS and PA along key environmental, demographic, and administrative variables. For this purpose, we converted available geo-referenced data to raster layers with a 500 m cell size and extracted the value of all cells for each of the following variables: land cover (CLC) and landscape dominant land cover; landscape mean altitude (m ASL) and ruggedness (measured as the Terrain Ruggedness Index (TRI) proposed by Riley et al. (1999)); population density; distance from the nearest road or track; administrative region (21 levels, including the small State of St. Marin); and presence of a SNS and/or PA.

We used land cover, altitude, and ruggedness as covariates, to test the assumption that SNS may complement PA globally by protecting lowland landscapes and agrarian matrices that are seldom included in PA (Bhagwat and Rutte, 2006; Bhagwat et al., 2005). Similarly, we considered population density and distance from the nearest road as proxies for human accessibility and possible human-induced influences, as also suggested by Reynolds et al. with regards to SNS in Ethiopia (2017). We also chose administrative region as the last covariate for two reasons. Firstly, it can be interpreted as a proxy indicator of the effect of space. Secondly, as the sources we used to

compile our database of SNS are organized by region, checking for region could allow to detect possible biases and disparities inherent to data collection. A similar consideration applies to PA, given that the creation and extent of local PA is decided by regional administrations on the basis of national directives.

The impossibility to clearly ascertain the borders and extent of each SNS, finally, represented a challenge for determining the extent of the SNS network. The 250 m buffer that we used to identify SNS is a conservative measure that we adopted to obtain a stricter selection, although it likely underestimates the area of influence of many SNS (*cf.* Frascaroli, 2013). Consequently, in the following analyses, we considered a buffer area of 1,250 m around each SNS as a tentative estimate of SNS' area of influence that may be closer to their real extent. We defined the SNS network as consisting of all raster cells that fall within the 1,250 m buffers for at least 50% of their extent.

2.4 Statistical analyses

To analyze the distribution and overlap of the SNS and PA networks, and how these may vary across the Country, we first calculated the number of SNS in each region and used a linear regression to test whether their abundance can be explained by the log transformation of the regional surface area. We then compared the density of SNS/km^2 and the proportion of surface area included in PA within each region. As a general measure of overlap between the two networks, we counted the number of raster cells that are part of both networks, both overall and across different altitude ranges.

To test what variables drive the density of the two networks, we built distinct GLMs. We modeled network density expressed as the proportion of cells belonging to the SNS or PA network, respectively, as a function of landscape altitude, ruggedness, dominant land cover, distance from road network, population density, administrative region, and a distance-weighted function of neighboring response referred to as 'auto-covariate'. Adding an auto-covariate to GLMs is one of

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the most common and flexible techniques for correcting the effects of spatial auto-correlation (SAC) in spatial regressions (Dormann et al., 2007). Since SAC is also known to be highly dependent on aggregation level (i.e., the size of the raster grid examined; Overmars et al., 2003), we tested our models at different aggregation levels. We considered progressive grids consisting of 5 x 5, 11 x 11, 21 x 21, and 51 x 51 cells (i.e., 2.5 x 2.5, 5.5 x 5.5, 10.5 x 10.5, and 25.5 x 25.5 km, respectively). We thus built a total of eight models, each of which analyzes the density of either the SNS or PA network at one of four possible scales. As the response variable always consisted of a proportion, we used a binomial distribution or quasi-binomial in case of over-dispersion of the data. We performed analysis of variance (ANOVA) on the models and compared the proportion of deviance explained by each variable across the two networks and four aggregation levels. We used Moran's I to test for SAC in the models' residuals.

Finally, to compare landscape-scale characteristics of the SNS and PA networks, and assess their potentials for conservation, we computed basic descriptive statistics: mean and standard deviation for numerical variables (altitude, TRI, distance from roads, and population density), frequency and relative frequency for categorical ones (CLC and CLC categories).

All analyses were performed with the GIS software QGIS v2.18 (QGIS Development Team, 2016) and the statistical software R v3.4.3 (R Core Team 2017).

3 RESULTS

3.1 Spatial distribution of sacred natural sites and overlap with protected areas

We identified a total of 2,332 SNS unevenly distributed across Italy (Figure 2.1a). Of all mapped SNS, ca. 43% are located in the northern regions of the Country, 35% in the center, and 22% in the south and islands. Although larger regions tend to host a significantly higher number of SNS (p < .001), this relationship accounts for only ca. 30% of the variability in the number of SNS per region, indicating that there are other factors driving SNS abundance. Further, SNS are clustered in

some areas, like the Northwest and Central Italy, or at the margin of reliefs, while less frequent in other areas, especially in lowlands (Figure 2.1b).

The PA network also has scarce coverage in lowland areas, but despite this similarity there is no apparent overlap between SNS and PA. Many SNS fall outside of PA (Figure 2.1c) and regions with a high percentage of surface area included in PA do not necessarily have higher density of SNS (Figure 2.2). Overall, there is greater regional variation in the density of SNS (coefficient of variation = .86 of a mean regional density of .01 SNS/km^2) than in PA coverage (coefficient of variation = .38 of a mean .22 regional surface being included in the PA network).
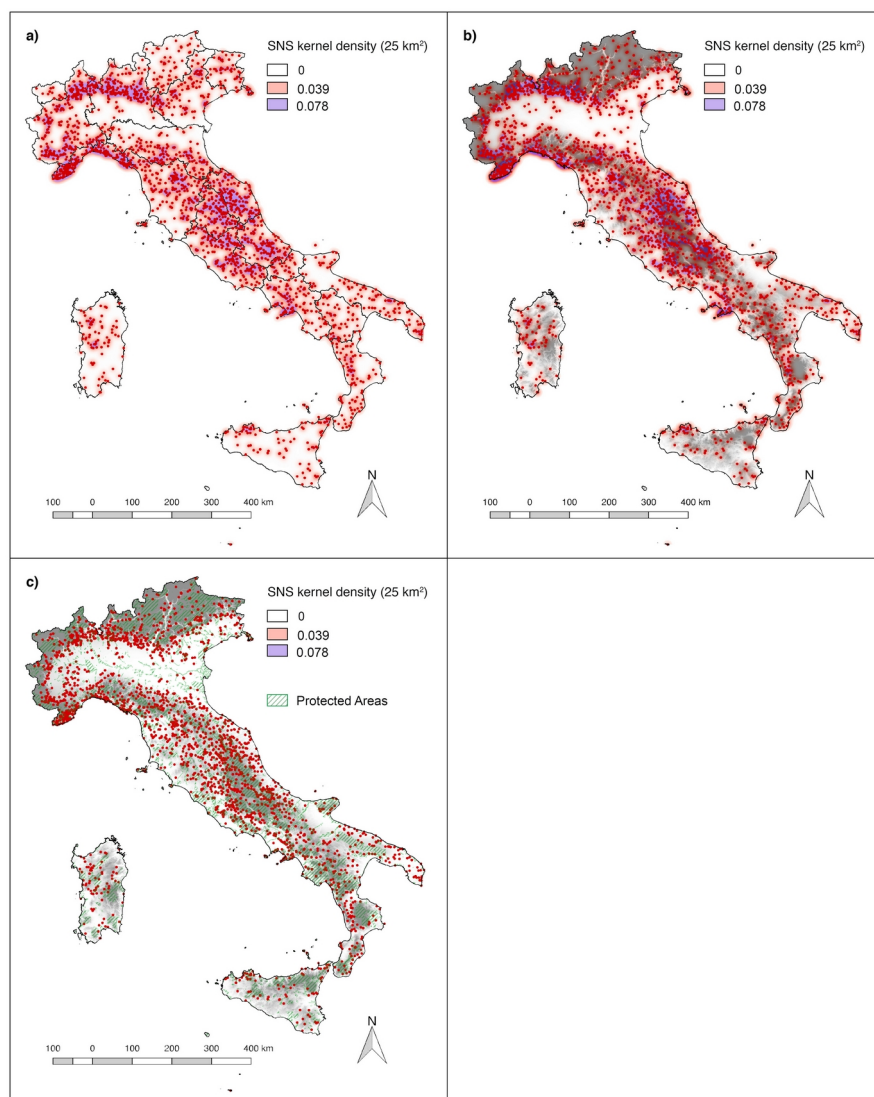
Figure 2.1. Distribution of sacred natural sites across Italy, highlighting: (a) site density across regions; (b) site density with respect to topography; and (c) overlaps with official protected areas.

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Based on our estimate, the areas under the influence of SNS would constitute a network of 10,314 km² (i.e., 41,256 raster cells), that is, ca. 3.4% of the national land surface (Table 2.1). In contrast, we counted the PA network as consisting of 65,493 km² (i.e., 261,973 cells), nearly corresponding to 21.7% of the national land surface. Overall, only 7,525 (0.6%) of all cells belong to both SNS and PA. Further, SNS and PA tend to have greater coverage in different altitudinal ranges: SNS below and PA above 1,000 m ASL, respectively.
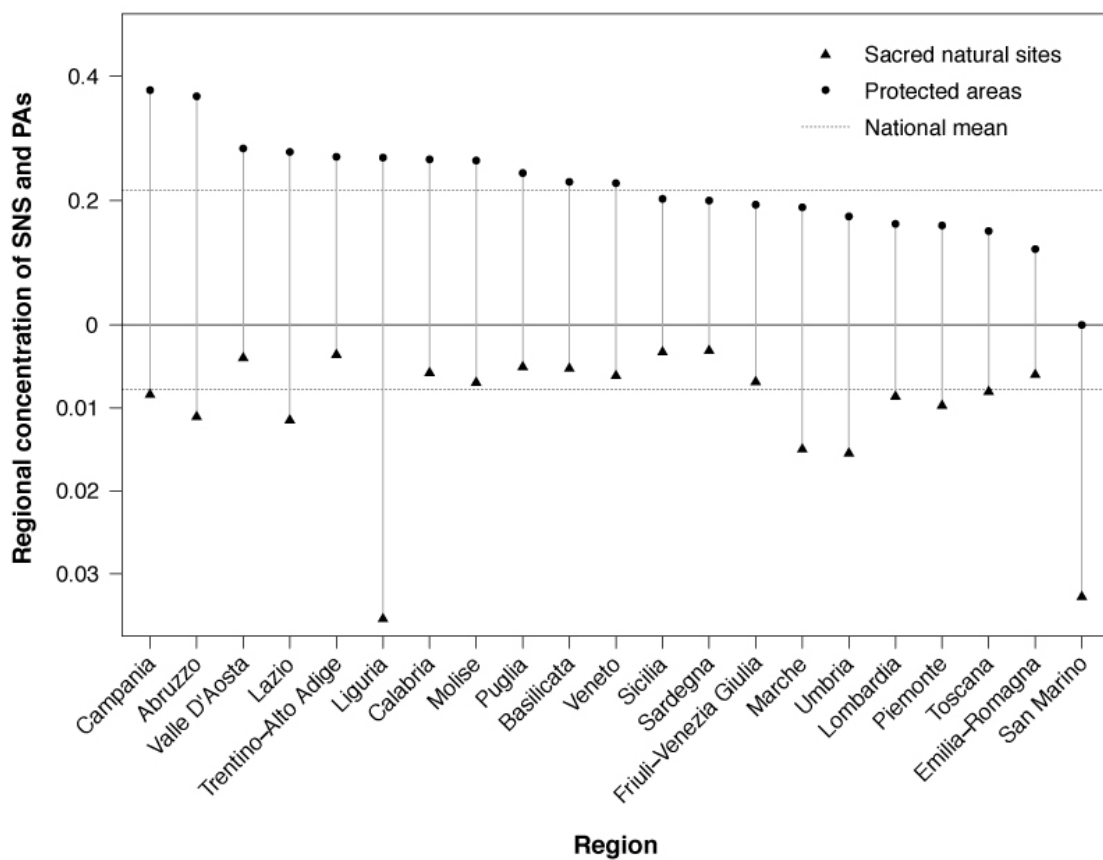
Figure 2.2. Comparison of the concentration of sacred natural sites and protected areas across regions, expressed as proportion of regional surface area included in protected areas and number of sacred natural sites per km², respectively.

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Table 2.1. Extension of sacred natural site and protected area networks. The table reports the amount of land (km^2) and relative proportion (in italics) that is included only in protected areas, only in sacred natural sites, and in overlaps of both networks, across altitudinal ranges.

	Altitude range									
	0 – 500 m	*500 – 1000 m*	*1000 – 1500 m*	*1500 – 2000 m*	*2000 – 2500 m*	*2500 – 3000 m*	*3000 – 3500 m*	*3500 – 4000 m*	*4000 – 4500 m*	*Total*
PA only	22,920	15,856	11,427	6,142	3,897	1,947	431	49	5	63,612
	0.122	*0.254*	*0.464*	*0.496*	*0.526*	*0.602*	*0.683*	*0.845*	*0.750*	*0.211*
SNS only	5,567	2,363	406	63	11	6	0	0	0	8,433
	0.030	*0.038*	*0.016*	*0.005*	*0.001*	*0.002*	*0.000*	*0.000*	*0.000*	*0.028*
Both	781	716	282	62	11	3	0	0	0	1,881
	0.004	*0.011*	*0.011*	*0.005*	*0.002*	*0.001*	*0.000*	*0.000*	*0.000*	*0.006*
Total	29,268	18,935	12,114	6,267	3,919	1,956	431	49	5	73,926
	0.155	*0.304*	*0.492*	*0.506*	*0.529*	*0.605*	*0.683*	*0.845*	*0.750*	*0.245*

3.2 Drivers of sacred natural sites and protected areas distribution

Comparing the deviance explained by our series of binomial models suggests that different factors drive the spatial density of SNS and PA, respectively, and that these effects vary across scales (Table 2.2).

Table 2.2. Synthetic overview of the deviance explained by auto-(quasi-)binomial models of sacred natural site and protected area densities across spatial scales, and residual spatial auto-correlation.

	Explained variability							
Variable	**Sacred natural sites**				**Protected areas**			
	2.5 km	5 km	10 km	25 km	2.5 km	5 km	10 km	25 km
sqrt(Landscape altitude)	0.0003	0.0011	0.0043	0.0195	0.1182	0.1425	0.1717	0.2318
sqrt(Landscape ruggedness)	0.0278	0.0301	0.0317	0.0302	0.0034	0.0027	0.0023	0.0022
Dominant land cover	0.0577	0.0759	0.1028	0.1714	0.0848	0.0775	0.0618	0.0343
sqrt(Distance from roads)	0.0180	0.0187	0.0183	0.0205	0.0053	0.0050	0.0035	0.0012
sqrt(Population density)	0.0243	0.0213	0.0177	0.0186	0.0110	0.0083	0.0054	0.0025
Region	0.0256	0.0452	0.0855	0.1981	0.0671	0.0793	0.0968	0.1352
Spatial lag (auto-covariate)	0.6947	0.5426	0.4312	0.3392	0.6539	0.6124	0.5820	0.5281
Dist. from roads : Pop. dens.	0.0007	0.0010	0.0021	0.0050	0.0000	0.0000	0.0001	0.0000
Altitude : Ruggedness	0.0008	0.0020	0.0027	0.0041	0.0001	0.0002	0.0004	0.0005
Altitude : Land cover	0.0004	0.0012	0.0024	0.0030	0.0001	0.0002	0.0002	0.0002
Ruggedness : Land cover	0.0003	0.0007	0.0003	0.0000	0.0000	0.0000	0.0000	0.0000
Altitude : Ruggedness : Land cover	0.0002	0.0006	0.0012	0.0018	0.0001	0.0001	0.0001	0.0002
Residuals	0.1492	0.2596	0.2998	0.1886	0.0560	0.0718	0.0757	0.0638
	Spatial auto-correlation of the residuals							
Moran's I	0.778*	0.950*	0.979*	0.982*	0.819*	0.961*	0.986*	0.991*

* p ≤ .0001

Dominant land cover at a landscape scale and regional location had a strong effect on the density of both SNS and PA. Altitude resulted to be the single most important factor for explaining the density

of PA, while it was very marginal for SNS. In contrast, there was a relatively stronger effect of landscape ruggedness (TRI) on the density of SNS than PA. Distance from the closest road and population density consistently had a very marginal effect in all the models.
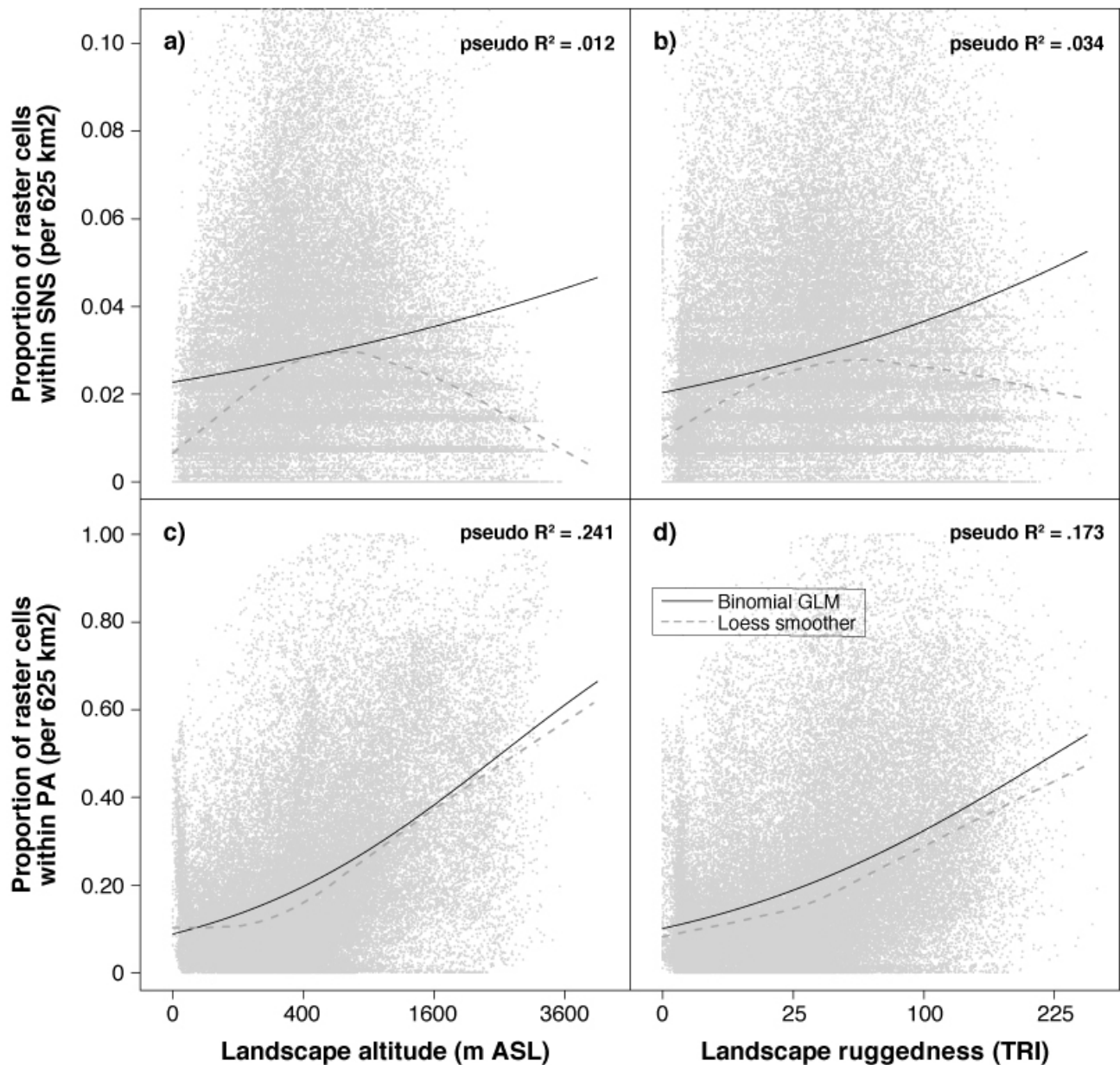

Figure 2.3. Proportion of 500 x 500 m raster cells included in sacred natural sites (a-b) and protected areas (c-d) at a broad scale (25 x 25 km), and their relation with landscape altitude (a, c) and landscape ruggedness (b, d). In each panel, a linear relation modeled on a quasi-binomial distribution is compared with a LOESS smoother. Only a random sample of 120,506 points (10% of the entire dataset) is displayed in the graphs. The 25 x 25 km scale was selected among the ones considered, as it is the one that showed the strongest relations between the variables.

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A closer look at the relationship with landscape altitude and ruggedness revealed that SNS density tends to slightly increase along both metrics, but this relationship could be best expressed as bell-shaped rather than linear (Figure 2.3a-b). In other words, SNS are more likely to be located at moderate ranges of elevation and topographic ruggedness, beyond which they become more sporadic. The relationship between PA density and both altitude and ruggedness, in contrast, was much stronger and approaching linearity (Figure 2.3c-d). Higher elevation and more rugged areas are more frequently protected, although the effect of the latter characteristic is weaker and does not explain additional variability as compared to altitude.

Table 2.3. Synthetic overview of key landscape attributes of the cells included in the sacred natural site and protected area networks, respectively. Mean and standard deviation are reported for numerical variables, frequency and relative frequency for categorical ones.

	Sacred natural sites		Protected areas	
Variable	*Mean*	*SD*	*Mean*	*SD*
Altitude (m ASL)	470 ±	*361*	892 ±	*732*
Roughness (TRI)	47 ±	*35*	63 ±	*47*
Population density	431 ±	*1273*	45 ±	*414*
Distance from roads (m)	1109 ±	*2995*	2309 ±	*3507*
Land cover type	*N. of cells*	*Prop. of cells*	*N. of cells*	*Prop. of cells*
Alien	328	0.008	1107	0.004
Extensive agricultural	15136	0.367	59833	0.228
Intensive agricultural	4658	0.113	28968	0.111
Natural	16486	0.400	160552	0.613
Urban / built	4075	0.099	2922	0.011
Urban green	82	0.002	290	0.001
Water	491	0.012	8301	0.032

In general, the models focusing on the smallest and largest spatial scales (i.e., 2.5 x 2.5 and 25 x 25 km, respectively) explained most variability. However, the majority of the explanatory power at the 2.5 x 2.5 km scale was contributed by the auto-covariate, indicating a particularly strong SAC. The weight of the auto-covariate, progressively decreased at larger scales, while other variables gained in importance, mainly land cover and region for SNS, altitude and region for PA. These results confirm that the spatial distributions of both SNS and PA tend to be clustered in non-random areas, with the effect of clustering becoming less marked at broader scales. Despite a lot of variability (ca.

70-94%) being explained by all the models, the outcome of Moran's test for SAC, as expressed by Moran's I, was always statistically significant ($p \leq .0001$), indicating that there models' residuals remain spatially auto-correlated and, consequently, additional explanatory variables are likely to be at play than those considered.

3.3 Landscape characteristics of sacred natural sites and protected areas

SNS are clearly associated with landscapes that are located at considerably lower altitude, less rugged, more densely populated, and closer to roads as compared to PA (Table 2.3). This is partly reflected in distribution across land cover categories, given that a larger portion of the SNS network consists of urban or artificial land cover (ca. 10% vs. 1%). Further, both SNS and PA have similar proportions of intensively agricultural land covers, while higher proportions of extensively agricultural land covers characterize SNS (ca. 37% vs. 23%) and higher proportions of natural land covers characterize PA (ca. 61% vs. 40%), respectively.

Substantial differences ($\geq 2\%$) in the proportions of cells occupied by original CLC categories offer more fine-grained insights into the differences between SNS and PA (Figure 2.4). Notably, a higher proportion of different natural land covers, both including several forest types and grasslands (CLC codes 3.1.1 and above), was found in PA, while SNS included more "discontinuous urban fabric" (1.1.2) and land covers that are distinctive of traditional livelihoods and agricultural systems (2.2.3, 2.4.2, 2.4.3, 3.1.14).
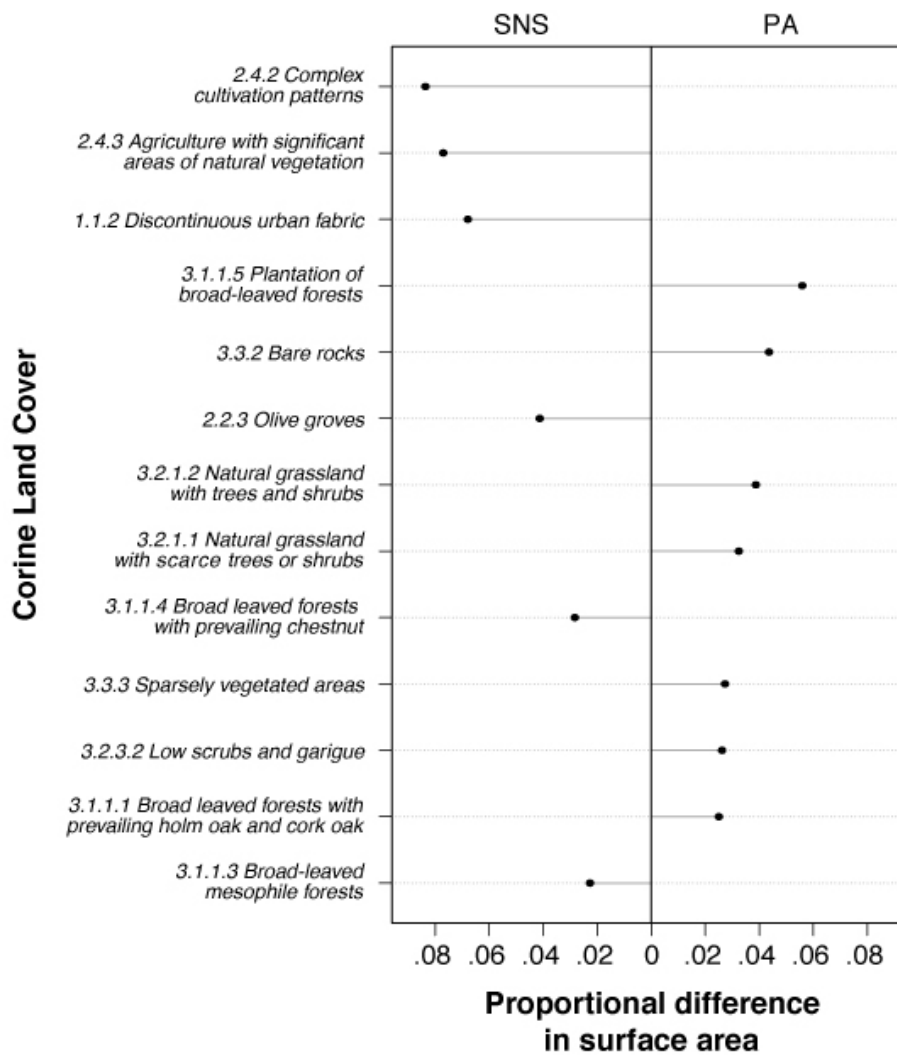
Figure 2.4. Difference in the proportion of cells that specific land covers occupy in the sacred natural site and protected area networks, respectively. Only Corine Land Cover levels for which the difference is substantial (≥ 2%) are displayed.

4 DISCUSSION

Our results support the view that SNS and PA constitute distinct networks with regards to both spatial distribution and landscape characteristics across all of Italy. While earlier studies underlined how SNS often overlap with PA (Dudley et al., 2009; Mallarach et al., 2012), our results revealed that such overlap is limited. The simplest explanation of this scarce overlap is that, in the Italian context, SNS and PA occur in different landscapes. Overall, the distribution of PA conforms to what is frequently indicated as a limitation of conservation networks worldwide, that is, being skewed towards high elevations and remote areas (Baldi et al., 2017). SNS, in contrast, are distributed at

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lower elevations and have widest coverage between 0 and 1,000 m ASL, that is, where the relative coverage of PA is most limited.

The occurrence of SNS at relatively lower altitudes than PA is also accompanied by a considerably lower proportion of natural land covers included in the SNS network, as compared to the PA network (i.e., 40% vs. 61% ca., respectively). However, two key aspects suggest that SNS still play an important role for conserving valuable biodiversity and landscapes, despite being characterized by lower levels of 'naturalness' than PA.

Firstly, SNS apparently provide effective defense against the expansion of intensive agriculture. The relative occurrence of intensively managed agricultural land is nearly the same for both SNS and PA. This is even more remarkable given that SNS are located at lower altitudes and in less rugged areas, which should be more immediate targets of agricultural intensification. Secondly, SNS encompass relatively large portions of complex agro-forestry matrices, semi-natural areas dedicated to traditional productions (i.e., olive and chestnut groves), and small-scale diversified agricultural systems. These constitute some of the most distinctive types of cultural landscape in Italy (Agnoletti, 2014; Agnoletti et al., 2015; Pezzi et al., 2017). Although they do not represent 'natural areas' in a strict conservation sense, cultural landscapes of this kind can harbor significant biodiversity (Nascimbene et al., 2016) and provide key ecosystem services (Bernués et al., 2014; Plieninger et al., 2014). The fact that SNS include a higher proportion of these land covers than PA further corroborates the hypothesis that SNS complement the PA network by protecting types of landscape that are less represented in PA (Bhagwat and Rutte, 2006).

Another difference between the SNS and PA networks is the greater extension of the former in artificial environments, especially urban peripheries and residential areas. While this finding also contradicts a perception of SNS as almost pristine areas, it further emphasizes their potential for sustaining biodiversity and ecosystem services in areas often neglected by PA and other landscape planning schemes (Soanes et al., 2018). Scholarly attention for SNS in urban areas is only a recent

development (Jackson and Ormsby, 2017). Nonetheless, several studies have already underlined how urban SNS can contribute to hosting species of conservation concern and providing cultural ecosystem services in highly urbanized settings (de Lacy and Shackleton, 2017; Gopal et al., 2018). The importance of this ecosystem service should not be underestimated, given that having access to direct experiences in and of nature is a challenge that urban populations are increasingly facing (Chiesura, 2004; Ward Thompson et al., 2012).

In all, our data portray a clear picture, for which SNS and PA markedly differ as to purposes and spatial characteristics. While PA are designed to mainly protect remote areas located at medium to high altitudes, SNS represent transitional areas between nature and culture. Therefore, they are characterized by important patches of nature, but in close proximity to agricultural landscapes, ways of transit, and inhabited areas. It has been highlighted that SNS in Italy often act as transitional spaces with regards to the social dynamics of local communities (Frascaroli, 2018). Our present results suggest that SNS occupy a similar place also at the spatial level, mediating between anthropogenic artificial and natural landscapes. As SNS largely consist of typical cultural landscapes and agro-forestry matrices, it may be hypothesized that they also offer a sort of "afterimage of the landscape and population distribution in rural Italy up to the 1960s, when rural abandonment began en masse (Blondel et al., 2010).

We are aware that our results may be partly influenced by biases in the source data. For example, it is not intuitive to explain the high variability in the density of SNS across regions. Similarly, we find no evident historical or religious reasons for the very high concentration of SNS in Liguria and North Italy and relative scarcity in the South. We assume that these disparities may reflect uneven research intensities used in the sources that have been used to build our inventory. Similarly, it is likely that our decision not to include local sources in the database compilation left out a number of the more obscure and remote SNS, which are likely to occur in settings that are more similar to PA or within PA themselves (*cf.* Frascaroli, 2013). While this omission allowed to provide a more

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conservative and perhaps realistic estimate of the coverage of the SNS network, it may have partly

exacerbated the differences between SNS and PA. These caveats aside, we believe that in virtue of

the large dataset analyzed our results remain robust overall. As such, they reveal important

differences between SNS and PA in Italy and help elucidate how the two networks can complement

one another.

While our findings strictly refer to SNS in Italy, future research should aim at assessing whether

they may also apply to comparable study systems in the rest of Southern Europe. The approach that

we used to map SNS demonstrates how identifying potentially key landscapes can be attained with

limited means, starting from non-specialized sources. A similar approach could hopefully be tested

and exported to other contexts.

5 IMPLICATIONS FOR POLICY AND PLANNING

Our results suggest that the inclusion of SNS into PA, which is sometimes advocated, may not

always be the most appropriate planning approach, at least not in our study area. Plain incorporation

of SNS into PA might have two main shortcomings. Firstly, in some cases it can underplay the

specific characters of SNS as compared to PA, possibly leading to homogenization rather than

valorization of their peculiar strengths. It is known that inclusion in PA can lead to discontinuing

some of the human activities that are key to maintaining cultural landscapes (Schmitz et al., 2012).

This concern is in line with the results of previous studies comparing the biological makeup of SNS

within and outside PA (Frascaroli et al., 2016), as well as broader experiences with cultural

landscapes in Italy (Agnoletti, 2014). Secondly, the inclusion into PA of SNS that retain functioning

governance structures can overlook one of the potential advantages of SNS and other OECMs, that

is, contributing to *in situ* conservation and human wellbeing, without burdening on scarce economic

resources (Bhagwat and Rutte, 2006).

While our results invite to some precaution by evidencing inherent differences between SNS and

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PA, they are per se not sufficient to support a specific course of policy action. Rather, we suggest that decisions should be taken on a case-by-case basis, ideally based on interdisciplinary assessments that consider not only the biological characters of SNS, but also their cultural and institutional contexts. As a general guideline, we believe that policies should aim to support the governance mechanisms inherent to SNS when those are still in place, as it is still the case for a number of SNS in Italy (Frascaroli and Fjeldsted, 2017; Frascaroli and Verschuuren, 2016), or help revitalize them whenever feasible. In such cases, and especially in the presence of demonstrated conservation values, SNS may additionally qualify to be recognized as OECMs (Borrini-Feyerabend et al., 2013; IUCN WCPA, 2017). This would grant them a form of legal recognition, even without incorporation into PA, and permit them to count towards the national achievement of Aichi target 11 (SCBD, 2010). In contrast, inclusion of SNS into the PA network may be a more appropriate course of action when the governance structures and management regimes relative to SNS have been lost altogether and cannot be revived, for example following abandonment or discontinued use. Many SNS in Italy also fall in this category (Frascaroli, 2013). Under similar circumstances, designation as a PA could help safeguard the combined natural and cultural heritage of which SNS generally consist, even in the absence of traditional governance structures. Although these general suggestions are based on the study of sns in italy, we believe that they may have wider applicability, especially in contexts where sns are not imminently threatened from destruction and encroachment, and therefore do not require immediate legal protection.

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CHAPTER III

Sacred Natural Sites and Protected Areas as drivers of forest cover changes in Italy from 1936 to 2018

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ABSTRACT

Forests will be critical to face climate and global changes, therefore knowledge on the drivers of forest cover changes are pivotal. While the drivers of deforestation are quite known, few is known about drivers of afforestation. Moreover, Protected Areas (PA) are known to effectively decrease deforestation. Other types of area-based conservation measures different to PA exhist, among these Sacred Natural Sites (SNS) have demonstrated their capacity to deliver positive conservation outcomes while making up an extensive "shadow network" of conservation. However, few is known on the capacity of SNS to regulate land use changes. Here we explored the role of SNS and PA as drivers of forest loss and forest gain in Italy between 1936 and 2018. We performed a descriptive analysis and modelled forest gain and forest loss by means of spatial binomial GLMs with residual autocovariates. The main drivers of forest cover changes were geographical and topographical factors such as latitude, longitude and elevation. Nonetheless, SNS and PA significantly decreased forest loss and increased forest gain. While the negative relation between SNS and forest loss is a desirable outcome, the positive relation with forest gain is at least concerning, as it points at an abandoment of such areas with consequent loss of rare open habitats. We call for a legal recognition of SNS and active management to help mantaining their role as informal PA. Finally, our study highlights the need for novel conservation planning, targeting less isolated areas than typical PA, to conserve a portion of such areas threatened by human activity and close an historical spatial gap in PA networks.

KEYWORDS: Forest Gain; Forest Loss; Protected Areas; Sacred Natural Sites; Spatial Modelling; OECMs.

1 INTRODUCTION

Forests are considered to play a pivotal role in face of climate change and, more generally, of global environmental change (FAO & UNEP 2020). Besides representing the primary carbon stock on land and driving a large portion of carbon fluxes throughout the biosphere, they host a major portion of terrestrial biodiversity. Moreover, they provide another number of ecosystem services such as soil protection, temperature and flood regulation, slope stabilisation, nutrients cycling, water and air purification, and supply a wide variety of products (e.g. timber, fibers, food and medicinals). Forests have also been crucial to the livelihoods of local populations and held important spiritual values since time immemorial (Blicharska & Mikusiński 2014, Stara et al. 2016). Hence, their status and dynamics are critical for human wellbeing and a sustainable future (Ellison et al. 2017).

When observed in small timeframes, forest cover appears to be quite static, whereas major fluctuations, typically due to natural dynamics, can be observed at broader time scales (i.e. centuries or more). However, at global scale there currently are areas where the overall forest coverage shows evident directional trends even at finer time scales due to the effects of human activities, such as urbanization, agricultural expansion and fires (FAO & UNEP 2020). The reduction of forest cover in natural or semi-natural forests is particularly relevant as it can lead to local losses of biodiversity and erosion of ecosystem services. The global reduction of natural forests has been seen as a major challenge for the near future (Chiarucci & Piovesan 2020) and several projects for large scale reforestation have been activated worldwide (see e.g., Cao 2008; Vadell et al. 2016; Wang et al. 2019; Yao et al. 2019). However, also forest gain is not necessarily free of troubles. If it is the consequence of tree planting, for example, it can be unsuccessful, especially when involving species unadapted to local climate, or have negative impacts on regional water availability (Xiao & Xiao 2019). At the same time, natural afforestation often occurs as a consequence of mountain and rural abandonment, leading to the homogenization of biodiversity-rich cultural landscapes and agro-forestry matrices (Amici et al. 2015, Agnoletti et al. 2019). In such cases, afforestation can result into a reduction, rather than enhancement, of landscape diversity, local biodiversity and environmental resilience.

Monitoring forest loss and forest gain, as well as identifying their spatial drivers, is therefore of utmost importance for the sound management of biodiversity and ecosystem services. While the drivers of deforestation have been extensively investigated, and are typically identified with commodity production, forestry, shifiting agriculture, wildfires, urbanization and agriculture trades (DeFries et al. 2010, Curtis et al. 2018), few studies have explored the spatial drivers of forest gain. Clement et al. (2009) found that in Northern Vietnam afforestation was positively correlated with proximity or presence of wood industry and distance from major roads, whereas it was negatively correlated with house allocation. Upton et al. (2014) identified a number of physical, economic and

policy drivers for land conversion to forestry in Ireland, with proportions both of different soil types and private forests being the most relevant predictors. In the province of Siena, Geri et al. (2010) found that forest gain was associated to higher elevation and steeper slopes, as a direct consequence of processes of mountain abandonment.

Protected areas (PA) are the cornerstone of biological conservation and are usually expected to positively affect forest conservation and management. Andam et al. (2008), for example, found that 10% of forests within PA in Costarica would have been cut without protection between 1960 and 1997, whereas according to Yang et al. (2019) without the establishment of PA in China deforestation would have increased up to 50% in those areas. PA in Sumatra not only had lower deforestation rates than unprotected areas, but a similar pattern was observed even for adjacent areas, up to 10 *km* from the borders (Gaveau et al. 2009). However, it must be noted that PA are often located in remote or inaccessible areas where human impact inherently tends to be lower, and when this pattern is not accounted for in the models, it can produce biased results (Joppa & Pfaff, 2009; Pfaff et al. 2015). In this wake, Pfaff et al. (2015) also showed that PA in Amazon that are closer to roads and cities have an higher impact on forest conservation, while Clemente et al. (2020) reported similar results for the Brazilian Cerrado. Besides their spatial location, other elements can influence the effect of PA on deforestation. In Brazilian Amazon, for example, Herrera et al. (2019) found significant variations across States and between state and federal PA. According to Miteva et al. (2019), the effect of PA in reducing deforestation in the Jucatan Peninsula (Mexico) changed across tenure regimes and forest types. Cropper et al. (2001), in contrast, did not record any significant relationship among National Parks and deforestation, whereas weak evidence emerged for Wildlife Sanctuaries halting deforestation. Leberger et al. (2020), finally, showed that forest loss is globally accelerating in PA across all IUCN categories, with certain regions exhibiting even higher rates of forest loss within PA than outside.

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Besides PA, a number of other more or less formal area-based conservation measures have also been in place, sometimes much longer than PA. Of these, Sacred Natural Sites (SNS) are probably the best known. SNS are defined as "areas of land or water holding special spiritual significance for people and communities" (IUCN-UNESCO 2008) and are often regarded as the oldest form of habitat protection in human history. Moreover, they represent a shadow network of unofficial PA which conserves biodiversity and mature forest stands, provides ecosystem services and potentially reinforces the official ecological networks (Dudley et al. 2009; Frascaroli et al. 2019). SNS are often found in association with forest remnants (see e.g., Cardelús et al. 2013) and have been shown to locally halt or reduce deforestation (Campbell 2004, 2005). However, large scale studies on the effects of PA and SNS in controlling forest cover changes are currently lacking.

In the present study, we assess whether PA and SNS are important factors regulating forest loss and forest gain processes by means of descriptive analyses and modelling. We take Italy as an ideal case study, thanks to the availability of nationwide data pertaining to both forest changes in a timeframe of 82 years (1936-2018) and the spatial occurrence of SNS. While focusing on the role of SNS and PA in respect with forest changes, the present work also provides insights on other drivers of forest changes at national level.

2 METHODS

2.1 Study Area

Italy is an elongated peninsula located in the middle of the Mediterranean Basin. It stretches approximately 8.2°E - 18.5°E in longitude and 35.5°N - 47.1°N in latitude. Its area is about 300,000 km^2, one fifth of which is made up of islands, mainly Sicily and Sardinia. It has a rugged topography, characterised by two main mountain chains, the Alps, in the North, and the Apennines, extending through the peninsula. The largest plane areas are in the North, the Po flood plain and the Venetian-Friulian flood plains, while other minor plains are scattered across the peninsula and the

islands. Its maximum elevation is 4,808 *m* a.s.l. (Monte Bianco). The Italian population is unevenly distributed, being concentrated in the urban areas along the coasts and in the northern plains. This pattern is the result of migrations that started at the end of the XIX century and dramatically accelerated after the II World War, following the shift from an agriculture-based economy to an industrial one, resulting in the abandonment of large portions of the Alps and the Apennines.

2.2 Data

2.2.1 Forest Cover Changes

To model forest gain and forest loss at national scale, we produced a map of forest cover changes (Figure 3.1) following the same approach of Camaretta et al. (2018) for the period 1936-2018. For this purpose, we used the Italian Kingdom Forest Map 1936 (IKFM) and the Corine Land Cover 2018 (CLC). We retained all the forest polygons unregarding of forest type (see Camaretta et al. 2018 for a list of employed land covers). Layers were then converted to raster format at 500 *m* resolution and overlaid, in order to produce a map of forest cover changes. The value of each pixel was determined as the combination of two binary states, i.e. forest or no-forest, one for each period, resulting in 4 possible outcomes: *no forest* (no-forest in 1936 and no-forest in 2018), *forest loss* (forest in 1936 and no-forest in 2018), *forest gain* (no-forest in 1936 and forest in 2018) and *forest conservation* (forest in 1936 and forest in 2018). In addition, we added inland waters and wetlands as a fifth category to mask areas unsuitable for the subsequent analysis, that is areas where no forest changes could be observed as they inherently cannot host forests.

2.2.2 Protection Status

PA in Italy mostly consist of two networks. One is the network of national parks and local reserves that are part of IUCN's Official List of Protected Areas (OLPA), with the first park in the country having been established as far back as 1922. The other network is the European conservation

scheme Natura 2000, which was set up in the EU countries in the 1990s, following the

implementation of the Habitats Directive. These two networks overlap greatly, with Natura 2000

containing most of OLPA. In addition, as a legacy of two millennia of Catholicism and earlier pre-

Christian cultures, Italy is dotted by a number of SNS, characterised by a wide spectrum of

environmental conditions and cultural uses (Frascaroli et al. 2016). Recently, Frascaroli et al. (2019)

assembled a geo-referenced list of SNS across Italy and compared their spatial and landscape

features with those of Italian PA, highlighting the complementarity of the SNS and PA networks.

Figure 3.1. Map of forest cover changes in Italy for the period 1936-2018. "No Forest" are areas which were not covered by forests in the 1936 nor in the 2018; "Forest Conservation" are areas which were covered by forests in both periods; "Forest Loss" are areas which were covered by forests in the 1936 but were not in 2018; "Forest Gain" are areas which were not covered by forests in 1936 but were in 2018.

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2.2.2.1 Sacred Natural Sites

We used the dataset assembled by Frascaroli et al. (2019), consisting of 2,332 occurences of SNS throughout the whole italian territory. Although Frascaroli et al. (2019) noted that the distribution of SNS in the data set is biased towards certain geographical areas (i.e. Center and NW Italy), we assumed the data set to be sufficiently robust for our analysis in virtue of its large size. Following the approach by Frascaroli et al. (2019), we considered a buffer of 1,250 metres radius around each SNS as its area of influence. This is an arbitrary size that in many cases does not properly reflect the real area of influence, as it can be smaller or bigger. At the same time, it is a pragmatic approach that enabled us to model processes on a large extent, without the need to retrieve specific - and hardly obtainable - information for each site.

2.2.2.2 Protected Areas

We considered as PA all the areas belonging to Natura 2000 network and/or OLPA. We aggregated the two datasets in a single layer and considered all the PA as a unique category of protection status, alternative to SNS. While PA network is actually made up of different protection categories, we did not take them into account as we wanted to focus on the differences between unofficially and officially protected areas, that is SNS and PA, rather than discriminating different protection regimes within PA.

2.2.2.3 Other variables

In addition to Protection Status, we extracted a number of potentially relevant variables to model forest gain and forest loss. A first set of variables consisted in geographical and topographical variables, i.e. *easting* and *northing, elevation*, s*lope* and *terrain ruggedness index* (*TRI*) (Riley et al. 1997). A second set of variables was composed by anthropogenic variables, i.e. *distance to nearest*

road and *population density*. In order to compare model coefficients, we standardised explanatory variables as Z-Scores by subtracting from each observation the variable mean and dividing by the variable standard deviation.

2.3 Data Analysis

2.3.1 Descriptive statistics

To first explore the frequencies of different forest cover changes and lacks of change across Italy, as well as within SNS and PA, we calculated a contingency table with protection status versus the four categories of the forest cover change map.

2.3.2 Modelling

Then, in order to investigate forest gain and forest loss, we modelled the two processes separately, by means of binomial GLMs. First, we removed current inland waters as we presumed they were mostly already present in 1936. We also removed pixels at elevations above 1,850 *m* a.s.l. as we assumed it as a reasonable treeline in the region. We converted the explanatory variables to raster format, aligning the resulting layers with the forest cover changes map. To reduce multi-collinearity, we checked correlation among all pairs of variables and we found that *elevation*, *slope* and *TRI* were all highly correlated (pearson's $\rho > 0.7$). We thus retained only the first one, assuming that it adds specific information on the geographical setting, compared to the other two. Then, we split our dataset into two subsets: areas that were not covered by forest in 1936 and areas that were covered. In this way observations for each dataset could have only two outcomes: no change and change. Notably, observations from the first dataset (unforested areas in 1936) could only be *no forest* (i.e. no change) and *forest gain* (i.e. change). Conversely, observations from the second dataset (forested areas in 1936) could only be *forest conservation* (i.e. no change) and *forest loss* (i.e. change). Then, we run some preliminary binomial GLMs to explore the validity of our approach and binned plots

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showed that binary response variables were unsuited for our models. Hence, we transformed the response variables by applying a moving window of 2.5 *km* and calculating the percentage of *forest gain* and *forest loss* within the moving window. Then, we modelled these derived response variables as a function of the selected explanatory variables. In order to reduce spatial auto-correlation (SAC) of the residuals, we added *easting* and *northing* quadratic and cubic terms to the models. In addition, as response variables showed an unimodal trend in respect to *elevation*, we also added its quadratic term to the models. Therefore, we checked for SAC in the residuals of the models by calculating Moran's I. As we still found highly significant auto-correlation of the residuals, we tried to remove it by adding a residuals auto-covariate term (RAC) to the models by calculating two different auto-covariates on the residuals of the two models, respectively. Finally, we calculated D^2 of the final models and the models without RAC and we checked again for spatial-autocorrelation in the residuals of the final models by calculating Moran's I. All analyses were performed with R 3.6.3 (R Core Team 2020). Data import and preparation were performed with *sf* (Pebesma 2018), *raster* (Hijmans 2020), *magrittr* (Milton Bache & Wickham 2014), *dplyr* (Wickham et al. 2020) and *rgrass7* (Bivand 2019) packages. D^2 and Moran's I were calculated by means of *modEvA* (Barbosa et al. 2015) and *spdep* (Bivand & Wong 2018) packages, respectively. We used *ggplot2* (Wickham 2016), *ggspatial* (Dunnington 2020) and *RStoolbox* (Leutner et al. 2019) packages to produce graphical outputs.

3 RESULTS

3.1 Descriptive statistics

Italy experienced forest gain in about 14.6% of its territory from 1936 to 2018, with even higher values in PA and SNS (21.7% and 18.8%, respectively). Forest loss occurred in about 4.6% of the whole territory and moderately greater values in PA and SNS, being 6% and 5.2%, respectively (Table 3.1).

Table 3.1. Summary of areal extension of forest cover changes (forest gain and forest loss) and lacks of changes (no forest and forest conservation) for the period 1936-2018 across the whole italian territory, its Protected Areas and its Sacred Natural Sites. Values are the percentages of each change or lack of change within the total area of its category, the total area is given in the headers. Inland waters were removed before all calculations.

	Italy *(298 083.25 km²)*	**PA** *(62 739.5 km²)*	**SNS** *(10 413 km²)*
No forest	65.37 %	45.15 %	56.83 %
Forest loss	4.61 %	5.98 %	5.24 %
Forest gain	14.54 %	21.75 %	18.77 %
Forest conservation	15.48 %	27.12 %	19.16 %

3.2 Modelling

3.2.1 Forest Gain

The model explained most of the observed variability ($D^2 = 0.89$, Table 3.2). However, this was largely due to the addition of the RAC, which explained ca. 50% of the entire variability (D^2 of the model without RAC = 0.38). Despite having added *easting* and *northing* (and their quadratic and cubic terms) along with the RAC, the residuals of the model were still spatially-autocorrelated (Moran's I = 6.63, *p*-value < 2e-16), most likely due to the large number of zeros in the dataset.

Table 3.2. Summary statistics of residuals auto-covariate binomial-GLMs for forest gain and forest loss in Italy for the period 1936-2018. Number of cells are the number of observations employed in each model and they consist of 500 *m* x 500 *m* cells. D^2 represents the fraction of observed deviance explained by each model, in parenthesis D^2 of models without the residuals auto-covariates. Pearson's *r* measure the correlation between the actual response variables and the fitted response variables. Moran's I and related *p*-values measure the degree of spatial auto-correlation within the final model residuals (*p* < .05 means that spatial auto-correlation was still present).

Model	**Number of cells**	**D^2**	**Pearson's *r***	**Moran's I**	**Moran *p*-value**
Forest gain	919856	0.89 (0.38)	0.93	6.63	< 2.2e-16
Forest loss	236345	0.78 (0.22)	0.91	8.91	< 2.2e-16
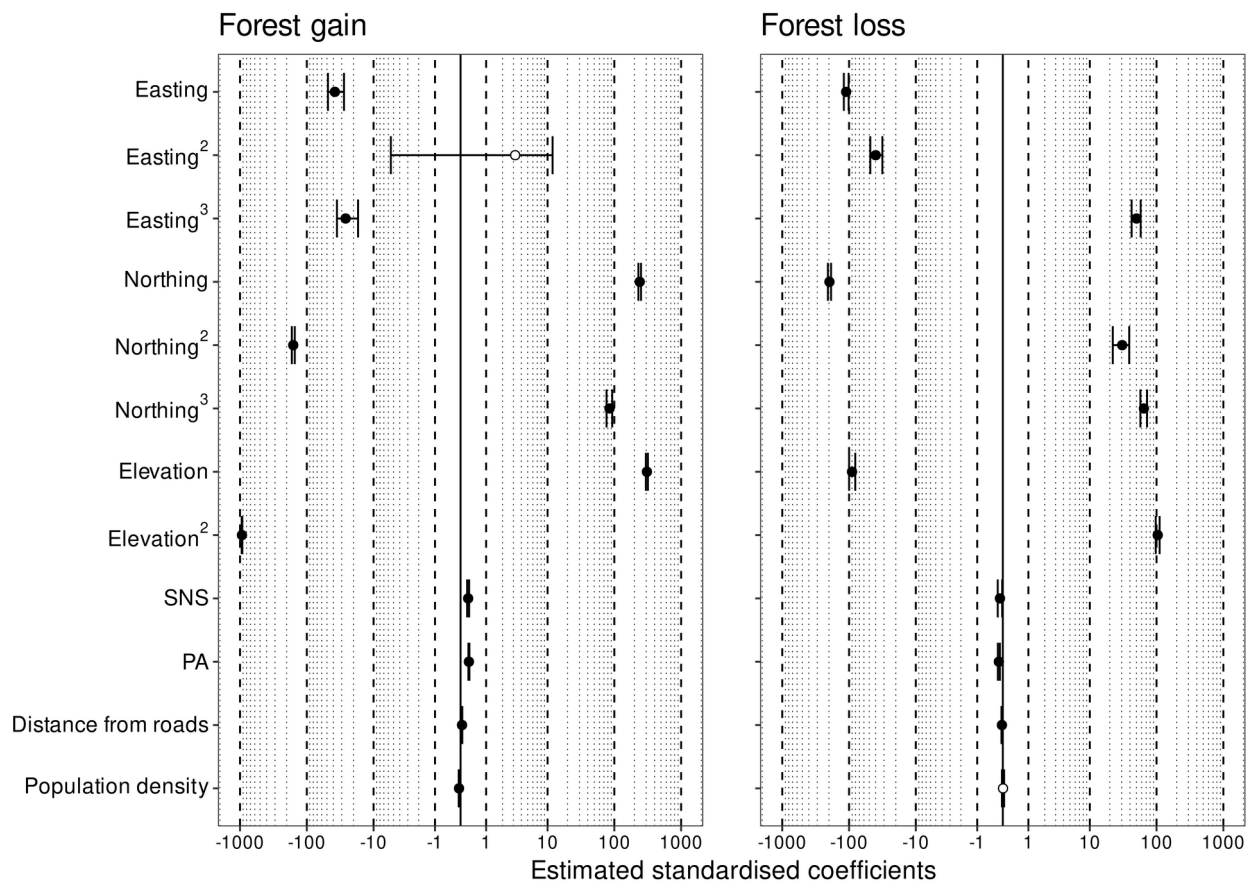
Figure 3.2. Standardised coefficients estimates for residual auto-covariate binomial GLMs modelling forest gain and forest loss in Italy for the period 1936-2018. SNS means Sacred Natural Sites and PA means protected areas. Solid dots represent statistically significant coefficients ($p < .05$) and empty dots represent statistically non-significant coefficients, whiskers are confidence intervals. X-axis is log10 transformed. Residuals auto-covariates estimates are not shown.

All coefficients, except for the quadratic term of *easting*, were significant (Figure 3.2). Moreover, the size of the coefficients varied greatly, with *easting*, *northing* and *elevation* terms being much greater than the others. In particular, forest gain was strongly associated with Northern and Western locations and with mid-altitude elevations. *Protection status* showed a positive effect, with PA having a slightly greater coefficient than SNS. *Distance to nearest road* had a positive effect, while *population density* had a negative effect (Figure 3.3).
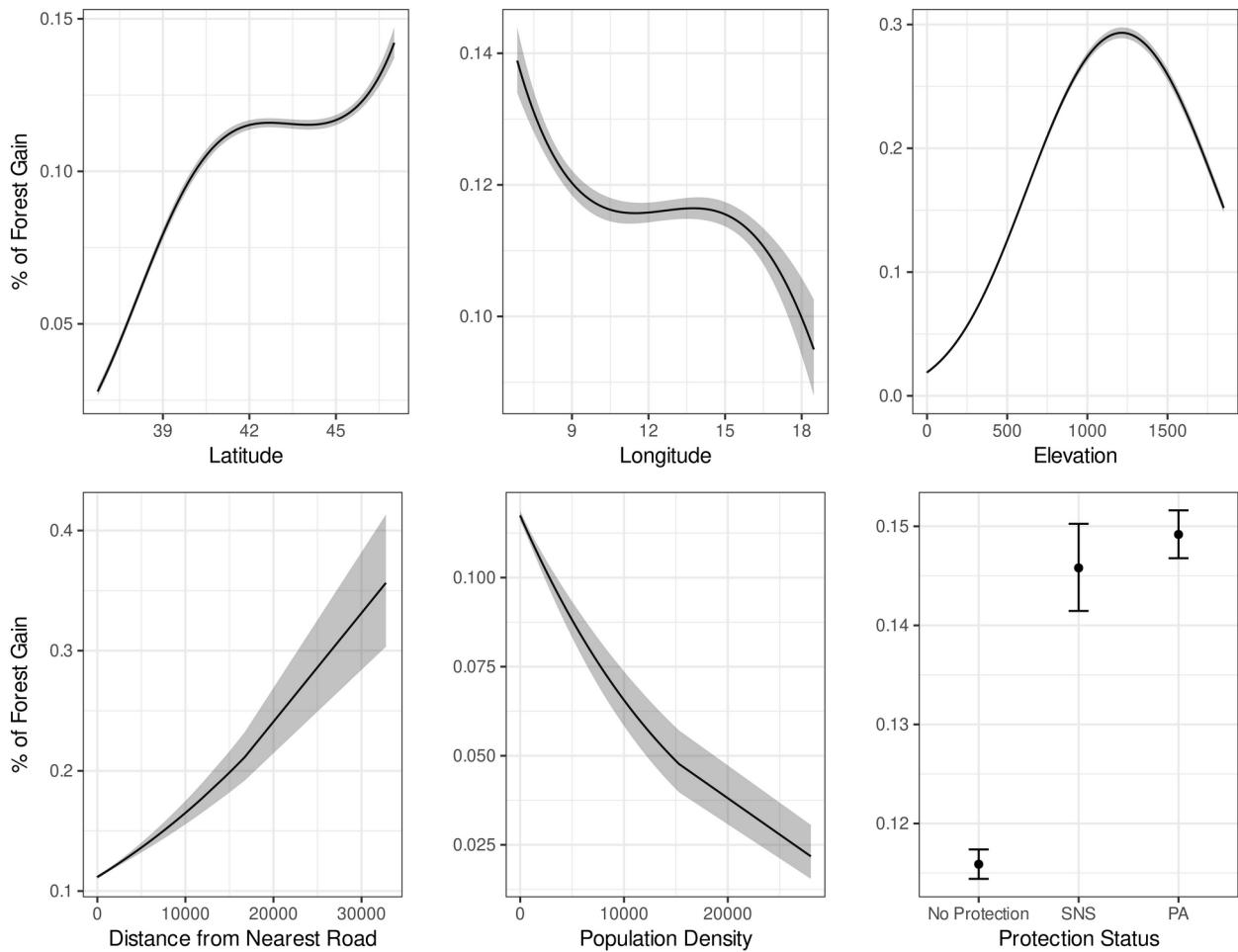
Figure 3.3. Marginal effect plots of forest gain residuals auto-covariate binomial GLM. SNS means Sacred Natural Sites and PA means protected areas. Residuals auto-covariate estimate is not shown. Continuous covariates were back-transformed from normalization before plotting.

3.2.2 Forest Loss

The model explained a large portion of observed variability ($D^2 = 0.78$, Table 3.2) but, similarly to the other model, this was mostly due to the addition of the RAC (D^2 of the model without RAC = 0.22). Still, we were not able to fully remove spatial-autocorrelation from the residuals of the model (Moran's I = 8.91, *p*-value < 2e-16). *Population density* was the only non-significant predictor (Figure 3.2). Again, *easting, northing* and *elevation* terms showed the largest absolute values, but with different trends (Figure 3.4). Forest loss was more strongly associated with Southern and Western locations and low- and high-altitude elevations. *Distance to nearest road* and *protection status* showed a negative effect on forest loss, with PA having a slightly larger effect than SNS.

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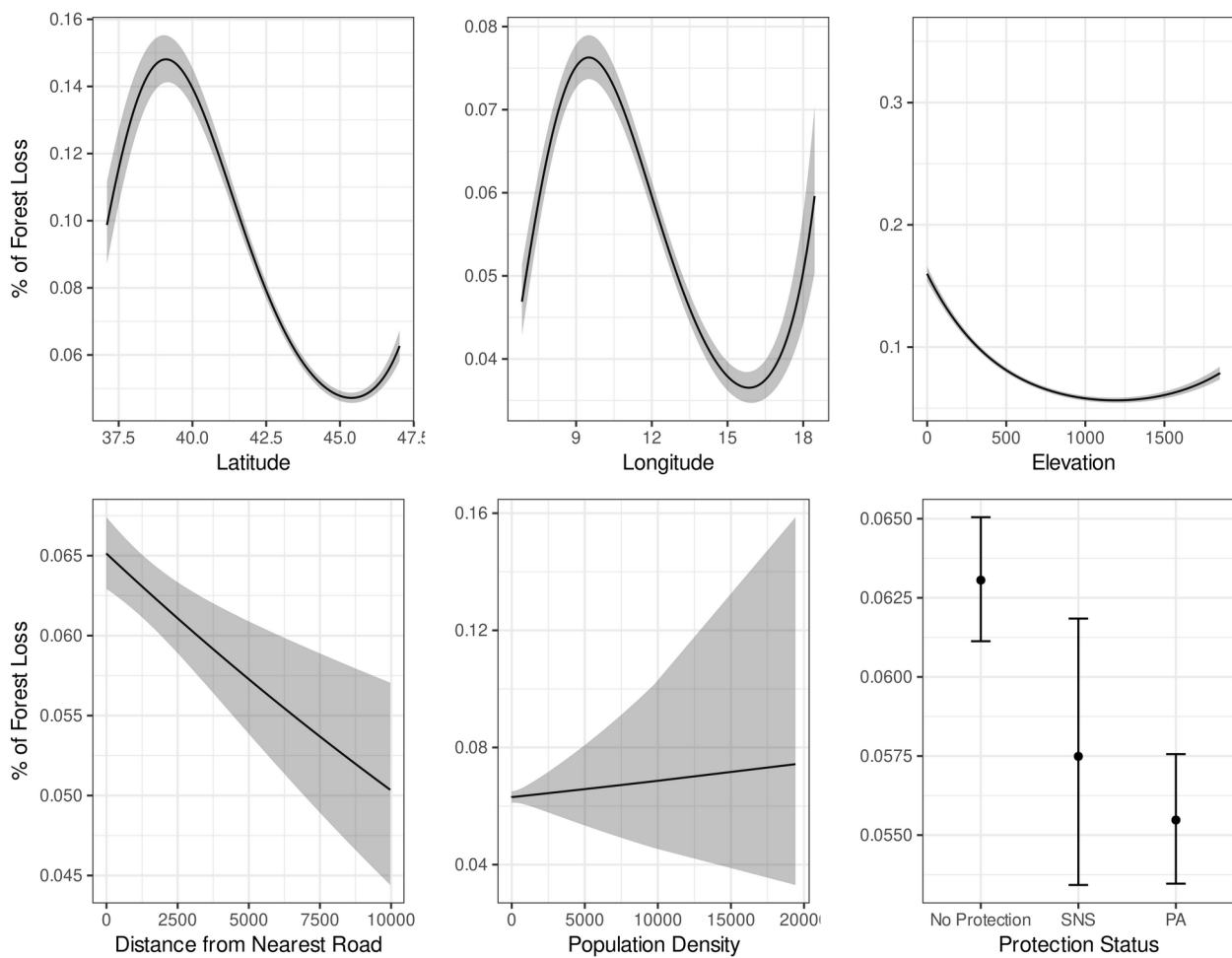
Figure 3.4. Marginal effect plots of forest loss residuals auto-covariate binomial GLM. SNS means Sacred Natural Sites and PA means protected areas. Residuals auto-covariate estimate is not shown. Continuous covariates were back-transformed from normalization before plotting.

4 DISCUSSION

In this work we produced one of the first attempt to model forest gain and forest loss at national scale and in relation to the protection status of the land surface. Overall, in the period considered, the main trend has been towards a net increase of forest cover throughout Italy (Antrop 2004, Mazzoleni et al. 2004, Falcucci et al. 2007, Camaretta et al. 2018, Garbarino et al. 2020). Rural abandonment since the 1960s and the subsequent afforestation of lands formerly dedicated to grazing and small-scale agriculture are commonly indicated as the main drivers of that tendency. This is quite clearly reflected in the land use dynamics of SNS, which in Italy are largely associated

with traditional rural landscapes and had a relatively higher increase of forest cover than Italy as a whole (37.9% vs. 30.0%) in the period considered. At the same time, both SNS and especially PA had relatively higher rates of forest loss.

However, our model of forest loss contradicts this last insight, demonstrating that SNS and PA in Italy significantly reduced forest loss from 1936 to 2018, after accounting for other geographical and anthropogenic factors. While this effect was already known for PA (see e.g., Andam et al. 2008; Gaveau et al. 2009; Pfaff et al. 2015), this represents a novel result for what concerns SNS, giving for the first time quantitative backing to a known phenomenon (see e.g., Cardelús et al. 2013). Despite being significant, however, the effect of SNS and PA in reducing deforestation is small compared to the geographical variables and the RAC. One of the reasons could be that a large portion of the effect of SNS and PA is inherent to their geographical location, and especially elevation. Notably, SNS in Italy are mostly found at mid elevations and PA at mid-high elevations (Frascaroli et al. 2019), which in turn are the elevation ranges where forest loss is at minimum. In certain cases, SNS and PA may also have failed to halt deforestation in the period considered. The loss of governance due to the abandonment of hilly and mountain areas may have lead to a marginalization of SNS and their role in land protection. As for the weak effect of PA, it could be due to the temporal scale of this study, as the vast majority of PA in Italy were established after 1936 and their effects on forest conservation may be lagging.

Our models also support the hypothesis that some form of land protection promotes forest gain. This is well in line with the descriptive results, which showed that SNS and PA in Italy had higher rates of forest gain than the whole territory. However, similarly to what we observed for forest loss, the effect of protection status was significant but relatively small. This again could partly be a consequence of elevation explaining portions of variability which could have been explained by protection status, as SNS and PA are mostly found at the elevations where forest gain is at its maximum, i.e. mid-high elevations. It must be noted that the positive effect of protection status on

forest gain is not necessary a desirable phenomenon, especially in such contexts where cultural landscapes have great relevance for biodiversity conservation. Indeed, the loss of open habitats such as traditional pastures or semi-natural grasslands can lead to local extinctions and weakining of ecosystem services such as pollination or pest control. Moreover, particular attention to forest gain should be posed in areas where open habitats are rare or particularly relevant for biodiversity such as mountain areas or islands.

Besides protection status, the models successfully explained most of the observed variability after the addition of the RAC. The great difference in explained variability between the RAC and the other covariates is most likely due to the large number of zeros in the response variables, which, in turn, reflects the intrinsically clustered arrangement of forests and anthropogenic land uses. This clustered arrangement probably increased the importance of geographical variables as well (i.e. *easting*, *northing* and *elevation*), which were the most important predictors after the RAC. The importance of geographical variables indicates that geographical location has been determinant in processes of afforestation and deforestation. This is interesting in that geographical location can be read as a proxy of historical and cultural factors.

Contrary to our expectations, the effects of the anthropogenic variables, *protection status*, *distance from nearest road* and *population density*, was rather weak and explained only a marginal portion of variability. Nevertheless, the effect was almost always significant and its direction concordant with the direction that we had hypothesised. In particular, *protection status* was positively associated with an increase in forest gain and a decrease in forest loss, both SNS and PA with similar magnitude of effects. However, the reason of this pattern is not clear. On the one hand, indeed, there is the possibility that active protection of these areas prevented cutting forested areas. On the other hand, the increased forest gain seems to hint the fact that land abandonment is the prevailing process. If this is the case, it would be worrisome as it could lead to landscape homogenization by reducing the typical patchiness of cultural landscapes. In addition, forest gain is likely to be, at least

partly, at the expense of grassland and pasture habitats, which are often relevant for biodiversity conservation in this geographical context.

It has to be noted that more sophisticated models, e.g., random forest or generalized boosted regression, are expected to perform far better than our models. These aforementioned models are machine-learning algorithms allowing classification and regression tasks, being non-linear, flexible, robust in respect with multicollinearity and able to model complex interactions makes them very useful and highly performing through a wide spectrum of ecological and spatial applications (Cutler et al. 2007, Elith et al. 2008). However, they somehow suffer difficulties of interpretation due to their complexity and, therefore, are not easily employable to make inference. On the other hand, despite being maybe a little simplistic, our models have the non-trivial merit of being easily interpretable and require limited computation time and resources.

5 POLICY IMPLICATIONS

There area three main areas of possible policy interventions that emerge from our findings. The first one concerns the official acknowledgement of the conservation role of SNS. This role has been already documented in a number of studies (see Dudley et al. 2010 for a review), especially with regards to the biological composition of local areas at a certain moment in time. This study adds another layer, demonstrating the importance of SNS also in maintaining forest cover at a national scale and over a broad timeframe. As evidence grows of the contribution of SNS to environmental conservation at different spatial and temporal scales, it also becomes increasingly important that this contribution is adequately recognized and ratified. Inclusion in PA can be an easy way to grant such ratification, although it might not be the ideal avenue in some cases. Fortunately, new policy instruments are currently gaining traction, such as the definition of 'Other Effective Area-based Conservation Measures' (OECMs), which could be more suited for SNS (IUCN WCPA 2019). Moving towards a more formal recognition of the conservation role of SNS at regional or national

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scales, whether by adopting existing frameworks or elaborating new specific ones, should in any case be seen as a priority for the coming years (Frascaroli et al. 2019).

The second indication we draw from our results regards the need for continued active management at SNS. Indeed, we identified a clear trend towards the afforestation of SNS. As in case of other cultural landscapes, this can be undesirable from a conservation perspective, as it can lead to the loss of rare habitat patches and complex landscape matrices of high conservation value. To avoid such outcomes, management continuity should be guaranteed at SNS. Notably, SNS can be associated with a variety of livelihood practices (in Italy, for example, pastoralism and agro-forestry), that have contributed to shaping and maintaining their biological makeup. Many of these practices are rapidly disappearing due to socioeconomic changes. Appropriate policies should incentivise them whenever possible as means to maintain SNS' biological specificity. Similar efforts should ideally rest on interdisciplinary research aiming to document and understand past and current management practices at SNS. Similarly, it is important that the need for continued management will be taken into consideration, when SNS are given legal recognition by inclusion in an official conservation scheme.

Finally, our study offers broader insights that go beyond SNS alone. Indeed we found that the effect of SNS and even PA on hampering deforestation has ultimately been marginal, compared to that of mere geographical and topographical factors. In other words, isolation and difficult access have been more important forms of protection than human norms and regulations. This means that future efforts to broaden and integrate conservation networks should primarily focus on more easily accessible areas, as these historically result to be more at risk. Also, this would help close a well-known gap in the spatial cover of PA (Joppa & Pfaff 2009), which is highly skewed towards isolated high-elevation areas at the expense of lowland and densely populated ones.

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CHAPTER IV

Sacred Natural Sites in Italy

conserve plant diversity

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ABSTRACT

Biodiversity is currently being eroded at an unprecedented rate, as Earth's biota has entered its sixth mass extinction. Protection of large portions of Earth has been called for to halt the current biodiversity crisis. However, establishment of large Protected Areas (PA) is typically difficult and in certain cases is not a viable option. Small PA and other area-based conservation measures can complement large reserves, while ensuring the connectivity of the networks of PA. Sacred Natural Sites (SNS) have demonstrated their potential for conserving biodiversity but studies on their role across Europe are few. We sampled plant assemblages at 30 SNS across Italy and related control sites (Reference Sites, hereafter RS) and compared them in terms of species richness, rarity, locally-exclusive species, beta and gamma diversity. The dataset comprised 243 plots grouped into 4 macrohabitats, 6510 occurrences and 1094 plant species. While species richness of SNS and RS were similar, we found that SNS had significantly rarer species assemblages, more locally-exclusive species and larger beta and gamma diversity. Moreover, woodlands and annual crops found at SNS showed greater conservation value in respect with RS. The observed patterns are likely due to traditional management practices, social taboos and unique natural features of SNS, calling for a legal recognition of these sites. The different trends we found across macrohabitats points at the need to apply habitat-based conservation measures.

1 INTRODUCTION

Biodiversity loss is currently one of the major threat humanity is called to face (Díaz et al. 2006, Cardinale et al. 2012, Ceballos et al. 2017), as Earth biota has entered its sixth mass extinction (Barnosky et al. 2011, Ceballos et al. 2015). While the magnitude of current extinction rates is still open to debate and different clades show different trends (McCallum 2007, Alroy 2015, Knapp et al. 2020), varying across different biomes (Woodruff 2001), there is large consensus about the need to conserve vast areas of Earth to halt the current biodiversity crisis, while restoring biota and ecosystems (see e.g., Locke 2013, Wilson 2016, Baillie & Zhang 2018, Dinerstein et al. 2019, Dinerstein et al. 2020).

Establishing large Protected Areas (PA) or nature reserves is usually difficult as it requires negotiations with local communities and stakeholders. Moreover, when such areas are planned to be strict nature reserves they have to be located in inhabited land or people has to be moved,

potentially creating conflicts. Therefore, large PA in certain cases are not a viable possibility. On the other hand, small PA have demonstrated to host important portions of regional biodiversity, especially when considering small to medium size organisms, and provide benefit to the functioning of large PA by enhancing connectivity (Volenec & Dobson 2020). In such perspective, other effective area-based conservation measures (OECMs), that are areas preserving biodiversity despite not being primarily devoted to it, have recently seized the attention of conservationists by matching the interests of local communities with conservation opportunities (IUCN WCPA 2019).

Sacred Natural Sites (SNS) have been often regarded as a type of OECMs or at least many of them hold the potential to be recognised as such (IUCN WCPA 2019). Indeed, these sites have proved to conserve biodiversity while being important to people and local communities because of their spiritual value (Bhagwat & Rutte 2006, Dudley et al. 2006, IUCN-UNESCO 2008, Dudley et al. 2010) and as such autonomously managed and preserved by people. It has been argued that SNS conserve biodiversity as a consequence of management practices, social taboos and the unique natural features occurring at their location (see e.g., Anderson et al. 2005, Frascaroli et al. 2016, Fungomeli et al. 2020). Moreover, their number, despite not being known, is estimated to be quite large and they have been found almost in every inhabited place. However, research up-to-date has largely focused on SNS in Asia and Africa (Dudley et al. 2010), with very few papers investigating the role of SNS for biodiversity conservation in Europe (Frascaroli 2013, Frascaroli et al. 2014, Frascaroli et al. 2016, Shepheard-Walwyn & Bhagwat 2018, Avtzis et al. 2018, Frascaroli et al. 2019, Nascimbene et al. 2019, Marini-Govigli et al. 2020). Moreover, only Frascaroli et al. (2016) investigated diversity patterns of biological communities at SNS in Italy, finding that SNS had higher species richness than control sites and conserved locally important species pools, while contributing to landscape diversity. However, the research was restricted to Central Italy and no other evidence based on field data collection has emerged for SNS across Italy. Moreover, we were not able to find any research in the literature whose area of study encompassed an entire country.

Therefore, in the present work we took advantage of a nation-wide survey of SNS (Frascaroli et al. 2019) to select a sample of sites scattered throughout Italy. After extracting a random set of comparable areas for each site, we carried out one of the largest data collection on the topic of SNS, resulting in 243 plots sampled at 30 SNS. We used a set metrics to compare different facets of biodiversity between SNS and control sites, by asking: do SNS in Italy contribute disproportionately to biodiversity conservation as compared to similar neighbouring areas? Are there any differences across different macrohabitat types?

2 METHODS

2.1 Data collection and preparation

Starting from the survey made by Frascaroli et al. (2019), we selected 30 sites scattered across Italy (Figure 4.1). For each site we considered a circular area of 250 *m* radius around the centroid of the site as pertaining to the site. Then, in order to compare SNS with control areas, for each SNS we also extracted 3 circular areas with a 250 *m* radius, that we called Reference Sites (RS). RS were randomly placed at increasing distances from the SNS, in order to have the centre of the SNS and those of the 3 RS at 666 *m*, 833 *m* and 1000 *m*. The random placement of the 3 RS was made in order to have the first RS placed at random direction and the angles made by each pair of RS with the SNS at the vertex being 120° wide. The 3 distances (666 *m*, 833 *m* and 1000 *m*) were randomly assigned to the 3 RS. Then, we mapped habitat patches within SNS and RS by using aerial photos. Within each circular area, which we refer to as "buffers", we sampled each unique habitat whose patch was at least 0.5 ha. When more than one patch of the same habitat was found in the same buffer, we sampled the largest one. We sampled 10 *m* x 10 *m* quadrats at the patch centroid or nearest accessible position. Within each plot we collected the list of vascular plant species. At five sites we were able to sample only 2 buffers of RS, because of lack of accessibility or for being located in transformed sites. Sampling was carried out during Spring-Summer of 2018 and 2019.

We reclassified each plot into four habitat macrocategories: woodlands and other natural or semi-natural closed habitats (W), grasslands and other natural or semi-natural open habitats (G), perennial crops (P), annual crops (A). Plots were classified as crops when the recent cultivation practices were evident, depending on crop type we classified them as A or P. As for W and G plots, given that the difference between closed and open habitats is sometimes fuzzy, we choose to define plots with > 50% coverage of shrubs or trees as W, otherwise we classified them as G. Finally, nomenclature was standardized according to Pignatti et al. (2017-2019) and occurrences were identified at species level.

Figure 4.1. Map of 30 Sacred Natural Sites (SNS) sampled in this work. Red dots represent SNS sampled in 2018, blue dots represent SNS sampled in 2019.

2.2 Data analysis

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All analysis were carried out and graphical outputs produced using R v. 3.6.3 (R Core Team 2020) with the following packages: *ggnewscale* (Campitelli 2020), *ggthemes* (Arnold 2019), *patchwork* (Pedersen 2020), *Rarefy* (Thouverai et al. 2020), *Rarity* (Leroy 2016), *raster* (Hijmans 2020), *RStoolbox* (Leutner et al. 2019), *sf* (Pebesma 2018), *tidyverse* (Wickham et al. 2019), *vegan* (Oksanen et al. 2019).

2.2.1 Descriptive analysis

We calculated the total number of occurrences, plots and species in the dataset. We also calculated the percentage of identified occurrences, the total number of plots and species pertaining to SNS and RS, respectively, as well as the number of exclusive species in each group.

2.2.2 Species richness

We compared species richness per plot between SNS and RS, we also compared it by splitting it in the 4 macrohabitat categories. As species richness was not normally distributed (Shapiro-Wilk: $p < .05$), we employed two-sample Wilcoxon Rank Sum Tests (also known as Mann-Whitney U test) to test if mean species richness of SNS and RS was different.

2.2.3 Rarity

Rarity sensu Leroy et al. (2012, 2013) was calculated at plot-level and compared between SNS and RS, as well as by splitting it by macrohabitat, similarly as to what we have done for species richness. Given thtat rarity was not normally distributed (Shapiro-Wilk: $p < .05$), we employed two-sample Wilcoxon Rank Sum Tests to test for significant differences.

2.2.4 Exclusive species

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Then, we compared the number of locally exclusive species at buffer level between SNS and RS, that is the number of species exclusive to a single buffer for a given site. However, due to having different numbers of plots in each buffer as a consequence of sampling design, we tried to correct it. In particular, for each buffer we calculated the number of locally exclusive species divided by its species richness. This resulted in a ratio, where 0 means no locally exclusive species in the buffer and 1 means only exclusive species. Then, as values were not normally distributed (Shapiro-Wilk: p < .05), we employed a two-sample Wilcoxon Rank Sum Test.

2.2.5 Rarefaction

As we wanted to compare beta and gamma diversity between SNS and RS, while having quite different sample sizes (74 plots vs 169 plots, respectively), we applied rarefaction techniques. In particular, we applied spatially-explicit rarefaction in order to reduce potential bias due to spatial arrangement (Chiarucci et al. 2009, Ricotta et al. 2019). However, due to sampling design, potential bias arisen both using spatially-explicit or non spatially-explicit rarefaction, as the 3 RS of each site taken together had on average more plots than the single SNS. This results in a steeper curve for SNS than that of RS, as it tends to move earlier from one site to other, exploring larger geographical gradients. In order to overcome this issue, we randomly extracted up to 2 SNS plots and 2 RS plots for each site and built the spatially-explicit rarefaction curves. Then, we repeated this procedure 100 times and we calculated average rarefaction curves and confidence intervals from these curves. Due to 4 sites having only 1 plot attributed to the SNS, whereas in all cases at least 2 plots were sampled in the RS, this resulted in rarefaction curves halting at plot 56 for SNS and at plot 60 for RS. Despite the limited number of plots available when splitting by macrohabitat, we calculated spatially-explicit rarefaction curves also for the 4 macrohabitats.

2.2.5.1 Beta diversity

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Rarefaction of beta diversity was implemented through calculating three different indices of beta diversity, capturing different dimensions of the same phenomenon (Schroeder & Jenkins 2018), Whittaker's species turnover (Whittaker 1960), Jaccard's dissimilarity index (Jaccard 1901) and Cody's index (Cody 1975). The first is a measure of the number of full species composition changes through the observations in the dataset, the second is a ratio of the dissimilarity of the whole dataset (where 1 equals maximum dissimilarity and 0 equals maximum similarity) and the third is a measure of species turnover typically employed along ecological gradients.

2.2.5.2 Gamma diversity

Rarefaction of gamma diversity was implemented by using species richness.

3 RESULTS

3.1 Descriptive analysis

We sampled 30 sites across Italy, being located in 19 out of 20 administrative regions. We collected 6510 occurrences of vascular plants, 95.87% occurrences were identified at species level, belonging to 1094 species. We sampled 243 plots, 74 belonged to SNS and 169 to SNS. On average, for each SNS we sampled 2.47 plots and 5.63 plots not belonging to the SNS. Given that we could not sample 5 RS buffers, we sampled 1.99 plots on average per RS buffer. We found 764 species in SNS, 165 of which were exclusive to them. In RS we found 929 species, 330 being exclusive to them.

3.2 Species richness

Despite not being significant, SNS showed moderately larger species richness per plot than RS (29.84 ± 15.84 vs. 25.46 ± 13.62, $p = .104$; Figure 4.2a). Moreover, SNS had marginally significant greater species richness than RS in case of W plots (27.5 ± 14.74 vs. 22.93 ± 10.36, $p = .098$; Figure

4.2b) and significantly greater species richness in case of A plots (22.23 ± 14.03 vs. 13.68 ± 11.45, *p* = .04).
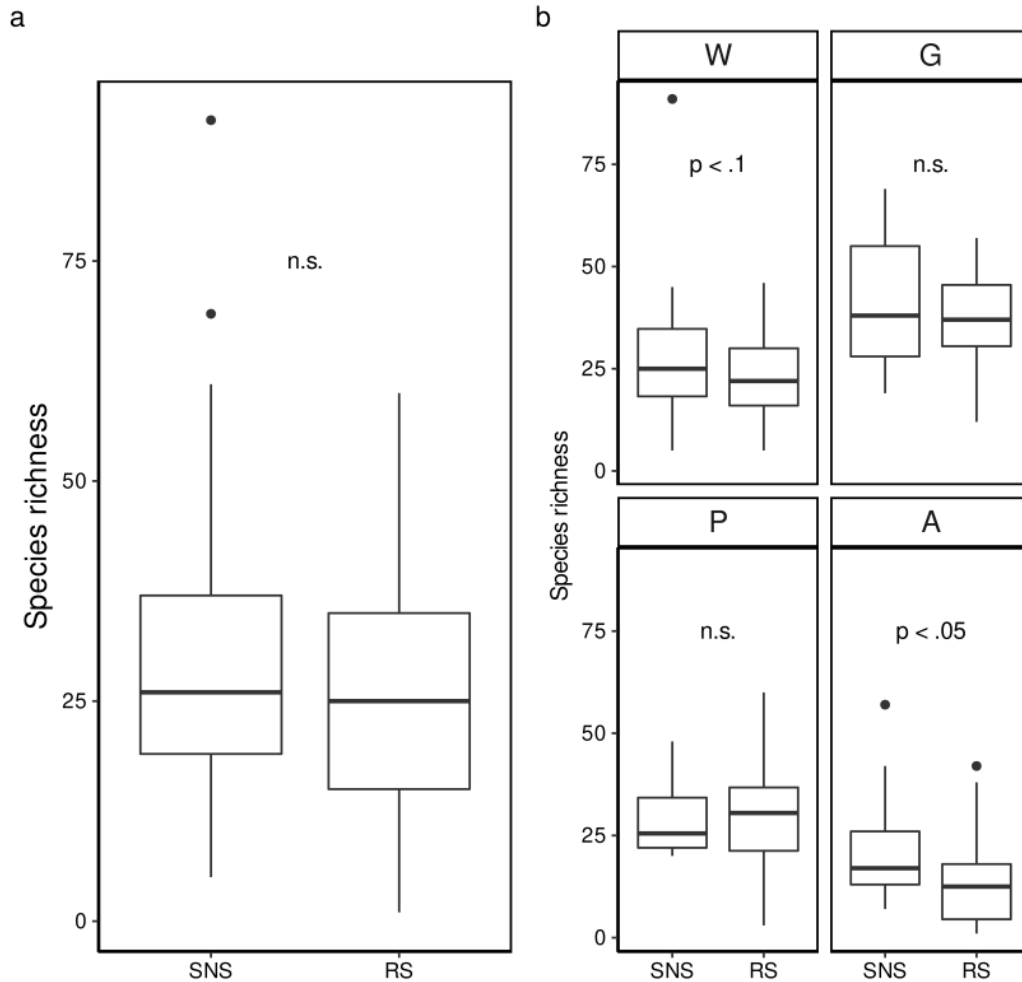
Figure 4.2. Boxplots comparing species richness between Sacred Natural Sites (SNS) and Reference Sites (RS). Text on the upper portion of the panels represent the *p*-values of two-sample Wilcoxon Rank Sum Tests (n.s. = not significant). In Figure 4.2a, all SNS plots are compared with all RS plots. In Figure 4.2b, panels are split according to macrohabitat classification (W = natural or semi-natural closed habitats, G = natural or semi-natural open habitats, P = perennial crops, A = annual crops).

3.3 Rarity

Rarity was significantly greater in SNS than RS (0.073 ± 0.088 vs. 0.054 ± 0.084, *p* = .027; Figure 4.3a) as well for W plots (0.081 ± 0.097 vs. 0.048 ± 0.07, *p* = .016; Figure 4.3b).
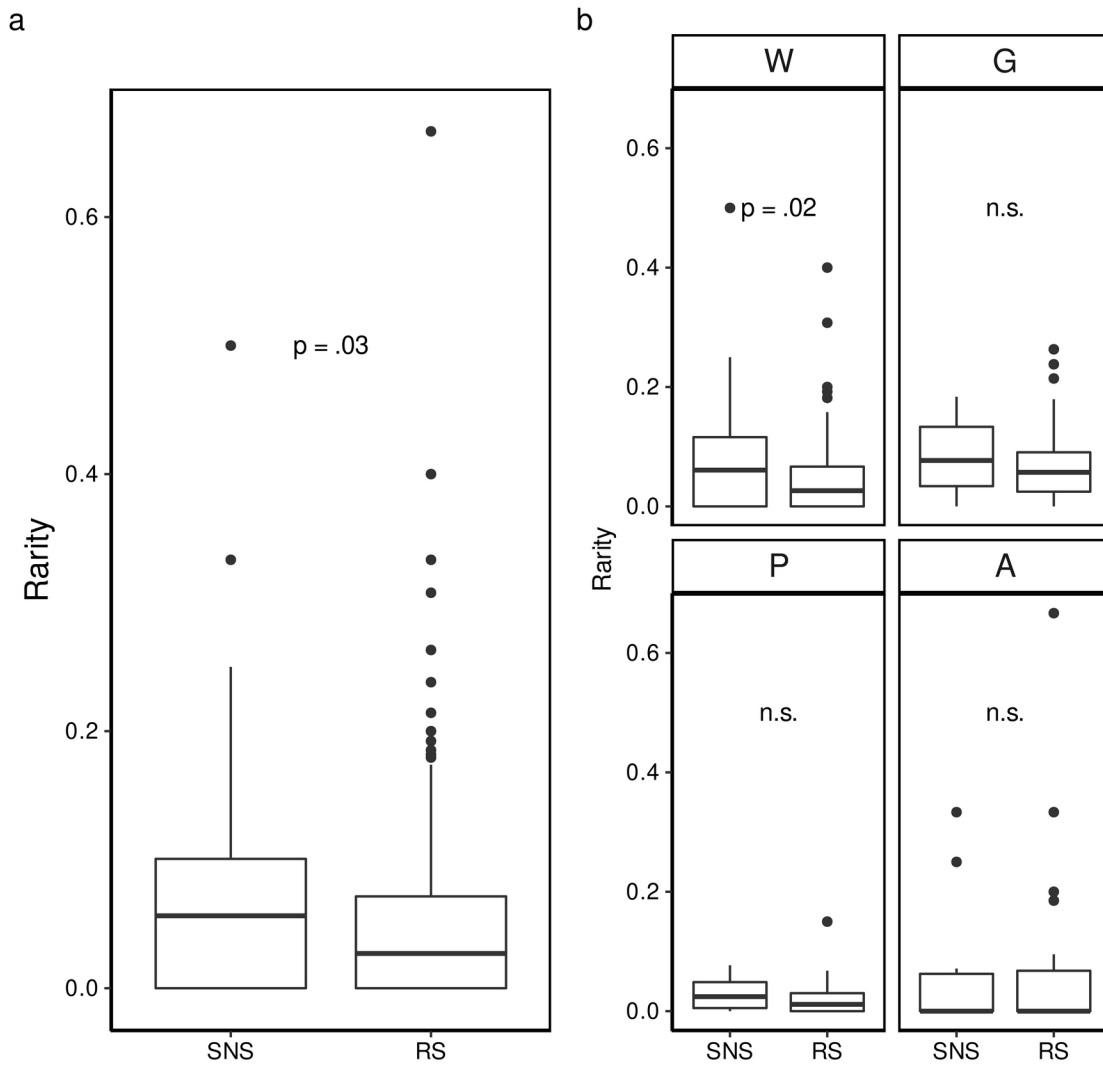
Figure 4.3. Boxplots comparing Regional Rarity between Sacred Natural Sites (SNS) and Reference Sites (RS). Regional Rarity values are rarity values calcultated according to Leroy et al. (2012, 2013). Text on the upper portion of the panels represent the p-values of two-sample Wilcoxon Rank Sum Tests (n.s. = not significant). In Figure 4.3a, all SNS plots are compared with all RS plots. In Figure 4.3b, panels are split according to macrohabitat classification (W = natural or semi-natural closed habitats, G = natural or semi-natural open habitats, P = perennial crops, A = annual crops).

3.4 Exclusive species

Locally-exclusive species per buffer ws significantly higher in SNS than in RS (0.496 ± 0.167 vs. 0.334 ± 0.164, *p* = 2.736e-05; Figure 4.4).
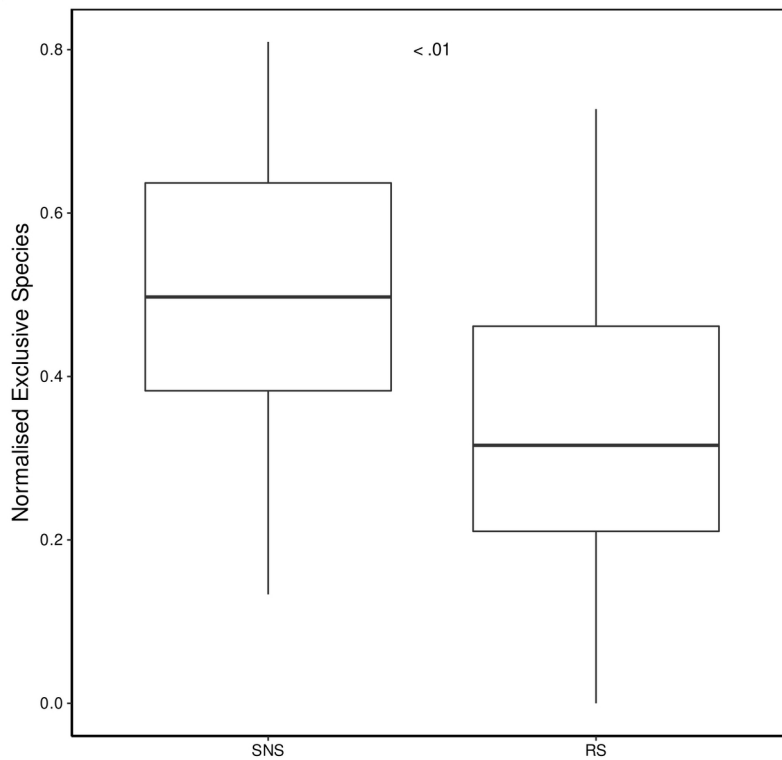
Figure 4.4. *Boxplots comparing Normalised Exclusive Species between Sacred Natural Sites (SNS) and Reference Sites (RS). Normalised Exclusive Species is calculated for each buffer (i.e. the circular area with 250 metres radius around each SNS and RS) by dividing its number of locally exclusive species by its total species richness. Text on the upper portion of the panels represent the p-values of two-sample Wilcoxon Rank Sum Test.*

3.5 Rarefaction

3.5.1 Beta diversity

Rarefaction of beta diversity showed no differences between SNS and RS when considering Whittaker's turnover and Jaccard's dissimilarity (Figure 4.5a). In contrast, rarefaction of Cody's index showed SNS had significantly greater beta diversity than RS, despite the difference was relatively small. Indeed, confidence intervals showed almost no overlapping and at 56th plot SNS reached 861.91 (95% CI: 851 - 872.82) and RS 802.45 (95% CI: 797.53 - 839.07).

Rarefaction of beta diversity by habitat showed that W and A plots of SNS were more diverse than the counterpart found at RS (Figure 4.5b), in particular Cody's index was larger for SNS both in the case of W and A plots, whereas Whittaker's turnover was larger only for W plots.
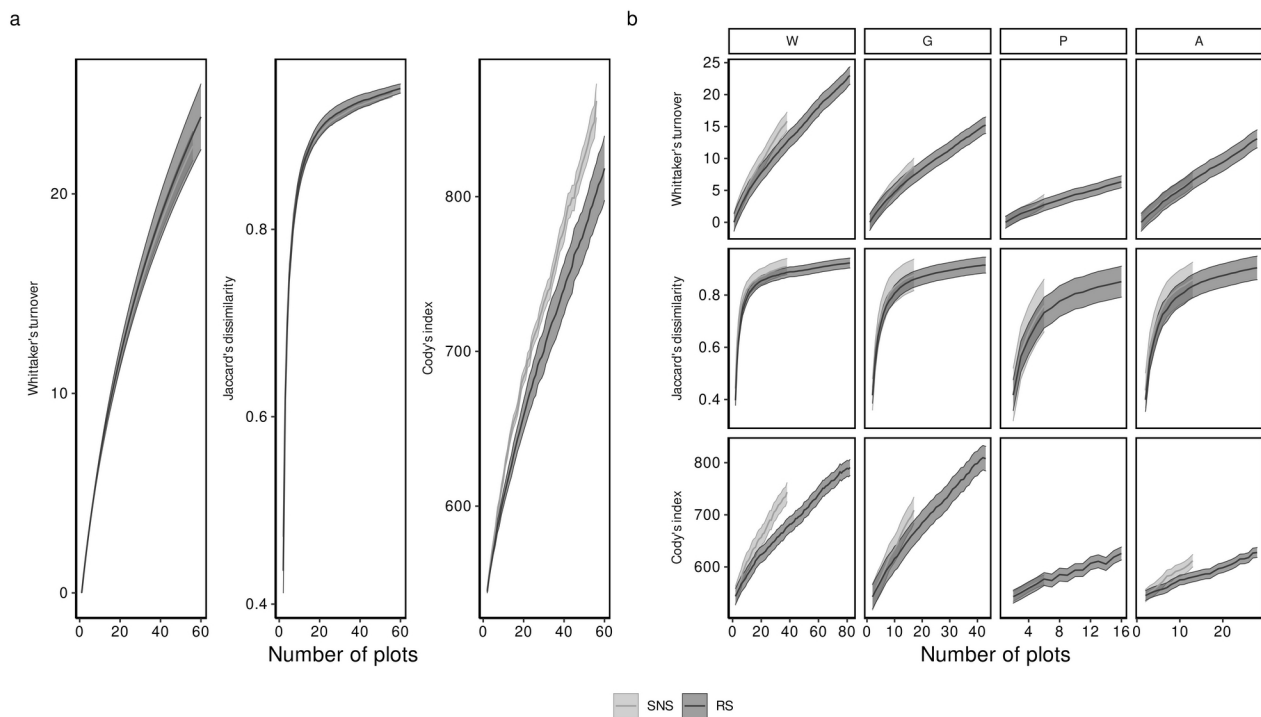
Figure 4.5. Spatially-explicit rarefaction curves of three indices of beta diversity of Sacred Natural Sites (SNS) and Reference Sites (RS). In Figure 4.5a, rarefaction curves of SNS and RS plots calculated through a resampling procedure in order to avoid bias due to sampling design. For each site we randomly extracted up to two plots of SNS and RS and built the corresponding rarefaction curves, we repeated the procedure 100 times, final curves and confidence intervals were calculated on the resampled curves. In Figure 4.5b, panels are split according to macrohabitat classification (W = natural or semi-natural closed habitats, G = natural or semi-natural open habitats, P = perennial crops, A = annual crops), classical spatially-explicit rarefaction was employed here.

3.5.2 Gamma diversity

As for beta diversity, rarefaction curves of gamma diversity showed that SNS were more diverse than RS (Figure 4.6a), with confidence intervals showing no overlapping and at 56th plot SNS reached 666.25 (95% CI: 637.35 - 695.15), whereas RS reached 548.53 (95% CI: 537.33 - 627.65). Also when calculating rarefaction curves of gamma diversity by macrohabitat, similar patterns as those of beta diversity emerged (Figure 4.6b). In particular, W and A plots showed significantly larger gamma diversity of SNS than RS, whereas G and P plots showed no clear differences.
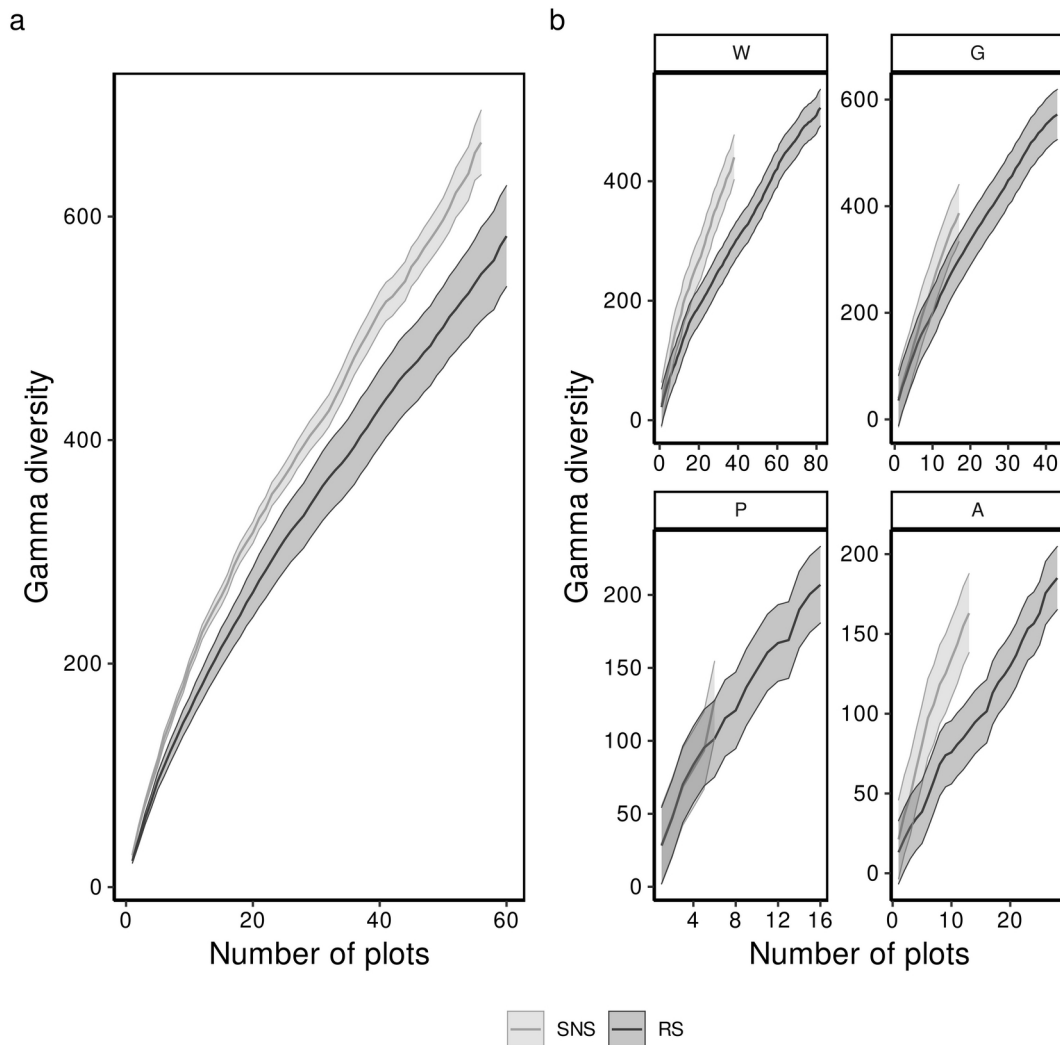
Figure 4.6. *Spatially-explicit rarefaction curves of gamma diversity of Sacred Natural Sites (SNS) and Reference Sites (RS). In Figure 4.5a, rarefaction curves of SNS and RS plots calculated through a resampling procedure in order to avoid bias due to sampling design. For each site we randomly extracted up to two plots of SNS and RS and built the corresponding rarefaction curves, we repeated the procedure 100 times, final curves and confidence intervals were calculated on the resampled curves. In Figure 4.5b, panels are split according to macrohabitat classification (W = natural or semi-natural closed habitats, G = natural or semi-natural open habitats, P = perennial crops, A = annual crops), classical spatially-explicit rarefaction was employed here.*

4 DISCUSSION

Generally, the results of this investigation comparing species diversity and conservation value of plant communities showed that SNS were more diverse than RS, across a range of sites spanning the whole Italian penisula. Indeed, an ever-growing body of literature has already revealed the opportunity provided by SNS to conserve biodiversity (Bhagwat & Rutte 2006, Dudley et al. 2006, IUCN-UNESCO 2008, Dudley et al. 2010). However, in the present work we evaluated it

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throughout Italy by taking advantage of one of the larger data collection in the topic, enlarging the scope of previous evidence focusing on Central Italy (Frascaroli et al. 2016).

According to our results, SNS in Italy are not significantly richer in species than RS, but they host rarer species assemblages and a greater proportion of locally-exclusive species. Moreover, the beta- and gamma-diversity accumulation curves of SNS were steeper than those of RS. Consequently, our hypothesis that SNS contribute disproportionately to biodiversity cannot be rejected.

Although Frascaroli et al. (2016) found SNS in Central Italy were richer in species than control sites, we observed no differences, which in turns seems quite common (see e.g., Mgumia & Oba 2003, Salick et al. 2007, Karun & Sridhar 2016, Avtzis et al. 2018, Kühnert et al. 2019). Moreover, this could be explained by the *intermediate disturbance hypothesis* (Connell 1978), which predicts an increase in species richness following moderate levels of disturbance. Indeed, SNS are typically visited and used by people, as well as their surrounding landscapes, the RS in our case. Therefore, disturbance has likely masked potential differences in species richness. Also the heterogeneity of the two datasets, the SNS plots and the RS plots, comprising very different plant communities, has potentially concealed differences in species richness. Indeed, some patterns emerged when splitting comparisons by macrohabitat, and thus limiting heterogeneity and comparing similarly disturbed habitats.

The rarer species assemblages as well as the larger proportion of locally-exclusive species and greater beta and gamma diversity of SNS confirm the significant contribution of SNS to regional biodiversity and highlight their fundamental role as OECMs. These patterns showed that the relevance of SNS for biodiversity conservation in Italy manifests both at local (i.e. landscape) and regional (i.e. national) scales. On the one hand, this could be the result of complex interactions among social taboos, traditional and continuous management, sustainable exploitation of natural resources and unique natural features. On the other hand, the patterns observed could rather be due to a more drastic exploitation of surrounding landscapes than that undergoing at SNS, therefore

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SNS would be areas where biodiversity is resisting rather than increasing. In both cases, it is clear that SNS in Italy are effectively acting as PA or at least as area-based conservation measures, therefore they should be taken into account in conservation planning. Moreover, a legal recognition of their role is desirable, to ease the persistence of governance structures and consequent management (Frascaroli et al. 2019). In contrast, assimilating the network of SNS into the existing network of PA should be avoided or at least done case-by-case following careful assessments, as it is known that incorporating SNS into PA can erode their governance structure, ultimately producing adverse effects on conservation (Dudley et al. 2009).

Woodlands and other closed habitats showed the most evident differences between SNS and RS, among the habitats taken into account. In particular, woodlands found at SNS seemed richer and more diverse than those found at RS, suggesting that SNS conserve a large spectrum of woodlands and similar habitats, which in turns are the typical focus for studies about SNS (Bhagwat & Rutte 2006, Dudley et al. 2010). Although previous evidence found that SNS in Central Italy conserve groves with not clear conservation value except for some large trees (Frascaroli et al. 2016), it seems that SNS in Italy can successfully conserve such habitats. Indeed, woodlands in Italy have historically being part of religious tradition. For instance the sacred groves of Camaldoli and La Verna have been managed by Camaldolite monks and Franciscan friars, respectively, for more than 8 centuries, although the current groves were planted at the end of 19th century (Manetti & Cutini 2006), and the sacred grove of Monteluco even dates back to the Roman Empire. Likely, the continuous traditional management of these woodlands has promoted great biodiversity and the existence of unique species assemblages. Therefore, the persistence of functioning governance structures and related management practices is going to be crucial to conserve biodiversity associated with these habitats in the future.

Annual crops also showed greater biodiversity when found at SNS rather than at RS. It is known that plant diversity of annual crops is largely affected by agricultural practices. In particular,

traditional extensive agriculture is known to promote greater diversity than modern intensive agriculture. In this framework, it seems that annual crops of SNS are more often managed in traditional ways than the annual crops of surrounding areas. Moreover, conservation of plant diversity of annual crops is of particular relevance both for sustainable agriculture purposes (Marshall et al. 2003) and for itself due to the presence of rare threatened species (Albrecht et al. 2016). While it seems that sound agricultural practices have been employed at annual crops of SNS, it is likely that additional conservation measures could not have any effect on their conservation value or even risk to erode it and hence should be avoided.

Differently from the woodlands and annual crops, grasslands and perennial crops showed no significant differences between SNS and RS. A possible explanation in the case of grasslands is that these habitats have similar disturbance regimes both at SNS and further from them, which in turns promote their great biodiversity, masking potential effect of SNS-related taboos and management practices. As for the perennial crops, it should be noted that few and very different plots have been sampled and classified under this category. In particular, milled and abandoned olive groves makes up the range of sampled plots. Therefore, the high heterogeneity and the small sample size are likely responsible of the absence of signal, rather than actual ecological or management-related causes.

5 CONCLUSIONS

In the present work, we demonstrated the conservation role of SNS in Italy, by sampling plant diversity at 30 sites across the country and related RS and comparing several biodiversity metrics. While SNS were not richer in species than RS, except for when we looked at particular habitats, they had greater beta and gamma diversity and they hosted rarer species assemblages and more locally-exclusive species. Moreover, we found that woodlands and annual crops at SNS had greater conservation value than those at RS, providing novel insights on SNS in Italy and more broadly in Europe. We call for a legal recognition of SNS in Italy, to help local community in maintaining this

precious biodiversity. However, incorporating SNS into actual PA should be avoided or at least carefully assessed case-by-case. Rather, we would like to stress the need for different conservation measures across different habitats, as we found SNS being effective in conserving only particular habitats.

Future research should cover areas and taxa that have not been studied in relation with SNS up to now, as large areas of Europe for instance. Moreover, a definitive evidence on the role of SNS for conserving biodiversity would also require to analyse other dimensions of biodiversity. In particular, phylogenetic and functional diversity of biological communities found at SNS have been seldom analysed, leaving a large gap that should be urgently filled.

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